Mine Environment Life-cycle Guide: epithermal gold mines

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The *Mine Environment Life-cycle Guide* has been developed with input from end-users including the Department of Conservation, Straterra, West Coast Regional Council, Waikato Regional Council, Northland Regional Council, New Zealand Coal and Carbon, OceanaGold, Bathurst Resources, Solid Energy New Zealand, Tui Mine Iwi Advisory Group – in particular Pauline Clarkin, Ngātiwai Trust Board, Ngāi Tahu, and Minerals West Coast. Thanks also to Malcolm Lane from Lane and Associates for advice on mine bonding arrangements.

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EXECUTIVE SUMMARY

Mining is an important economic activity in New Zealand, and it has a long history that is often linked with important cultural developments. The process of mineral extraction inevitably affects the surrounding environment, but there are few tools available to help mining companies and regulators assess and predict the environmental impacts of mining operations for New Zealand's geology and environmental conditions. The *Mine Environmental Life-cycle Guides* (MELGs) follow on from the *New Zealand Minerals Sector Environmental Framework* and the *Mine Drainage Framework* and have been developed as part of a collaborative research programme with key mining partners to assist with the planning of future mine developments in New Zealand. A separate MELG has been developed for different generic mine types: mesothermal gold, epithermal gold, and potentially acid-forming and non-acid-forming coal.

The MELGs bring together previous minerals sector environmental planning tools and provide guidance for minerals sector companies as they proceed through operations to closure and post-closure. They include guidance on stakeholder engagement (particularly iwi engagement) and seek to reduce the uncertainty surrounding the achievement of agreed post-mining outcomes for future, operating, and closed mines during planning, operations, and closure.

The MELGs result from a collaborative research programme between CRL Energy Ltd, University of Canterbury, University of Otago, University of Auckland, O'Kane Consultants (NZ) Ltd and Manaaki Whenua – Landcare Research, and they have been developed in conjunction with end-users including the Department of Conservation, Straterra, West Coast Regional Council, Waikato Regional Council, Northland Regional Council, New Zealand Coal and Carbon, OceanaGold, Bathurst Resources, Solid Energy New Zealand, Tui Mine Iwi Advisory Group, Ngātiwai Trust Board, Ngāi Tahu, and Minerals West Coast.

Specifically, the MELGs examine the geochemistry of mine waste rock and mine drainage, the freshwater ecology of mine-affected catchments, and the effectiveness of water treatment techniques and terrestrial rehabilitation techniques for different mine types and environments to achieve intended outcomes. The guides highlight innovations within the minerals sector in New Zealand using case studies, and incorporate current knowledge from the global minerals sector to provide guidance on state-of-the-art mine environment management practices. In addition, the economics of alternative environmental management options are assessed.

Increasingly, a social licence to operate is required to enable successful minerals sector development, which requires meaningful participation from iwi, local communities, and other stakeholder groups. Changes to mining legislation and regulatory guidance also place increased focus on iwi participation and engagement. For this reason, stakeholder engagement – particularly iwi engagement – from an early stage of mine planning is a focus in the MELGs. In New Zealand, 'everywhere is someone's back yard' compared to other countries, where mining often occurs in unpopulated or sparsely populated areas. Further, for iwi, rohe and areas of interest can be complex and overlapping. Although no generic approach to participation and engagement will be appropriate for all sites, there are some common themes that have been articulated by Māori, mining companies and regulators, and these are incorporated in the MELGs.

The MELGs are 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, assessments of environmental effects (AEEs) for resource consenting, and the development of resource consent conditions. The document should also help internal decision-making by mining companies and in Resource Management Act processes.

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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
As	Arsenic
CMA	Crown Minerals Act
CMER	Centre for Minerals Environmental Research
DOC	Department of Conservation
EC	Electrical conductivity
Eh	Oxidation/reduction potential
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies)
Fe	Iron
LLB	Limestone leaching bed
ICP-MS	Inductively coupled plasma - mass spectrometry
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
NZP&M	New Zealand Petroleum and Minerals
PAF	Potentially acid-forming rock
RMA	Resource Management Act
Sb	Antimony
TSS	Total suspended solids
WRD	Waste rock dump
XRF	X-ray fluorescence
Zn	Zinc

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). It is important to note that ABA testing often does not consider the potential for metalliferous drainage, and additional tests are required to confirm such effects.
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 acid rock drainage, but AMD arises because of human-induced changes to the rock mass through mining, mainly by exposing minerals to oxidation processes. Rocks at mine sites that are considered to be potentially acid forming (PAF) should be handled and disposed of in ways that minimise acid formation.
Acid Mine Drainage Index	A biotic index based on the relative abundance of specific aquatic macroinvertebrate genera, developed specifically to determine the impact of acid mine drainage.
Acid-neutralising capacity (ANC)	A 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	Encompasses mining- and non-mining-related sources of acidic drainage. This term could be used to describe acid drainage from a mine site or natural acidic drainage from PAF rocks, or acidity resulting from landslide that exposes PAF rocks to weathering.
Acidity	Acidity associated with acid mine drainage (or acid rock drainage), comprising mineral acidity (the hydroxide ion demand by cations of Fe, Al, Mn and others) and hydrogen ion acidity (measured as mg/L H_2SO_4 or similar units).
Adit	A horizontal passage leading into a mine for the purposes of access or drainage.
Baseline data	Environmental data used to establish baseline conditions for a proposed mining operation.
Benthic layer	Inorganic and organic material forming the streambed.
Biofilm	The community of algae, bacteria, and fungi within a matrix of polysaccharides adhering to the surface of the streambed substrata.
Coagulation	The addition of chemicals to reduce the net electrical repulsive forces at particle surfaces, promoting the consolidation of particles.
Colloid	A 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	A group of plant and animal populations interacting within a given location.
Contaminant	Any physical, chemical or biological substance that is introduced into the environment. This term does not imply an effect and usually refers to substances of anthropogenic origin.
Deposit	A 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, animal and micro-organism communities and their non-living environment interacting as a functional unit.
Eh	The Eh of a water sample indicates the electric potential difference of a system during the supply or uptake of electrons during oxidation and reduction reactions. Eh is measured in millivolts. 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.
Engineered landform	A landform comprising mining waste rock that has been designed, with specific construction criteria, to minimise oxygen ingress and the infiltration of water

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.
Flocculation	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 rate	Typically, the unit of interest when assessing the potential impacts of mine drainage. It is expressed in units of m^3/s or L/s.
Food web	A 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	A group of rocks that are recognisable over a large area (typically over tens of square kilometres) and that have similar characteristics such as age, composition, geological history or depositional environment.
Grazer	An organism primarily feeding on living algal tissue.
High wall	The sloping face of an open-cast mine – may also be referred to as the pit wall. More commonly used in coal mining to indicate the higher edge of the pit.
Historical data	Environmental data that have been collected previously. This may include data from locations affected by historical mine drainage or pristine sites.
Hydrogeology	Physical and chemical processes of water movement in rocks.
Hydromining	Mining of coal using a high-pressure water jet to extract coal, and gravity fluming to transport coal from the active mine face.
Iron oxyhydroxides	A general name that includes a wide range of orange and brown iron oxide minerals ('rust'), many of which are poorly crystalline. Iron oxyhydroxides form naturally in the weathering environment, but can be enhanced by mining activity, and commonly accompany acid mine drainage and neutral mine drainage. Also called 'yellow boy'.
Kinetic test	An analytical method for releasing dissolved components from rock, whereby the testing regime monitors changes in rock chemistry over 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 (MPA)	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.
Mine drainage	A collective term for groundwater, surface water runoff, and mine process water at a mine site.
Mine waste	A collective term for mine tailings, mine water, and mine waste rocks.
Non-acid forming (NAF)	Non-acid-forming rocks have excess potential to neutralise acid rather than produce acid. Non-acid-forming rocks can still generate metalliferous drainage.
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. In environmental geochemistry, oxidation often involves the addition of atmospheric oxygen. Oxidation occurs concurrently with reduction, whereby electrons are gained by an atom.
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+}, and Mn^{2+}.
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.
Potentially acid forming (PAF)	Potentially acid-forming rocks are rocks that release more acid than they consume, typically during oxidation or weathering.
Precipitation	The condensation of a solid from a solution.
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 physicochemical conditions and concentrations in the downstream environment.
Rehabilitation	The repair or improvement of a damaged ecosystem to meet defined ecosystem functions or attributes (e.g. native ecosystem, pasture system).
Remediation	A physical, chemical or biological action to remove contaminants with the goal of reducing and managing the risks to the environment or to humans. Remediation includes rehabilitation actions aimed specifically at treating or otherwise removing pollution or contamination.
Residence time	The length of time mine water spends in a passive treatment system.
Restoration	The process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (as defined by the Society for Ecological Restoration, in defining ‘ecological restoration’). In the past the term has been more strictly defined as ‘Return of damaged ecosystem to original state and function’, but this is very difficult to achieve. Most projects in New Zealand aim to achieve a healthy or functional condition, more or less equivalent to rehabilitation.
Semi-Quantitative Macroinvertebrate Community Index	A biotic index based on the relative abundance of specific aquatic macroinvertebrate genera.
Shredders	Organisms that primarily feed on coarse particles (>1 mm) of dead or decaying vegetation, including both leaves and wood.
Sod	Intact surface layer of ground containing vegetation attached to underlying roots. Sods may be 100 to 700 mm deep depending on equipment capacity and depth of the majority of the root mass.
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. A common example is gypsum ($CaSO_4 \cdot 2H_2O$).
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.
Suspended solid	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, a common unit of identification among similar individuals. It is 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, 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	The functional classification of organisms according to their feeding relationships. The basal level consists of the primary food resource (plants, algae, detritus), followed by herbivores, predators, etc.
Turbidity	A measure of the amount of light scattered by suspended particles in a sample, typically reported in nephelometric turbidity units (NTU).
Unconformity	The boundary between a group of older rocks and a group of younger rocks. An unconformity normally represents a long time gap in the geological record, where uplift and erosion have occurred. Alluvial gold can accumulate on unconformities during erosion.
Vegetative direct transfer	Precise excavation of large ‘sods’ or ‘slices’ of vegetation together with attached topsoil and root mass, their transfer in a single layer to a rehabilitation site, and placement in a way that maintains the integrity of individual sods. Trees and shrubs over 3 to 5 m tall are pre-felled prior to transfer of remaining vegetation and stumps with attached root plates.
Waste rock	Rock that does not contain any minerals in sufficient concentrations to be considered ore, but which must be removed in the mining process to provide access to the ore. Waste rock usually has no immediate value and is stored on the mine site or used in the construction of engineered landforms such as tailings dams. Waste rock may be *in situ* or excavated.
Waste rock dump	The closest point to an active pit for planned storage of waste rock.
Waste rock pile	Unplanned disposal of waste rock. In old documents, also referred to as ‘mulloch’ or ‘mullock’, being waste rock from which the mineral has been extracted or that is associated with extracting ore
X-ray fluorescence (XRF)	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 a lower atomic mass than fluorine.

1 Introduction

New Zealand has a long history of mining, starting with the extraction of pounamu (greenstone) by Māori, followed by European settlers extracting coal and gold in the 1860s. Most residential, commercial, and industrial products use minerals directly or during their production, and so mining will remain an important industry in the future. However, the removal of minerals from the ground may result in significant negative impacts on the environment. Mining companies are required to reduce and minimise these impacts, often in an increasingly stringent manner as our understanding of surrounding ecosystems and landscapes improves.

The *Mine Environment Life-cycle Guides* (MELGs) provide guidance for mining companies, regulators, and stakeholders. They are intended to provide consistency and transparency in decision-making for proposed mining operations, particularly those related to preventing or minimising environmental impacts. The MELGs are 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 (DOC), and in the resource consenting process, such as during consultation, assessment of environmental effects (AEE), and the setting of resource consent conditions. The MELGs may also be useful in developing future regional plans for water quality and terrestrial ecosystems.

MELGs have been developed for different mine environments: coal mining in potentially acid-forming and non-acid-forming regions, epithermal gold (this guide), and mesothermal gold. These guides build on the previous *New Zealand Minerals Sector Environmental Framework* (Cavanagh et al. 2015) and the framework for predicting and managing the water quality impacts of mining on stream ecosystems (Cavanagh et al. 2010) by incorporating new science, economic considerations, and stakeholder engagement – particularly with iwi.

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

- the interpretation of rock geochemistry and the prediction of mine drainage chemistry
- sampling strategies for rocks and streams, and analytical 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
- the optimal selection of active and passive treatment systems
- the identification and exploration of mitigation options with communities
- the identification of resources for rehabilitation and rehabilitation techniques.

1.1 Epithermal gold

1.1.1 Summary of activity

Epithermal gold deposits are located in the North Island and are mostly hosted in the 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. The 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 today at large open-cast and underground mines. Important mineral deposits mined in the last 50 years include the Tui deposit, Golden Cross, and Martha and related underground deposits. Tui is an andesite-hosted polymetallic epithermal-style deposit, which was 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 open-cast and underground methods, which ceased operations in 1998. Golden Cross is west of Waihī, at the south of the Coromandel Peninsula. The Martha deposit is a large quartz vein system, which occurs in andesitic rocks currently mined by open-cast methods. The Martha deposit has several satellite vein systems (Favona, Corenzo, etc.), which are mined by underground methods. Other small underground mines have been active in the last 10 years. Currently the New Talisman operation in Karangahake Gorge is producing from vein and alteration zones around historical workings.

Other epithermal-style mineralisation occurs in Northland and the Taupō 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 Taupō area rare vein mineralisation is hosted by ignimbrites. Northland and Taupō epithermal deposits are currently being explored.

1.1.2 Geology

Hard-rock epithermal deposits occur in Northland, and in Coromandel, extending southwest towards the Taupō 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 Taupō 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. Geothermal power stations exploit the same parts of the geothermal systems in which gold is deposited, and gold-rich precipitates have been found in modern geothermal power stations (Brown & Simmons 2003). 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 the formation of epithermal gold deposits involve circulation of groundwater, derived from rain, above a shallow heat source associated with molten volcanic rocks. As a result, there has been extensive fluid flow through fractures and cavities in the host rocks, which 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, which can be up to several metres across. These veins commonly have some calcite associated with them but not sufficient to compensate for the large amount of calcite removed from the 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, which 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). Deeper parts of epithermal deposits can be enriched in copper, lead and zinc. 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 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 (Simpson et al. 2016).

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, which is essentially identical to that part of the orogenic (mesothermal) process (Cavanagh et al. 2018). 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 content. Underground epithermal mines generate substantially smaller amounts of this altered waste rock than open-cast mines.

Mining of these deposits can occur as either large open cuts or in underground tunnels. Open-cast gold mines produce large amounts of waste rock. At New Zealand's largest epithermal open-cast mine, the waste rock generated is used to construct embankments for the tailings impoundment. These tailings impoundments are complex in design, with different types of waste rock used for different parts of the dam structure depending on the geotechnical and geochemical properties of the waste rock. Tailings at these mines include all material that is crushed and processed to liberate gold. Tailings volumes are also large, and these are deposited as a slurry within the impoundment, and water is separated for recycling through the ore processing circuit.

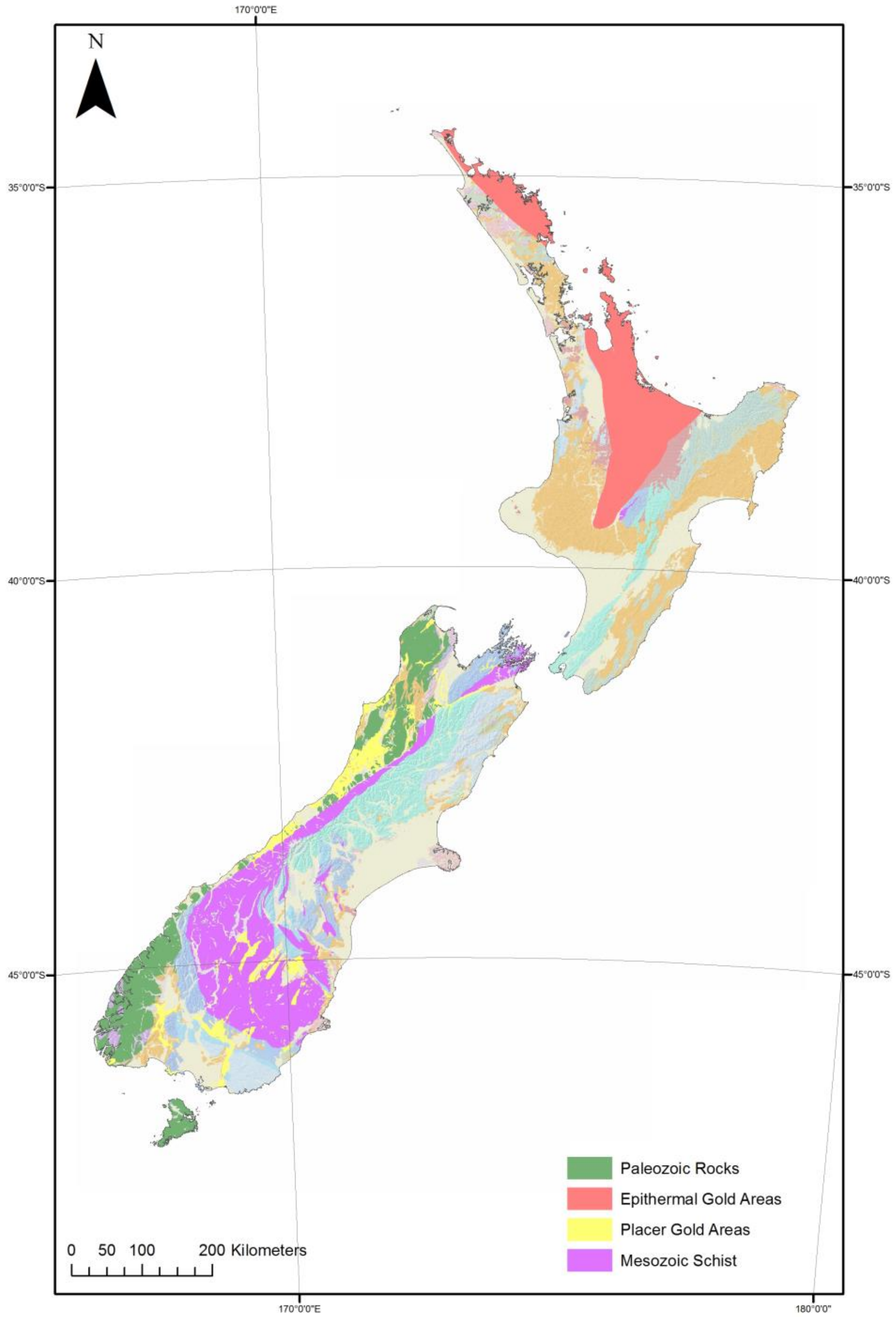

Figure 1. Location of gold provinces in New Zealand. The pink areas outline where the rocks that host epithermal gold deposits occur.

1.2 Regulatory requirements

The following section provides an overview of regulatory requirements associated with mining operations in New Zealand, particularly as they relate to environmental impacts on streams and the terrestrial environment. This information is not intended to replace New Zealand Petroleum & Minerals (NZP&M), regional council, New Zealand Heritage or DOC advice, and interested people should contact the relevant council and DOC to discuss any specific enquiries. The information provided in this document is correct as at July 2018, and users are advised to check with NZP&M, relevant councils, New Zealand Heritage, and DOC for any updated requirements.

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

- a permit granted under the CMA[1]
- an access arrangement negotiated with all landowners and occupiers[2]; for land administered by DOC there is a formalised application process (discussed further in section 1.2.2)
- resource consents (e.g. covering the use of land and water, discharges to water and air, changes to traffic patterns and roads, and other aspects of the operation), which are granted by district and regional councils under the RMA.

In addition, permits are required for any activities that may affect surface or underground archaeological sites (defined as pre-1900) through New Zealand Heritage, and any activities that affect native animals defined in the Wildlife Act 1953. Where native animals are identified, permits (from DOC) and mitigation actions may be required for drill-site exploration as well as mining (see section 1.2.3).

The CMA and RMA require that all persons exercising functions and powers under the Acts take into account the principles of the Treaty of Waitangi. More detail on how this should occur under the CMA is contained within the *Minerals Programme* developed under the CMA – the current minerals programme was released in 2013 (NZP&M 2013). Specifically, the *Minerals Programme* outlines the ways in which the Crown's responsibility to have regard to the principles of the Treaty are met in the context of mining. This includes a requirement that Tier 1 permit holders submit an annual report on their engagement activities with iwi and hapū whose rohe are directly affected by the permit. Early and meaningful engagement with affected iwi about planned activities by mining companies is more likely to result in the development of productive relationships with mutually beneficial outcomes. Further discussion on iwi engagement and participation is provided in section 1.3.4.

The purpose of the CMA is to promote prospecting for, exploration for, and mining of Crown-owned minerals for the benefit of New Zealand. Thus permits or licences granted under the CMA are largely related to the technical feasibility and economic aspects of mining. However, the granting of permits includes consideration of good industry practice, feasibility, and project economics, which in turn requires recognition and management of potential environmental impacts. Granting of access to land (by DOC) and resource consents have a greater focus on the management of negative environmental impacts from mining activities.

The MELG is intended to facilitate robust mine planning through guidance on information requirements and interpretation to inform mine feasibility studies, project economics, good industry practice with respect to minimising the environmental impacts of mining activities[3], access arrangements, and resource consents, as well as outlining optimal methods and approaches to identify and meet agreed post-mining environmental outcomes.

1.2.1 Mining permit

NZP&M (under the Ministry of Business, Innovation and Employment) manages the Government's oil, gas, coal, and minerals under the CMA. Anyone wanting to prospect, explore or mine the Government's resources must obtain a permit from NZP&M. Specific regulatory requirements are also provided in Crown Minerals (Minerals Other than Petroleum) Regulations 2007 and the *Minerals Programme* (NZP&M 2013), and further information is available at http://www.nzpam.govt.nz/cms.

As an overview, three main types and two tiers of permits are available. The three types of permits are for:

- prospecting
- exploration
- minerals mining.

Under the CMA, 'prospecting' means 'any activity undertaken for the purpose of identifying land likely to contain mineral deposits or occurrences, and includes geological, geochemical, and geophysical surveying, aerial surveying; taking samples by hand or hand held methods; taking small samples offshore by low-impact mechanical methods'.

'Exploration' is defined in the Act as 'any activity undertaken for the purpose of identifying mineral deposits or occurrences and evaluating the feasibility of mining particular deposits or occurrences of one or more minerals; and includes any drilling, dredging, or excavations (whether surface or subsurface) that are reasonably necessary to determine the nature and size of a mineral deposit or occurrence.' Granting of an exploration permit includes consideration of whether the proposed exploration is in accordance with good industry practice, and whether the proposed exploration work programme will enable a commercially justifiable decision to be made on the development of a particular mineral deposit before the permit expires.

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[1] If the minerals are privately owned, a permit under the CMA is not needed, but all of the other permits and consents are still required, together with the consent of the mineral owner.

[2] In some cases negotiation with landholders may be required to cross their land to access the land in which the minerals exist.

[3] Noting that good industry practice as defined in the CMA (section 2[1]) does not include any aspect of the activity regulated under environmental legislation.

For a minerals mining permit, 'mining' is defined in the Act as meaning 'to take, win, or extract, by whatever means, a mineral existing in its natural state in land, or a chemical substance from [that mineral]'. In determining whether to allow a permit, the Minister will consider the applicant's mining feasibility studies, which include mine design, scheduling and production; resource recovery; and economic viability / project economics – in particular the financial viability and technical constraints, and the proposed level of expenditure in relation to the scale and extent of the proposed operations, and whether the proposed mining operations are in accordance with good industry practice.

The two tiers of permits are:

- Tier 1: includes all prospecting permits, exploration permits for which anticipated work programmes will exceed a specified total expenditure for a given mineral, and mining permits for which annual production or royalties exceed specified amounts over the durations specified in the *Minerals Programme* (NZP&M 2013)
- Tier 2: all those permits that are not Tier 1 (includes alluvial gold).

Permits will not include land unavailable for prospecting, exploring or mining. These areas may include defined areas of land requested by an iwi or hapū that have particular importance to the mana of the iwi or hapū (these areas are listed in Schedule 3 of the *Minerals Programme*), Crown land described in Schedule 4 of the Act for all activities except certain activities described in section 61(1A), and other land that is not available due to a legislative requirement. NZP&M hold a list for the latter land, which is available on request. In addition, there are areas in New Zealand where the mineral rights are private and are usually attached to the occupancy title, and at these sites permits issued by NZP&M are not required, although resource consent and access arrangements are required.

Before a Tier 1 exploration or mining permit is granted, the ability (capability and systems) of an applicant to meet the environmental requirements for the type of activities proposed under the permit is considered.

After a permit is granted, annual reporting is required for all permit holders. For mining permits this includes feasibility studies that relate to mine design, scheduling, production, resource recovery or economic viability, and a summary of the status of consents or applications for consents, including access arrangements and resource consents that affect the ability of the permit holder to give proper effect to the permit. The MELG is intended to assist with concept to feasibility studies, providing guidance on information requirements and interpretation to identify and meet agreed post-mining environmental outcomes.

1.2.2 Land access arrangements

Anyone wishing to undertake mineral-related activities requires permission from the landowner and occupier for access to the land. The access arrangement requirements of private landholders, including Māori, will vary. The *Minerals Programme* (NZP&M 2013) sets out the general requirements. For minimum impact activities, written consent of the landowner/occupier is required for certain classes of land, including land regarded as wāhi tapu (a place sacred to Māori). The CMA contains further specific provisions for entry onto Māori land for minimum impact activities that are outlined in the *Minerals Programme*. This includes that reasonable efforts have been made to consult with those owners of the land able to be identified by the Registrar of the Māori Land Court, and at least 10 days' notice is given to the local iwi authority prior to entry onto the land. Other activities may not occur except in accordance with an access arrangement agreed in writing between the permit holder and each landowner/occupier.

Conservation land

In the case of operations occurring on public conservation land, permission is required from the Minister of Conservation, and in some cases access decisions are made jointly, by or on behalf of the Minister of Conservation and the Minister of Energy and Resources. The considerations, and the privileged position of mining relative to concessions, are discussed in a 2010 report *Making Difficult Decisions: Mining the Conservation Estate*, by the Parliamentary Commissioner for the Environment.[4] There is no formal link between requirements for access agreements with DOC and resource consents.

Permission for minimum impact activities is granted by DOC by way of a consent, while an access arrangement is required for all non-minimum-impact activities (mining, non-minimum-impact exploration, and prospecting). In 2018 there were 113 approved mining operations on conservation land, and 54 of these are active. However, in November 2017 Prime Minister Jacinda Adern announced that there would be 'no new mines on Conservation land'. Options for implementing the policy will be consulted in a document that was to be released in September 2018[5] - although to date it has not been released - and until then the status quo continues; previously only Schedule 4 land was excluded from mining.

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

[4] https://www.pce.parliament.nz/media/1301/making-difficult-decisions.pdf.
[5] https://www.beehive.govt.nz/release/public-views-be-sought-%E2%80%98no-new-mines-conservation-land%E2%80%99-policy-implementation

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

Public conservation land is divided into different categories, ranging from stewardship land to national park status. Each regional district (ex conservancy) has a conservation management strategy, which provides a plan for land managed within the specific region and describes the inherent values and purpose of each category of land managed. The conservation management strategy also outlines the policies associated with granting access arrangements.

Under Section 61(A) of the CMA, applications for access are unable to be accepted for land that is listed in Schedule 4 of the Act (e.g. wilderness areas, nature reserves, national parks). Access to the minerals (and consequently land containing the minerals) under a minimum impact activity consent or access arrangement can only be granted to land within a minerals permit. For any mining-related activities on public conservation land that are outside the minerals permit area (e.g. access roads, buildings, processing facilities), a concession may be required. It is recommended that you contact DOC staff in the relevant district prior to commencing the application process. DOC has also developed *Mining Activities on Public Conservation Land: An Applicant's Guide* [6], which provides information and guidelines on the steps for DOC processing and administration of CMA access arrangements and minimum impact activity consents.

For further information on access for mineral-related activities on public conservation lands and the relevant application forms and guidelines, visit the DOC website[7], or contact the Hokitika Shared Service Centre, which processes all mining-related access arrangements located on public conservation land and can be contacted by:

- email: permissionshokitika@doc.govt.nz
- post: Private Bag 701, Hokitika 7842
- phone: +64 3 756 9117.

1.2.3 Resource consents

Resource consents will be required for mining activities. The RMA set outs the responsibilities of regional and local authorities and applicants. Regional councils issue four types of resource consent, covering:

- land use and disturbance
- water use
- discharges to water, land, air (including noise)
- any activity in the coastal marine area.

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

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

General guidance on applying for resource consents has been developed by the Ministry for the Environment (*An Everyday Guide to the RMA: Applying for a Resource Consent*, MfE 2015). Individual councils provide information on their websites about obtaining resource consents and may indicate specific information requirements. For example, the West Coast Regional Council (WCRC) requires an annual work programme to be completed as a condition of the consent. Templates for coal and gold mining are provided on the WCRC website[8]. These require information on the general operation of the site, rehabilitation, and water management. Similarly, Waikato Regional Council has templates for mining and quarrying activities[9]. Consultation with the relevant council will provide specific information on requirements, templates for applications, and relevant plans. As with access arrangements, applicants should discuss their application with the relevant regional council to ascertain current requirements.

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[6] http://www.doc.govt.nz/Documents/about-doc/concessions-and-permits/mineral-exploration/applicant-guide-mining-activities-conservation-land.pdf
[7] http://www.doc.govt.nz/get-involved/apply-for-permits/mining/
[8] http://www.wcrc.govt.nz/consents/mining.htm
[9] http://www.waikatoregion.govt.nz/consentforms/

Implementing consent conditions and management plans is the primary means by which environmental impacts from mining activities are managed. Conditions should be linked to specific monitoring and annual reporting of performance and compliance. Bonds are set at a level to ensure compliance with the resource consent conditions in the event that a mining company stops works, and these may form part of the negotiations during resource consenting (see section 3.8 for further details on bonding). In setting consent conditions, a balance needs to be met between prescriptive conditions that are straightforward to verify, and conditions that allow for flexibility in how the desired outcomes are achieved. Consent conditions can specify that environmental values lost through mining are compensated, or mitigated elsewhere – this may be through payments to a specific fund or organisation, or completion of specific off-site projects. Such environmental or biodiversity compensation (or offsets) is typically required when the mine site affects native ecosystems or habitat of native wildlife. Such consent conditions can be covered with targeted bonds.

1.2.4 Other statutory instruments to be considered

Wildlife Act

Prospecting and mining activities may also need a wildlife authorisation from DOC under the Wildlife Act 1953. This is needed to catch, handle, release or kill native wildlife, or to hold dead specimens of wildlife. Schedule 7 lists native terrestrial and freshwater invertebrates declared to be animals under the Act. This includes most wētā, some native weevils and beetles, katipō and Nelson cave spiders, and native land snails. A decision-making tool to help decide which authorisation is needed is provided by DOC.[10] Any vegetation clearance (such as drill-site prospecting) in native ecosystems may be anticipated to affect native lizards, snails and beetles, if present, and so requires an authorisation.

National Policy Statement for Freshwater Management

In 2014 the Government enacted the National Policy Statement for Freshwater Management (NPS-FM). The NPS-FM outlines a series of national objectives and policies that require the maintenance or improvement of water quality, including the protection of outstanding water bodies and wetlands. The NPS-FM includes objectives to safeguard life-supporting capacity, ecosystem processes, and indigenous species, including their associated freshwater ecosystems.

The NPS-FM also included a National Objective Framework. The Framework defines water quality attributes (e.g. nutrients, dissolved oxygen) for which national bottom-line values are proposed. Freshwater ecosystems that exceed the bottom-line values need to be managed to improve their water quality. Additional attributes continue to be developed. The National Objective Framework requires local government to manage all freshwater systems for 'compulsory national values' (e.g. ecosystem health, human health, and mahinga kai). The NPS-FM and Framework will have direct effects on the management of freshwater systems receiving mine waters. At the time of writing, no metal attributes have been proposed as part of the Framework.

Where historical mines have negative effects on surface waters through acid and metal discharges, the potential for modern open-cast mining to mitigate existing degraded surface waters can be a benefit, and may be useful to explore when considering new mining operations. The types of activities that may reduce impacts include removal of overburden, capping that prevents oxygen and water entering underground workings, or diverting acid drainage into treatment facilities. Such 'cleaning up' of historical mine sites can also include revegetation and weed control, and can be useful compensation, especially where enduring benefits result across environmental and cultural heritage; for example, by supporting tourism (see Case Study 1, on Globe Progress mine, p. 23).

Regional plans and guidelines

Individual regional councils may have regional plans containing rules that apply directly to mining operations and their effect on the environment, including outstanding natural landscapes. These rules must be taken into account in a resource consent application. Regional plans for the management of water will also typically outline the classification of different aquatic systems according to the water quality classes set out in Schedule 3 of the RMA. Many regions also have specific standards and requirements for the control of storm-water discharges from earthworked areas. These standards may reference the erosion and sediment control guidelines 'TP90' (Technical Publication 90, Auckland Regional Council 1999), which have been updated by GD05 (Guideline Document 2016/005, Auckland Council 2016), or the Environment Canterbury equivalent.[11]

Each region also has a unique pest (plant and/or animal) management plan. These plans identify and categorise pest plants and animals that must be controlled in the manner and to the extent prescribed by the relevant plan, and may cover areas planned to be mined. For example, the West Coast Regional Council Pest Plant Management Plan, updated in 2018, identifies weeds to be eradicated (e.g. woolly nightshade), excluded (e.g. tree privet, bushy asparagus and hornwort), and species to be controlled via

[10] http://www.doc.govt.nz/get-involved/apply-for-permits/which-authorisation-do-i-need/.
[11] http://ecan.govt.nz/publications/General/FullErosionandSedimentControlGuideline.pdf).

'Good Neighbour Rules', for example, by maintaining a defined 10 m boundary clearance for broom and gorse where the adjacent property is managing these species.

However, there are advantages to managing a broader range of weeds on mine access roads and mine sites to a more stringent level than that required by this current plan, because many weeds are relatively straightforward to exclude, and if allowed to establish will smother and slow, prevent or alter the natural succession of native ecosystems. The Buller coal measures, in particular, can be vulnerable to weed invasion by species tolerant of acidic and low-fertility conditions once disturbed; these include gorse, broom, *Juncus squarrosus*, knotweeds, cotoneaster, Spanish heath, wilding conifers, silver birch and *Agapanthus* (among others). A very broad range of weeds can grow and reproduce on the West Coast, but because of low population have not yet established.

Heritage New Zealand Pouhere Taonga Act 2014

Heritage NZ have a statutory role to protect archaeological sites (i.e. where any site is pre-1900). DOC is a signatory to the International Council on Monuments and Sites charter and has an interest in more recent structures or remnants that reflect the heritage of a site, and can produce heritage assessments or provide management advice, etc. In the context of mine sites, including the rehabilitation of abandoned mines (which are often located on conservation land), the identification and retention of any heritage fabric should be an integral part of site assessment, exploration and mine planning. Consideration of surface and underground mining heritage will often go hand in hand with rehabilitation and compensation.

The first part of assessing the heritage of a site is to map the area. This can include establishing a transect baseline with right-angle distances that locate each piece of iron, beams, and other relics, each of which is recorded with photography and a description. Three-dimensional scanning and drone photography may also be used (see Case Study 1). Subsequent activities can range from salvaging objects, to the restoration and development of interpretive signage to provide the historical context and enhance tourism.

Case Study 1: Conserving historical mining sites

New mining operations are often in areas of historical mining activity. Management of historical mining artefacts can be a highly valued part of mine rehabilitation and compensation. The Globe Progress open-cast mine is in an area of historical underground coal and gold mining within the Victoria Conservation Park. This mining heritage includes walkways, abandoned mining equipment, buildings and other mining history, stretching from Reefton to the Waiuta area (https://www.reefton.co.nz/discover).

Construction of the tailings dam affected parts of the Alborns Coal Mine. Alborns was an underground mine that started in about 1880 and operated at a large scale from about 1935 before being abandoned in the 1950s (https://www.reefton.co.nz/see-and-do/walks-and-hikes/alborns-coal-mine-walk). The area was not protected under Heritage New Zealand Pouhere Taonga. A six-part mitigation package mine was agreed with DOC to preserve the legacy of the 'old time' Alborns coalminers and enhance a resource for tourism.

Figure C1. Fully restored Alborn mine coal truck on original track.

The six parts are:

- a full, high-resolution photographic and survey record using a laser scan, which allows the area to be digitally visited, much like Google street view (Figure C2)
- relocation of all recoverable artefacts, in consultation with DOC, to publicly accessible areas of track, ideally in areas that provide suitable context
- conservation of remnants of a make-shift winch constructed from a c. 1915 Leyland truck by building a roofed shelter over the truck to slow its degradation
- full restoration of a coal truck by a heritage specialist in Christchurch, which was then placed back on old tram tracks with the Leyland truck that would have winched it up (Figure C1)
- development of interpretation material explaining the truck, its role in the tramway winching operation, and the mining operations
- re-routing and upgrading of the Alborns Coal Mine / Buck Gully Mine walking track.

The first two steps are typical minimum requirements for preserving the heritage of a site.

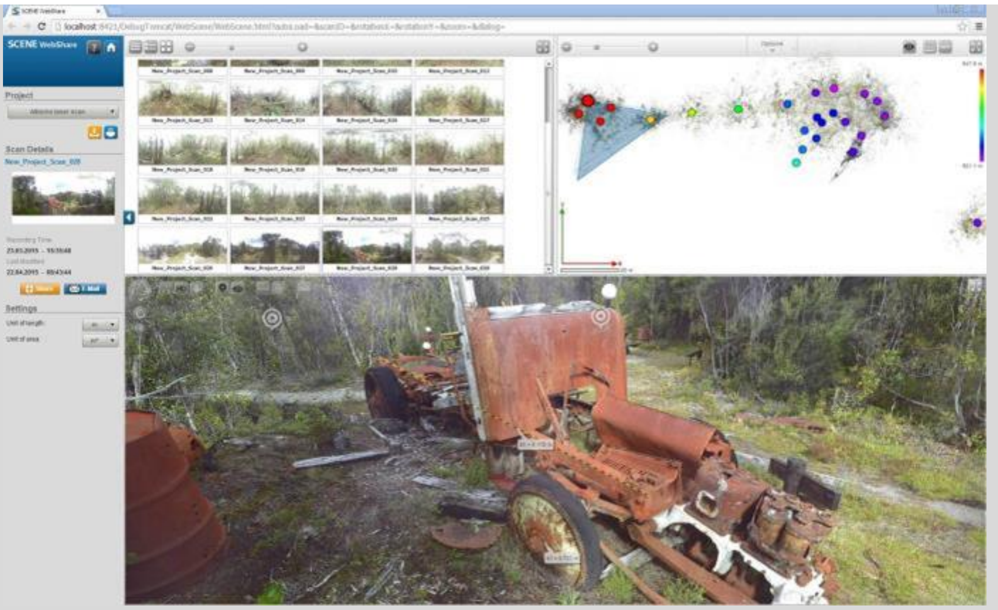

Figure C2. Part of a high-resolution digital photographic record for Alborn Mine.

Key findings relevant to mine rehabilitation:

- The Heritage New Zealand Pouhere Taonga Act 2014 requires an authority before works begin in any place with pre-1900 human activity (http://www.heritage.org.nz/), but even relatively recent machinery (1950s) can be part of significant heritage. Only a small proportion of archaeological sites in New Zealand are in reserved areas, so consult DOC or Heritage NZ if your site has surface or underground history.
- Protecting historical artefacts and sites from deterioration, enhancing their condition, safe accessibility and/or interpretation can create legacy assets for the local community. The latter includes using techniques to capture virtual images.

Key references:

Jones K L 2007. Caring for archaeological sites. Department of Conservation. https://www.doc.govt.nz/globalassets/documents/science-and-technical/sap243entire.pdf

OceanaGold 2015. Mitigation of the historic Alborn's coal mine and remnants. Minerals West Coast presentation.

Historic heritage publications and fact sheets from the West Coast: https://www.doc.govt.nz/our-work/heritage/heritage-publications/by-region/west-coast/

Figure C3. Historic explosive magazine foundations with interpretation signs, Globe Progress

1.3 Decision-making

Decision-making related to minerals sector developments/proposals or to compliance with resource consents is usually completed through the RMA process, which might include regional council notified hearings with commissioners or through the Environment Court. An alternative process for projects of national significance is available through the Environmental Protection Authority in New Zealand, and this involves a hearing with a board of commissioners. For large minerals sector operations and the setting of environmental bonds, decision-making is typically completed by an expert review panel comprising representatives from the mining company and regulators. These panels usually meet annually to assess changes in the monitoring data from the mines, or changes in the scale of operation with respect to bonding.

A critical component to feed into decision-making is early and meaningful engagement of the mining company with affected iwi and hapū, in recognition of their role as kaitiaki of the land. These early conversations may identify specific 'no go' areas due to their cultural significance or where the impacts of mining activity are unable to be satisfactorily mitigated. These conversations, and those with a wider set of stakeholders, also help to identify 'what matters'. This is the basis on which the conditions (e.g.

approaches for rehabilitation) that will deliver 'what matters' can be identified. Starting with the end in mind guides mine planning to enable agreed post-mining outcomes to be achieved cost-effectively. Further information on iwi and stakeholder engagement is provided in sections 1.3.4 and 1.3.5.

The purpose of the MELG with respect to decision-making is to provide guidance on what information is required at different stages of mining activity to best ensure that environmental impacts are minimised throughout mining operations in the most cost-effective and satisfactory manner. The following sections provide an overview of the environmental effects of mining epithermal gold regions, how they can be mitigated, and the consequent cultural and economic impacts.

1.3.1 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 (usually downstream) from the mining activity. For example, environmental effects that are largely restricted to the mine site and nearby areas include those arising from clearing vegetation and habitat modification, modification of soil profiles and drainage regimes and slope, or subsidence. These effects include impacts on animals with territories or home ranges that intersect with stripped or disturbed land, particularly some native birds such as kiwi, fernbird and pipit, and bats.

Environmental effects that occur away from the mine site include increased noise, light (where lighting is used at night), dust deposition, and impacts on downstream water quality and ecosystems. Adjacent and 'upstream' ecosystems can be impacted if activities affect the passage of fish, invertebrates or terrestrial animals (both pests and native animals) through the mined area. Some changes to topography and surfaces have visual effects beyond the mine site, particularly if ridgelines and/or tall trees or rocky outcrops are removed. Such environmental impacts are also likely to have an impact on cultural values, including the life-giving properties of water, such as the ability to nourish plants and animals that are potentially food (mahinga kai) or fibre. Sites with specific spiritual guardians (e.g. taniwha) are particularly important (see also section 1.3.2).

In some cases, major effects of mines are associated with new access roads, especially in steep or remote land within the conservation estate. The roads can be sources of sediment, barriers to fish passage (where they cross watercourses), and long-term passages for pest plants and animals to expand their ranges.

Current expectations of mining operations are that environmental effects, whether on or off site, will be avoided, mitigated or minimised (e.g. ACSMP 2011). 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. In addition, where mining occurs on conservation land, mining is expected to provide a net conservation benefit. This recognises that the greatest threat to conservation land is not mining but introduced plant and animal pests, that current and projected public funding is not adequate to protect native biodiversity on public (DOC) lands, and that mining can provide revenue and/or additional pest control over parts of the conservation estate, provided the compensation is additional to existing conservation funding, and the duration or quantum of mitigation is adequate (PCE 2010).

A particular focus for the MELG is the effects on downstream aquatic systems arising from mine drainage, because they can be severe and are among the most difficult mining-related environmental impacts to predict, mitigate, manage or remediate. The presence or absence, severity, extent and duration of chemical impacts on downstream ecosystems depend on a complex variety of local, regional, natural, and anthropogenic factors. Operational management, including timely and effective rehabilitation, or water treatment, is often required to prevent downstream aquatic impacts.

Open-cast mining removes terrestrial ecosystems in order to access ore, and its effects are generally easier to understand and predict than downstream 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, as well as the remediation techniques that are adopted. This MELG focuses on predicting the quality of mine drainage, potential ecological impacts caused by mining and mine drainage, the management/treatment of mine drainage, and optimal rehabilitation techniques at New Zealand mine sites to achieve agreed post-mining outcomes.

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 the potential to chemically interact with mineralised rocks. Mining results in increases in the reactive surface area of rocks and the exposure of minerals to surface water and oxygen. The vast volume of material moved and processed at mine sites in combination with reactive minerals means that mine drainage waters develop variable compositions that are distinctly different from natural background waters (Plumlee & Logsdon 1999a, b).

Epithermal gold 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), copper (Cu), zinc (Zn), lead (Pb), manganese (Mn) and/or other 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 lower concentrations of dissolved components.

AMD 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 c. 1,000 mg/L), extremely elevated trace element concentrations, low alkalinity, and increased chemical oxygen demand. Trace elements that can have elevated concentrations in epithermal gold mine drainage, include Zn, Cu, Pb, Mn, cadmium (Cd), mercury (Hg), arsenic (As), antimony (Sb), or others. At low pH these metals are very soluble in water and are transported downstream in dissolved form. Some of these metals become less soluble with increasing pH and can precipitate or adsorb to substrates at various pH thresholds. The precipitates are commonly Fe 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 2).

Figure 2. **AMD-contaminated water discharging from Adit 5, Tui mine.**

Epithermal gold mines and rocks that host gold mineralisation in New Zealand can typically produce both acidic and neutral drainage enriched with trace elements, commonly Zn, Cu, Pb, Fe, Mn, Cd, Hg, or others. Where natural weathering processes release these trace elements, concentrations remain relatively low and dilution or attenuation in the downstream environment is rapid. Where mining releases these trace elements the concentrations are higher and treatment of mine-impacted water is often required prior to discharge into the wider environment (Goldstone & MacGillivray 2002; Pope & Trumm 2015; Trumm & Pope 2015). Where there are historical workings or wastes related to epithermal gold mining, often the concentrations of trace elements is extreme (both in solid materials and nearby water) and the related geochemical and mineralogical systems are complex (Haffert 2008, 2009; Druzbicka 2015; Fairgray et al. 2016; Fairgray & Webster-Brown 2017). Based on understanding epithermal mineralisation, information from current mining activities and case studies at historical mines, geo-environmental models can be constructed that provide predictive value for future resource development related to this type of mineralisation.

Aquatic biological effects

This assessment of the response of biological communities to mining has focused on stream (moving water) ecosystems. Only one study, which sampled 31 ponds and pit lakes as a Master's study (Bunder 2015), has been undertaken to assess how lake and standing water communities respond to mining activities.

Stream ecosystems support diverse and often distinct communities of plants and animals. Most people probably think of fish, but almost all streams also have large communities of microbes, algae, plants, and invertebrates. These organisms interact to form food webs (Figure 3), which support animals higher up the food chain. Almost all New Zealand freshwater species depend on other freshwater plants and animals as their food, and comparatively few rely on terrestrial prey. As a result, any disruptions or contamination of waterways by mining activities can lead to reductions in the number and diversity of the higher animals in our freshwater ecosystems.

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 they 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. reducing the decomposition of organic matter) or the habitat of stream organisms (Figure 3).

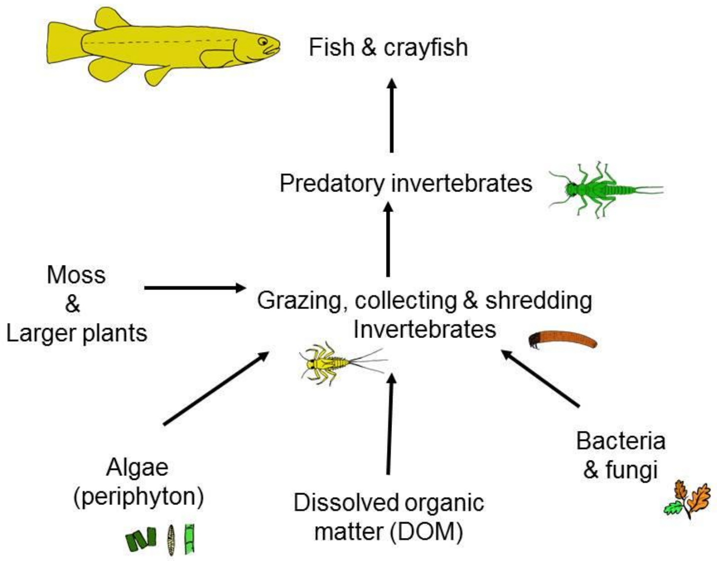

Figure 3. A simplified food web of a stream ecosystem. Effects of mine drainage on fish can be direct, such as toxicity from low pH, or indirect, such as effects on algae, which in turn affects invertebrates and fish.

Biological effects from epithermal gold mining may arise from high metal concentrations and/or high sediment loads. 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. 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 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, high metal concentrations have the most potentially severe impacts on life in streams. Toxicity from dissolved metals probably has relatively little effect on microbes, algae and higher plants and mosses, but it can have effects on invertebrates and fish, reducing diversity and changing community composition.

Aquatic biota (e.g. algae, invertebrates, and fish) are useful for monitoring the influence of mining and the subsequent treatment of stream ecosystems because they reflect a history of water quality across an organism's life. Macroinvertebrates (invertebrates larger than 0.5 mm) are commonly used for biological monitoring in New Zealand because they:

- are easy to sample and relatively easy to identify
- are long lived, and thus reflect water quality changes over time
- have variable tolerance to stressors such as low pH and elevated metal concentrations.

In New Zealand, fish and algae are more challenging to use for biological monitoring. Algae are difficult to identify, and ecological knowledge on their responses to mine drainages is still being developed. Fish are charismatic components of stream 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 stream may not be indicative of current or past water quality. Under certain circumstances riverine birds may also be useful indicators, although no studies have been done on these for New Zealand mine sites.

This document focuses on macroinvertebrates for biomonitoring, both in-stream and also for toxicity testing (e.g. to assess the toxicity of treated or untreated mine drainage). In the foreseeable future other useful biological indicators of stream health might become available, and these could be used to complement macroinvertebrate data.

Multiple mine inputs can occur along a river system, so impacts can be cumulative. These cumulative impacts may result in mild impacts higher up a river system and severe impacts downstream. However, if marked point-source mine 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.

Terrestrial (biological) effects

Assessment of the response of terrestrial biological communities to mining in the MELGs has focused on native forest/shrubland and exotic pasture ecosystems, as these have been the most common types of vegetation removed by mining activities. Pasture and cropping land is straightforward because:

- measures of success are established, such as crop production or pasture kilograms of dry matter per year relative to regional averages or unmined ('control') areas on the property
- desirable soil characteristics and profiles are often available and quantified in terms of rooting depth, soil fertility, and water supply
- fertiliser regimes are readily used to boost production in the short term.

In some cases, production of crops in rehabilitated land can be improved, especially if post-mining soil and topography overcome factors that limit production pre-mining (e.g. drainage or site uniformity is improved). In addition, whole-farm efficiencies can be achieved by raising land above flood levels or reconfiguring races, paddocks, and water reticulation, or by adding shelter belts or stand-off pads. New water supplies (from dredge or sediment ponds), relocated farm tracks, and stream crossings can not only improve farm efficiency but also improve environmental outcomes by excluding stock from waterways. Most farm plans do not address amenity, but this is readily addressed through landscaping, particularly where trees are included, and watercourses/lakes are required to be designed to deliver aesthetic as well as engineering outcomes.

In contrast, the magnitude of effects of mining on native forest and shrubland ecosystems is typically severe and lasts for many decades to centuries, as tall trees, vertical layers of plants (structural diversity), and deep layers of leaf litter and fallen logs that typify native forest take from decades to centuries to develop. Further, many native vertebrate and invertebrate species are adapted to the humid, sheltered and stable environment of a forest, not young, rehabilitated areas. In addition, most native forests include many different plant species, unlike pasture or cropping, and these have diverse mycorrhizal and invertebrate associations.

As with aquatic effects, indirect effects can occur if the removal of habitat affects the food supply (e.g. invertebrates for birds), vital habitat (e.g. large cavity-forming trees for bats or birds), or the ability to move through the landscape (e.g. due to roads or high walls, or bare areas. Indirect effects include the flows of pest plants and animals, which are often adapted to disturbance: this is particularly problematic for wind-blown weeds such as pampas, silver birch, pine or willows, which can quickly colonise bare ground if upwind of a new site. Such biosecurity risks are increasingly important, especially with the spread of new organisms such as myrtle rust and kauri dieback, and awareness of potential impacts on New Zealand's unique fauna and flora from pests hitch-hiking on equipment, people and plants grown off-site in nurseries.

This document focuses on plant, soil, and topography monitoring as tools to assess the potential difficulty of, progress in, and success of rehabilitation. While bird and lizard (skink and gecko) monitoring, and some specific rehabilitation of habitat in mined areas, is increasingly common for larger mine sites, and native frog monitoring has been part of gold-mining assessment in the Coromandel for over 20 years, monitoring of terrestrial invertebrates is relatively uncommon. An exception has been mines in the Brunner coal measures, where intensive monitoring and some captive rearing of *Powelliphanta* snails (*P. augustus* and *P. patrikensis*) has occurred over the last 10 to 12 years. Most rehabilitation has assumed that if the desired plant species are present and healthy, the invertebrate and other components of the ecosystem will gradually re-establish.

Reducing impacts – management and treatment

Impacts on stream ecosystems from epithermal gold mining can be mitigated through management, particularly of mine waste (mine water, waste rocks, processing residues, soils and soil amendments); prevention of sulphide oxidation; minimisation of water movement mobilising any oxidation products; water treatment techniques; or a combination of these. In general, best management practices to prevent or reduce the formation of contaminated mine drainages and high total suspended solids (TSS) will be more cost-effective than ongoing treatment of mine-affected drainages. The most common practices to reduce the formation of AMD or high TSS runoff are to separate 'clean' and potentially contaminated runoff (using diversions); minimise the area that is bare, or 'open', and therefore vulnerable to generating sediment; and minimise contact of water (and air) with materials that may generate contaminants. 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.

The impacts of mining on terrestrial ecosystems can be mitigated to some extent by adopting a mitigation hierarchy; i.e. first avoid harm by minimising the footprint (area) and extent of impact, and then focus impacts on ecosystems that are more resilient to removal and rehabilitation (e.g. young shrubland, pasture or non-native shrubland). Both direct and indirect impacts can be magnified or mitigated by the rehabilitation methods used, especially the extent to which soils, trees, and wood are replaced, and particularly soils with intact litter layers and living plants (known as direct transfer or community translocation). Operators of mines in high-value ecosystems are increasingly committing to maximising direct transfer, as this is one of the most effective methods to quickly re-establish a range of plant communities with their soil and litter invertebrates.

Where harm is unavoidable, conditions of wildlife permits will require specific organisms to be surveyed, and sometimes translocated, and this is usually restricted to specific seasons to reduce impacts on breeding populations or maximise translocation success. The residual ecosystem components should be salvaged, separated, stored or stockpiled in conditions, and for durations, that optimise rehabilitation. In nearly all cases, a priority is to salvage topsoil and favourable growing media, avoiding its degradation by using it immediately on areas ready for rehabilitation, or stockpiling it without mixing with lower-quality materials or allowing pest plants to establish. The speed of recovery of rehabilitated native ecosystems is also influenced by the characteristics of the adjacent and surrounding land, including access roads. Adjacent land in similar ecosystems is often an important source of natural plants and invertebrates, so it is beneficial to manage these areas to reduce the impacts of pest animals, and to limit the edge effects associated with disturbance (such as eroded sediment, or barriers like high walls).

Management to minimise AMD during operations – overburden management

Activities during mine operations to minimise the formation of AMD are a critical first step to minimise mining impacts on adjacent streams. Activities to reduce mine drainage impacts focus on preventing or reducing the contact of oxygen and/or water reaching acid-forming materials, minimising water entering the mined area, and neutralising or reducing the concentration of contaminants present in any mine drainage. Methods to achieve these goals include evaluating the factors that influence mine drainage at each site and applying appropriate site-specific management options to reduce the amount of affected mine drainage (e.g. 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 mine waste management techniques. 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 adopted, and that monitoring and performance data enable strategies to be optimised or 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. This includes mine wastes and overburdens used to construct access/haul roads, and places where mine wastes are temporarily stored (in case these are used longer term or become permanent).

Water 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), power, and regular operation and maintenance, and are reliable. Their main advantages are that they are effective at removing acid and metals from mine drainage 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 operational 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 the (bio)geochemical processes to occur, which means 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 so sufficient residence time in the system for this dissolution to occur is required (Skousen et al. 2000). In the long term, treatment using passive systems is typically more economical than using active systems, especially after mine closure (Skousen & Ziemkiewicz 2005). At closed and abandoned mines the water chemistry and flow rate are typically more stable than at active mine sites, and land is usually more readily available for treatment systems – factors that fit well with passive treatment.

For drainage from epithermal gold mines, elevated trace element concentrations are typically the primary component that must be removed. However, the specific elements that are most enriched, or most difficult to treat, are highly variable both between mineral deposits and at different seeps on the same deposit. Treatment of mine drainage should be considered for all mine sites, both during operations and after closure. This requires assessment of the predictions about mine drainage chemistry and flow rates, and assessment of the land available for treatment systems, since some systems require more land than others. This document shows how to decide between active and passive treatment, and how to decide among the various types of systems within each category.

Terrestrial rehabilitation

Rehabilitation of the terrestrial environment is the main way that recovery of areas affected by mining is achieved and the long-term effects of mining are minimised. Broad rehabilitation principles and key specific outcomes (e.g. areas to be avoided, features to be reinstated) should be planned during mine feasibility studies. Rehabilitation long-term outcomes and (shorter-term) closure success criteria should be proposed alongside a concept rehabilitation landscape plan as part of environmental impact assessment (see Case Study 2). These will be finalised during consultation, resource consenting and mine access permitting, and in most cases should be explicitly referred to in consent and access conditions.

Case Study 2: Linking long-term rehabilitation goals to closure criteria and short-term measures

Mining is usually consented based on agreed long-term rehabilitation outcomes. These may be described, shown in concept plans, and/or based on 'reference' sites. Where the reference condition takes a long time to develop, as for forests and most native ecosystems, shorter-term outcomes are needed. These shorter-term outcomes usually include 'closure' conditions, when the mining company can relinquish resource consents. Closure conditions need to be supported by measures at the time of landform and root zone construction, and plant establishment. The Cypress mine, an open-cast coal mine in native forest and red tussock wetland ecosystems, is an example where long-term, closure, and early revegetation conditions were developed.

Long-term outcomes across landscape were illustrated in a concept plan (Figure C4) and described as 'ecosystems similar in plant and animal species diversity and functioning to those premining ... with blending of constructed landforms' [into the natural landscape]. Exclusions were identified: coal measures sandstone pavements were not rehabilitated, and high walls would be created. Rehabilitation closure criteria were developed to be achievable within 10 to 15 years of initial revegetation and were specific to different types of rehabilitated vegetation (Table C1). The very low tolerance of weeds reflects the high ecological values of the area pre-mining and its vulnerability to weeds. The closure criteria ensure rehabilitated areas have a high certainty of developing naturally in the desired ecosystems with minimal ongoing maintenance. In addition, the rehabilitated footprint needed to support 1,000 *Powelliphanta patrickensis* snails is an acid test of success, as these mainly eat earthworms. The closure table was supported by conditions including various management plans and monitoring, both reviewed by independent peer reviewers.

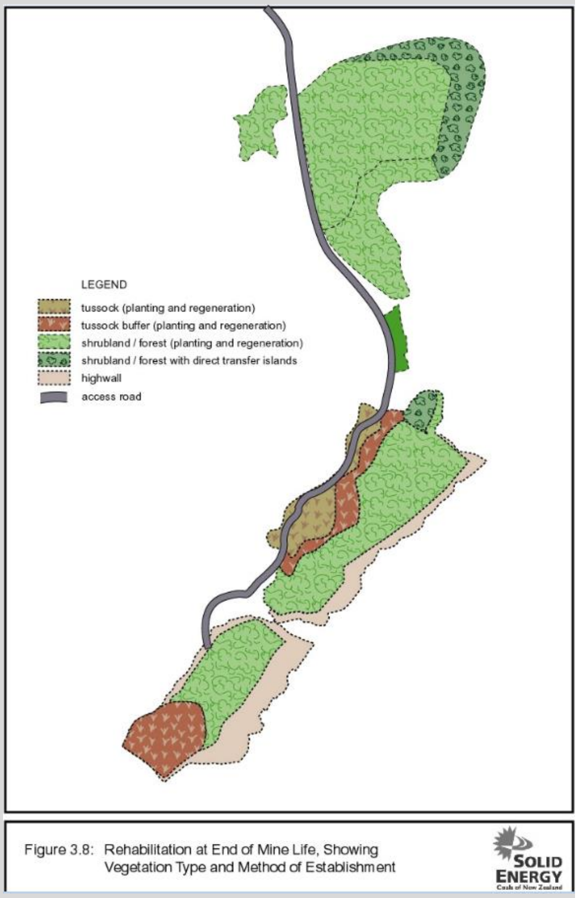

Figure C4. Concept plant for Cypress mine.

The closure criteria were supported by, and are consistent with, short-term descriptive outcomes: stable landforms with native-dominated plant cover and erosion-resistant surfaces, and physical and chemical conditions favouring sustainable plant communities. These were in turn supported by specific short-term criteria, which were not part of the consent conditions (Table C1).

Table C1: Closure criteria and interim rehabilitation criteria for the Cypress mine, 2011

Major landform	Short-term aim (3 or 12 months)	Closure
Tussock planted into backfill	Low-sloping landform (<5 degrees) Required hydrological characteristics Min. topsoil depth = 300 mm Min. tussock density = 2/m^2 Min. tussock basal diameter = 50 mm	Mean native vascular plant or rock cover >90%, including >75% tussock that is >0.3 m height. *Juncus squarrosus* cover <2%
Forest/shrubland <18 degrees slope (except high walls)	Min. topsoil depth = 100 mm; total rooting depth 500 mm over minimum 1.5 m non-acid forming overburden Minimum plant density = 1/m^2 Minimum of 3 native species/10 m^2 Cover of coarse wood or rock = 5 to 40%	Mean plant, rock or wood cover >90% at 0.5 m height or >75% cover at 1 m height. Minimum 5 native species/10m^2 No visible flowering or seeding gorse, broom; *Juncus squarrosus* cover <2 %

Figure C5. Cypress area in 2004 with red tussock wetland in the foreground and forest on the hills.

Key findings relevant to rehabilitation:

- Long-term, broad rehabilitation outcomes should be presented as a concept landscape plan with supporting descriptions, including reference sites, where relevant. These must be consistent with closure criteria.
- Closure conditions need to be supported by measures that can be assessed at the time of landform and root zone construction, and plant establishment. These allow early identification of potential problems.

The concept plan and closure success criteria will inform a draft closure plan, which should be prepared at the beginning of mining and updated regularly as more information becomes available. These may be required to be updated annually, alongside the annual mine plan, to reflect changes in information and in the economics of the mine. Target long-term rehabilitation outcomes may be based on reference sites, but the success criteria for mine closure are usually an intermediate condition, especially for native forest ecosystems that take many decades to develop. While success criteria may be relatively prescriptive, the best outcomes are often achieved by allowing flexibility in the way outcomes are achieved. Rehabilitation is just one opportunity to minimise the impacts of landscape clearance and change associated with minerals access and extraction. At each site there are also opportunities to avoid the area, shorten the duration or minimise the extent of impact, and (often) to achieve off-site mitigation. Mining offers opportunities to change land uses, mainly by manipulating land topography, drainage and runoff, the surface soils or covers, and the plants.

There are two key principles. The first principle is to *start with the end in mind* by developing an agreed concept of what successful rehabilitation will achieve in the medium and long term. This is useful to show as a landscape plan, supported by photographs and descriptions of the different sub-sites/ecosystems that are described. These help remove doubt about the overall intent of rehabilitation, so that when mine plans change, the concept on which agreements were based can be maintained. From these, the intermediate points of mine closure, and the starting point for initial landscape, soil profiles and plant coverings, can be developed. Developing an agreed concept needs engagement from the ultimate landowners and/or custodians/kaitiaki. The agreed concept will be based on identifying 'what matters'.

From this, the conditions that will deliver 'what matters' can be identified. 'What matters' and how these might be rehabilitated is likely to require technical knowledge to be integrated with local knowledge. At the same time, it can be very useful to define specific outcomes that are not wanted, or are unacceptable; for example, high walls over a specific height or area (due to landscape and fragmentation effects), or pit lakes, or slopes over specific gradients that mean a paddock is not trafficable, or streams that cannot support mahinga kai such as tuna (eels). In the Brunner coal measures, preventing final landforms with acid-generating overburden that are exposed to water and/or air is critical, and this is managed by identifying PAF materials so that they can be specifically managed.

Rehabilitation options are nearly always constrained by the resources available for rehabilitation, the space available to store these resources until areas ready for rehabilitation are available, and what is needed to rehabilitate the target ecosystems. The available resources (people, equipment, soils, plants, beneficial overburdens and rocks, fauna), post-mining landforms (dictated by pre-mining topography, swelling factors and geotechnical stability) and the mine schedule influence and potentially dictate the range of outcomes that can be achieved. This range of possible outcomes forms the envelope within which specific post-mining rehabilitation landforms, ecosystems and success criteria can be agreed. Sometimes historical surface activities compromise rehabilitation options, most commonly because surface soils and vegetation have been removed or contaminated, and pest plants have established seed banks. Any rehabilitation involving forest or tree growth, or deep leaf litter layers, will generally take a long time to develop, compared to rehabilitation of pasture or seral native ecosystems.

The second principle is to *create a flexible rehabilitation plan* that can adapt to a changing mine plan. It is rare to have full knowledge of the characteristics of the geology of the resource, or overburden, or specific rehabilitation methods for site ecosystems before mining. Economic cut-off grades for resource extraction or the resource-to-overburden ratio depend on resource value and input costs (diesel, labour), which may rapidly change. They may also be influenced by site-specific costs and risks of rehabilitation, accumulated as rehabilitation progresses. Consequently, mine footprints (including access roads and ex-pit stockpiles) may extend or contract.

In larger, longer-term developments it is not unusual for personnel, contractors, and companies to change. Written records of the broad rehabilitation principles (the agreed concept plan), priorities, and success criteria, along with key specific outcomes, are therefore important. Adaptive management informed by principles and the results of monitoring gives miners in longer-term projects (10 years) the flexibility they need to achieve agreed outcomes in the most cost-effective manner. This is why it is important to have some areas rehabilitated very early in a mine's life – usually the access road (cuts, fills and stockpiles) and initial stockpiles of stripped soils and overburden. The stockpiles and ex-pit dumps are where the growth potential of potential root zones would be confirmed. They help inform the main risks to successful rehabilitation and develop responses that are triggered if these adverse or unexpected outcomes occur (e.g. a topsoil or shortage of non-acid-forming material). This process helps focus on aspects where responses are unproven. Triggering of bonds is typically a last resort action and rarely results in good environmental outcomes.

Rehabilitation is generally based on creating landforms that are, first, 'safe' (for people and specific animals) and geotechnically stable to specified design earthquake and storm events. Landform design and development should ensure that root zones and contours that support the post-mining land use(s) can be created and maintained. This must include the development of suitable hydrological/drainage regimes and water bodies (wetlands, ponds, lakes and streams), as well as ensuring agreed water quality. Ecosystem restoration may require the construction of specific habitat features (e.g. wetlands, tarns, log piles, wood habitats in or out of flood zones, rock outcrops or rock refugia, or specific plants), or the avoidance of specific barriers (high walls or roads). Farm rehabilitation is likely to include specific requirements for water reticulation, paddock sizes and fencing, and soil chemical fertility (e.g. Olsen phosphorus).

Finally, vegetation establishment and change in plant species and composition over time (succession) influence post-mining values, in combination with the types and intensity of maintenance. Maintenance is critical for areas with components vulnerable to pest animal browsing or predation, pest plant invasion, or fire (especially under climate change scenarios, and for sites with public access). Maintenance needs usually reduce rapidly once a dense plant cover has developed, as the plants eliminate bare soils with high light levels, and this limits the establishment of new species.

This MELG outlines a process, and the information required, for effective planning and implementation of mine rehabilitation to selected outcomes: farming (pasture), plantation forestry, and native ecosystems.

1.3.2 Cultural impacts

Environmental effects arising from mining activities are likely to have an impact on a range of cultural values, on public land, or adjacent to private or Māori land, where mining activities can impact landforms, natural resources, waterways and special places, under various forms of management, including co-managed areas. Many of these areas are ancestrally connected (through whakapapa, ngā uri or descendants, and ancestral lands associated with stories of tīpuna) and are significant to local people (tangata whenua), especially where areas contain sacred, spiritual, or culturally significant sites. In many cases, local kaitiaki and hapū will be responsible for the area long after the mining company has gone, even if they do not own or jointly manage the site.

Mining activities can therefore seriously impact the life-giving properties, energy, or life essence (mauri), of water (wai) and land (whenua), including the ability to support and nourish important plants and animals (taonga species) that are culturally significant, including food harvest sites (mahinga kai), areas for growing fibre materials (e.g. harakeke, raupō), sacred sites (wāhi tapu, urupā), or sites associated with specific spiritual guardians or kaitiaki (e.g. taniwha). Specific issues may also arise where impacts have the potential to affect natural resources and taonga such as greenstone (pounamu).

A central goal for most tangata whenua/kaitiaki groups is maintaining or enhancing the mauri of these cultural values and sites through kaitiakitanga (guardianship). The location where mining activities are proposed will be an important consideration for iwi/hapū, particularly with regard to determining whether mining should proceed, what post-mining consequences and effects are likely to occur, and how that relates to Māori aspirations and desired outcomes. In some circumstances, maintaining the mauri of the location may be addressed through offsetting activities during mine operations. Some areas considered for mining may be wāhi tapu (sacred sites of special spiritual significance to iwi/hapū), in which case there is likely to be considerable resistance to that land being mined. Early and effective engagement resulting in the development of a meaningful values-based relationship with iwi/hapū will help to identify the aspects, issues, and priorities of concern, and to develop appropriate approaches and guidelines for redress and action (see section 1.34).

1.3.3 Economic impacts

Economic impacts arising from mining can be wide ranging. The Total Economic Value framework (e.g. Pearce & Turner 1990), shown in Figure 4, provides an economic view on how the financial gains (profits) from a mine relate to its economic, social, intangible and cultural values. The framework distinguishes between use value, (quasi-)option value, and non-use value. Use values derive from actively using a mine site. The mineable mineral resource has consumptive direct-use values, meaning that extracting the minerals depletes this value-generating attribute of the mine site. These values are directly related to the financial gains from a mine and are used to pay for certain social effects, such as jobs and addressing complaints, and to effects on the local economy from, for instance, service and supply contracts. Before mining commences, the site may be used for non-consumptive active uses: perhaps people use it for weekend hikes or ceremonies. During the operational phase of mining these values may be strongly diminished, but rehabilitating the site may partially or fully restore them. Targeting rehabilitation may also provide for new use values that were not available before the start of mining.

The site may also generate indirect use values, reflecting benefits that can be enjoyed without actively pursuing them. A hilly or mountainous range provides a nice view, for instance, or may be an element in local culture (e.g. the maunga could be an ancestor) and folklore. If extracting minerals requires the hilltop to be removed, these non-use social or cultural values are reduced and rehabilitation may never mitigate the loss.

A mine can also create indirect use disbenefits and associated negative values. During the operational phase a mine can create higher traffic volumes, noise and dust, and raise demand for schooling or housing. These disbenefits stem from mining activity rather than the site *per se*, but given that the one is inseparable from the other, these effects must be considered. Disbenefits include the creation of tailings dams that contain toxic materials, or mine waste overburden landforms that generate acid leachate.

Option value and quasi-option value are values that come from waiting for some form of further information. While they may be useful concepts to use in dynamic optimisation problems (such as optimising extraction rates over time), their relevance to mine environments is limited and is disregarded in further discussion.

Non-use values derive from mental perceptions and have no bearing on physical experiences. Consequently, non-use values are also sometimes referred to as intangible values. For a landscape that contains minerals, two well-known non-use values are bequest value (the desire that future generations can enjoy the landscape as it currently is) and existence value (a sense of pleasure that the landscape is what it is). Other benefits that can be included in non-use values arise when people (individually or as a group) identify with the landscape.

Indirect use values are likely to overlap with non-use values. If an individual's culture is linked to the landscape, then that person will probably want his or her children to be able to experience it as well. There is little guidance in the cultural heritage literature about the delineation of social, cultural, heritage, and historical value (Eppink et al. 2017).

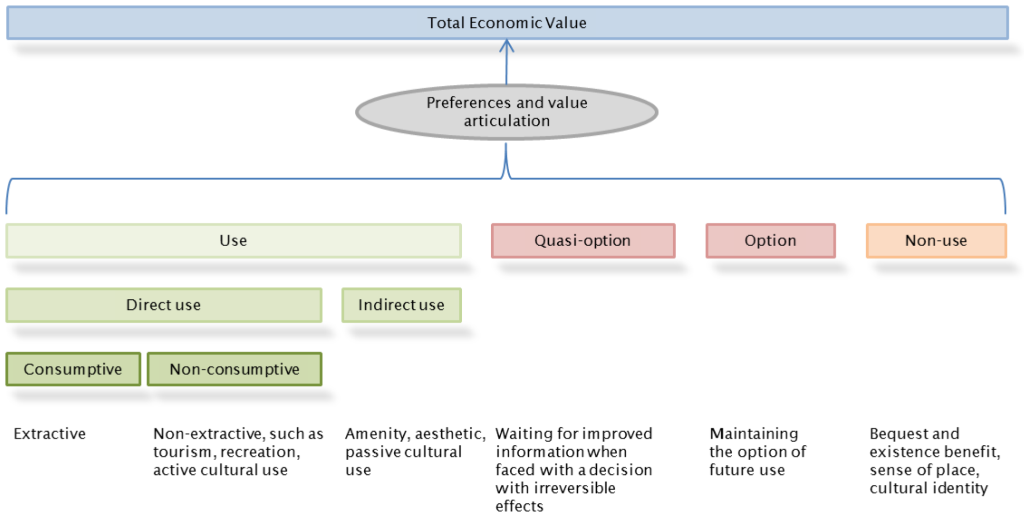

Figure 4. **Illustration of the Total Economic Value framework.**

In the context of this guide, the focus of economic discussion is on project economics relevant to bonding and economic tools that can assist in the establishment of consent conditions.

Bonding

Generally, companies are required to lodge a bond with a regional council to provide financial assurance that the funds required to complete site closure are available even if the company that operates the site is unable or unwilling to close it (e.g. Peck & Sinding 2009). The practice of bonding minimises the exposure of councils and ratepayers to the short- and long-term environmental and financial risks that are created when a mine is opened

The financial risk associated with mines in New Zealand is in recent practice covered by two bonds that cover distinct types of risk. One is the 'performance' bond, which reflects the rehabilitation cost if a site needs to be closed at the time the bond size is calculated. The other is the 'post-closure' bond, which reflects long-term risks that will remain after the mine has been closed. This post-closure bond includes extreme occurrences such as a failing tailings dam, and more likely issues such as future seepage from waste rock stacks and corrective actions that would be required after, for instance, 20- or 50-year rain events, or earthquakes.

If there are foreseeable ongoing costs, such as those that would arise from ongoing treatment of mine drainage or regularly cleaning out drains, these are included in the post-closure bond. For these costs, bond conditions dictate that, upon successful closure of the site and release of the bond, the share of the bond that covers these costs that are expected to continue in perpetuity is relinquished to a trust (or similar entity) that is charged with administering this ongoing care. Further details on bonding arrangements are provided in section 3.8.

Changes in the mine permitting regime now place greater emphasis on tighter cost analyses, which means quantification of the costs associated with environmental management and achieving final closure objectives for the project during early phases of resource development is required. However, there will always be a higher level of uncertainty around these costs during the pre-feasibility stage (mine permit application), which reduces as mining operations proceed and more information on the resource and environmental management is gathered.

1.3.4 Iwi engagement

The value of early and genuine engagement with iwi/hapū cannot be overstated. This engagement should recognise the important role of tangata whenua (local people) as kaitiaki of the land, and allows for the early identification of cultural values, issues, and culturally significant sites such as wāhi tapu (sacred places), wāhi taonga (treasured places, flora fauna, habitats) or wāhi tūpuna (places of ancestral significance). In all these areas there is likely to be strong opposition from tangata whenua/iwi/hapū to mining because of the cultural importance of these places, concerns of impact, and the significant values attached.

Best practice and early engagement will lead to clarification of issues and articulation of desired culturally desirable post-mining outcomes. This engagement can be considered effective when a values-based and respectful relationship develops from the onset between the iwi and the mining company – with ongoing engagement leading to respect between parties, and meaningful dialogue and contributions from all parties. This in turn requires that the knowledge and language (e.g. terminology) used to express the effects of mining operations is understood and shared by all parties within a collaborative learning environment.

Iwi engagement in mining activities has been given greater effect since changes to the CMA in 2013, which require that Tier 1 permit holders provide an annual report (iwi engagement report) to NZP&M describing the holder's engagement with iwi and hapū where the tribal rohe (area) includes some or all of the permit area, or who otherwise may be directly affected by the permit. Guidance on engagement is available at http://www.nzpam.govt.nz/cms/iwi-communities/working-with-iwi-hapu/industry-engagement-with-iwi, which includes links to best practice guidelines for engagement with Māori in relation to mining, developed by Te Rūnanga o Ngāti Ruanui Trust, and based on their experiences of the development of the petroleum and minerals industry in the Taranaki region.[12] These guidelines provide an excellent starting point for planning engagement and echo many of the sentiments expressed by iwi representatives in the development of this guide. Specific guidelines have also been developed by other iwi; for example, 'Kā Rūnaka expectations for petrol and oil companies in East Otago' (Ruckstuhl et al. 2017), which outlines expectations for the manner of engagement, and also some of the specific concerns for the iwi relating to resource extraction activities within their tribal rohe.

Protocols outlining the way in which the Ministry for Business, Innovation and Employment will consult with specific iwi and hapū have also been developed under the CMA and can provide a useful starting point for mining companies to identify appropriate points of contact. These protocols cover the areas shown in Figure 5. Where no protocols exist, Te Puni Kōkiri, which has around 20 regional offices, is a good starting point to identify which iwi should be contacted.[13]

[12] http://www.ruanui.co.nz/environmental.aspx

[13] https://www.tpk.govt.nz/mi/

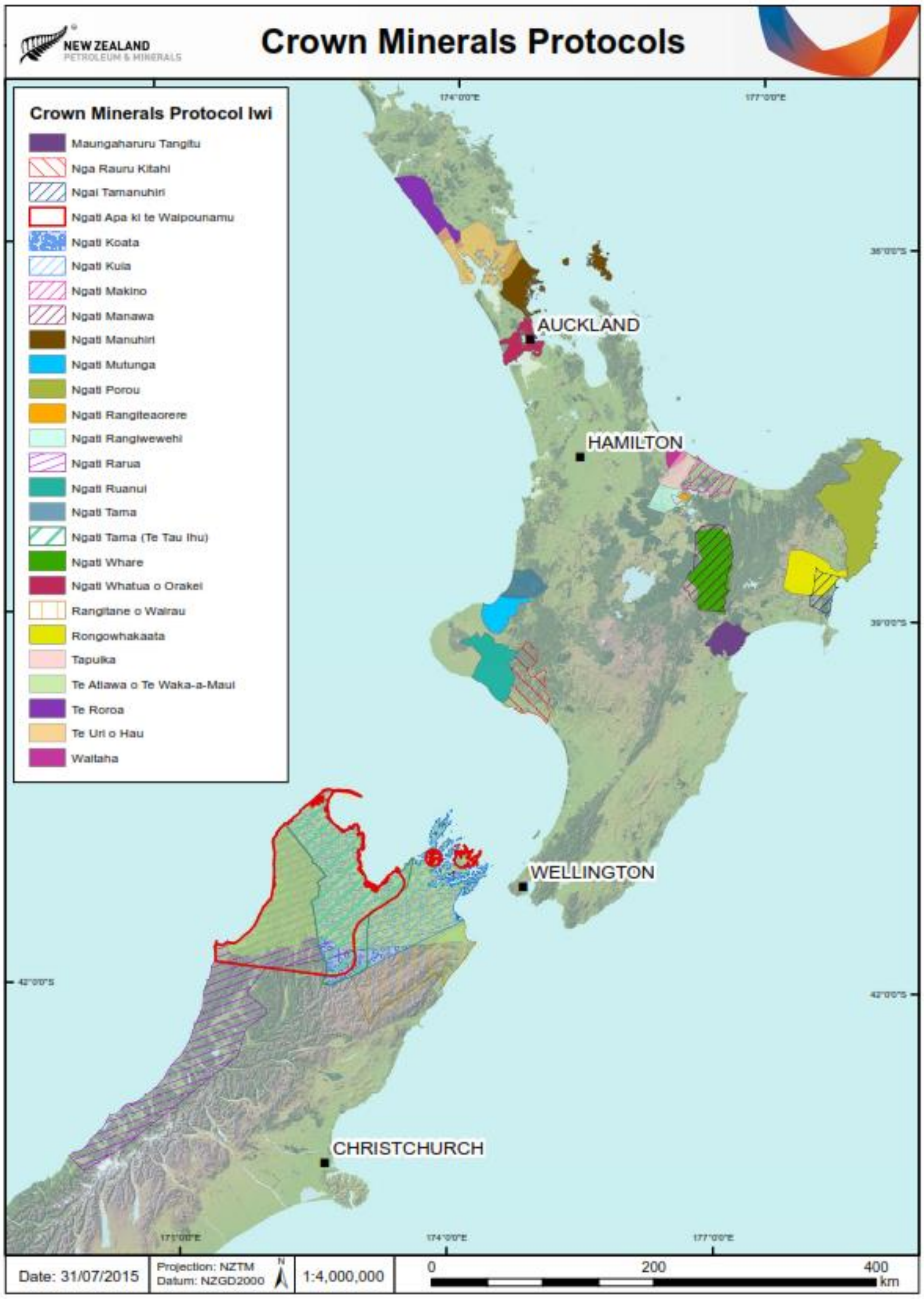

Figure 5. Crown minerals protocol areas, which include the rohe moana (areas of sea) within the territorial sea adjacent to rohe whenua (areas of land) shown on the map (source: Crown Minerals).

1.3.5 Stakeholder engagement

As with iwi, the value of early and genuine engagement cannot be overstated. It is through this engagement that issues of concern to stakeholders can be raised and addressed. Through this process, the desirable outcomes post-mining can be identified and agreed by the relevant parties, along with an acceptable level of impact, and acceptable rehabilitation can be agreed upon by the relevant parties.

The principle of early and genuine engagement is emphasised in regulatory guidance on consultation or engagement for resource consents (MfE 2015) or mine permitting (NZP&M 2013), which also contain further details on general processes for undertaking consultation or engagement. Planned rehabilitation outcomes will include short-term and closure criteria, which are established with input from landowners, 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 (particularly for farmland), specific biodiversity criteria (for conservation land), and indicators of land resilience.

Engagement during consenting may identify opportunities and priorities for early rehabilitation. Off-site mitigation or offsetting to compensate for impacts is also valuable, even when mining has a short-term effect. For example, upgrading of a road to allow mine vehicles may be an opportunity to replace existing culverts that are barriers to native fish passage; riparian works may be an opportunity to exclude stock, remove weeds, and establish a flood buffer.

Ecosystem services

One structured way to engage with stakeholders (communities) about the impacts arising from mining operations is through consideration of ecosystems services. There are four main groups of ecosystem services:

- provisioning (e.g. food, water, raw materials)
- regulating services (e.g. flood control, water treatment)
- cultural services (e.g. recreation and tourism, aesthetic, spiritual)
- habitat services that support species and biodiversity.

The ecosystem service schedule provided in Appendix 1 provides a structure for stakeholder engagement that ensures all potential values that exist in a pre-operations landscape can be accounted for consistently in the financial viability analysis, mine design, reports for consent applications, resource consent(s), and closure plans. These pre-operation documents are related through the externalities of the mine and the mine's impacts on (in)tangible social, cultural, and environmental values. An ecosystem service assessment enables companies and communities to be more specific about which values exist in the pre-mining landscape, how they arise, and what their (non-) market value is. This knowledge can inform precise reporting and costing of avoidable impacts – impacts that can be mitigated – and those values that will be (partially) lost, thus improving the mine's performance from design to closure.

Finally, ecosystem services-based engagement can help formulate better resource consent conditions. A list of ecosystem services that can reasonably be expected to exist post-closure constitutes a set of conditions that is specific and verifiable. Moreover, such a list would be supported by current practice: many ecosystem services can be linked to familiar water quality indicators, the presence of flora and fauna, and safety and accessibility indicators. If a given environmental characteristic is of poor quality, then the level of related ecosystem services is likely to be poor as well.

There are several benefits that can arise from such an early, structured engagement with stakeholders. For a start, it is likely to identify what different parts of the community care about – this informs 'avoidance' and 'rehabilitation' strategies, as well as indicating where further effort in surveying and research should focus. It may also strengthen the social licence to operate. Higher levels of social licence to operate can be achieved if stakeholders see that mine planning explicitly aims to minimise the disturbance of the landscape they know, and their use of it. It may minimise the need for biodiversity offsets, which should be considered only after opportunities for on-site avoidance and mitigation are exhausted.

While ecosystem services are specifically tied to the environment of the mine site itself, engaging in a similar manner about the post-closure social and economic viability of surrounding communities should be considered. The arrival and departure of a mining company can be significant shocks to local social structures and economies. Early thinking together with communities about how these shocks can be mitigated allows potential solutions and sources of economic, infrastructural, and succession planning support for them to be identified.

1.4 The *Mine Environment Life-cycle Guide*

The *Mine Environment Life-cycle Guide* (MELG) considers environmental information in the framework of a conventional mine life-cycle, taking account of the current mine-permitting regime, including land access arrangements now operating in New Zealand and resource consents required under the RMA. The key stages of a mine life-cycle and associated permitting/consenting activities, and how these relate to the stages considered in the mine environment life-cycle guides, are outlined in Table 1, and form the basis for the structure of this document.

Table 1. **Summary of a conventional mine life-cycle, the mine permitting/consenting operating in New Zealand, and their relationship to the mine environment life-cycle stages used in this guide**

Mine life-cycle stage	Permitting/consenting regime	Mine environment life-cycle stage
Exploration	Prospecting (land access) Exploration (land access)	Exploration – prospecting and exploration activities
Mine planning	Mine permit – feasibility study Resource consent	Pre-operations – all activities leading up to the development of an operational mine, specifically activities required to obtain land access and resource consent. Engagement with iwi and stakeholders initiated.
Operations (including commissioning and construction)	Annual reporting	Operations – all activities that are undertaken over the time the resource is being extracted (i.e. the mine permit is active)
Decommissioning	Permit relinquishment	Closure – when resource extraction ceases and rehabilitation activities are working towards the agreed post-mining environment. Under a staged mine plan, some areas of the mine may be in closure while other areas are still operational. A mine is considered to be fully in the closure phase when the mining permit has been surrendered.
Post-closure	Agreement with regulators that appropriate conditions, including allowances for ongoing treatment, have been met to the extent that agreed post-closure outcomes will be met. The mine company is no longer involved on-site.	Post-closure – agreement with regulators that appropriate conditions, including allowances for ongoing treatment, have been met to the extent that agreed post-closure outcomes will be met. The mine company is no longer involved on-site.

The MELG builds on the previous *New Zealand Minerals Sector Environmental Framework* (Cavanagh et al. 2015) in the exploration and development stage and extends this to incorporate economic considerations and stakeholder consultation, in particular iwi engagement. The MELG also includes information required during operations to ensure the post-mining outcomes can be achieved in a cost-effective manner. During operations, field trials and mass-balance experiments under local environmental conditions can be completed, whereas only laboratory tests or glasshouse trials can be completed during the exploration and pre-operational stages.

Specifically, the MELG aims to identify the critical elements that will influence the likelihood of success and/or the cost of achieving the agreed post-mining outcomes at closure, and the information that can be used to reduce the uncertainty of these outcomes. For mining companies, collection of these additional data sets during operations will translate to reduced bonds, and for other stakeholders it ensures a greater level of certainty in achieving post-mining outcomes. Ongoing stakeholder engagement is required during operations – particularly with the ebb and flow of operational projects reflecting factors such as global commodity prices – to provide assurance that specific agreed post-mining outcomes will be achieved.

An overview of the different life-cycle stages, how these relate to the type of information available, the degree of effort that should be put into different activities (stakeholder engagement, geochemistry, operational management and treatment, rehabilitation and stream ecology), and how information provided at different stages may inform the bond quantum is shown in Figure 6. Further details of the activities covered in each section are shown in Figure 7.

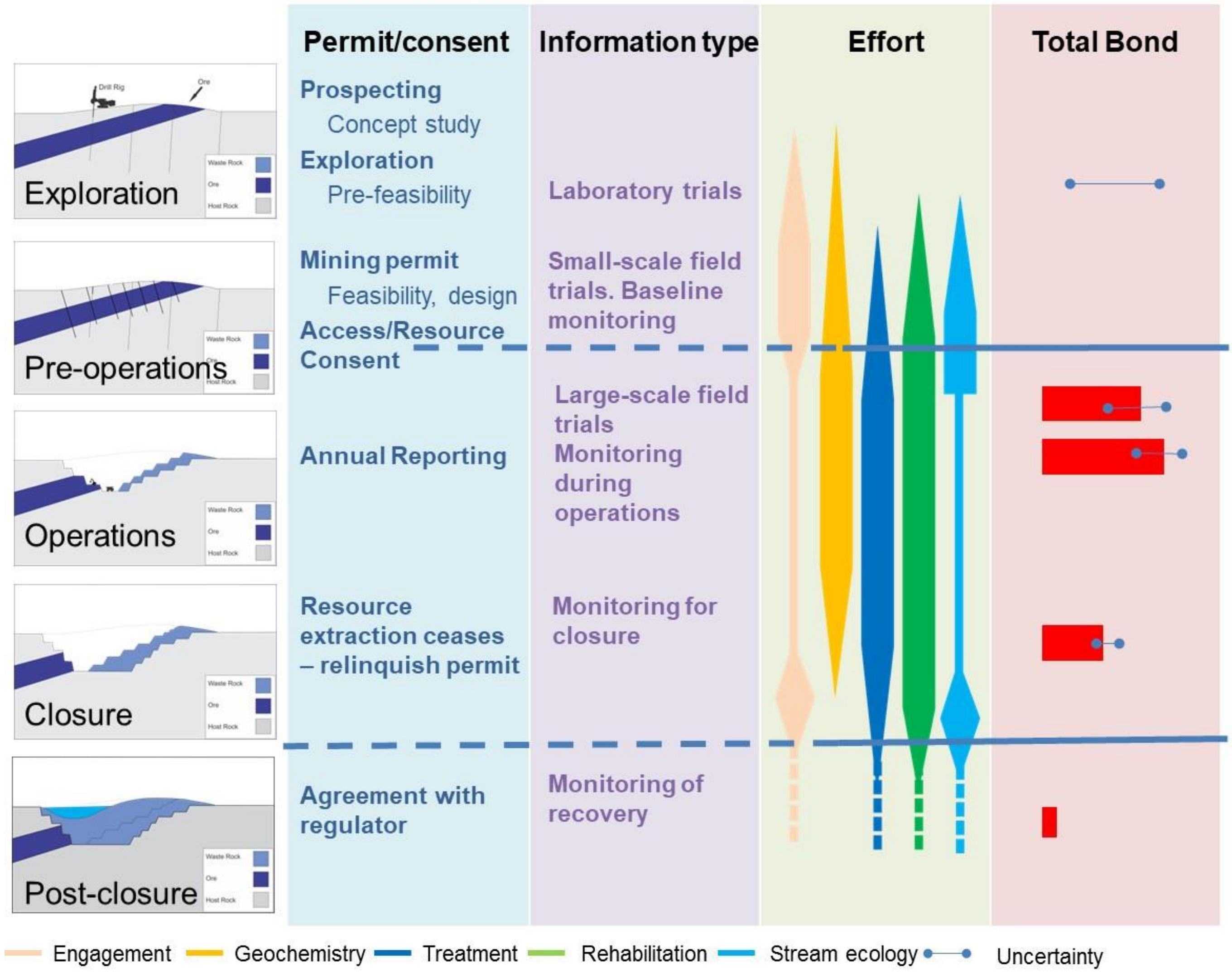

Figure 6. Summary of the relationship between the mine life-cycle, the type of information able to be obtained at each stage of the mine life-cycle (e.g. laboratory studies, field trials), and the focus of effort required in different science streams to reduce the uncertainty associated with the bond.

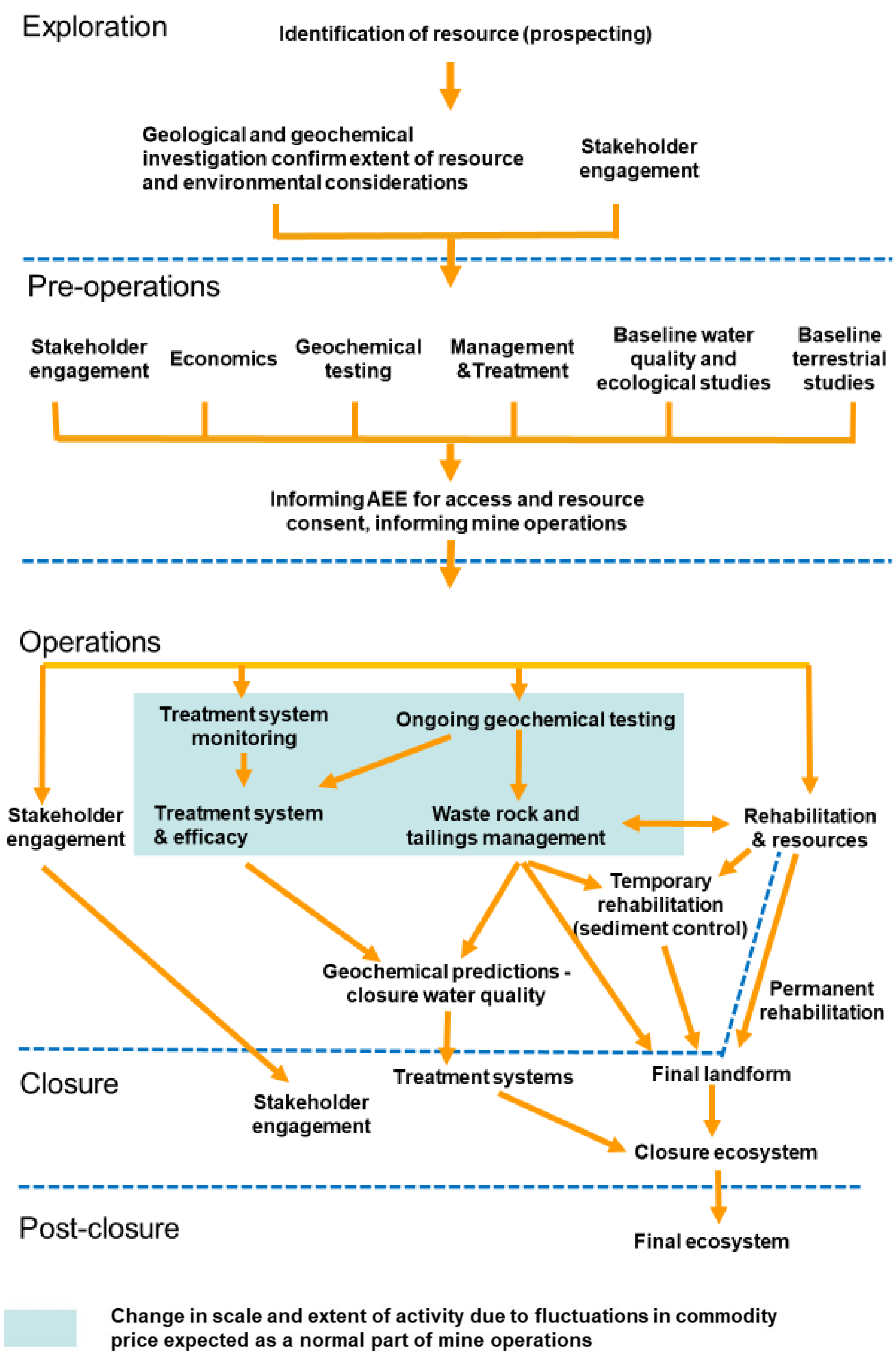

Figure 7. Outline of the information required at different stages of the mine environment life-cycle provided by this guide.

2 Exploration

2.1 Prospecting

Prospecting activities include geological, geochemical, and geophysical surveying, aerial surveying, and collection of samples by hand or hand-held methods to identify land likely to contain mineral deposits (NZP&M 2013). From both a mine permit and land access perspective, prospecting activities are usually considered to have minimal environmental impact, and so this document does not consider prospecting further from an environmental impact perspective. However, early stakeholder engagement, particularly with iwi and landowners/administrators, will be beneficial for the development of longer-term relationships if prospecting reveals potential for a mineable resource.

Activities in areas with high cultural values and/or previous negative mining experiences may have perceived impacts that are much greater than those perceived by the prospector (e.g. Puhipuhi, Karangahake). During prospecting, general geochemical signatures of rocks hosting mineralisation are established, and analogy to other nearby mineral deposits is established from the literature. Prospecting is often undertaken by different companies that ultimately develop a mine.

2.2 Exploration

Exploration activities may range in intensity, with early-stage exploration activities being similar to prospecting activities and considered minimal impact, and later-stage exploration involving drilling. Where this occurs in sensitive environments, wildlife permits may be required, along with increasingly stringent mitigation and rehabilitation activities (including biosecurity measures). This includes most Brunner coal measures land administered by DOC and all sites where vegetation is cut or cleared. The selection of drill site locations should include the ability to minimise environmental impacts by, first, avoiding areas where vegetation needs to be cleared (i.e. targeting gaps in shrubland or forest sites) or that are sensitive ecosystems (e.g. wetlands), and, second, using techniques that minimise disturbance, such as drilling platforms, which, when removed, allow vegetation to regrow.

It is useful to document and photograph vegetation in the vicinity of drill sites before and after drilling to provide evidence of impacts. Plants that are cut may be suitable to use as a resource to rehabilitate the drill site. In natural ecosystems, ensure biosecurity practices are adequate to prevent weed establishment, primarily through ensuring equipment and boots are clean of soil and seeds.

For the purposes of this document, activities to inform assessment of potential environmental impacts are covered under Pre-operations (chapter 3).

2.3 Engagement

Engagement with iwi and stakeholders is a critical – though sometimes difficult – task during mineral exploration. There is a tension between iwi and stakeholders, on the one hand, who wish to know what is planned for their environment, and companies, on the other hand, who are private regarding their exploration plans, because there is commercially sensitive information, ideas and data involved. Iwi and stakeholders will wish to know how large any possible mine will be, where it will be, and whether there will be vibration, dust, surface or groundwater impact.

During exploration, companies will have a target in mind, but also a lot of uncertainty around the future shape of any mine. At times during exploration companies may not wish to disclose sample plans, proposed drill-hole locations, preliminary results, or aspects of long-term plans because this could give advantage to other parties. Therefore discussions between stakeholders and companies during exploration should be set at a mutually agreeable high level, with key local issues and local sensitivities identified by the community, and the generalised size, shape, and location of the exploration target, along with the uncertainties, described by the company.

Companies come in different forms (public, private) and sizes (large corporate to junior), and their ability to disclose information might differ. Public companies are required to make disclosures to the stock market about any factor that might influence their share value, whereas private companies have no such disclosure requirements and the level of detail disclosed is determined by the shareholder(s). Large companies might have many representatives with internal expertise covering many aspects of environmental management, communication, and community engagement. Junior companies might only have two or three full-time employees and some contract staff, and might have little expertise in environmental management and engagement. Any of these types of companies, and possibly other types of entities, could operate a mineral exploration permit, and the level of expertise in community engagement and perceptions regarding confidential information will differ with each individual company.

2.3.1 Iwi engagement

For Māori, face-to-face (kanohi-ki-kanohi) engagement is expected, and this includes asking key tangata whenua/iwi/hapū groups how they wish to engage during the exploration phase and who should be key contacts. Engagement is usually carried out under a tikanga (values) framework of protocols and establishing the correct process for dialogue. During exploration activities, the early identification of values and sites of cultural significance (e.g. wāhi tapu, wāhi taonga, wāhi tūpuna) is a critical stage at the beginning. Genuine engagement with iwi/hapū at this early stage is essential for any mining company wanting to develop a respectful long-term and meaningful relationship with iwi/hapū to help exploration activities progress towards developing a proposal for mining and achieve agreed outcomes for all parties.

Guidance on engagement is available at http://www.nzpam.govt.nz/cms/iwi-communities/working-with-iwi-hapu/industry-engagement-with-iwi, which includes links to best practice guidelines for engaging with Māori in relation to mining. This was developed by Te Rūnanga o Ngāti Ruanui Trust based on their experiences of the development of petroleum and minerals industry in the Taranaki region.[14] Similar guidelines have been developed by Kā Rūnaka to outline expectations for oil and gas companies in East Otago (Ruckstuhl et al. 2017) and provide insight into expectations for the manner of engagement and cultural considerations within their tribal rohe. Protocols outlining the way in which the Ministry for Business, Innovation and Employment will consult with specific iwi and hapū have also been developed under the CMA and provide a starting point for contact people and/or the manner in which those iwi and hapū wish to engage. Where no protocols exist, Te Puni Kōkiri, which has around 20 regional offices, is a good starting point to identify which iwi should be contacted.[15]

2.3.2 Other stakeholder engagement

For non-iwi stakeholders, early and genuine engagement is also important. Engagement during the exploration phase is challenging due to the tension between the commercial sensitivity of exploration activities and a desire by stakeholders to understand the potential environmental impacts – and benefits. The principle of early and genuine engagement is emphasised in regulatory guidance on consultation or engagement for resource consents (MfE 2015) or mine permitting (NZP&M 2013), which also contain further details on general processes for undertaking consultation or engagement. These documents provide a useful starting point for engaging with non-iwi stakeholders. Section 1.3.5 also outlines an ecosystems services review approach for providing a structured approach to engaging a broad range of stakeholders.

[14] http://www.ruanui.co.nz/environmental.aspx
[15] https://www.tpk.govt.nz/mi/

3 Pre-operations

3.1 Introduction

The pre-operations phase covers the steps required to determine the feasibility of the mining project, and the activities associated with obtaining land access and resource consents. Determining the feasibility of the mining project is a key stage in determining the viability of any project progressing and needs to consider all aspects of the project. For instance, management of potential environmental effects for the project is an important cost to be captured in the financial model. This section highlights the information required to identify and manage potential environmental impacts that can be collected prior to mine operations to ensure the desired post-mining outcomes are identified and can be achieved. This information will assist with the development of environmental management plans typically developed during resource consenting processes.

An overview of the process to determine the potential extent of environmental impacts of mining and options to minimise these impacts is shown in Figure 8. The key impacts considered are impacts on the terrestrial environment and impacts on aquatic systems within the mine area and downstream of the mine. Effective rehabilitation can 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 process outlined in Figure 8 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 influence decision-making. Instead, the MELG provides a robust scientific basis for this decision to be made during consenting processes. Stakeholder engagement is critical to identify relevant post-mining outcomes that determine whether the rehabilitation outcomes are acceptable.

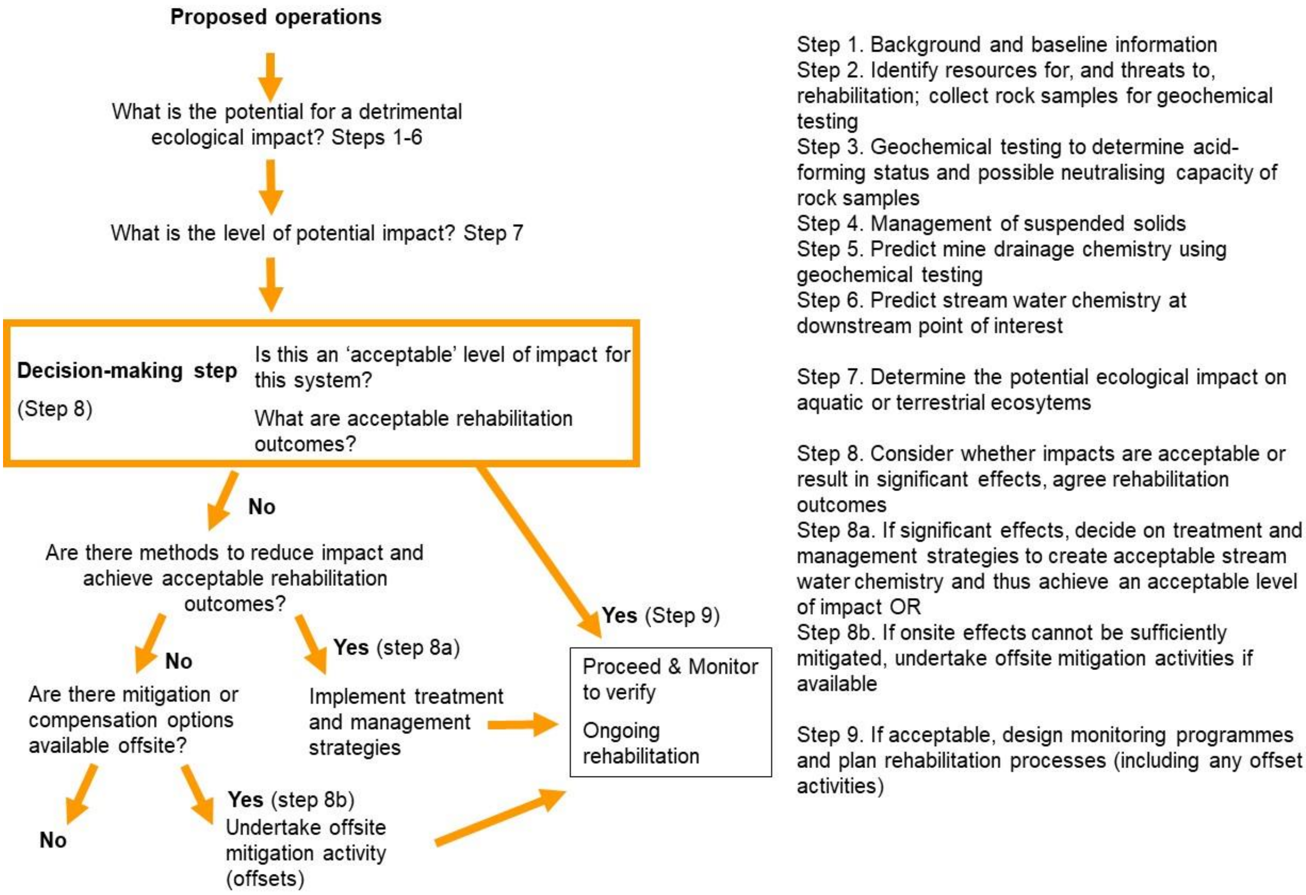

Figure 8. A general framework and detailed step-by-step guide for predicting and managing water quality (pH and metal) impacts on streams resulting from mining.

3.2 Engagement

Early and effective stakeholder engagement is required to enable post-mining outcomes to be identified and agreed, and to establish a good relationship between stakeholders and the mining company. Early conversations may identify specific ‘no go’ areas due to their cultural significance to local iwi (e.g. wāhi tapu) and should build upon those started during the exploration phase (more detail on initial engagement is provided in sections 1.3 and 2.3). These conversations help to identify ‘what matters’, from which the conditions that will deliver 'what matters' can be identified. Starting with the end in mind guides mine planning to enable agreed post-mining outcomes to be achieved cost-effectively. In the conversation it is important that what matters to different stakeholders is understood by all parties, and that agreement on what matters is reached. For this, the language used, and the way in which information is presented, is very important. For example, engineers and scientists are very familiar and comfortable with technical information and numbers, and may readily understand the significance of numbers (such as metal concentrations in water) with respect to potential environmental impacts. However, for many stakeholders the desired environmental outcomes may be articulated in more general terms, such as ensuring that certain species, such as tuna (eels) or trout, are unaffected by mining operations, or that after mining other species (e.g. kererū) will return to the rehabilitated area.

More-focused discussions with stakeholders, including iwi, helps to identify indicators that can be used to provide a measurable link between the articulated and agreed outcomes and the operation of the mine, including the performance of waste rock management or treatment systems and rehabilitation. Principles for rehabilitation should be developed during early engagement and consenting activities. Specific principles for rehabilitation to native forest that are likely to be raised by DOC or iwi include the use of locally collected propagules for vegetating areas, and minimising the export or import of rehabilitation materials from outside the site or rohe. Constructing natural landforms that will blend the mined landscape into adjacent unmined areas may be preferred. Principles for rehabilitation to farmland may include ensuring all final slopes are gentle enough to be safely negotiated around the contour by quad bikes, and every 10 ha has a minimum area of shelter-belt coppicing for firewood.

Planned rehabilitation outcomes will include short-term and closure criteria, which are established with input from landowners, administrators, and regulatory authorities. All these parties – but particularly landowners and ecologists (who may not have experience with mining) – are likely to need help to understand the impacts and options that are available. Visits to similar mine sites or large earth-worked sites (e.g. dams, highways) with known rehabilitation methods are usually extremely helpful to guide discussions.

Sites with a range of rehabilitation ages are particularly useful where native ecosystems are affected, especially if they include areas near 'canopy closure' or near the condition at which bond release is achieved. 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 (particularly for farmland), specific biodiversity criteria (for conservation land), and indicators of land resilience. Ensure that stakeholders understand the uncertainties inherent with mining a resource, and discuss the 'maximum' and 'minimum' areas that may be affected, depending on the results of further geological proving of the resource.

Engagement during consenting may identify opportunities and priorities for early rehabilitation and off-site mitigation. Offsetting to compensate for impacts is also valuable, even when mining has a short-term effect. For example, upgrading a road to allow mine vehicles may be an opportunity to replace existing culverts that are barriers to native fish passage; riparian works may be an opportunity to exclude stock, remove weeds, and establish a flood buffer.

3.2.1 Iwi engagement

For Māori, face-to-face (kanohi-ki-kanohi) engagement is expected, and includes asking key tangata whenua/iwi/hapū groups how they wish to engage during the course of operations. Māori place a high value on ongoing engagement that is respectful and meaningful, where constructive dialogue and contributions can be made by all parties. Agreed goals and outcomes are often a true test of whether a value-based relationship has been established and the engagement has been effective. This in turn requires that the knowledge and language used to express the effects of mining operations on aspects important to iwi is clear to all parties. Māori values and environmental values and goals may sometimes be similar, and in these cases synergies can be delivered in both data collection and mitigation by including project ecologists in discussions with tangata whenua. Sometimes ecologists can act as 'intermediaries' by helping translate environmental impacts and mitigation options to achieve desirable cultural outcomes. Genuine engagement includes transparency around planned activities, and engagement at appropriate levels between the mining company and iwi.

During pre-operations, conversations may intensify as iwi seek to understand the potential impacts and benefits. Questions for the mining company may include those outlined as Ka Rūnaka expectations:

- What is the company's history with indigenous populations elsewhere?
- What long-term negative legacies are likely? For epithermal mining this includes legacy risks of AMD, tailings mobilisation and consequent contamination of surface waters). Also, what positive legacies are likely? Local kaitiaki feel a responsibility to future generations who will live in the area, and so their assessment of acceptable risk may be different.
- Are there employment opportunities for local people now and in the future, such as research, consultancy, rehabilitation and its maintenance, monitoring and reporting/auditing (using both mātauranga Māori and traditional science)?
- In what ways might cultural values and mātauranga Māori be implemented within mining and rehabilitation – from day-to-day activities (e.g. water sampling), for new employees (e.g. induction to relevant aspects of Māori culture).

The pre-operations phase may include the development of a cultural impact assessment, which should have extensive involvement with, if not being led by, local iwi. It may be relevant to build in consent conditions (e.g. for cultural monitoring or cultural balance plans) that provide iwi with surety of a minimum level of ongoing engagement and the opportunity to have input during mining operations to enable mutually desirable outcomes to be reached during and after mining operations. While these activities may initially be stimulated through consent conditions, this should be immaterial if a value-based relationship develops.

3.2.2 Other stakeholder engagement

The ecosystem services framework provides one way in which to conduct initial stakeholder engagement to draw out the environmental aspects that are of most importance to the relevant stakeholders, which can then also be used to assist in developing mine plans and resource consent conditions. Environmental issues of concern for non-iwi stakeholders are likely to be similar to those expressed by iwi for native ecosystems, and include issues such as impact on trout fishing or development of suitable productive land post-mining. Guidance on general processes for undertaking consultation or engagement for resource consents (MfE 2015) or mine permitting (NZP&M 2013) is also a useful starting point.

3.3 Historical and baseline data

The potential ecological impacts of a proposed mining operation are assessed by first defining the maximum probable mine footprint. This must include not only the surface area under which the mineral resource lies (for open-cast mines), but also areas required for associated infrastructure. Schedule 4 of the RMA requires that alternative locations or methods for undertaking an activity should be considered. The mineral cannot be shifted, but the location of ex-pit overburden dumps, roads and processing plants (ROM pads), offices and other infrastructure can usually be located to minimise environmental impacts. In remote sites, the area affected by access roads and power lines can be greater than the mine footprint, especially in steep landscapes, and while the mine footprint may be rehabilitated, roads may be permanent features that fragment areas and affect the movement of animals. The environmental footprint of a project usually extends beyond the 'cleared' boundaries due to edge effects and has an impact on home ranges or the migratory pathways of animals.

To predict the potential ecological impact of a mining activity on aquatic and terrestrial ecosystems, information is required on the site's water quality and current ecological status/productivity. 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 to identify any naturally elevated concentrations of elements in surface water that might be present in mineralised areas. Field surveys should also identify features such as threatened ecosystems and species, and any impacts from historical mining activity or other industries. These data may also influence subsequent consenting, land access arrangements (and wildlife permits), and compliance or management decisions. Collation of historical data also enables any heritage features of the site to be identified and subsequently managed or enhanced.

This section provides an overview of the data required to determine the potential impact of mining on downstream water quality and to assist in determining what is achievable regarding rehabilitation of on-site terrestrial ecosystems. The type and scope of field data collected are likely to be influenced by the historical data from previous work, in addition to any aspects that come to light in the early planning stages. Where mining is likely to negatively affect rare and/or threatened native species and ecosystems, and rehabilitation cannot replace all values, additional mitigation or compensation will probably be required, and this could be outside or inside the mine area. Field data may therefore be required to establish the distribution and abundance of threatened ecosystems, plants and animals *outside* the proposed mine site, and to establish the potential benefits of different mitigation activities.

In May 2018 the Environment Institute of Australian and New Zealand released the second edition of *Ecological Impact Assessment Guidelines for New Zealand* to help raise the standard of ecological assessment for both terrestrial and aquatic ecosystems.[16] The standards are intended to help council ecologists and planners who process resource consents to check if all expected information is present and treated in an appropriate way. The updated standards stress ecological description and analysis as a basis for impact assessment and management (not individual species). They include considering the role of tangata whenua values. Please refer to this valuable document, as it will be updated, and it will be used by regional councils.

3.3.1 Historical data

Collating 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. Discussion on the use of historical information to provide a qualitative desktop assessment of potential mine drainage chemistry is provided in section 3.4.5.

[16] https://www.eianz.org/resources/publications/2018---ecological-impact-assessment-guidelines-for-new-zealand-2nd-edition.

Relevant historical information will include geological maps, soil maps, and previous data on water quality, stream flow, terrestrial ecology, rainfall, stream ecology, and climate. The extent of historical information available will influence the amount and type of baseline data that need 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
- NZ Mine Plans (https://mineplans.nzpam.govt.nz/), a purpose-built database with an interactive GIS webmap, which provides free access to a searchable catalogue of mine plans
- data collected during previous mining operations and exploration, as reported to NZP&M and available via the internet (http://www.nzpam.govt.nz/cms, tools section) or from the NZP&M library – registration is required to access the technical data from the website, although there is no associated cost
- iwi management plans (on the Quality Planning website) and cultural impact assessments or other cultural values reports for sites, species, habitats or ecosystems – Manaaki Whenua – Landcare Research's Nga Tipu Whakaoranga (https://www.landcareresearch.co.nz/resources/data/nga-tipu-whakaoranga-maori-plant-use-database) and NIWA's Kaitiaki Tools Mahinga Kai Species (https://www.niwa.co.nz/freshwater/management-tools/water-quality-tools/kaitiaki-tools)
- the Terrestrial and Freshwater Biodiversity Information System (TFBIS) programme, which provides information about projects funded to access biota and biodiversity (http://www.doc.govt.nz/tfbis); this includes links to the New Zealand Virtual Herbarium (http://www.nzherbaria.org.nz/virtherb.asp), and interactive identification keys for specific groups of fauna and flora
- soil maps (S-map), Landcover Database 4, New Zealand Land Resource Inventory Land Use Capability, and soils data from the Land Resource Information Systems (www.landcareresearch.co.nz/reources/data/lris)
- ecosystem attributes from Land Environments of NZ (LENZ, https:\\lris.scinfo.org.nz), which includes climate, soil attributes and native vegetation from the National Vegetation Survey databank
- the New Zealand Plant Conservation Network (http://nzpcn.org.nz), which provides information on the identification and distribution of plant species, and also holds NZ Botanical Society newsletters (http://nzpcn.org.nz/page.aspx?publications_Botsoc_journals)
- regional council maps of Significant Natural Areas
- site and regional pasture and plantation forestry production data – historical and forecast data are available from https://www.dairynz.co.nz/feed/pasture-management/pasture-growth-data/, which has links to their Pasture Growth forecaster and 'Farmwatch': in the 1970s and 80s, a seasonal pasture production profile was developed for 30 sites throughout New Zealand, based on an average of 11 years of monthly data; most regions were included, with variation mainly reflecting differences in temperature and seasonal moisture deficits, and the results were published in the *NZ Journal of Experimental Agriculture* (e.g. Radcliffe 1974; Radcliffe & Cossens 1974; Roberts & Thomson 1984)
- geological maps – available from GNS Science at http://www.gns.cri.nz/store/publications/maps.html
- industry reports and scientific literature, as published
- the Centre for Minerals Environmental Research web page: http://www.cmer.nz/
- 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.

Following are some further sources of information.

- Regional councils may hold relevant information in technical reports, reports supporting resource consent applications, or monitoring data, including aquatic and terrestrial biological data, flow and water quality data, and data on quantity monitoring. These can be obtained by contacting the relevant council directly.
- Currently, few comprehensive and freely available databases exist for water chemistry, stream algae or invertebrates. However, the development of the Land & Water Aotearoa (LAWA) database provides limited information from local government (https://www.lawa.org.nz/explore-data/west-coast-region/river-quality/lower-buller/burkes-ck-@-sh69/). 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.
- DOC uses a freshwater classification system: Freshwater Environments of New Zealand (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).
- DOC reports, particularly survey reports for the Protected Natural Areas Programme, Recommended Areas for Protection, and some Conservation Advisory Science Notes (http://www.doc.govt.nz) may be useful.
- The NZ Threat Classification systems and lists are updated from time to time and are available on the DOC website. Threatened land environment classification is available on the Manaaki Whenua – Landcare Research website.

- NZ Birds online includes cultural values (http://nzbirdsonline.org.nz), and the Ornithological Society website (http://osnz.org.nz) includes information on native birds.

3.3.2 Baseline Information

Baseline information on the state of both aquatic and terrestrial environments, including the impacts of any legacy mining activities, is a critical first step in assessing the potential impacts of a new mining development and opportunities for preventing or mitigating those impacts. Baseline information might form the basis for consent application details where consent conditions are related to maintaining previous water quality or the state of terrestrial systems. For aquatic systems, baseline data may subsequently be used in models to predict downstream water chemistry at a specific site. Specific information for baseline water quality includes:

- surface- and ground-water quality, flow rates and variability (site hydrology)
- climatic data, including rainfall, evaporation, evapotranspiration, and wind speed
- the ecological status of the streams (upstream, within the mine, and downstream)
- any previous mining activity and effects (legacy issues)
- any other activities that take place in the catchments, and assessment of their impacts on baseline information.

Similarly, collection of data on current terrestrial ecosystems, whether native, pasture or plantation forestry, informs an assessment of the impact of mining at the site. In native and pastoral ecosystems, areas that have been eroded or cleared of vegetation (e.g. slip scars) can be particularly useful to indicate vulnerability to erosion, edge effects, weed invasion, and natural colonisation processes. Specific information to determine the baseline status of terrestrial ecosystems includes:

- land cover use at the site and adjacent areas (considering how adjacent land can affect the site under consideration, either negatively, such as being a source of pests, or positively, such as through native bird and plants re-establishment)
- geology, soils, hydrology, landforms and how infrastructure interacts with these
- terrestrial communities/ecosystems and vegetation types and how they interact, which includes the dominant plant species in height tiers, plant cover estimates, condition and evidence of threats (identify high-value ecosystems)
- terrestrial fauna, both vertebrates and invertebrates (identify high-value species)
- specific habitat features; e.g. boulder outcrops, logs, large trees with cavities (bat/bird roosts)
- what is not present that would be expected to be present (and why)
- evidence of impacts of current management or uses or values, especially in relation to tanagata whenua values.

Focus on the issues likely to influence management of the site, including ‘fatal issues’, and do a gap analysis to identify where adequate information is unavailable. Inadequate information may be due to constraints on the timing of any surveys, as many species may be particularly visible or active for only parts of the year or in specific weather conditions. A common failing of ecological surveys is a lack of analysis on species that have a disproportionate impact on the success and cost of rehabilitation, especially weeds and their pathways into the site. Finally, initial baseline surveys provide an opportunity to identify mining relics to be considered with respect to their heritage value.

Site hydrogeology

Surface-water hydrology data, including detailed knowledge of streams that could be affected by the operation (both upstream and downstream), and measurements of flow volume, are required for characterising baseline catchment conditions. Ultimately, a hydrological model is required of the catchment to be disturbed by mining, which should include probabilistic analysis of the effect of rainfall events on stream flow and chemistry. In particular, hydrological models that include calculations 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, often with the assistance of probabilistic modelling software (e.g. GoldSim™).

At some mines (e.g. underground mines, or possibly deep, open-cast 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 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 if mining proceeds.

Collection of flow data

A suite of stream-flow measurements that characterise the variability of stream flows at the site is required to predict the effects of mixing mine drainage with other catchment water. Generally, stream flow measurements are required for all tributaries that contribute more than c. 10% of surface flow volumes, or c. 10% of any dissolved components to the main stream of interest.

The frequency and duration of sampling will depend 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 selected sites in each catchment are required to understand flow variability.

Additional information on flow rates, and their variability with season, rainfall and drought, is required 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, and can be matched with historical climate data to provide long-term hydrological models that capture seasonal, annual, and even decadal changes in stream flow.

Climate data

The collection of climate data permits extrapolation of environmental variability through the extremes of wet, dry, hot or cold average annual conditions, so that the year(s) in which other environmental monitoring data are collected can be put in the context of longer-term climate variability. For example, if the baseline stream or ecological data are collected in a particularly warm, wet year, then animal behaviour might be abnormally active and stream flows would be unusually high. If it is not recognised as an unusually warm, wet year, then consent conditions might be put in place that are difficult to meet under normal climate conditions.

Climate data are best collected through an automated weather station positioned at the site where mining is intended. These stations can collect or derive a variety of climate data, including rainfall, windspeed and direction, temperature, dew point evapotranspiration, and humidity. The instruments require calibration, maintenance, and regular checking to ensure good-quality data are collected. Once a significant data set (>1 year) has been collected at the mine site, it can be compared to data from the nearest or most appropriate climate monitoring site. Often these monitoring sites will have 50 to 100 years of climate records, and a relationship can be established between the data measured at the mine site and the data from the monitoring site. Once the relationship is established, the climate record for the mine site can be extrapolated to match the records from the monitoring station. There are still uncertainties with this approach, but these can be quantified and managed if good data are collected.

Climate variability and its impact on mine site environments can be modelled with software such as Goldsim™. This software uses statistical analysis to identify the probability of different types of climate events and can be useful for the design of appropriate mine infrastructure. Climate variability and modelling can also be used to predict the impact of climate change on future weather patterns. Applications such as HIRDS (High Intensity Rainfall Design System; https://hirds.niwa.co.nz) can be used to estimate the impact of more extreme weather (which is likely with climate change) on the probability of different weather events.

Baseline chemical water quality

The flow volume and chemistry of the receiving environment can have a significant effect on water quality downstream of a mine drainage discharge. For example, if the background water has sufficient alkalinity, the addition of small volumes of AMD (with or without trace elements) might have minimal downstream effects. In contrast, streams with low alkalinity or low trace elements could be significantly affected by small volumes of AMD.

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)
- current sources of mine drainage (legacy issues)
- background or baseline physiochemical properties (temperature, pH, electrical conductivity, dissolved oxygen, etc.)
- concentrations of dissolved anions (sulphate, chloride, bicarbonate, phosphate and possibly others) and cations (Ca, Mg, Na, K, and possibly others)
- concentrations of dissolved metals and other trace elements, including Fe, Al, Ni, Zn, Mn, As, Co, Cu, Cd, Pb, and Sb
- limits of reporting – often water quality data are obtained, yet the limit of reporting (detection limit) is too high and above typical regulatory guidelines, which means the data have limited application and a repeat analysis may be required.

It is essential that when data are gathered on water quality, flow rates are also obtained in order to derive loads. This is an integral part of any conceptual model and probabilistic analysis of effects.

Collection of baseline water quality data

A representative suite of samples collected from the entire catchment of a proposed mine site will provide an initial characterisation of background water quality, although, sampling should also include a temporal component to understand seasonal effects and climate-related events.

There is no hard-and-fast rule for sampling frequency. However, 1 year of monthly monitoring data enables good characterisation of stream chemistry. Baseline monitoring could be usefully supplemented by continuous monitoring of pH, electrical conductivity, flow and/or turbidity. If the rock geochemistry is characterised (section 3.4.3), these water samples are also important for predicting downstream water chemistry, if mining proceeds. 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 c. 10% of flow volume, or c. 10% 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, usually filtered to 0.45 µm (though sometimes 0.22 µm or 0.1 µm filters are required), of all relevant components, and samples should be collected according to standard methods (e.g. Standards New Zealand 1998a, b).

For epithermal mineral deposits, a selection of analyses of trace elements such as Zn, Cu, Pb, As, Sb, Cd, Hg, or Mn should be carried out. However, the trace metals associated with epithermal deposits are variable, and screening analysis should be completed for other metal and metalloid elements. 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 deposits can be established by analysing data from other similar operations, although the variability of trace element concentrations within and between epithermal deposits means that these data are only a guide. Where mines are planned in sequences of rock that have no historical mining activity, a cautious approach to trace element analyses is recommended, and often it is in the interest of the applicant to identify baseline chemistry for elements that might be naturally elevated rather than identifying that they are elevated after mining commences when baseline conditions cannot be tested.

Other parameters such as dissolved oxygen, 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, and it also provides a basis for more detailed geochemical modelling, which might prove useful for water quality predictions at some 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 legacy mining activities

Sites that are affected by historical mine drainage should have a baseline chemical and hydrological survey completed. This survey should include the chemistry and volume of the historical mine drainage, in addition to inputs from unaffected 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. Drainage from historical mining activities should be quantified well enough to provide a mass balance of dissolved components related to historical activity compared to natural concentrations (under different flow regimes) (e.g. Trumm et al. 2016, 2017). This information is important so that any impact assessment of future resource development includes a thorough assessment of prior impacts. It is possible that new developments could remove or mitigate impacts from historical workings and lead to an improvement or reduced level of impact overall. There are many studies of historical mine drainages on water quality throughout New Zealand (for an example, see Case Study 3 below).

Case Study 3: Assessment of contaminant load from the historical Bellvue coal mine – the 'seesaw effect'

Drainage from historical workings can provide valuable clues about how drainage from new mines might affect nearby waterways. Sometimes these drainages can generate high flow rates and contaminant loads, with significant delays following rainfall, which can result in a highly variable impact on the environment.

The Bellvue coal mine, located on the West Coast north of Greymouth, ceased production in 1970 but continues to discharge AMD at an average flow rate of 0.93 L/s into nearby Cannel Creek. The chemistry of the AMD has low pH (2.6) and elevated concentrations of dissolved metals (69 mg/L Fe, 39 mg/L Al, 0.76 mg/L Mn, 0.32 mg/L Zn, 0.15 mg/L Ni). Acid load is positively correlated with flow. The water quality in Cannel Creek changes from a near-neutral pH stream with low metal concentrations upstream of the confluence, to a stream with a pH of approximately 3.1 with elevated concentrations of dissolved metals. Chemistry and flow rates of the AMD and Cannel Creek were quantified during and between rainfall events to identify potential changes in acid loads and resulting effects on the stream during rainfall.

The results show that Cannel Creek (CC flow) and AMD flow (Bellvue flow) rates show an asynchronous pattern in response to rainfall events. Cannel Creek flow peaks 24 hours after maximum rainfall and the AMD flow rate peaks 24 hours later. The acid load to Cannel Creek is therefore greatest at the tail end of storm events, when the AMD flow rates are still elevated but Cannel Creek is returning to base level. This situation may be common for abandoned underground mines that are near the surface and affected by rainfall (through fracturing, etc.), which discharge AMD to surface water streams, and it may even occur with large overburden dumps. The response of flow rates from waste rock dumps (WRDs) and underground mines to rainfall events compared to the surface stream response should be considered in the early planning stages.

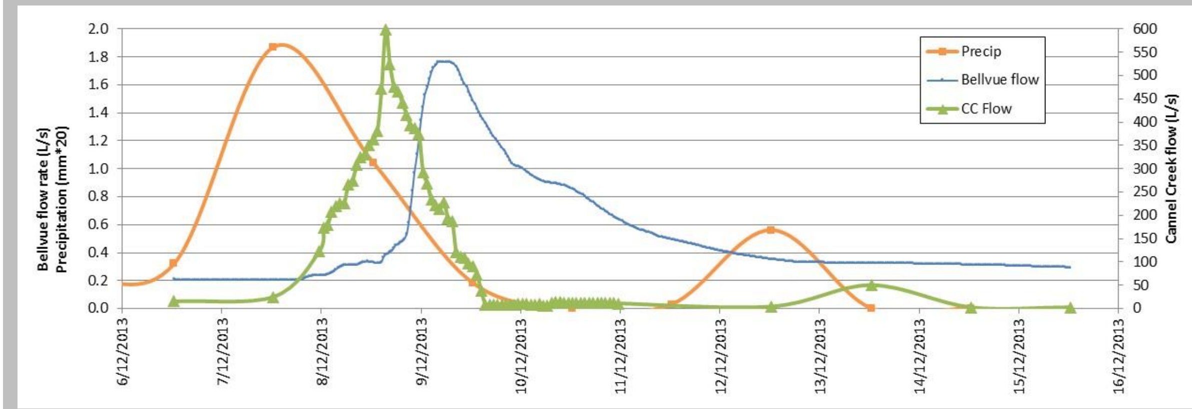

Figure C6. Variation in rainfall (precip.) and flow in Cannel Creek (CC flow) and discharge from the Bellvue mine adit (Bellvue flow) over a rainfall event.

Key findings relevant to mine management:

- Flow rates from underground mines, and potentially from large WRDs, can show a delayed response to rainfall events compared to surface streams.
- An asynchronous pattern of flow rates between AMD and surface streams can result in variable acid load, and hence variable effects on downstream water quality coupled to rainfall events.
- Planning for mine water treatment should consider the potential increased impact to the environment after rainfall events and should include enhanced treatment to offset this impact.

Key references:

Trumm D, Cavanagh J 2006. Investigation of remediation of acid-mine-impacted waters at Cannel Creek. Landcare Research Contract Report LC0506/169.

Trumm D, Pope J, West R, Weber P 2017. Downstream geochemistry and proposed treatment – Bellvue Mine AMD, New Zealand. Proceedings of the International Mine Water Association Conference, Lappenranta, Finland, 25–30 June 2017. Pp. 580–587.

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. It is essential to conduct baseline monitoring prior to any mining because 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 4).

Because freshwater organisms live most of their lives under the water, these communities reflect an integrated record of water quality over time. Fungi, bacteria, meiofauna (e.g. plankton and mites), algae, macroinvertebrates, and fish are all important in natural waterways and each is vulnerable to mining impacts. In this document we focus on using macroinvertebrates (Figure 9) for biomonitoring. Sampling procedures are generally simple and well developed, and there is considerable knowledge of macroinvertebrate ecology, including how they respond to mine drainage. 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 9. 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: the 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 (e.g. the community is 20% mayflies)
- *quantitatively,* where the abundance of each species is determined in a sample of known stream bed area; abundance is usually expressed as number of animals per square metre of stream bed.

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 in Cavanagh et al. (2015).

Of the three levels of 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 in Cavanagh et al. 2015). However, qualitative data *will not* detect changes in the relative abundance of key species, and therefore overlook a potentially significant component of change in waterway communities. Results can also be influenced by the sampling effort, as greater effort in 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 greater detail, which 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.

Of note is the Acid Mine Drainage Index (AMDI), developed using data collected from 91 sites on the West Coast of New Zealand (Gray & Harding 2012). The AMDI was developed by associating water chemistry and benthic invertebrate community data with AMD indicator scores for 57 taxa, calculated using weighted averaging. Site scores can range from 0 (severely impacted) to 100 (unimpacted), and sites can be categorised as 'severely impacted', 'impacted' or 'unimpacted' (Appendix D.5 in Cavanagh et al. 2015). Gray and Harding (2012) found that the richness of mayflies (Ephemeroptera), caddisflies (Trichoptera), and stoneflies (Plecoptera), along with total richness, may be useful companion metrics to the AMDI, and suggested that reporting multiple metrics was valuable for determining any impacts on aquatic systems.

In addition to biological sampling, assessing the physical properties and water chemistry of a site 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 environmental effects (AEE). Sampling should include both 'reference' and potentially affected sites to enable the subsequent detection and quantification of mining-induced change. A reference site is a site that represents a typical non-mine-affected site in the region. Ideally, it should also be physically comparable (e.g. similar size, elevation and substrate) to affected sites.

It is absolutely essential that reference sites be included in any baseline survey, as data from these sites provide the ability to detect any changes that might occur independently of mining activities; for example, changes caused by large floods, droughts, vegetation regeneration or other factors. Ideally some reference sites will be outside a future potential mined area. A critical aspect of designing a biological monitoring programme is selecting the location and number of sampling sites, because they directly influence the ability to detect and monitor change. The most important considerations are:

- the location and comparability of potentially affected and reference sites
- replication, both between and within impact and reference sites.

To enable comparisons over time and between sites, 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 (BACI) comparisons, which are widely accepted as the standard method to detect and quantify ecological impacts (see Figure 10 for an example of the sample design, and Appendix D.6 in Cavanagh et al. (2015) for more detail).

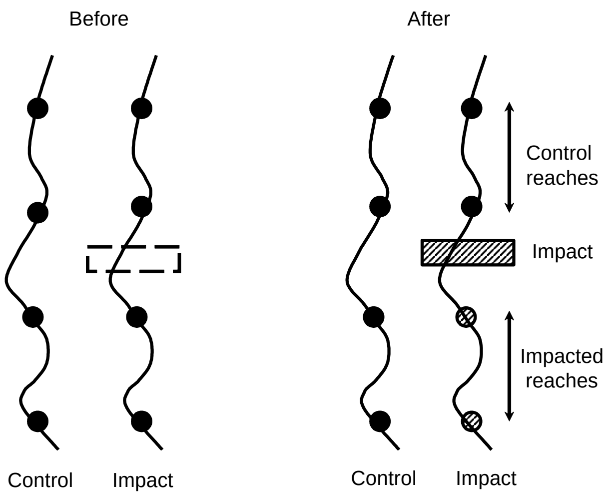

Figure 10. Before-after-control-impact (BACI) sampling designs for assessing the environmental effects of human impacts on stream ecosystems. Each circle represents a sampling site. A rigorous BACI design includes multiple sampling sites in affected reaches, and upstream and additional stream control (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, because 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 to January), so it is preferable to conduct surveys during these months.

Baseline biological surveys often focus on biodiversity, and so the emphasis is often placed on collecting qualitative or semi-quantitative data across multiple sites rather than quantitative data over fewer sites. Although this guide focuses on macroinvertebrates, baseline assessments would also normally qualitatively sample fish communities. 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.

Baseline monitoring of terrestrial systems

A meaningful description of the terrestrial ecological status of the site needs to set the site and the potential impacts of the proposed project within the surrounding ecosystems, landscape, catchment and region. Baseline information helps identify the ecological and/or production values within areas impacted by a proposed mine. This area is likely to be significantly larger than the stripped footprint, especially in natural ecosystems, because of edge effects, impacts on parts of home ranges (e.g. kiwi ranges may be 10 ha or more), and impacts on connectivity, which affect how animals move across a site (including seasonal migration / seasonal uses).

Surveys often focus on vegetation as a surrogate for biological diversity – both the common and ‘significant’ parts of a region’s biodiversity – as these inform consultation on values that may be impacted (i.e. ‘what people care about’). Plant and animal diversity, abundance and distribution are important, with special attention paid to locally or nationally threatened native species,[17] ecosystems or environments.[18]

The features that influence the significance of native ecosystems or species are usually defined in regional council plans. These can include the landscape pattern at the site and its surrounding area (e.g. the connectivity or fragmentation of ecosystems). A common failing of ecological surveys is a lack of analysis of pest and weed species that are likely to have a disproportionate impact on the success and cost of rehabilitation (especially weeds) and their pathways into the site.

Production values can be measured by crop yields, monthly pasture dry matter, and stocking rates. The seasonal pattern of crop or grassland production across a farm can be important to the whole-farm production. For example, on the West Coast, drier paddocks can be particularly valuable because they provide winter stand-off areas resistant to pugging damage, as well as early spring production. Similarly, a variety of ecosystems can enhance the value of a natural area, particularly when covering an intact drainage or altitudinal range.

Baseline monitoring should inform rehabilitation planning by providing the following information.

- *The pre-mining condition of farmland and production forest, including the land-use capability:* this indicates the main constraints to productivity and versatility, and links them to the current management system. Where a farm or plantation forest is impacted, map the area with the landowner or manager, and identify factors that influence productivity and value across the broader forest and farm operation. This information can be useful to identify innovative outcomes; for example, creating a level log-processing site prior to harvesting, or a winter standoff pad for a dairy operation, or upgrading access, such as bridging watercourses that have restricted access. Such work is done by a farm or forestry consultant, who can then also explore alternative management options that rehabilitation can deliver.
- *Plants and ecosystems:* there are three common national frameworks for terrestrial ecosystems: ecological regions and districts (McEwen 1987), Land Environments of New Zealand (LENZ, Leathwick et al. 2003), and the New Zealand Land Resource Information System (NZLRI). The latter two databases are online, accessed through http://lris.scinfo.org.nz/. Most regional councils have mapped Significant Natural Areas.
- *Desk-top surveys:* these should use aerial photographs and topographical maps to identify the main ecosystems or land cover classes, which are then ground-truthed to allow more detailed mapping. Manaaki Whenua – Landcare Research (National Vegetation Survey Databank) and DOC provide information on standard methods for ground surveys.[19] Surveys need to include measures that allow an assessment of the regional significance of species and ecosystems present. Significance criteria can vary from region to region, and an ecologist cannot assign mana whenua values to ecological features. Section 5 of *Ecological Impact Assessment* (EIANZ 2018) identifies how to assign ecological values in New Zealand and proposes a method encompassing four ‘matters’ : representativeness, rarity/distinctiveness, diversity and pattern, and ecological context. This supersedes Myers (2011) and Sawyer & Stanley (2012), which also provide a useful discussion of criteria with respect to the RMA.

[17] http://www.doc.govt.nz/nztcs
[18] http://www.landcareresearch.co.nz/resources/maps-satellites/threatened-environment-classification
[19] http://www.doc.govt.nz/our-work/biodiversity-inventory-and-monitoring/vegetation/

Table 2. Ecological matters to consider when assigning value

Criterion	Description
Representativeness	• Structure and composition, indigenous species dominance, species assemblages, etc.
Rarity and distinctiveness	• Amount of remaining habitat or vegetation, national priority for protection, unusual species or assemblages, endemism, distinctive ecological features
Diversity and pattern	• Level of natural diversity, abundance and distribution • Biogeographical pattern and complexity, and temporal considerations
Ecological context	• Site history, essential characteristics, size, shape and buffering, condition and resilience, ecological linkages, pathways, and exchange of genetic material

Source: based on the much more detailed Table 4, EIANZ 2018

- *Animals:* 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 some fauna and flora; for example, rats are commonly monitored using wax chew tags, and bird monitoring often uses a 5-minute bird count method. Monitoring is often weather or seasonally dependent; for example, invertebrates are usually sampled during summer months, when activity is greatest. DOC has developed biodiversity inventory and monitoring standards.[20] Some regional councils may have specific requirements.
- *Age and complexity of ecosystems:* when combined with species, this indicates the time frame for, and difficulty of, rehabilitation . The age of an ecosystem can be assessed from tree ring cores, the presence or absence of plant and invertebrate species with very slow dispersal, soil/landscape assessments, and the history of disturbance (inferred from burning, clearing, aerial photographs and cultural records). In natural sites areas with erosion scars, old drill sites or earthworks can be useful sites.
- *Potential rehabilitation materials, and their salvageability and vulnerability to degradation when stockpiled:* potential rehabilitation materials depend on the objectives of rehabilitation. The ability to salvage and store these materials largely depends on the mine schedule, the mining equipment available, and access constraints (e.g. steep slopes). Within a mine site there are also competing uses for some materials; for example, rock that resists erosion and does not generate acid may be sought-after as a road-surfacing material, for line diversion drains to protect them from erosion, and for creating rock outcrops and habitat features within rehabilitated areas. Similarly, NAF overburden that can be compacted to low permeability will be useful to cap and seal PAF, and as a root zone, especially if topsoils are scarce.

Heritage considerations for legacy sites

Management of historical mining artefacts can be a highly valued part of mine rehabilitation and compensation. Rehabilitation of abandoned mines can enhance both heritage and environmental outcomes, which can support future recreation and tourism activities. Retention of the heritage fabric and environmental remediation/rehabilitation activities should go hand in hand, as the latter can involve disturbing, or uncovering, features of heritage value. Where these include activities from pre-1900, the Heritage New Zealand Pouhere Taonga Act 2014 requires an authority before works begin,[21] but more recent activity can also be of heritage value, so consult DOC or Heritage NZ if your site has surface or underground history (see Case Study 4 below).

Case Study 4: Enhancing the heritage value of Bellvue Mine

Bellvue mine is an underground coal mine that ceased production in 1970 and was simply abandoned (as was practice at the time). It has a discharge of acid mine drainage (pH 2.4 to 2.9) into Cannel Creek, which is lifeless below the discharge point due to elevated metals and low pH (Trumm et al. 2016). It is about 15 km northeast of Greymouth on the West Coast of the South Island, accessed from the main coast highway. The Centre for Minerals Environmental Research (CMER) is rehabilitating the Bellvue mine in partnership with local organisations (Grey District Council, the West Coast Coal Heritage Trust, Minerals West Coast, Tai Poutini Polytechnic, Go West Coast). Rehabilitation has included treating AMD from the Bellvue adit using passive mussel shell reactors, and revegetation of part of the coal storage area along with parts of the road, which were widened to allow access.

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[20] http://www.doc.govt.nz/our-work/biodiversity-inventory-and-monitoring/
[21] http://www.heritage.org.nz/

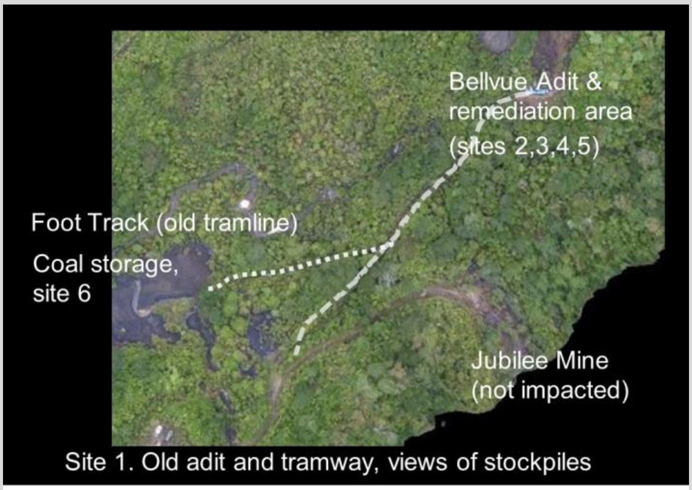

Figure C7. Overview of identified heritage areas.

Many relics of historical mining activities remain on site. The first step in assessing the heritage of the site was to work with DOC to assess and map the area. This identified six specific sites with heritage features. In addition, the remediation planning and induction programme ensured all workers knew how to treat any mining relics (pieces of iron, wood, etc.).

An additional driver for enhancing heritage aspects is the potential of the site to support a mountain bike cycleway from the Strongman mine area to Greymouth, via the James Mine sites and the Point Elizabeth track. This would be a tourism asset and was proposed by the local mayor, and would complement the new Paparoa Track opening in 2019 (https://www.doc.govt.nz/parks-and-recreation/places-to-go/west-coast/places/paparoa-national-park/things-to- do/tracks/paparoa-track/).

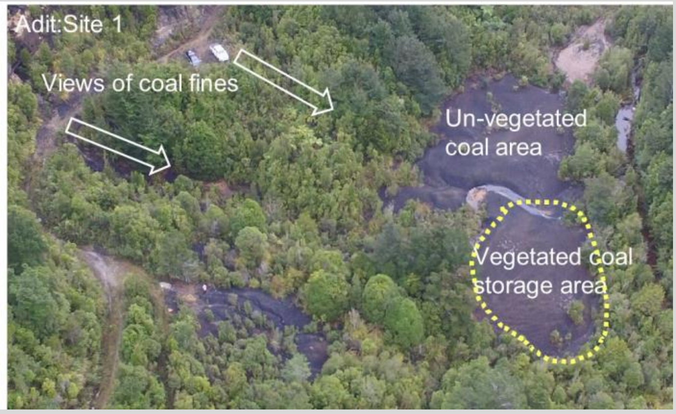

Figure C8. Detail of heritage sites 1 and 6, the latter showing area of coal fines being actively revegetated (vegetated).

These sites reflect different elements of the operation. Specific actions were identified to ensure heritage values were conserved and could be built upon in the future. Heritage site 1 is the location of an adit, and has remnants of an aerial tramway and views into the tramways destination of a coal storage area (heritage site 6) (Figure C8). The site is suited to future interpretation of mining heritage. The main action required was to ensure road fill was not dumped at the site and the road was not widened in this area. Adjacent weeds (mainly gorse) were treated. At heritage site 6, an unvegetated section of the coal fines area was retained to provide historical context, while the other section is being revegetated to enhance the terrestrial environment and reduce ARD flowing from the coal fines.

Heritage site 2 is the location of Bellvue coal bins (Figure C9), with the key action being to ensure no damage to the external structure of the bins. At heritage sites 3 (adit), 4 and 5 (below), numerous small metal and wood items from historical mining activity were present. Here, the main actions were to ensure these materials were salvaged and placed in adjacent areas where they were accessible, so they could be the basis for a restorative work in the future.

Figure C9. View of heritage sites 2 (coal bins), 3 (adit), 4 and 5 - the latter showing small metal and wood items.

Key findings relevant to mine rehabilitation:

- Protecting historical artefacts and sites from deterioration, enhancing their condition, safe accessibility and/or interpretation can create legacy assets for the local community.

Key references:

Trumm D, Cavanagh JE 2006. Investigation of remediation of acid-mine-impacted waters at Cannel Creek. Landcare Research Contract Report LC0506/169, CRL Report 06-41101, https://www.wcrc.govt.nz/Documents/Environmental%20Management/BellvueMineRemediation.pdf

Trumm D, Pope J, West R, Weber P 2016. Bellvue Mine AMD – downstream geochemistry and proposed treatment. AusIMM https://www.researchgate.net/profile/James_Pope2/publication/317612962_Bellvue_Mine_AMD_-_Downstream_geochemistry_and_proposed_treatment/links/5943337aa6fdccb93ab275b6/Bellvue-Mine-AMD-Downstream-geochemistry-and-proposed-treatment.pdf

3.4 Prediction of mine drainage and downstream water chemistry for proposed new operations

3.4.1 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. Further, pre-mining activities provide the opportunity to identify the resources required for rehabilitation, which is critical in determining the feasibility of operations. Pre-mining forecasts of both operational and closure water quality are an important aspect of the mine planning and consenting process and clarify to regulators and stakeholders what the environmental effects of the project will be. During mine planning and operations, detailed predictions of mine drainage chemistry and evolution over time will assist with water management and treatment. Predicting mine drainage chemistry can help to predict 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.

3.4.2 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 units 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 only, and detailed analyses of the various rock types that will be disturbed at a proposed mine site are required to provide quantitative assessment. 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 are the minerals that are reactive after mining-related disturbance, including acid-forming or neutralising rocks and minerals that release trace elements.

Rocks that host epithermal gold mines occur in the active volcanic areas of New Zealand (Taupō Volcanic Zone), the Coromandel Region and Northland. Historically and currently, mining of epithermal deposits has only taken place in the Coromandel area, although exploration programmes have been conducted throughout Northland and the Taupō Volcanic Zone. The host rocks for these deposits are volcanic, typically with intermediate to felsic geochemistry, which have been uplifted and eroded since deposition. The most up-to-date compilation of geological maps for New Zealand, showing these units, is the 1:250,000 Q Map series.[22] More detailed geological maps for specific areas may be available from GNS, university theses, published reports and scientific papers, or mineral exploration reports.[23]

Epithermal deposits can form either or both acidic or neutral mine drainages depending on the mineralogy of the rocks that are mined. Sulphide mineral precipitation occurs throughout epithermal mineral deposits, and the specific mineral types depend on the geochemistry of the geothermal fluids that were present during deposition of the mineral deposit. Different sulphide minerals precipitate at different depths within the geothermal system (Figure 11) controlled by mineral solubility, temperature, boiling and other factors. In addition, epithermal systems can have several spatially overlapping generations of mineralisation, which means it can be difficult to predict which minerals will be present in an epithermal mineral deposit (Mauk et al. 2011). Carbonate alteration

[22] GNS website: http://www.gns.cri.nz/research/qmap/aboutqmap.html
[23] Crown Minerals website: http://www.crownminerals.govt.nz/cms

is a common feature of many (but not all) epithermal deposits, and if it is present, neutral mine drainage with or without trace elements can occur.

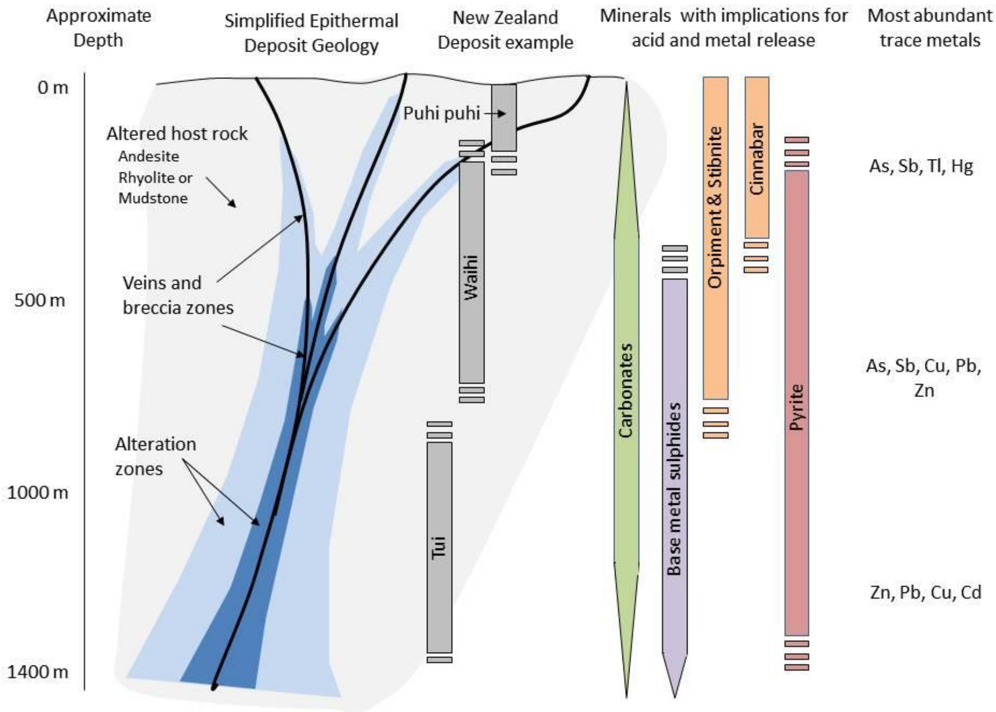

Figure 11. **Schematic of epithermal mineral deposit with a generalised model for the distribution of selected minerals that release or neutralise acid or contain trace elements.**

Both acidic and neutral mine drainage from epithermal deposits can have enriched concentrations of trace elements (Pope & Trumm 2015). Pyrite is often present in epithermal deposits and is an important source of acid. However, the variety of other sulphide minerals that can be present in epithermal mineral deposits means that the chemistry of epithermal mine drainages is more variable than drainages from either coal deposits or mesothermal deposits. In addition, neutralisation (or partial neutralisation) by carbonate minerals causing precipitation of secondary Fe oxyhydroxides and adsorption of trace elements to a variable degree adds another layer of complexity to drainage chemistry from epithermal mines.

Where proposed new mine operations overlap or are adjacent to historical mines, information on drainage chemistry 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 and alteration/mineralisation style. Different geological formations with different mine drainage characteristics can occur adjacent to each other. In addition, mine chemistry evolves with time, and therefore direct comparison of historical mine drainages to future mine drainage chemistry could be difficult.

3.4.3 Analysis of rocks from the proposed mine site

Field observations and measurements 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 selecting 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 the 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.

Important minerals and field observations

The type, abundance, distribution and reactivity of minerals within rocks strongly influence the chemistry of mine drainages, the associated potential environmental impacts, and the management or treatment strategies required. The most important groups of minerals that influence mine drainage chemistry are sulphides and carbonates (Plumlee & Logsdon 1999a, b; Appendix C3 in

Cavanagh et al. 2015). Oxidation of sulphide minerals often 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, and these influence the acid-producing or neutralising characteristics of rocks.

The distribution of sulphide, carbonate and secondary minerals varies within and between rock types in a gold deposit. Therefore a general geological/mineralogical description of any samples collected is essential for the interpretation of analytical data. Details of what should be included in a general geological description are provided in Appendix C.4 in Cavanagh et al. (2015).

Field or hand-specimen observations of minerals in rocks to be disturbed by mining is a qualitative tool for assessing mine drainage potential. These observations are useful for selecting rocks for further analyses and interpretation of data once the analytical results are obtained. Important observations include:

- a general geological description of a rock type or sample (Appendix C.4 in Cavanagh et al. 2015)
- the presence of primary sulphide minerals, particularly pyrite, chalcopyrite, orpiment, realgar, sphalerite and galena,
- the presence of carbonates, particularly calcite, ankerite and siderite
- the presence of secondary minerals that indicate the reactivity of rocks when exposed to the surface, such as Fe^{3+} (ferric) oxides, hydroxides and hydroxysulphates, Al hydroxides, and sulphate minerals.

Further details, including photos of different rock types, are provided in Appendix D.5 in Cavanagh et al. 2015. Field observations of minerals in hand specimens or outcrops are qualitative and do 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 data, and so that previously unidentified rocks that may have implications for mine drainage chemistry are identified, analysed, and appropriately managed.

Sampling strategies for the geochemical characterisation of rocks

Quantitative laboratory analysis of rock geochemistry is required to make more robust predictions of likely mine drainage chemistry from a proposed new mine site. Trace element concentrations in rocks can be determined by techniques such as X-ray fluorescence (XRF) or inductively coupled mass spectrometry (ICP-MS).

Quantitative testing can be undertaken for several purposes, and sample collection strategies will differ accordingly. For the purposes of characterising 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. Samples collected during site characterisation for acid–base accounting (ABA) 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
- identification and sampling of specific geological features that have anomalous acid-producing or neutralising characteristics including:
    - a representative and statistically significant suite of samples from the rock types to be disturbed.
    - epithermal gold deposits, which could have anomalous acid-producing or neutralising minerals or trace elements that are localised in
        - different rock types
        - mineral alteration zones
        - mineralised zones
        - faults, veins or breccia zones
        - hanging wall vs foot wall zones
- collection of sufficient data so that statistically meaningful ABA values can be identified or calculated for important rock types.

In general, an early phase of 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 can be selectively mined and managed appropriately (see also section 3.6). Additional sampling of rocks is required throughout resource development and mining so that rocks continue to be appropriately managed (see also section 4.3).

How many samples?

The density of samples required for an assessment of mine drainage chemistry can be less than that required to identify and define a gold resource and can accompany exploration or resource definition drilling. There is little exact guidance on the number of samples, although sample density guidance from the Canadian Mine Environment Neutral Drainage (MEND) programme can be used as a starting point (Table 3).

Table 3. **Guide to the number of samples for geochemical testing.**

Tonnes waste rock	No. samples
10,000	3
100,000	8
1,000,000	26
10,000,000	80
100,000,000	236
1,000,000,000	711

Source: Modified from Price 2009 to cover a disturbance of up to 1 billion tonnes using the relationship ABA no. = $0.0346x^{0.4792}$

However, guidance on the number of samples that need to be collected should be provided by a suitably experienced geochemist because there might be circumstances where lower or higher sampling densities are justified prior to consent depending on the geological complexity of the mineral deposit. Several factors could influence sample density, including:

- project phase
- geological or geochemical complexity
- quantities of critical materials (e.g. NAF for capping of overburden dump)
- quality assessment and quality control for geochemical data.

This is reflected in Figure 12, which shows a comparison of suggested sampling density with the actual number of samples collected from New Zealand projects or mines during resource consenting and/or operations. There is a significant increase in the number of samples collected during operations of the Escarpment mine compared to that used during resource consenting. This is a common trend, but also reflects the geological complexity at this mine, as well as limitations on the availability of critical materials, necessitating a higher level of certainty for characterisation. In contrast, additional samples collected during resource consenting for Te Kuha did not change the conclusions of the proposed overburden management plan, as the site has simpler geology and less limitation on critical materials.

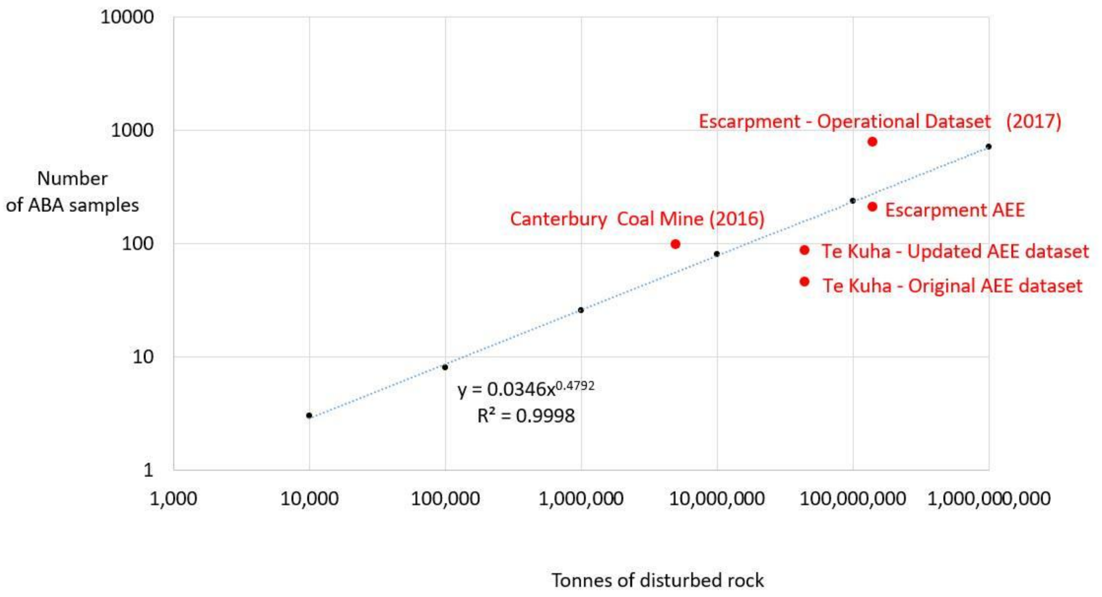

Figure 12. **Guideline for number of samples for ABA analysis based on tonnes of waste rock disturbed, and the number of samples collected from different New Zealand mines for assessment of environmental effects (AEE) for consenting or during operations.**

Density of sampling should ensure that the complete sequence of rocks to be disturbed by mining is sampled. Sufficient samples of each rock type encountered in the sequence that is to be disturbed by mining should be submitted for ABA analysis to enable classification of the rock with the desired statistical certainty.

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 the results of ABA tests.

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

- a statistically significant suite of samples from each rock type to be disturbed by the proposed mining
- rocks from mineralised areas and alteration zones
- rocks from close to faults
- rocks from near the surface
- rocks from below the oxidised zone in alluvial sediments
- representative samples of overburden types.

If the mine site overburden management plan requires rock types with specific geochemical properties that are in short supply, then the volumes of these rocks should be calculated and modelled to a similar level of certainty as the volumes of mineralised rocks. For example, if oxidised rock or NAF rock is required for capping, or low metal content rocks are required to line waterways, the volumes of these rock types that are available on the mine site should be calculated accurately during mine planning.

We recommend calculating the amounts of these materials that are required to deliver an environmental plan to a similar level of certainty as mineralised rocks that have volumes calculated to define their mineral resources (AusIMM 2012). In addition, mine scheduling should be completed to ensure these materials are available on the mine site when they are required. Once approvals have been given for the project and operations commence, further testing can be undertaken to address uncertainties. This is often part of the transition from the pre-mining conceptual AMD management plan to the operational AMD management plan.

Metal and sulphur contents

The trace element concentration of rock samples can be determined by a number of analytical methods, including XRF and ICP-MS. 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 can be released into solution. In contrast, trace elements in silicates or some oxides are less reactive and less likely to be released into solution. To identify trace elements that are reactive and environmentally available, kinetic tests with trace element analysis of leachate are required.

Acidity and various metals are the main aspects of environmental significance in drainages from epithermal gold mines. Salinity can also be elevated in mine-impacted drainage compared to natural catchments. Mineralised areas that are not mined will have naturally elevated concentrations of components that occur in mine-impacted drainage after mining operations commence. These areas lead to natural biodiversity, and therefore thorough baseline sampling programmes are required to identify natural background variability in the pre-mining landscape, so the post-mining rehabilitated landscape will also target generation of biodiversity.

The gold content of ore rocks in epithermal mineral deposits is usually in the range of 1–50mg/kg, but the trace elements content can be several orders of magnitude higher, even up to g/kg in highly mineralised zones. Trace elements that might be enriched to this level include Fe, Zn, Pb, Cu or As, and there is a suite of other metals that could be enriched by several orders of magnitude above unmineralised rocks, including Cd, Hg, Sb, Tl, Se and Mo. Processing rocks that host epithermal gold will inevitably produce waste that is elevated in trace elements.

The waste streams include residue from ore that is processed for gold (tailings) and material that must be disturbed to access the ore (waste rock). Waste rock can be either unmineralised or mineralised but without recoverable gold concentrations. Mineralised waste rock is likely to occur in a halo around the ore zone and might require selective handling, management and storage during mine operations. It is likely that the mineralisation and alteration system will selectively concentrate some trace elements but not all elements, and that the pattern of relative enrichment for the different trace elements might be complex and not necessarily richest where the gold is richest.

Often the enriched trace elements will be associated with sulphide minerals, and break-down of these minerals will also release sulphate to the environment. Sulphate can become enriched in mine-influenced drainages to concentrations that exceed recommended discharge limits. However, the enriched trace elements can also be associated with a variety of other minerals, including carbonates, sulphates and sulphosalts.

Acid–base accounting (ABA) and analysis of stored acidity

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. At an epithermal mine, acidification and/or trace element enrichment of mine waters is likely to require on-site management and treatment. The main potential sources of acid or trace element generation are:

- mineralised rocks in pit walls and underground excavations
- ore stockpiles
- sulphide concentrates
- sulphide-rich tailings
- residues (solids and waters) from ore processing.

There are several reasons to undertake ABA analysis, including to:

- determine the presence or absence of PAF materials
- confirm the presence of sufficient acid-neutralising rocks
- predict the pH and acidity of mine drainage chemistry
- establish relationships between specific rock types and acid production or neutralisation
- optimise management of waste rock or overburden/interburden with respect to mine drainage chemistry
- select rock types for more detailed geochemical analyses.

ABA 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.

ABA 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. ABA 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 to the interpretation of the results are listed below; further details are available in Appendix C.6 in Cavanagh et al. 2015.

Maximum potential acidity (MPA)

- Total sulphur (S) is often used as a conservative approach to determine the maximum potential acidity (MPA) that could be generated by a rock sample. It is calculated assuming all S is sourced solely from the mineral pyrite (FeS_2), with the results expressed as kilograms of H_2SO_4 per tonne of rock (kg H_2SO_4/t).
- Usually MPA values are between 0 and 200 kgH_2SO_4/t.
- MPA analyses are commonly combined with acid neutralising capacity (ANC) analyses for interpretation.
- There are some important limitations to MPA testing that should be understood when interpreting the results of MPA analyses (see Appendix C.6 in Cavanagh et al. 2015).

Acid neutralising capacity (ANC)

- ANC is the amount of acid that can be neutralised by a rock sample. It generally 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 kg H_2SO_4/t so that it can be directly compared with MPA.
- ANC values for rocks are commonly between 0 and 200 kg H_2SO_4/t, although highly carbonate-rich materials such as limestone have higher ANC values (>200 kg H_2SO_4/t).
- Rocks with less than 1 kg H_2SO_4/t should not be relied on for ANC, and sometimes data can even indicate negative ANC values (e.g. Weber et al. 2005a,b), which therefore requires assessment by a competent geochemist.

Net acid production potential (NAPP)

- Net acid production potential (NAPP) is calculated as NAPP = MPA − ANC, and is reported in units of kg H_2SO_4/t.

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 sample.
- 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 kg H_2SO_4/t, and the end pH of the NAG solution is also measured.
- Samples with NAG pH > 4.5 are defined as NAF.
- There are some important limitations to 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 in Cavanagh et al. 2015.

There are some important limitations in the applicability of NAG analysis, and the 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 in Cavanagh et al. 2015).

Paste pH

- Paste pH is a field-based analysis that provides an indication of the readily soluble acidity in crushed rock. It 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 at either 5 min, 12 hr, or 24 hr.
- A sample with a pH less than 5.5 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.

Soluble acid leach testing

- Soluble acid leach tests are designed to identify acidity that is stored in the sample and may become mobilised through interaction with water. This differs from the ABA tests above (MPA, NAPP, NAG, etc), where the objective is to quantify the potential acidity if minerals in rocks interact with water and oxygen. The distinction between these types of tests is subtle, but important for management practices.
- In general, soluble acid leach testing is most applicable to oxidised or partially oxidised rock.
- Methodologies for these tests are borrowed from the assessment of acid sulphate soils, which are common in Northern New South Wales and Queensland, Australia (Ahern et al. 2004). The tests include:
  - 1 Mol KCl digestion and analysis for dissolved cations/anions, followed by titration to pH 7.0 to determine the titratable actual acidity; acidity is related to soluble oxidation products such as melanterite
  - 4 Mol HCl digestion to measure the sparingly soluble oxidation products such as jarosite; the liquor is analysed for sulphate and is then used in conjunction with 1 Mol KCl data.

Interpretation of acid–base accounting analyses

MPA, ANC, and NAG analyses are the primary analyses that are useful for determining the acid-forming status of rock samples. Commonly, ABA data are plotted with NAG values on a *y*-axis, and NAPP (Figure 13A) or the ratio of MPA:ANC on an x-axis. 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 in Cavanagh et al. 2015). The geological description of the sample (Appendix C.4 in Cavanagh et al. 2015) is often the most important piece of information used to interpret samples that plot in the uncertain quadrants.

An alternative approach, adopted in the MEND guidelines (MEND 2009), utilises the ANC:MPA ratio (Figure 13B). Typically, material with an ANC:MPA ratio of less than 2 is classified as NAF material, while an ANC:MPA ratio of less than 1 indicates potentially acid-generating material. If the ANC:MPA is between 1 and 2, the sample classification is uncertain, possibly PAF if ANC is insufficiently reactive or is depleted at a faster rate than sulphide oxidation and subsequent acid generation.

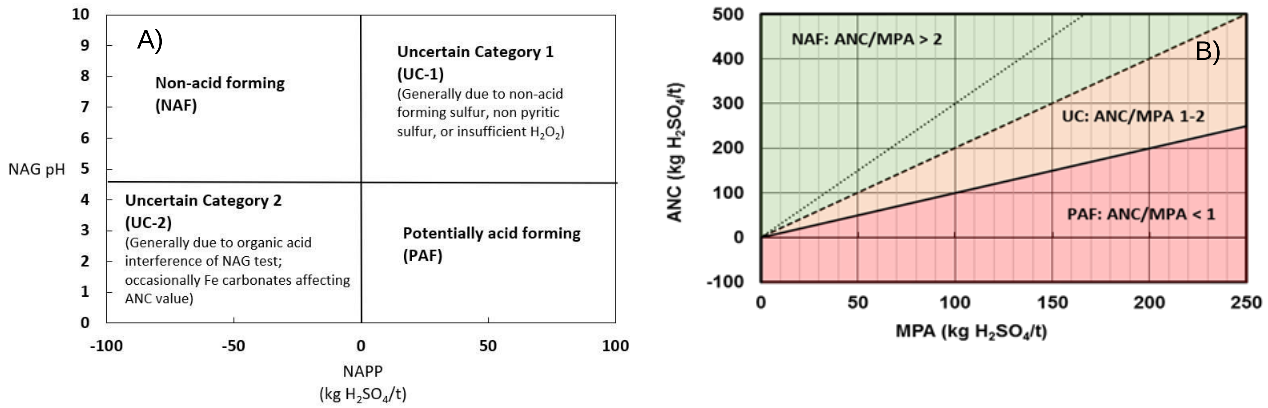

Figure 13. A: NAG pH vs NAPP graph. Samples that are acidic according to both NAG and NAPP tests plot in the PAF field, while those that are non-acidic in both tests plot in the NAF field. Samples where one of the analyses indicates acidic and the other indicates non-acidic plot in the uncertain field and require further investigation. B: ANC vs MPA graph. Material with an ANC:MPA ratio <2 are classified as non-acid forming material, while ANC:MPA <1 indicates potentially acid-generating material. If the ANC:MPA is between 1 and 2, the sample classification is uncertain (source: Olds et al. 2015).

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 and the mine plan (including mine scheduling information and kinetic test data, see section 3.4.5 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.
- 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 equal in quantity to 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.

Ongoing monitoring of the rocks disturbed during mining 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. As ABA data are collected and collated, an understanding of the relationships between the geochemical parameters of a suite of rocks and the environmental implications of the measurements can be developed. As the geochemical database increases there can be opportunities to streamline analysis and interpretation, as described in Case Study 5 below.

Case Study 5: Geochemical classification of waste rock using a process flow approach

Geochemical classification of waste rock involves a range of acid–base accounting (ABA) tests. The established approach to classifying mine waste rock is to use a simple combination of ABA tests, such as NAPP and NAG, to provide two methods to assess whether the sample is PAF or NAF. This matrix-style classification system (e.g. Figure 13 in the main text), which is often a requirement of resource consent conditions, is expensive and time consuming when applied as a blanket approach to classification. Matrix-style classification can lead to a large proportion of samples with conflicting results or incorrect classification of samples. An example of the original Escarpment coal mine classification scheme is provided in Table C2.

Table C2. Original resource consent matrix-style classification for the Escarpment coal mine

Classification	Paste pH	NAG pH	NAPP acidity (kg $CaCO_3$ eq./t)
NAF	>4.5	>4.5	<0
Low risk	>4.5	>4.5	<5
PAF	<4.5	<4.5	>2

An alternative waste rock geochemical classification methodology is to use a process flow approach to optimise the classification and testing regime. This is an iterative approach that relies on detailed knowledge of site geology and geochemistry, and the completion of a suitable sampling programme, incorporating ABA. For the Escarpment coal mine, total sulphur was determined to be a key variable, with samples with total sulphur less than 0.015 wt% being immediately classified as NAF, while samples with total sulphur between 0.45 and 0.7 wt%, and over 0.7 wt%, classified PAF and high-acid-forming, respectively (Figure C10). Additional data or testing are required on samples with total sulphur between 0.015 and 0.045 wt% to enable classification (Figure C10). Confirmation of the correct classification of materials was determined by kinetic testing. This process flow approach resulted in far fewer samples being classified as uncertain compared to the current resource consent matrix-style classification (Figure C11) and at a lower theoretical cost (Table C3).

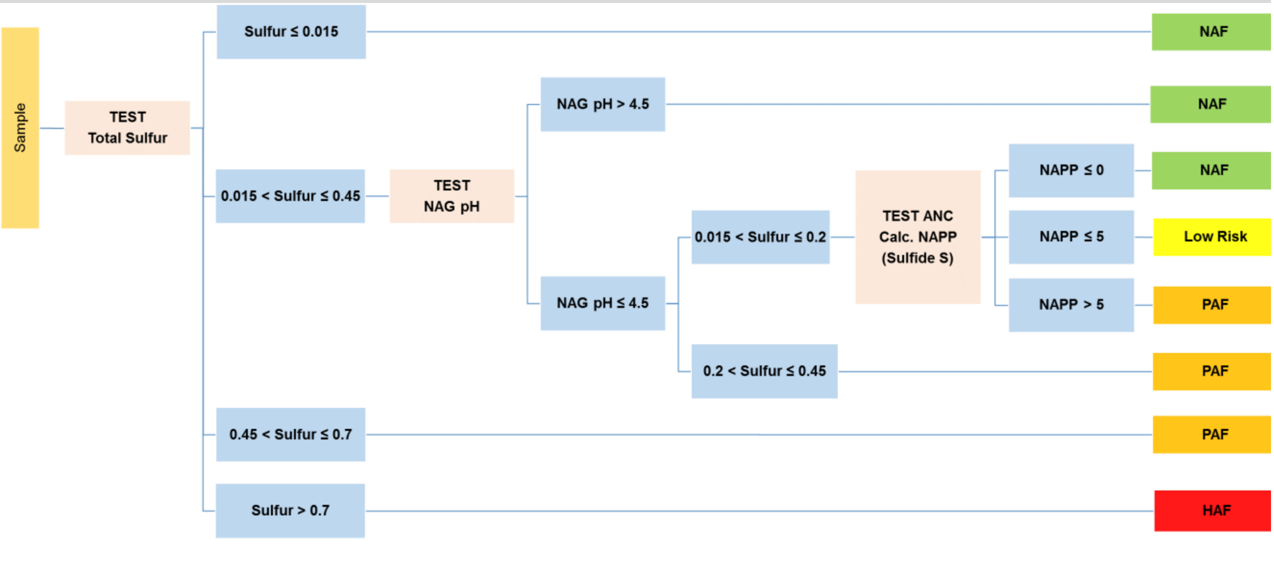

Figure C10. Escarpment coal mine waste rock geochemical classification using the process flow approach. Sulphur values are wt%.

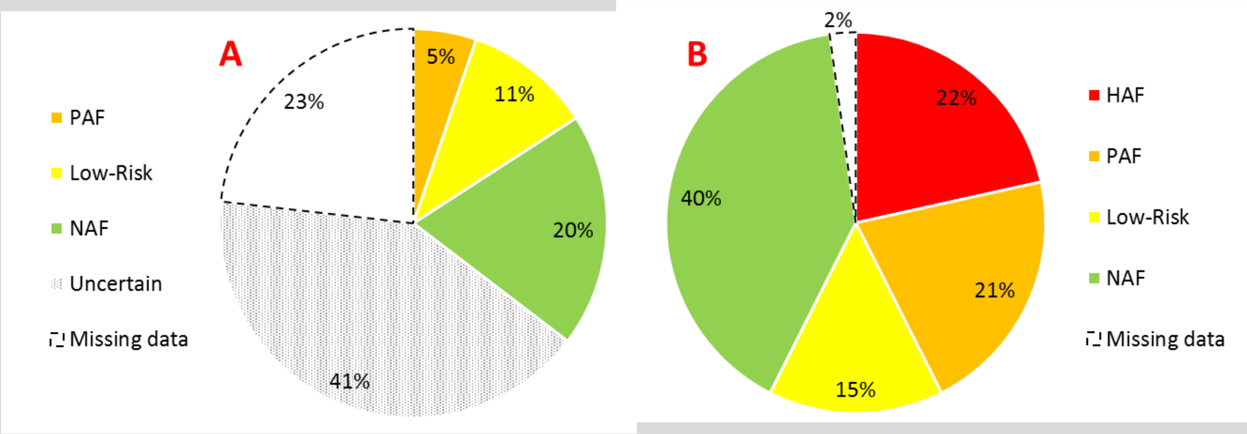

Figure C11. Sample classification by the (A) matrix-style and (B) process flow methodologies.

Table C3. Escarpment coal mine ABA testing and cost differences between the process flow approach and the matrix-style approach (resource consent approach)

	Test cost	Process Flow		Matrix style	
	$/analysis	No. tests	Cost	No. tests	Cost
Total S	$20.00	726	$14,500.00	726	$14,500.00
NAG pH	$35.00	430	$15,100.00	726	$25,400.00
ANC	$45.00	179	$8,100.00	726	$32,700.00
Sulfide S	$35.00	179	$6,300.00	-	-
Paste pH	$15.00	-	-	726	$10,900.00
Total	-	**1514**	**$44,000.00**	**2904**	**$83,500.00**

Key findings in relation to geochemical testing are:

- Process flow methodology can increase the certainty of classification of materials and reduce the number of tests required, and also the cost of testing for samples collected after determining the classification process
- Development of a process flow approach requires an initial detailed understanding of the geochemistry of the rocks and its ABA data.

Key reference:

Olds WE, Bird B, Pearce JI, Sinclair E, Orr M, Weber PA 2015. Geochemical classification of waste rock using process flow diagrams. New Zealand Annual AusIMM Branch Conference, Dunedin, 31 August – 2 September 2015. Pp. 307–318.

Acidity that is mobilised through soluble acid leach testing is the most easily mobilised form of acid that is associated with a rock. The minerals that contain this kind or acid often form as partial reaction products from the oxidation of sulphides. As a result, these tests have most application during assessment of the oxidised part of a mineral deposit, during the assessment of acid runoff from pit walls, during leachability assessments of metals from tailings, or when old waste rock must be re-handled.

These tests indicate the amount of acid and trace elements that will be rapidly released from the rock just by interaction with water. The different reagents indicate relative mobility, but there are no standardised approaches to interpreting the results. Assessment of the applicability of these tests and their interpretation should be completed by an experienced geochemist, and geochemical testing to assess the trace element content of rocks should be carried out.

Other contaminants of concern

During mining, nitrate can be released at elevated concentrations because of the chemistry of the explosives used to blast waste rock prior to excavation. The concentration of nitrate that remains in waste rock and the concentration that occurs in mine-influenced water is difficult to predict. The discharge of nitrate from mine sites is an emerging issue, however, and the importance of this impact will depend on local conditions such as rainfall (Morin & Hutt 2009), background concentrations of nitrate, and uptake by plants or organisms.

Waste rock models

ABA data and geological information are used to develop a waste rock distribution model based on the classification system (e.g. Figure 13). A waste rock distribution model provides the materials quantity list for the mining project, including topsoil, NAF, PAF, and other materials. The waste rock distribution model enables development of a mine schedule that describes where different types of materials will report during mining. This schedule ensures there are sufficient quantities of the right materials available to implement the overburden management plan during mine operations. Such data are essential for the scheduling and placement of PAF and high-PAF waste rock within the core of a waste rock dump (WRD), or the construction of NAF caps or other waste rock management activities. Initially the waste rock distribution model might be based on geological information and interpretation, but the geological model might be upgraded to a geo-statistically defined model (e.g. Sinclair 2018), such as a block model as the project advances.

3.4.4 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, the lag period preceding acid generation, and the 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 are 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 the following:

- sulphide oxidation rates, acid generation and mine drainage evolution trends
- amounts and rates of metal or metalloid mobilisation into mine waters
- simulation of the impact of ore processing and generation of tailings on mine drainage
- amounts and rates of sulphate mobilisation into mine waters
- 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 (e.g. testing different mixing ratios of limestone to PAF material in WRDs).

Kinetic tests should commence on selected samples of interest during initial geochemical assessment of a site because 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 that no treatment is required.

Prior to operations, usually only small volumes of material are available and kinetic tests are limited to leach testing on samples with a maximum mass of about 50 kg. Additional information on kinetic testing is presented in chapter 4 (‘Operations’) below. During operations, more complex kinetic testing can be completed.

3.4.5 Prediction of water quality downstream of an epithermal gold mine

Groundwater, surface-water runoff, and mine process water at a mine site all have the potential to chemically interact with mineralised rocks or tailings, and these mine waters develop distinctly different compositions from the natural background waters. 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 the predicted chemistry and volume of potential mine drainage to predict the water quality downstream of a mine.

Site hydrogeology and background water quality information are integrated with information on mine drainage to predict downstream water quality using reactive transport modelling (Figure 14). Reactive transport modelling is required 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 that is based only on dilution ratios of different mine drainage components is inadequate. Required information may include:

- flow rates
- alkalinity, acidity and pH
- dissolved Fe (preferably both Fe^{2+} and Fe^{3+}), Al
- metals likely to be released from mine wastes (Zn, Ni, Mn, Cu, Cd, Cr, Co, As, Pb, etc.)
- dissolved oxygen

- fine-grained particulate Fe- and Al-oxide and hydroxide mineral abundance
- major cations and anions (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, SO_4^{2-}, Cl^{-}, HCO_3^{-})
- redox conditions.

Often hydrogeological parameters are input into probabilistic process flow modelling software such as GoldSIM™, which allows a full range of climate variables, hydrogeological properties, runoff to infiltration, evapotranspiration, and other factors to be modelled and assessed through time. The outputs form an assessment of the physiochemical model, can be matched up with mine water management infrastructure to ensure the infrastructure will handle the variability in flow and chemistry that is likely for the site.

This type of modelling is a powerful tool for predicting outcomes at mine sites prior to operations, and these models are also often useful for the mine site once the operation is established. For best results, probabilistic modelling is calibrated to field data such as stream-flow variability, rainfall variations, infiltration and runoff measurements, along with other measured parameters. GoldSIM™ probabilistic modelling does not take into account geochemical reactions that occur within the mine environment. To model geochemical reactions, additional software is required that completes chemical equilibrium modelling, or reactive transport modelling, such as PhreeqC or Geochemists Workbench.

Probabilistic modelling, chemical equilibrium modelling, and reactive transport modelling require 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).

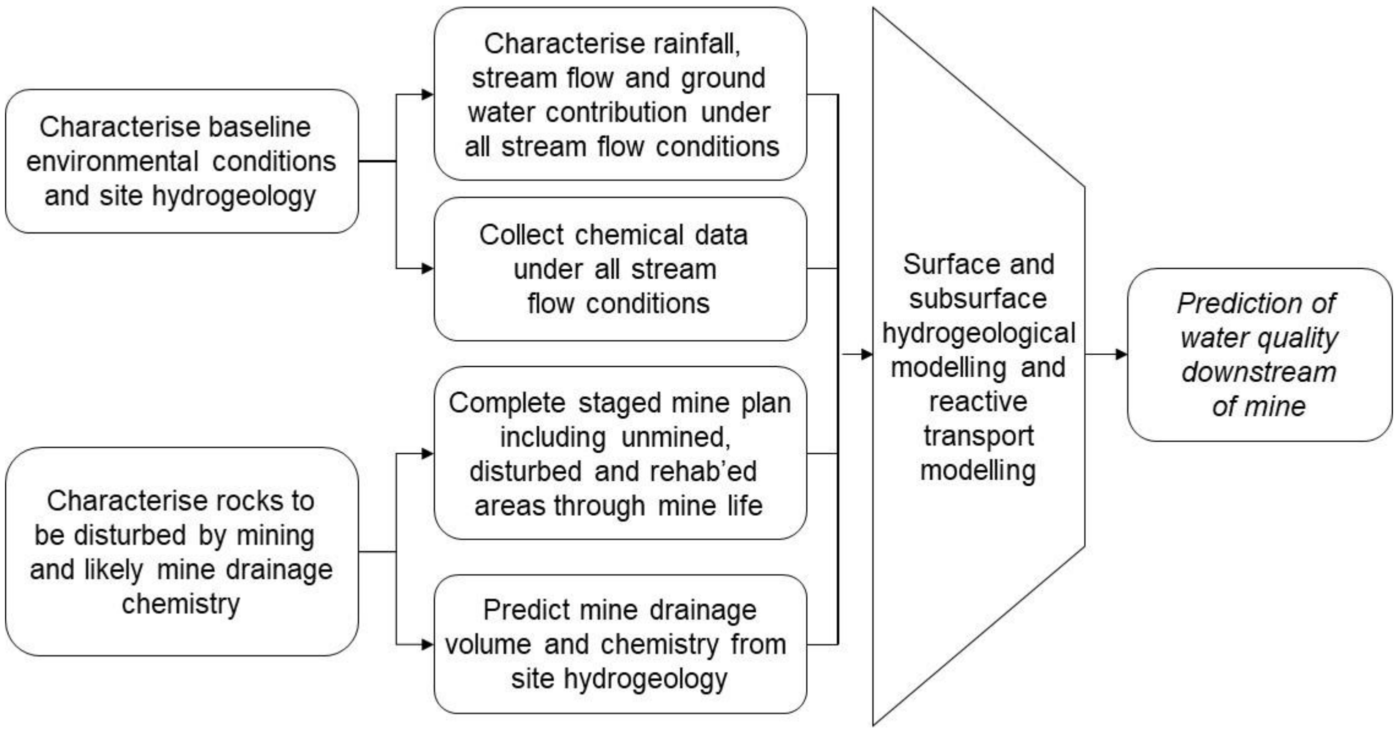

Figure 14. **The basic process for determining water quality downstream of a mine.**

The determination of likely mine drainage chemistry and collection of relevant site hydrogeological and background water quality for epithermal gold mines is outlined below.

Water quality at epithermal deposits reflects the mineralogy that characterises these deposits: pyrite, carbonates and various other sulphide minerals (Craw 2001; Christie et al. 2007). 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 are available are Martha, Golden Cross and Tui, although some of the historical mines have also been studied (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 (Giles et al. 2010). Golden Cross was mined during the 1980s and 1990s and rehabilitated in the late 1990s, water quality was predicted through acid–base accounting testing (Miller 1987), and treatment systems were developed (Peterson & Kindley 1993. 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 operational and has active water treatment and a progressive rehabilitation programme in progress, along with trials of passive treatment systems that might be used at closure (Trumm et al. 2018). The geochemistry of waste rock and tailings at these deposits and the waste rock and tailings management options selected are the main factors that control water quality at these sites.

Often open-cast epithermal mines will have a pit lake at the end of operations, and this will be included in closure planning. Pit lake water chemistry will have to be predicted during pre-operations planning phases and is dependent on the types of rocks that form the walls of the pit (Castendyk et al. 2005), the rate of pit filling, and ground-water chemistry. Pit lake chemistry has been predicted for Martha mine (Castendyk & Webster-Brown 2007a, b, 2010). The outcome of the pit lake water chemistry predictions including chemical stratification and lake turnover, as well as the desired final use or discharge criteria for the pit lake, will determine if treatment pit lake water is required.

During pre-operations, decisions are made related to mine design, ore processing, as well as waste rock and tailings storage and management. All of these decisions have implications for the water quality that will be produced by mining and will influence the water management and treatment options that can be used during mine operations and after operations cease. Through application of acid–base accounting and kinetic testing, sufficient information can be obtained to make predictions of mine drainage quality, and when this is site hydrological data, downstream water quality can be predicted for the site.

Site hydrogeological data for predicting downstream water quality are similar to those used for determining baseline hydrogeology (section 3.3.2), but include a projected value of mine drainage volume. The volume of mine drainage relates to the type of mining (open-cast 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 in waste rock and tailings, 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.

Downstream water quality

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 concentrations of trace elements, and it may be more relevant to consider the flux of trace elements 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 in Cavanagh et al. 2015. These elevated background concentrations will influence the regional flux of trace elements, 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 that can dilute trace elements to low levels indistinguishable from the regional flux, and natural attenuation of trace elements by Fe oxyhydroxide. Fe oxyhydroxide forms naturally around gold mines through oxidation of iron-bearing minerals, and provided it is constantly renewed this is can be an effective mechanism to remove trace elements from mine drainage. This adsorption process is the principle behind the common use of ferric chloride solutions in active treatment systems at gold mine sites. Long-term attenuation of dissolved metals and stabilisation of metal-bearing mine wastes can occur by adsorption and subsequent crystallisation of relatively insoluble minerals.

These attenuation processes occur naturally in areas where geology that hosts mineral deposits are present and could lead to low dissolved concentrations of trace elements in water, but potentially to enriched concentrations of trace elements in sediment. Long-term enrichment of sediment with trace elements from historical epithermal mines is an issue at Tui mine, where management was poor during operations and remediation only took place decades after mining ceased (Fairgray et al. 2016, 2017).

Future and current mining practice should demonstrate that secondary enrichment of sediments downstream of mines does not occur, in addition to demonstrating that dissolved water quality is adequate.

3.5 Predicted ecological impact

Moving from a prediction of water quality to an assessment of likely ecological impact, the first step is normally a comparison with water quality guidelines designed to protect aquatic life. The most commonly used guidelines in New Zealand for assessing the impacts of mine water are the ANZECC (2000) guidelines. The USEPA (2005) methodology for the derivation of site-specific criteria may also be used to derive guidelines for a particular environment if the ANZECC guidelines do not appear to be relevant, due to the specific conditions or the type of aquatic life present.

The ANZECC & ARMCANZ (2000) guidelines are widely used to set discharge consent conditions in NZ, and provide a reasonable assessment of the potential effect of an individual contaminant or change (such as pH or turbidity) on the receiving environment. However, the guidelines do not take account of the cumulative effect of multiple contaminants and/or changes in water chemistry. There is also concern that some NZ native organisms are poorly protected by these guidelines, give the scarcity of robust toxicity data. Consequently, the results of ecological assessments downstream of mine sites are needed to validate (or otherwise) predicted effects and impacts. Consideration should also be given to the quality of sediment downstream of a mine as it may contain elevated concentrations of trace elements, which in turn could accumulate in aquatic biota.

3.5.1 Aquatic impacts

Some of the most common trace elements that can be elevated in drainages downstream of abandoned epithermal deposits are Zn, Cd and Mn, while other trace elements such as Pb may be elevated in the sediment. The impact of these trace elements on aquatic ecosystems is an area of ongoing 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 invertebrate diversity declined from 26 to 12 taxa (Jon Harding and Kevin Simon, pers. comm. and Case Study 6). Freshwater crayfish (kōura) were present in non-impacted upstream sites but completely absent from downstream sites with any dissolved metals present. Further, mayfly genera were completely excluded from the Tunakohoia stream, which received high Zn concentrations (2.0–4.0 mg/L) from the Tui mine (Harding & Simon, pers. comm.). This finding is consistent with patterns seen overseas, where mayflies are often not tolerant of dissolved metals but some caddisflies can survive.

In other studies, toxicity testing using rainbow trout showed an effect concentration at which 50% of test species were affected (EC50) at Zn concentrations as low as 1.45 mg/L. 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 has shown high tolerances to Zn. Īnanga, 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). Results from the Tui mine discharge into the Tunakohoia showed that eels were present in waters of 2 mg/L Zn, whereas no other fish species occurred in these reaches. This finding seems inconsistent with laboratory ecotoxicological results and may be due to the Tunakohoia having a cocktail of dissolved metals present.

Case Study 6: The Tui mine: the impact of zinc from a remediated metal mine on the ecology of receiving streams

In New Zealand a number of metal mines occur in the Coromandel Peninsula and Kaimāī Ranges in the North Island. One of these was originally opened as the Champion mine (near Te Aroha) in 1884 to provide smelting fluxes but was closed due to high zinc concentrations. In 1967 the mine was reopened as the Tui mine to extract zinc, copper and lead sulphides (Webster 1995). The mine was abandoned without remediation and comprised underground workings and four portals, a waste rock ore dump and mine tailings. Mine tailings were stored behind a dam, which contained approximately 90,000 cubic metres of acidic, sulphide-rich, tailings. The tailings had significant amounts of zinc and cadmium. The portals and mine tailings dam discharged into two streams: the Tui and Tunakohoia. In 2007 it was estimated that 5,000 kg of heavy metals, primarily zinc, iron, manganese, arsenic, cadmium, and lead, was released into Tunakohoia stream per annum (AECOM 2010). In response, a remediation project was completed in 2013. The water quality objectives of the remediation were not linked to any ecological objectives or improvements.

Studies of the water chemistry and stream invertebrate and fish communities were conducted on the Tui and Tunakohoia streams between 2014 and 2016 (Gregersen 2016). They showed high concentrations of zinc (e.g. 2.6 mg/L) in the water column on occasions, and elevated lead and copper in the sediment. As a result, stream invertebrate communities were negatively impacted (Figure C12).

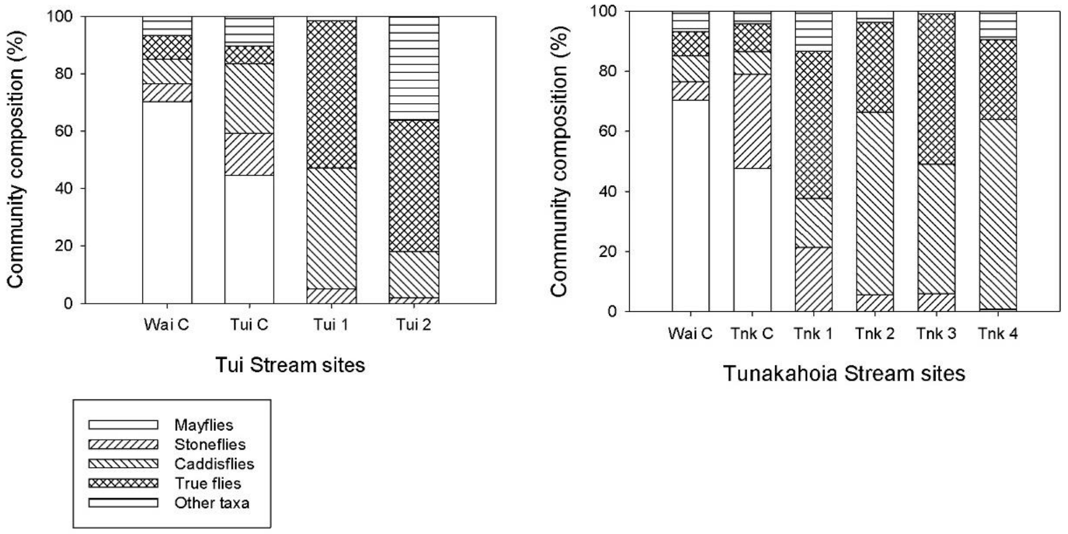

Figure C12 . Macroinvertebrate community composition at different sites in the Tui and Tunakohoia streams.

In particular, mayflies (which are indicators of a healthy stream) were completely absent below the mine discharges. In contrast, net-spinning caddisflies and some cased caddisfly species were able to tolerate elevated zinc concentrations. Control sites (WaiC, TuiC and TnkC) have nearly 50% of their communities dominated by mayflies, whereas the sites below the mine discharge are dominated by caddisflies and true flies.

A survey of fish communities in the Tui and Tunakohoia found that eels and banded kōkopu (pictured) were common in the lower reaches of the two streams. However, both species declined as zinc concentrations increased upstream.

Figure C13. Banded kōkopu were common in the lower reaches of the Tui and Tunakohoia streams.

Key findings relevant to mine management:

- Metal toxicity will impact the biodiversity of streams, but it will also impact different species in different ways. Understanding what species may be more or less affected can inform better decision-making on remediation.
- Remediation actions should have clearly defined water quality goals. If these aim to improve the ecological health of receiving waters, then water quality goals need to match ecological requirements.

Key references:

AECOM New Zealand Limited 2010. Tui Mine – phase 1 & phase 2 remedial works assessment of environmental effects. Report to Department of Conservation.

Gregersen RG 2016. The response of stream ecosystem function to acid mine drainage remediation. The University of Auckland.

Webster JG 1995. Chemical processes affecting trace metal transport in the Waihou River and estuary, New Zealand. New Zealand Journal of Marine & Freshwater Research 29: 539–553.

Further research is required to better identify metal concentrations that don't impact aquatic ecosystems in New Zealand. However, in the first instance it is recommended that the ANZECC & ARMCANZ (2000) guidelines be used. Ecological assessments, and toxicity testing can be undertaken during operations (or remedial activities) to better establish the concentrations below which there are no significant adverse effects.

Case Study 7: Emerging methods for assessing ecological health in mine waters

Traditional stream monitoring and assessment tools have focused on sampling stream invertebrates and using well-established metrics to assess impacts. These traditional tools include measuring biodiversity (i.e. counting the number of species), calculating the number (or percentage of the community) of pollution-sensitive species (e.g. the number of mayflies, stoneflies and caddisflies, or EPT), or calculating a pollution metric using the presence/absence or abundance of different species. Examples of this type of metric in New Zealand are the Macroinvertebrate Community Index (Stark 1985) or the Acid Mine Drainage Index (AMDI; Gray & Harding 2012).

However, new tools continue to be developed, and one emerging technique is the use of food webs. Food webs involve identifying all the organisms that live in a mine-impacted waterway and determining which organisms are prey and which are predators. From this a food web can be constructed that identifies the important components of energy transfer in the waterway. Food webs can be very complex and may consist of hundreds of species, including bacteria, fungi, algae, higher plants, mosses, benthic invertebrates, fish, and birds. Food webs can also include the biomass and abundance of each of these organisms. However, simpler (and less accurate) food webs can be constructed. This has been done for a number of mine drainage streams (Hogsden & Harding 2012, 2014).

The diagram below from Hogsden & Harding 2012 shows simplified food webs for four types of streams.

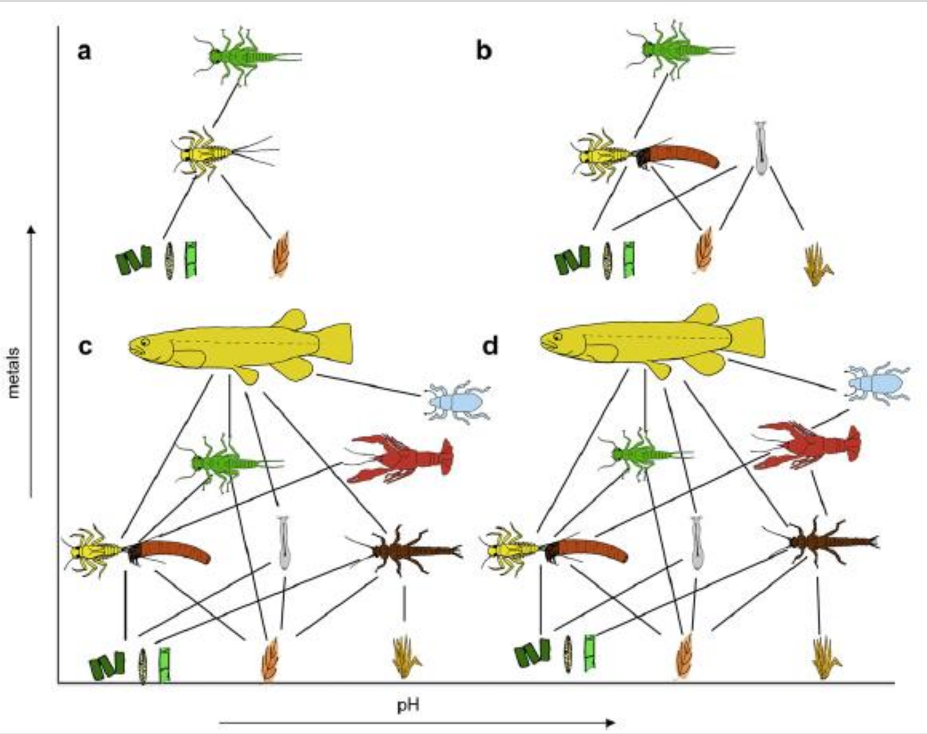

Figure C14. Simplified food webs for four types of streams (Hogsden & Harding 2012). a: A stream impacted by acid mine drainage has acid-tolerant plant species, such as some diatoms and filamentous algae. Leaves and wood often fall into these streams and are decomposed and act as food for invertebrates. However, bacteria and fungi are also affected by acidic and metal oxides, and so normal decomposition rates are often greatly reduced. As a result, food quality can be poor for those benthic invertebrates that can survive in AMD waters. Fish are usually absent. b: In natural streams with neutral pH but elevated metals, the food web is more complex. Algal food quality is better and more benthic invertebrate species are present. Often fish might still be absent or in low diversity. c: In naturally acidic streams, but with few metals, the food webs are even more complex. Normally a wide range of algal and plant species might occur, with diverse benthic invertebrate species and a number of fish species (although pH might still exclude some species). d: Neutral pH, low metal streams will have the most complex food webs, with a wide range of species.

The great advantage of identifying the food webs of mine-impacted waterways is that it can provide us with an understanding of what components of the stream we need to fix to regain a healthy ecosystem. For example, if the basic food supply in the food web is dominated by filamentous algae (which most benthic invertebrates cannot eat), then we can target remediation to reduce these algae and encourage other species.

The challenge with conducting food web studies has been that collecting these data has traditionally been time consuming and expensive, involving many hours of laboratory analysis. Several techniques have sped up this process. One is the use of stable isotopes to identify which organisms eat each other and where and what types of food an organism eats (Hogsden & Harding 2014). The figure below (from Hogsden & Harding 2014) shows the isotopic comparison of deltaN15 and deltaC13 ratios for a neutral pH stream (pH 7.1) and an AMD stream (pH 2.8). The neutral stream has a large isotopic area, containing organisms with a range of deltaN15 and deltaC13 components; in comparison, the AMD stream has a very small isotopic range.

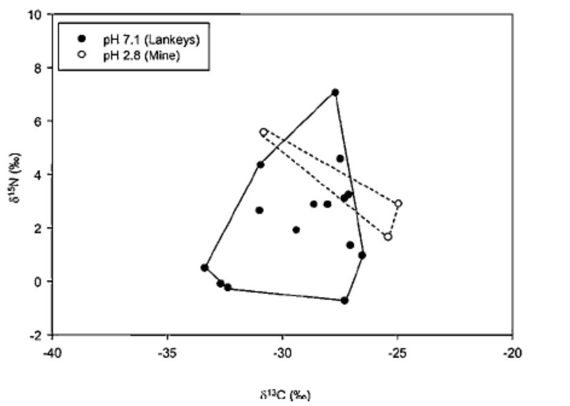

Figure C15. Comparison of deltaN15 and deltaC13 ratios for a neutral pH stream (pH 7.1) and an AMD stream (pH 2.8) (source: Hogsden & Harding 2014).

More recently a number of computer software programs have been developed that enable food webs to be inferred by modelling from existing data sets (e.g. WebBuilder; Gray et al. 2015). This involves much less laboratory time, but it will be less accurate and robust, although these tools should improve over time.

Another emerging approach to assessing mine impacts also uses food webs but focuses on the body size of organisms in the waterway. Body size is an important trait for animals in freshwater ecosystems. In New Zealand freshwaters, the larger the animal, the more dominant the role it plays in the ecosystem. In New Zealand all large freshwater animals are predators, and the largest are the top predators (usually large eels or trout).

A comparison of body size across a gradient from non-mining to mine-impacted streams focusing only on benthic invertebrates shows that invertebrates in mine-impacted streams have constricted body sizes (Pomeranz et al. 2018). The figure below shows the body size range for benthic invertebrates in 25 streams. The first 12 streams are non-mine impacted, while the remaining streams have been impacted by mining activities. The x-axis shows body length and the y-axis abundance. Benthic invertebrates in mining streams have fewer larger-body-sized and fewer smaller-body-sized animals than in non-mining streams.

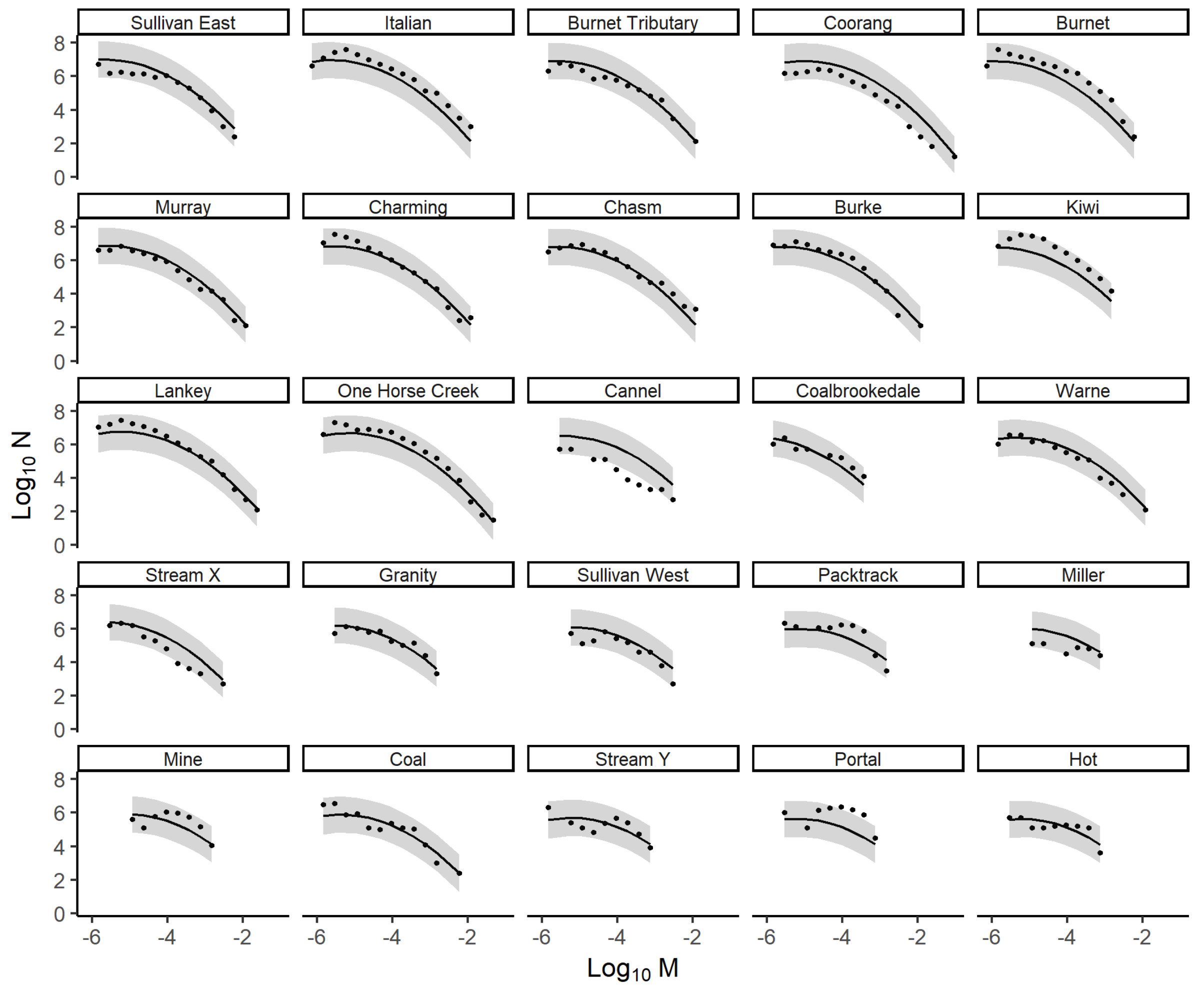

Figure C16. Comparison of body size of invertebrates across a gradient from non-mining to mine-impacted streams focusing only on benthic invertebrates, showing that invertebrates in mine-impacted streams have constricted body sizes (source: Pomeranz et al. 2018).

As a result, the range in body sizes of benthic invertebrates in mining streams is truncated. This research is still developing, and the processes causing this truncation are unclear. However, the consequences of this pattern may provide further insights into remediating mine drainage waterways in the future.

Key references:

Gray C, Figueroa D, Hudson L, Ma A, Perkins D, Woodward G 2015. Joining the dots: an automated method for constructing food webs from compendia of published interactions. Food Webs 5: 11–20. DOI: http://dx.doi.org/10.1016/j.fooweb.2015.09.001.

Gray PD, Harding JS 2012. Acid Mine Drainage Index (AMDI): a benthic invertebrate biotic index for assessing coal mining impacts in New Zealand streams. New Zealand Journal of Marine & Freshwater 46: 335–352. DOI: 10.1080/00288330.2012.663764.

Hogsden KL, Harding JS 2012. Anthropogenic and natural sources of acidity and metals and their influence on the structure of stream food webs? Environmental Pollution 162: 466–474.

Hogsden KL, Harding JS 2014. Isotopic metrics as a tool for investigating the effects of mine pollution on stream food webs. Ecological Indicators 36: 339–347.

Pomeranz JPF, Warburton HJ, Harding JS 2018. Anthropogenic mining alters macroinvertebrate size spectrum in streams. Freshwater Biology.

Stark JD 1985. A Macroinvertebrate Community Index of water quality for stony streams. Water and Soil miscellaneous publication 87. Wellington, National Water and Soil Conservation Authority.

Suspended solids

High levels of suspended sediment can be a significant issue associated with mine discharges. Impacts on aquatic ecosystems arising from high turbidity are largely physical, such as smothering of benthic organisms and reduction in light penetration. Elevated suspended sediment concentrations may have direct or indirect effects on benthic invertebrates and fish. Direct effects might be caused by the scouring and abrasive action of suspended particles that damage gill tissues or reduce respiration by clogging gills, leading to susceptibility to infection or disease, reduced growth, or mortality. Both invertebrate and fish eggs and younger fish, including sac fry, smolts, and juveniles, may be more sensitive than adults, for which direct lethal effects may not occur until extremely high concentrations that are uncommon in natural environments. Direct effects also include smothering of food resources such as algae and organic matter, benthic organisms (Figure 15) or eggs of some species, clogging of refugia, and physical 'armouring' of the bed. Excessive deposited sediment reduces habitat for stream life. In contrast, indirect effects include reduction in algal growth (which is food for invertebrates) 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. There have been a few studies on the effects of suspended sediment on the ecology of streams, but the results are highly variable. Nonetheless, Quinn et al. (1992) recommended that benthic invertebrate diversity could be protected if NTU < 20 above reference levels could be maintained. Attempts at large-scale field surveys have been generally inconclusive, and so attempts have been made to test the impacts through experimental trials (Boubée et al. 1997; Rowe & Dean 1998; Cavanagh et al. 2014). A number of these trials focus on turbidity (measured in NTU), which will not always correlate with suspended solids.

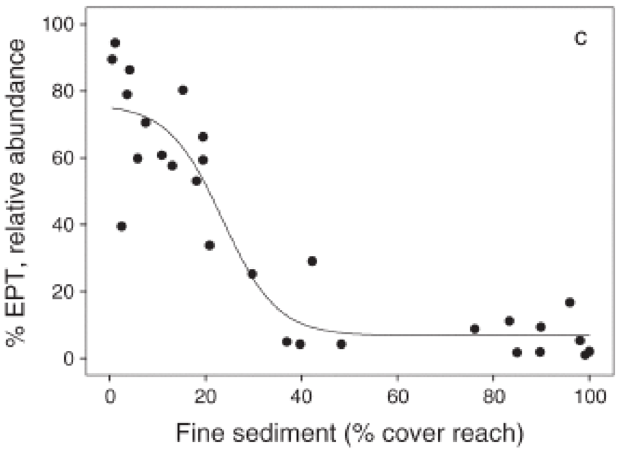

Figure 15. Fine deposited sediment exceeding 20% cover on the bed of streams has been shown to negatively affect stream invertebrates (i.e. mayfly, stonefly and caddisfly taxa) (source: Burdon et al. 2013).

Indirectly, fish may be affected by suspended sediments through decreases in water clarity, which can impact feeding success, and habitat quantity and quality, leading to decreased growth rates and changes in community structure and population size. Laboratory trials at the University of Canterbury using four fish taxa (īnanga, kōaro, brown trout, and eels) suggest that growth (length) of īnanga over 21 days may be affected at turbidities between 5 and 15 NTU, and between 15 and 50 NTU for kōaro. There was no apparent effect on weight of these fish species with turbidities up to 200 NTU. Similarly, no significant effect on growth (body length) of brown trout was observed up to 200 NTU, although a decrease in the mean fish weight was observed for trout above 5 NTU.

Finally, eroded 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, and 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.

3.6 Prevention and management of AMD

Prior to operations, several steps for managing AMD should be considered, including prevention, minimisation, control and treatment. Prevention of AMD is achieved by preventing the interaction of the principal constituents of the AMD production processes (sulphide minerals) with oxygen (and to a lesser extent water). Essentially, this can be considered prevention of the oxidation processes.

Where prevention is not possible, the objective is to minimise or decrease the contaminant load reporting to the receiving environment. This often involves minimising the interaction of oxidation products with net percolation of water through waste rock or tailings. Any remaining AMD needs to be controlled to prevent the release of untreated AMD-influenced water to the receiving environment. An important component of understanding whether control strategies are effective is monitoring water quality and quantities, enabling the calculation of contaminant loads.

In the event of AMD migration from mine waste storage facilities, there is a requirement to treat AMD-influenced water prior to discharge to the receiving environment. Depending on a number of variables, passive or active treatment systems may be appropriate. In addition, total suspended solids could also be an important part of mine drainage management, and additional strategies may be required to manage sediment.

This section provides an overview of different techniques that can be employed to minimise the formation of AMD, and information that should be collected to determine potential mine drainage management techniques. Generally, there are more options and opportunities to identify cost-effective options earlier rather than later in the life of the mine (Figure 16).

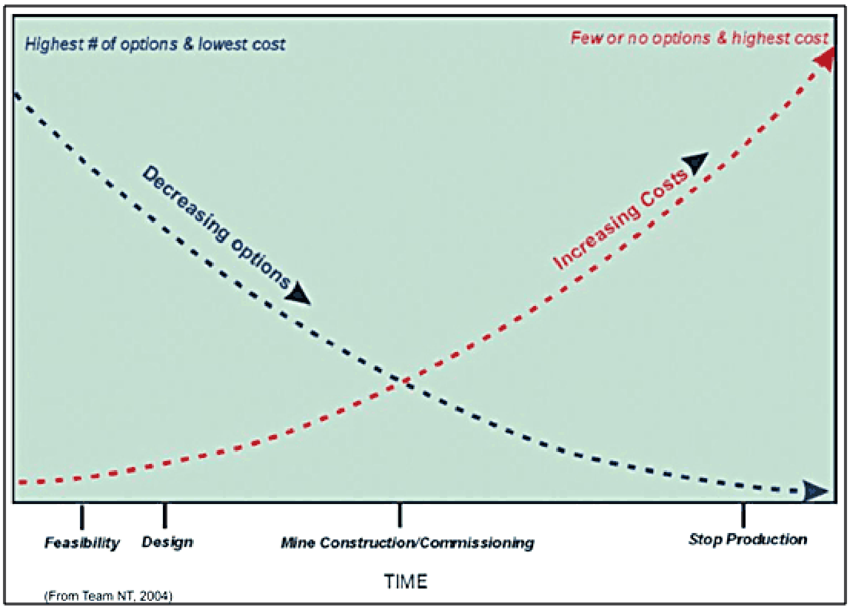

Figure 16. **Change in number of options available to effectively mitigate AMD and cost in relation to when option is implemented (source: TEAM NT 2004).**

3.6.1 Prevention of AMD through minimising oxygen ingress

There are a number of opportunities to minimise oxygen ingress into a waste rock dump (WRC) or tailings storage facility, which should be considered during design. Options are site specific and could include the following.

- Construction of the basal foundation of the WRD or tailings storage facility is an important aspect of the design and needs to ensure PAF waste rock is out of possible drainage paths and the basal layer is not an ingress point for oxygen advection. WRD and tailings storage facility design should consider this on a site-specific basis.
- For WRD, grain-size segregation should be minimised by paddock dumping or the use of short lifts 2–4 m in height. However, each site is material specific, and higher tip heads can be used if the waste rock has the potential to produce finer-textured material.
- At tailings storage facilities, the length of tailings beaches will influence the time that tailings are exposed to air and therefore the time available for oxidation. Short beach lengths minimise the potential for AMD. There might be other mine site considerations that influence the length of tailings beaches, such as available freeboard to manage heavy rainfall, process water requirements or oxygenation of process water.
- In a WRD designed to limit the advective ingress of oxygen, there is benefit in placing high PAF materials in the centre of the WRD, furthest from atmospheric oxygen. This means the pathway for oxygen diffusion is greatest, which significantly slows down oxidation rates.
- In a tailings storage facility, construction of the embankment with impermeable materials to prevent oxygen and water ingress or escape from the tailings prevents oxidation and discharge.
- Advective barriers can also be constructed around the perimeter of the WRD. Such barriers limit the advective ingress of oxygen, both laterally and horizontally, to the inner core of the WRD that might be constructed by high end-tip practices.
- Cover systems are also part of the toolbox to minimise the ongoing oxidation of sulphides. These can include:

- erosion-protection systems
- store-and-release systems
- enhanced store-and-release systems
- barrier-type systems
- alkaline covers
- cover systems with engineered layers
- saturated soil or rock cover systems.

Some further details on covers are provided below. Additional information on covers is available in various documents, including MEND 2012, and INAP 2009.

- Encapsulation and layering involve the placement 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, their effectiveness is governed by the availability, type, and reactivity of acid-consuming materials, the balance between acid-forming and acid-neutralising materials, the deposit geometry, the nature and flow of water through the deposit, potential chemical armouring of alkaline materials, and potential passivation of pyritic materials.
- Waste-rock types of varying acid-forming and acid-neutralising potential can be blended or mixed to create a deposit that generates a discharge of acceptable quality. The effectiveness of this depends on the availability and grain size of materials, the mine plan, the geochemical properties, the reactivity of the waste-rock types, the 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 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).
- Water covers can be used to prevent oxygen ingress. Water covers can be developed in tailings storage facilities, in old pits, or, when possible, constructed to flood PAF waste rock. Sometimes the WRD can be placed in a deep pit or in underground workings below the groundwater level, and upon recharge of the groundwater the waste rock becomes inundated. The flooding of underground or surface mine voids may result in the release of acidic discharge arising from the dissolution of stored oxidation products, as in a final void pit lake. Inundation has the potential to prevent the supply of oxygen to PAF materials . 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 partially oxidised, because the secondary minerals can store acid and the water cover could become acidic.
- Hydrogeological controls are 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 the movement of air and water through mine workings.
- Addition of alkaline amendments to engineered landforms can prevent the formation of AMD through neutralisation and maintenance of high pH conditions, which prevents catalytic oxidation of pyrite by FeIII. There are several approaches to this (Table 4), and selecting the most effective method at each mine site will require field trials.

Table 4. The benefits and limitations of alkaline amendments (source: INAP 2009)

Configuration	Benefits	Limitations
Liquid amendment	• Excellent initial control of solution pH • Versatile – allows localised treatment • Proven to work	• Time – alkaline materials are consumed by even pH-neutral water • Cost and availability of reagents • Particle size and release of alkalinity • Effort of mixing or blending
Layering	• Easy to implement and manage	• Difficult to obtain mixing of alkaline and acid leachate due to preferential flow
Encapsulation and alkaline cover	• Easy to implement and manage • Versatile – allows localised treatment	• Cost and availability of material • Time and release rate of alkalinity • Alkaline materials are consumed by pH-neutral water
Blending	• Excellent pH control • Proven to work	• Cost • Effort of mixing and blending • Availability of materials
Injection and mine filling	• Proven to work	• Cost and availability of material

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

3.6.2 Minimisation of AMD by controlling the interaction of water with PAF rocks

Where prevention is not possible, the objective is to minimise or reduce the contaminant load reaching the receiving environment. This often involves minimising the interaction of oxidation products with net percolation of water through waste rock or tailings. It can also involve offsetting acid generation by mixing or layering on-site PAF and NAF rock in WRDs, or by alkaline addition to WRDs or tailings. In AMD formation, water acts as a solute transport mechanism and a reactant.

Water management is often the most cost-effective method of minimising AMD. Water management techniques can be implemented during the construction phase as well as at later stages of mining operations. Some techniques can be applied to prevent oxidation and also to minimise infiltration, so there is overlap between techniques used for prevention and minimisation.

Following are some of the techniques that have been developed for minimising AMD (MEND 2001; DITR 2007; INAP 2009):

- 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 waste rock, tailings, and PAF rocks in high walls, as rapidly as possible via ditches or pumping.
- PAF waste rock is elevated off the pit floor, where net percolation often flows laterally to discharge as seeps from the WRD.
- Basal drainage paths are constructed to route AMD-impacted water from the waste rock or tailings to one or a few discharge points. The drainage paths should be constructed from NAF material. The discrete discharge points enable capture of the AMD-impacted water for management (and treatment), as required. These are sometimes referred to as foundation drains.
- Net percolation of rainfall into the WRD is controlled by:
    - compaction of waste rock to encourage runoff
    - limiting the amount of WRD surface area to limit high net percolation rates
    - constructing cover systems to reduce net percolation
    - progressive rehabilitation and minimising the time to final landform.
- Ensure the slope angle and length of the rock pile are such that they minimise permeability and erosion and are appropriate for the climatic conditions.
- 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 WRDs.

Waste-rock and tailings management is an important part of minimising AMD, primarily through minimising infiltration of water. It is also an integral component of rehabilitation, as waste-rock piles and tailings dams can ultimately become parts of the final landform after mining. Appropriate management is required to ensure the root zone is suitable for plants. Management may include rock pile design and the use of dry or wet covers.

Dry covers are typically earthen, organic, NAF rock wastes or synthetic materials placed over PAF rock wastes to reduce oxygen diffusion and minimise water infiltration. NAF and root zone must be deep enough to protect the capping layer from roots breaching the cover. Some NAF materials are suitable by themselves, or amended, as rooting media. Favourable root-zone materials are NAF and have physical properties that store and supply adequate water and oxygen for plant roots. Such materials also provide sufficient anchorage for selected plants.

Common dry covers include the following.

- *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 are often limited by the 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:* these can increase the 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 ($<10^{-8}$ m/s) 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 so 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 the capillarity of the NAF needs to be taken into account. Toe-slopes of WRDs are particularly vulnerable to leachate entering the root zone.

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

- water store covers – low-permeability materials that maintain saturation, encourage runoff of surface water and reduce oxygen infiltration
- 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, while NAF tailings are used as cover above the level of the pond (Figure 17)

Of 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.

Figure 17. Water cover over tailings storage facility at Golden Cross mine site. NAF rock was used for surface rehabilitation, and pasture was planted for a farming operation.

3.6.3 Tailings management options

Tailings may form sources of AMD, and so appropriate management of these materials is required. Mine tailings are residues from any processing of ore on site. 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:

- discharging as a slurry into a tailings dam or underground workings
- dewatering:
    - disposing of in tailings piles or mixing with waste rock – this material can be directly planted in, or will naturally revegetate if properties are favourable
    - using to line landfills
    - using to make low-grade fuel such as briquettes.

Details on both of these methods is provided below.

Discharge

Fine tailings are conventionally discharged from a processing plant as a water-rich slurry, and they are accumulated behind a dam so that the solids settle. 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 in Cavanagh et al. 2015). The design of tailings dams will be specific to each site, as the design must take into account pathways for water, 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 in Cavanagh et al. 2015). 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 the 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. One benefit is 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 (National Research Council 2002):

- plugging of pumping systems

- incomplete knowledge of available storage space
- increasing water flow from underground workings – which then must be managed
- increase of hydraulic head on bulkheads and other barriers in the workings – which can result in blowouts.

Dewatering

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 in terms of both capital and operating costs, and therefore is likely to be completed only when site conditions, such as limited space/volume for a 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. Tailings may contain sulphides, which, when exposed to air and/or water, may result in AMD, and thus will 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, sulphide content, volume of air, and permeability (Kolling & Schuring 1994).

If they are adequately dewatered, mixing tailings with waste rock and placing them in waste-rock piles can be a suitable method of disposal. However, if sulphide 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. Removal of sulphides from 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 the end of the mine's life.

3.6.4 Data collection

More than one method sequentially, or a combination of methods at any one time, may be required to achieve prevention or minimisation of AMD, and different methods may be applicable during different stages of the mine's life. To assist in determining which of the above techniques may be needed over the life of the mine, the following data should be collected:

- geochemistry of waste materials, so that selective handling and disposal of PAF material can be planned (e.g. construction of cells for PAF rock or placement of PAF material in the centre of WRDs)
- land area available to construct WRDs and tailings storage facilities using paddock dumping
- the availability of fine-grained, low-permeability material on-site or nearby that can be used to construct adequate covers for WRDs
- local availability of other amendments that can prevent or minimise the oxidation of PAF material (e.g. biosolids, compost, zeolites)
- local availability and grain size of limestone or other neutralising amendment (e.g. mussel shells, cement kiln dust, steel slag) that can be used as an alkaline amendment to WRD or tailings storage facility covers, or mixed/layered with PAF material within the WRD, or used as stemming material during blasting
- the depth to groundwater at the site, and the potential for waste rock or tailings to be inundated to prevent oxidation.

In addition to the above reference data collected during the planning and pre-operations stage, experiments can be conducted to optimise waste rock management strategies. These involve testing potential prevention and minimisation strategies in kinetic leachate columns and lysimeter field trials (see section 3.4.5). For example, limestone can be added to PAF material in a leachate column to test the effectiveness of neutralisation of AMD formed from oxidation of the sulphides. Columns with various mixing ratios and various grain sizes can be used to determine optimal parameters. Other columns can be constructed in a fully saturated state to test inundation strategies. Data to be collected from column experiments include: water chemistry, dissolved oxygen content, intrinsic oxidation rate, and analysis of secondary minerals formed in the columns.

Ultimately, the mining plan and scheduling should be planned to accommodate the WRD and tailings storage facility design selected for the site, which will prevent or minimise the formation of mine-impacted drainage.

Once management techniques have been initiated or treatment has 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 4 ('Operations').

3.6.5 Treatment of AMD

Adequate treatment of mine-impacted waters can be critical for achieving regulatory compliance at downstream monitoring sites. Even if overburden management strategies are used to prevent or minimise the formation of mine-impacted drainage, there are many uncertainties in the application of these strategies, and mine-impacted drainage may still occur. Treatment of AMD may therefore be required and contingencies for treatment should be included in the mine plan before mining begins.

Treatment can only be undertaken on point-source discharges, which requires effective water collection systems to be 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.

The overall aim of water treatment is to raise the pH and lower the concentrations of dissolved metals. Treatment can be accomplished using either active or passive treatment systems (Ziemkiewicz et al. 2003). Typically, active treatment is utilised during mining operations and passive treatment is used during mine closure. However, sometimes active treatment continues well after mine closure, and conversely, under the right conditions, passive treatment can be used both during mining operations and after closure.

Many factors must be considered when choosing the most appropriate technology for any given mine. These factors include the predicted chemistry and flow rates from the site, assessment of potential treatment media that might be readily available near the site, the potential land area that will be available for treatment systems, and the results of active treatment and/or passive treatment trials.

Further, nearly all mine sites produce high suspended solids, which need to be managed. Management and treatment of high TSS should therefore also be considered during the pre-operations stage. Ideally, sufficient land area can be reserved for passive treatment systems utilising settling ponds, but if only a small area of land is available, it should be assumed that an active treatment system for TSS control may be required, and the mine plan should allow for this.

To identify treatment options that may be best for the planned mine, both for during operations and at closure, the anticipated flow rates and water chemistry from mining operations (water from open pits, from underground mines, and from WRDs and tailings storage facilities) and the anticipated flow rates and water chemistry from the site at closure should first be considered. As explained in section 3.6.1, AMD is generated through the interaction of oxygen and water with sulphide minerals. Oxygen ingress rates and net percolation rates into WRDs or tailings storage facilities can be estimated to predict potential contaminant loads reporting to the environment. Acid load produced by a WRD or tailings storage facility is a function of acidity generated by: (1) sulphide oxidation, which is limited by oxygen ingress and requires net percolation, and (2) acidic salt solubilisation, which is limited by net percolation (Weber et al. 2014). Oxygen ingress rates (or oxygen flux) can be estimated using Fick's First Law, and net percolation can be estimated using the GoldSim™ model. Together, these data can provide estimated water flow rates, water pH, dissolved Fe concentrations, and sulphate concentrations. The potential concentrations of trace elements can be estimated using the results of kinetic leachate tests (section 3.4.5). These data can be used to suggest if passive or active treatment is recommended for the site (see below).

The availability of treatment media or neutralising chemicals near the site will have a direct impact on the economics of active and passive treatment. Active systems require the regular input of neutralising chemicals, whereas passive systems are constructed with a lifetime supply of treatment media, which can include waste mussel shells, waste steel slag, biosolids, limestone, compost, or general organic waste, among others. The proximity of these materials to the site should be considered when deciding between active and passive treatment, or the types of systems.

The third factor to consider is the availability of land at the mine site. Passive systems require much more land than active systems, and so are often not used during the operational phase since much of the land area is used for the mining operation. Most mine sites aim to use passive treatment upon closure, therefore ample land area should be reserved early in the mine planning stages for the eventual construction of passive treatment systems upon closure. If an active treatment system is selected during the operational phase, or even at closure, one must bear in mind that these systems generate sludge on a continuous basis due to the neutralisation of the acidic water and precipitation of metal hydroxides. This sludge can be disposed of on-site if allowances for this are included in the mine plan.

Treatment options should be included in the mine plans. This includes identification of collection areas from WRDs and tailings storage facilities or other sources of AMD reserving land area for the treatment system, and identifying discharge points to the environment. It should be assumed that passive treatment will eventually be used at closure, if not sooner, and ample space allowed for these systems.

Choosing between active and passive treatment

Several factors will influence the decision whether to use active or passive treatment. Briefly, if mine drainage exceeds the thresholds provided in Figure 18, large amounts of neutralising materials are required to ensure appropriate treatment and large passive systems can be prone to failure, and so active treatment may be a better choice. (Details of how each parameter influences the choice of active or passive treatment is provided in Appendix F in Cavanagh et al. 2015.)

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, and several active mine sites are using passive treatment for some or all of their AMD. Often passive treatment can be used to complement active treatment as a final polishing step to improve water quality to meet the 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
- can be accommodated at small sites.

The main advantage of passive treatment systems is that they are more economical (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 (commonly 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).

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 rainfall events (due to dilution), and in other cases it increases (due to dissolution of acid-sulphate salts or to more rapid sulphide oxidation).

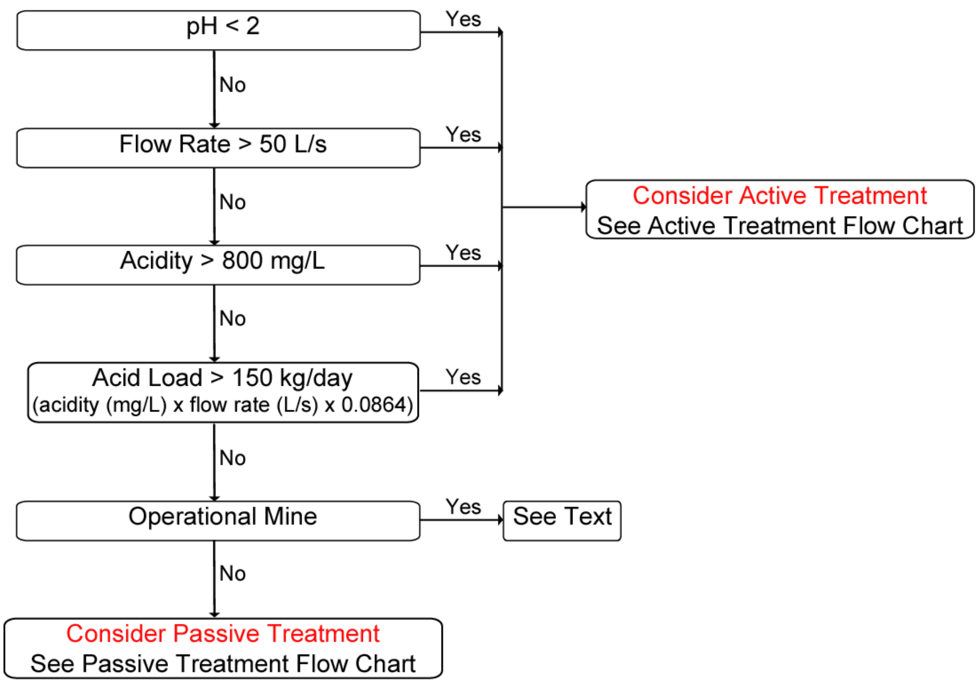

Figure 18. Flow chart to guide selection between active and passive treatment for AMD (source: modified from Waters et al. 2003).

Work in New Zealand is underway to evaluate the overall hypothetical cost of passive treatment and active treatment over long-term treatment of AMD at New Zealand sites (Eppink et al. 2017). This work is considering factors such as capital expenditures, operating expenditures, replacement cycles for active treatment plants and passive treatment media, and discount rates over decades of costs.

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 19), while efficacy of treatment and particularly costs (primarily of the chemicals required) will influence the final selection (Appendix F.2 in Cavanagh et al. 2015). Once an active treatment system has been selected, a computer program such as AMDTreat (Means et al. 2003; Cravotta et al. 2015; Appendix F.4, Cavanagh et al. 2015) 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 pre-treatment, 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).

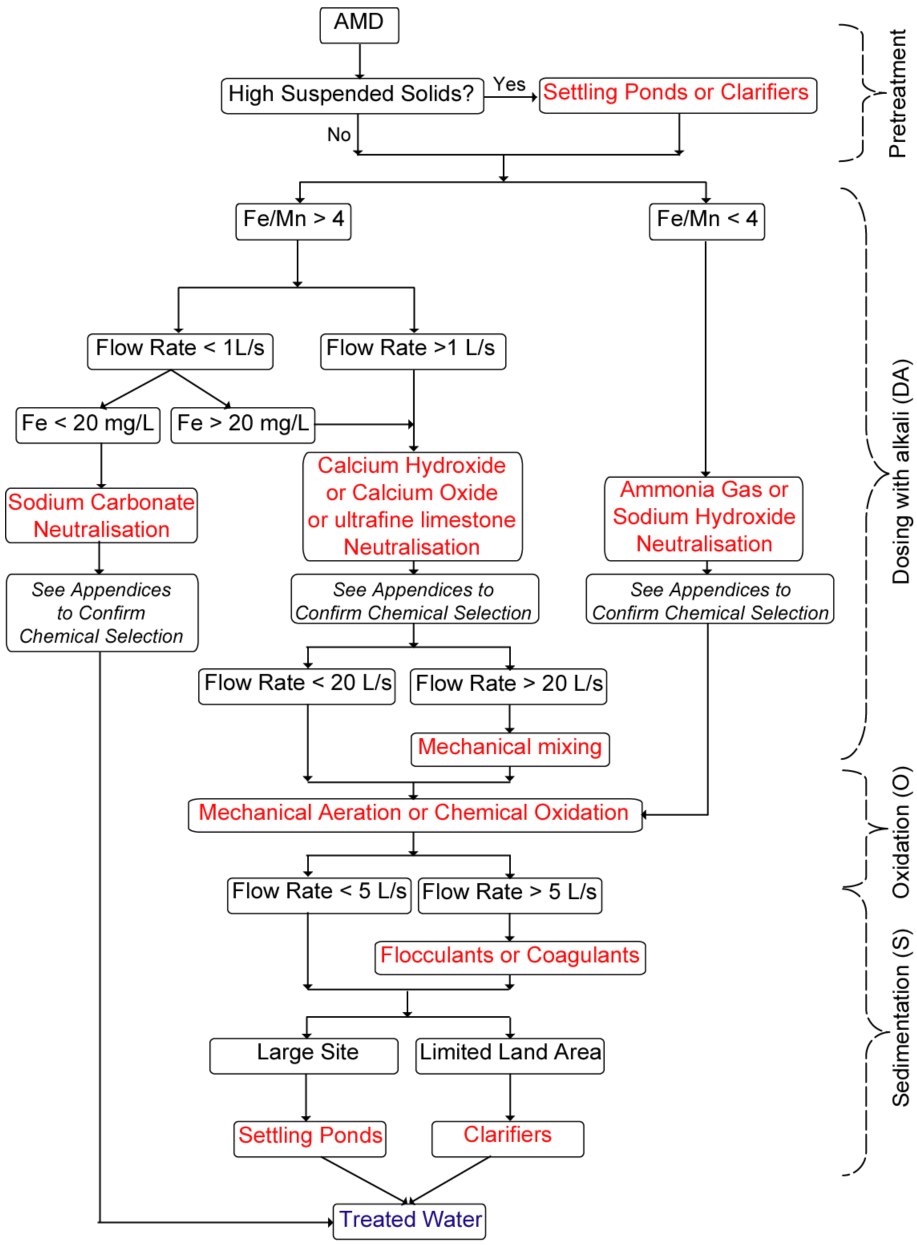

Figure 19. Flow chart to design a site-specific active treatment system for AMD (modified from Rajaram et al. 2001). The generic treatment step is shown on the side of the diagram. Note: for treatment of suspended solids see section 3.6.6. For appendices refer to Appendix F in Cavanagh et al. 2015.

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 section 3.6.6.

Dosing with alkali

Dosing with alkali raises the pH of the AMD, which helps to reduce dissolved metal concentrations through the formation of metal oxides. Various chemicals can 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 20 presents a summary of the key benefits, with more details provided in Appendix F.1 in Cavanagh et al. 2015. An illustration of the costs of active treatment using various chemicals is provided at the end of this section.

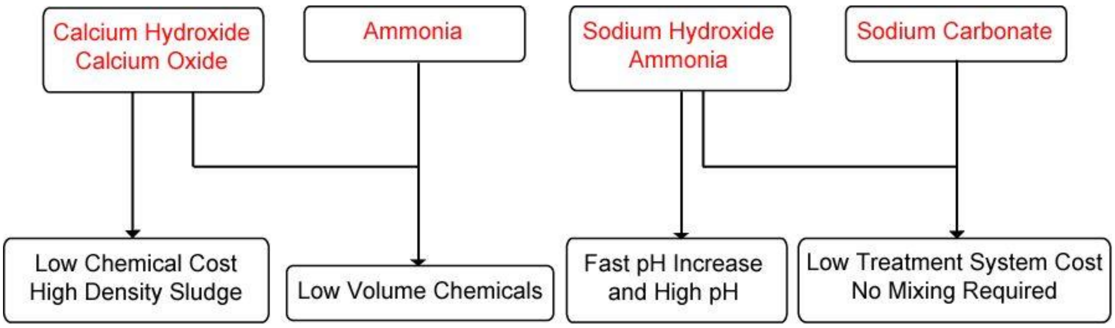

Figure 20. **Summary of the key benefits of the five most commonly used chemicals for the dosing-with-alkali step.**

Oxidation

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 in Cavanagh et al. 2015 includes 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

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. Examples of different sedimentation systems are shown in Appendix F.2 in Cavanagh et al. 2015.

Disposal of the sludge produced during this step is an important consideration, particularly with regard to the potential leaching of trace elements from the sludge. Standard tests are available to determine potential leaching of trace elements (see Appendix F.2 in Cavanagh et al. 2015). 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.2) may be required. If disposal to landfill is planned, 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 of an active treatment system, and therefore 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 turn depends on the efficacy of the chemical for neutralisation. Figure 21 illustrates the relative costs of active treatment using different chemicals over 20 years of treatment. Treatment using calcium oxide is the least expensive, and so all other treatment systems are compared to a calcium oxide system. These comparisons were determined using AMDTreat (Means et al. 2003) and are based on the relative costs of chemicals in New Zealand and default parameters for labour and construction costs provided in AMDTreat, although in the example provided the latter are typically less than 6% of the total costs over 20 years. A study is underway in New Zealand to determine the cost breakdown of hypothetical active and passive treatment at coal mines in New Zealand (Eppink et al. 2017), which will also aid with determining costs for epithermal gold mines.

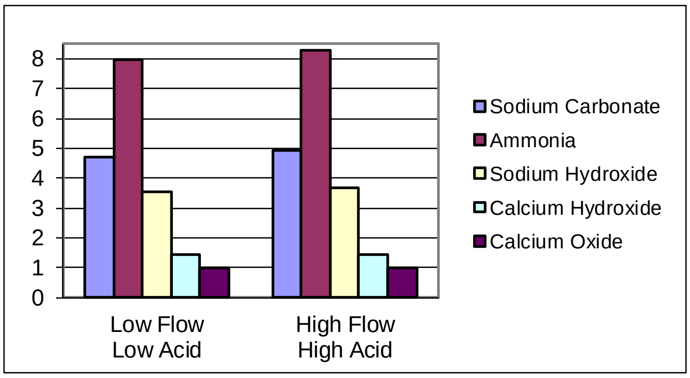

Figure 21. Comparison of potential costs for the active treatment of two hypothetical AMDs over 20 years using the five most commonly used chemicals in the USA. Treatment using calcium oxide is the least expensive, therefore, all other systems are scaled

to a calcium oxide system. For example, treatment using ammonia is approximately eight times more costly than treatment using calcium oxide. These comparisons were determined using AMDTreat (Means et al. 2003) and are based on the relative costs of chemicals in New Zealand 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, 1,000 mg/L) option over a 20-year treatment period.

Selection of passive treatment systems

In the long term, treatment of AMD using a passive treatment system is typically more economical than using an active treatment system. However, careful system choice and design are required to ensure 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, 2010). 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, it is 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 by iron- and sulphate-reducing bacteria, 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 content and the ferrous:ferric (Fe^{2+}:Fe^{3+}) iron ratio. The dissolved oxygen content of mine drainage will be unknown for a new mine, although it is likely to be high for an open-cast mine and either high or low for an underground mine. Similarly, the Fe^{2+}:Fe^{3+} ratio will be unknown for a new mine, although it is likely to be low for an open-cast 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 dissolved oxygen concentration and the ratio of Fe^{2+} to Fe^{3+}.

For highly oxidised AMD (saturated dissolved oxygen and Fe, mostly as Fe^{3+}), the oxidising strategy is most appropriate. Typical treatment systems that employ the oxidising strategy are open limestone channels (Ziemkiewicz et al. 1994), open limestone drains (Cravotta & Trahan 1999), limestone leaching beds (Black et al. 1999), slag leaching beds (Simmons et al. 2002), and diversion wells (Arnold 1991; PDEP 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 dissolved oxygen 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; 'The science of acid mine drainage and passive treatment' 2001; O'Sullivan 2005), anoxic limestone drains (Hedin & Watzlaf 1994), sulphate-reducing bioreactors (Mattes et al. 2007), and successive alkalinity producing systems (Kepler & McCleary 1994), also known as vertical flow wetlands, or reducing and alkalinity producing systems (Zipper & Jage 2001). It should be noted that although anoxic limestone drains only work with reduced AMD, they do not result in the biochemical reduction of Fe and sulphate and the formation of sulphides. Instead, they rely on the fact that pH can be raised to neutral without armouring of the limestone by Fe hydroxides, since Fe^{2+} will not form a hydroxide below pH 7.

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

Figure 22 and Figure 23 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 in Cavanagh et al. 2015, which also includes a comprehensive description of each of the common passive treatment systems).

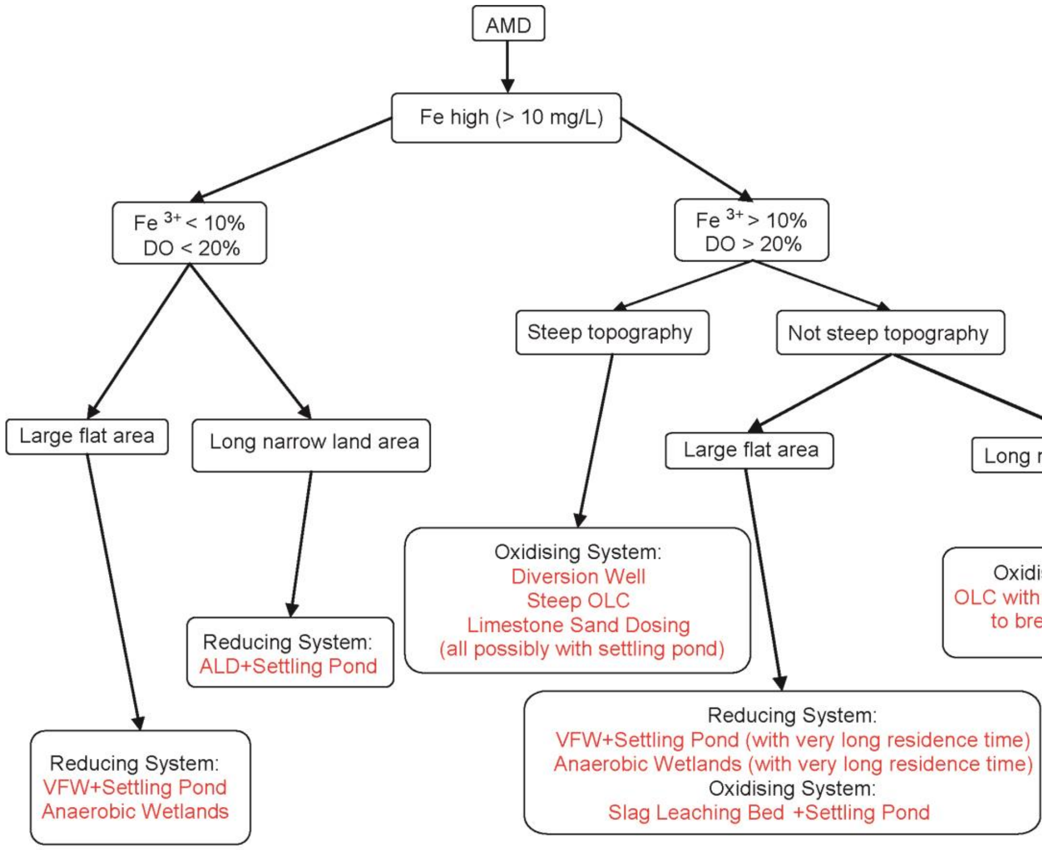

Figure 22. Flow chart to select among AMD passive treatment systems based on water chemistry (high Fe), topography, and available land area (source: Trumm 2010). Notes: DO = Dissolved oxygen; ALD = anoxic limestone drain; OLC = open limestone channel ; VFW = vertical flow wetland.

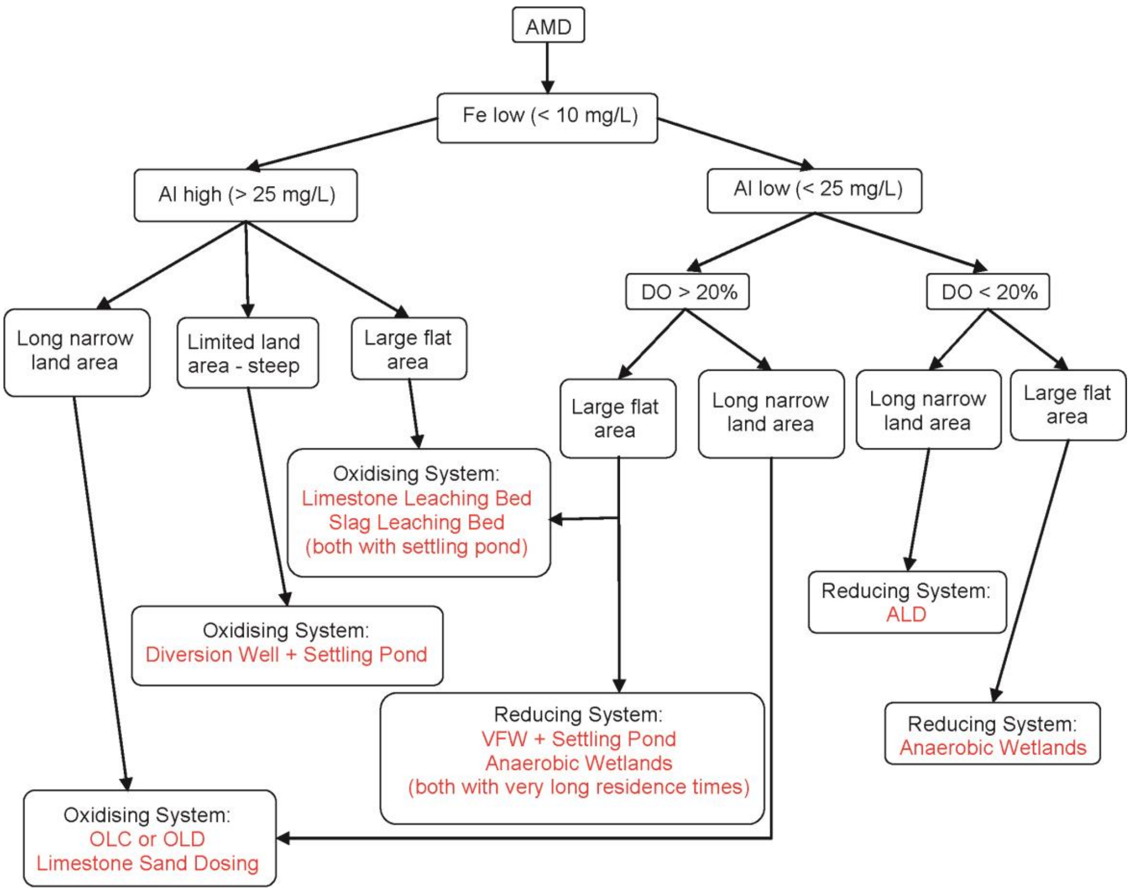

Figure 23. Flow chart to select among AMD passive treatment systems based on water chemistry (low Fe), topography, and available land area (source: Trumm 2010).

Fe concentration

Iron is the most difficult metal to remove from AMD using passive treatment technology. This is 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 because, under low dissolved oxygen 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. Al concentrations, dissolved oxygen, and land area are used to further decide between treatment strategies and passive treatment systems.

Al concentration

Al is a much less problematic metal than Fe in the treatment of AMD. It precipitates out of solution as an amorphous white material 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). Al concentration will influence treatment selection when Fe concentrations are low (<10 mg/L, Figure 23).

Dissolved oxygen concentration

The dissolved oxygen content of mine drainage will be unknown for a new mine, although it is likely to be high for an open-cast mine and either high or low for an underground mine. In general, a highly reduced AMD (dissolved oxygen < 20%) is best treated using a reducing strategy, whereas an oxidised AMD (dissolved oxygen > 20%) is best treated using an oxidising strategy. However, available land area may limit the 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 dissolved oxygen and for reducing conditions to become established. Oxidising strategies can be used for AMD with low dissolved oxygen concentrations, but these systems should be constructed with cascades to add dissolved oxygen to enable oxidation reactions to occur.

Available land area

Available land area descriptions in Figures 25 and 26 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 24 illustrates the relative costs of the different systems based on a hypothetical AMD that could be treated using any of the treatment systems. In reality, some treatment systems would be unable to treat certain mine drainages. The least expensive system is an aerobic wetland, therefore, all other systems are scaled to the cost of an aerobic wetland. 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 in Cavanagh et al. 2015).

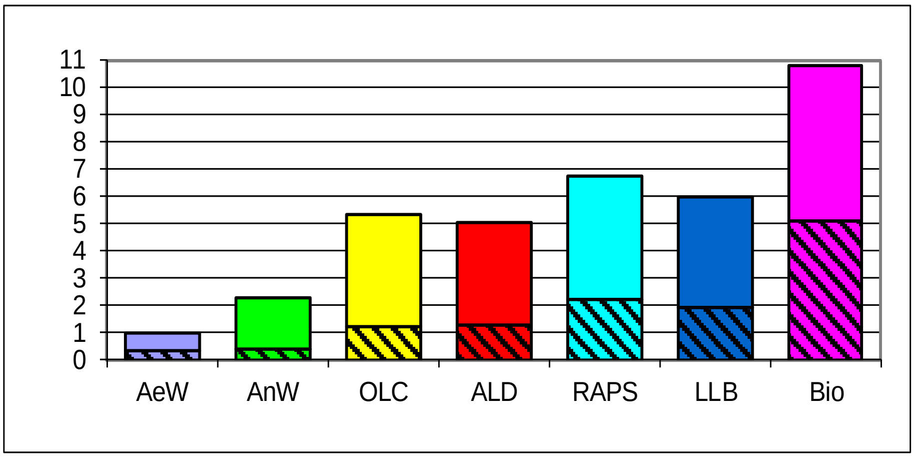

Figure 24: **Relative treatment system costs for a hypothetical AMD determined using AMDTreat (Means et al. 2003) using the cost of limestone in New Zealand and model default parameters for labour and construction costs. The least expensive system is an aerobic wetland, therefore, all other systems are scaled to the cost of an aerobic wetland. For example, treatment using an ALD is approximately five times more costly than treatment using an aerobic wetland. Aew = aerobic wetland; Anw = anaerobic wetland; OLC = open limestone channel; ALD = anoxic limestone drain; RAPS = reducing and alkalinity producing system; LLB = limestone leaching bed; Bio = bioreactor. The hatched area indicates the non-chemical costs.**

Case Study 8: Mussel shell reactors – new technologies for passive treatment

Passive treatment of mine drainage is an area of ongoing research. New technologies continue to evolve as researchers strive to find more cost-effective treatment techniques. To help reduce cost, passive treatment commonly uses waste products, such as steel slag from the steel-making industry, spent mushroom compost, Fe oxides from AMD sites, and biosolids. One such waste product in New Zealand is the green-lipped mussel shell from the seafood industry. This industry is the largest of our seafood exports, generating 12,000 tonnes of shell waste per month, much of which ends up at landfills. The shells are composed of $CaCO_3$, a prime ingredient for acid neutralisation, and the waste also contains organic material at approximately 10% by weight.

Early experiments by researchers in New Zealand found that waste mussel shells outperformed limestone in bioreactors. This was attributed largely to the higher surface area of mussel shells. Later experiments at the Stockton coal mine found that reactors constructed of just mussel shell waste, without the other organic ingredients of standard bioreactors, worked effectively at removing Fe, Al, Zn and Ni, and restoring pH to neutral. These reactors were constructed with downflow configurations. Analysis of the data found that the pH increased vertically downward through the reactors (Figure C17). Sequential extraction and bio-geochemical studies found that Fe was removed as a hydroxide at the top of the reactor, Al precipitated as a hydroxide below the Fe layer as the pH reached Al solubility limits, and Zn and Ni precipitated as sulphides in the deepest regions of the reactor.

These systems act as hybrid oxidising and reducing systems. Oxidising reactions occur at the top, removing Fe and Al, while reducing reactions occur at depth, with sulphate-reducing bacteria using the waste organic matter to reduce sulphate to sulphide, resulting in the precipitation of trace element sulphides. These reactors are very effective at treatment of AMD, but if the Fe concentrations are high they can be prone to failure as the permeability is reduced through accumulation of hydroxides. However, periodic removal of the precipitates can restore permeability.

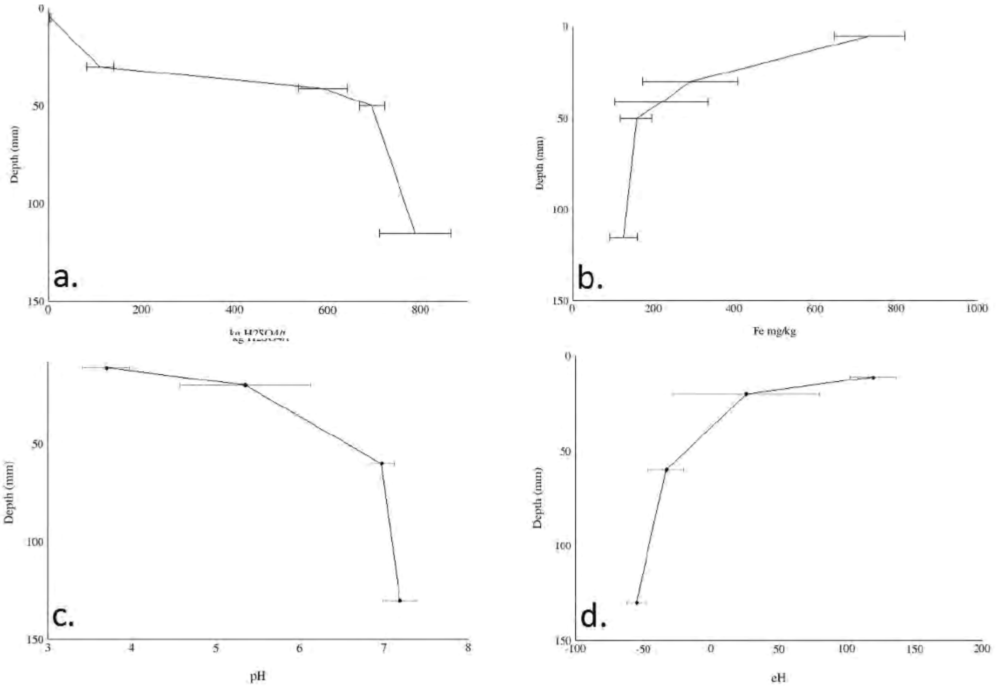

Figure C17. Data for a downflow mussel shell reactor as a function of depth. a: ANC; b: Fe(III) concentrations; c: pH; d: eH.

To address sites with high Fe concentrations and sites with high trace element concentrations, small-scale mussel shell reactors were constructed with *upflow* configurations. In these reactors, incoming AMD encounters a highly reduced environment at the base of the reactor, resulting in the precipitation of Fe sulphides rather than Fe hydroxides. The sulphides have a more compact structure than the hydroxides, which means they take up less space in the reactor and are distributed throughout the reactor rather than in a concentrated layer as the Fe hydroxides are in downflow reactors. This results in much less permeability loss with time. In addition, trace elements are removed as sulphides in the highly reducing environment of the upflow reactor. As with any bioreactor, however, reactive sulphides must be managed appropriately at the end of the lifespan of the reactor.

Based on the success of the upflow reactors, a full-scale upflow reactor was constructed at the abandoned Bellvue coal mine, a second full-scale upflow reactor has recently been completed at an active coal mine, and a third is planned for another active coal mine.

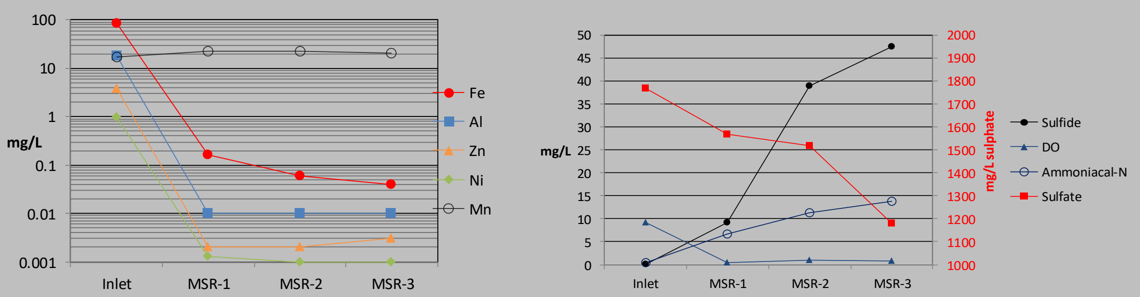

Figure C18. Changes in metal concentrations, sulphide, dissolved oxygen, ammonia, and sulphate through an upflow mussel shell reactor. MSR-1, 2, 3: outlet concentrations from three tanks installed in series.

Key findings relevant to mine management:

- Passive treatment technology is an active area of research, often utilising waste products to reduce cost.
- Waste mussel shells in New Zealand are an ideal passive treatment medium.
- Mussel shell reactors can be constructed in downflow or upflow configurations.
- Downflow mussel shell reactors work best for low Fe sites, otherwise Fe hydroxides will accumulate at the top, reducing permeability with time.
- Upflow mussel shell reactors work best for high Fe sites and high trace element sites, since these reactors remove metals as sulphides distributed throughout the reactor, and permeability is not compromised.

Key references:

Trumm D, Ball J, Pope J, Weisener C, West R 2015. Passive treatment of ARD using mussel shells – Part III: Technology improvement and future direction. Presented at 10th International Conference on Acid Rock Drainage, April 20–25, 2015, Santiago, Chile.

Weber PA, Lindsay P, Hughes JB, Thomas DG, Rutter GA, Weisener CG, Pizey MH 2008. ARD minimisation and treatment strategies at Stockton Coal Mine, New Zealand. In: Bell LC, Barrie BMD, Baddock B, MacLean RW eds. Proceedings of the Sixth Australian Workshop on Acid and Metalliferous Drainage. Brisbane, ACMER. Pp. 113–138.

Weber P, Weisener C, Diloreto Z, Pizey M 2015. Passive treatment of ARD using mussel shells – Part I: System development and geochemical processes. Proceedings of the 10th International Conference on Acid Rock Drainage and IMWA Annual Conference. April 21–24, 2015. Santiago, Chile.

Weisener C, Diloreto Z, Trumm D, Pope J, Weber P 2015. ARD passive treatment using waste mussel shells – Part II: System autopsy and biogeochemical investigations. Presented at 10th International Conference on Acid Rock Drainage, April 20–25, 2015, Santiago, Chile.

3.6.6 Treatment for suspended solids

Management of overburden and rehabilitation activities to prevent erosion and the generation of suspended sediment should be integrated, as necessary, into mine discharge planning prior to release into any receiving environments. However, it is likely that these management activities will be insufficient to completely mitigate suspended sediment, and treatment will be required. Default trigger values in New Zealand for turbidity are 4.1 NTU for upland rivers and 5.6 NTU for lowland rivers (ANZECC & ARMCANZ 2000).

Settling ponds to capture TSS in mine discharge prior to release to any receiving environments should be considered a bare minimum. In some instances active treatment will be required. This could occur if very fine dispersive clays are present (see Craw et al. 2008 for geological factors that control the characteristics of suspended sediment). Research on the paleoplacer gold deposits in Central Otago has found that schist and greywacke basement rocks altered to kaolinitic clays are the primary cause of high turbidity in water from placer gold mines (Druzbicka & Craw 2011).

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 25). 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) can 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 of Cavanagh et al. 2015.)

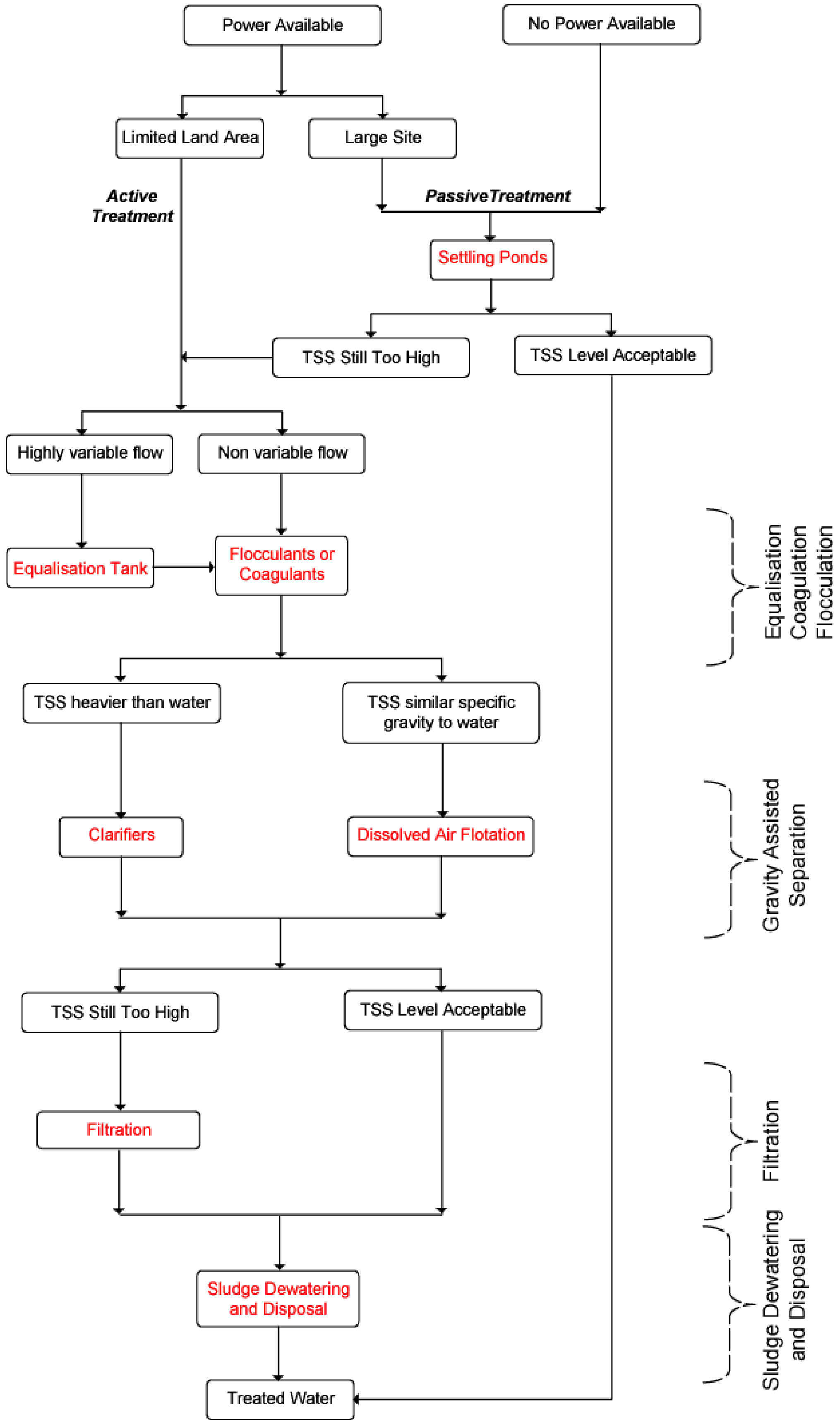

Figure 25. Flow chart to select a treatment system for total suspended solids (TSS).

3.7 Terrestrial rehabilitation

Broad rehabilitation principles and key specific outcomes (e.g. areas to be avoided, features to be reinstated) should be planned during mine feasibility studies. Long-term rehabilitation outcomes and (shorter-term) closure success criteria should be proposed alongside a concept rehabilitation landscape plan as part of environmental impact assessment (see Case Study 2, p.30). These will be finalised during consultation, resource consenting and mine access permitting, and in most cases should be explicitly referred to in consent and access conditions.

The concept plan and closure success criteria will inform a draft closure plan, which should be prepared at the beginning of mining and updated regularly as more information becomes available. They may be required to be updated annually, alongside the annual mine plan, to reflect changes in information and the economics of the mine. Targeted long-term rehabilitation outcomes may be based on reference sites, but the success criteria for mine closure are usually an intermediate condition, especially for native forest ecosystems, which take many decades to develop. While success criteria may be relatively prescriptive, the best outcomes are often achieved by allowing flexibility in the way outcomes are achieved.

Mine rehabilitation is the primary method for remedying effects of mining that cannot be avoided. All ecosystems can be rehabilitated, but for native ecosystems it may take many decades to centuries to develop above-ground structural complexity with logs and deep leaf-litter layers, and the time taken for rehabilitation means that other ecological mitigation offsite is required to prevent a net loss. In 2010 the Parliamentary Commissioner for the Environment, Jan Wright, investigated ‘Mining the conservation estate’, and concluded that such mining must ‘go beyond compensating for damage’ and provide net conservation benefit, which could be achieved by funding predator control in other areas of the DOC estate (PCE 2010). Such off-site compensation may be linked to the quality of on-site rehabilitation.

Effective on-site rehabilitation minimises the long-term effects of mining and can enhance cultural heritage (if present prior to mining) and access for long-term uses compatible with the conservation estate (pest control, and tourism, such as walking and cycling tracks). For long-term infrastructure such as roads, bridges and structures to have ongoing use, a source of funding is needed for maintenance. For this to happen, rehabilitation outcomes should be developed during pre-mining consultation, using a landscape concept plan showing a long-term vision with specific success criteria. This long-term aim is often based on reference sites. However, the rehabilitation criteria in resource consents at ‘closure’ – being the time when rehabilitated areas have reached a condition at which the mining company can relinquish resource consent responsibility for a site, and bonds are released – will be much earlier. The vegetation and landscapes will be immature, typically 5 to 20 years old.

Criteria for closure need to allow flexibility in the way outcomes are achieved. 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. Capabilities to access, strip and move rehabilitation resources will change as equipment or contractors change. This is particularly important for direct transfer, in which soil and vegetation are moved together in large sods. Many different combinations of equipment have been used – all influence the outcome that can be achieved by influencing the size of sod, size of tree, extent of disturbance, and accuracy of placement. Short-term closure criteria and long-term concepts will be developed with input from landowners, 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), specific biodiversity species and ecosystems, 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 (see section 3.7.2). A generic flow chart that shows both the common resources available for rehabilitation and what influences their salvage and reuse is provided in section 4.5 (Figure 33). Rehabilitation resources, especially living sods and 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, including many individual species (insect and plant), and to control erosion. Its use is limited by mine scheduling, because final (backfilled) landforms with NAF substrate have to be available. Timely access to stripping areas also has to be available to salvage mānuka slash, and where forests are present trees need to be pre-felled before removal of stumps and understorey using direct transfer.

Early rehabilitation is vital to demonstrate on-site capability and allow site-specific techniques and costs to be developed through adaptive management. In native ecosystems, such rehabilitation should occur in the first year along mine access roads and bunds. Such rehabilitation, especially of fill sites, is important to minimise edge effects, which are the negative effects created on adjacent, undisturbed areas. Maintenance and monitoring of these, and later rehabilitated areas, are important components of successful rehabilitation, despite monitoring rarely being done effectively or consistently at New Zealand mine sites. The failure to record what is done, and to apply adaptive management, results in less than optimal outcomes – wasting both dollars and years (see Case Study 9, below).

Case Study 9: Overburden characterisation is critical to predict and understand plant performance

Overburden generated from mining is notoriously heterogeneous in composition and physio-chemical properties. Different rock types, local stratigraphy, and weathering can contribute to high variation in surface layers. New Zealand has many historical mine sites, where any retrospective rehabilitation is faced with recontouring to deliver stable slopes, an absence of soil, and a high cost of importing amendments or soils. At the Wangaloa open-cast coal mine in south Otago, a variety of substrates were exposed from coal-bearing sequences between 1945 and 1989 (see Figure C19). Final rehabilitation was initiated in 2002, using recontouring followed by planting over 100,000 nursery plants into 10 ha of unamended overburden.

Figure C19. Heterogeneity of overburden in rehabilitated area (left); dieback of koromiko (*Hebe salicifolia*) after 10 years (right) (photos 2018).

The University of Otago's Geology Department has characterised and monitored the Wangaloa overburden and waters since 2001. Combined with tracking plant performance, this study has been able to identify key drivers of revegetation at the site and associated time frames. Some of the answers were not apparent until after 10 years; some plants initially performed well before dying back year after year (e.g. *Hebe saliciflora*); others started slowly and have improved over time (e.g. mānuka).

Large losses of nursery plants occurred on the overburden (60–65% mortality in the first 3 years), with some areas unsuccessfully replanted three times up to 2010 (Todd et al. 2009). Overburden amended with imported topsoil and raw overburden rich in Cretaceous siltstone had the highest plant survival due to higher pH (4.5–5) and water retention compared to other types of overburden. Quartz-rich overburden also had relatively benign root zones, except when quartz pebble content exceeded 35% volume, at which point nutrient and water availability reduced to growth-limiting levels (Craw et al. 2007; see Figure C20). Seedlings planted into coal-rich and pyrite-bearing overburdens died within 1 year, repeatedly. These two hostile root zones had AMD (pH 1–3.5) from sulphate oxidation, with delayed onset in some areas due to earthworks inadvertently bringing acid-generating material to the surface. High boron in the coal (400–500 mg/kg) and very dry conditions exacerbated plant stress, with some plants having up to 230 mg/kg B (Rufaut et al. 2015).

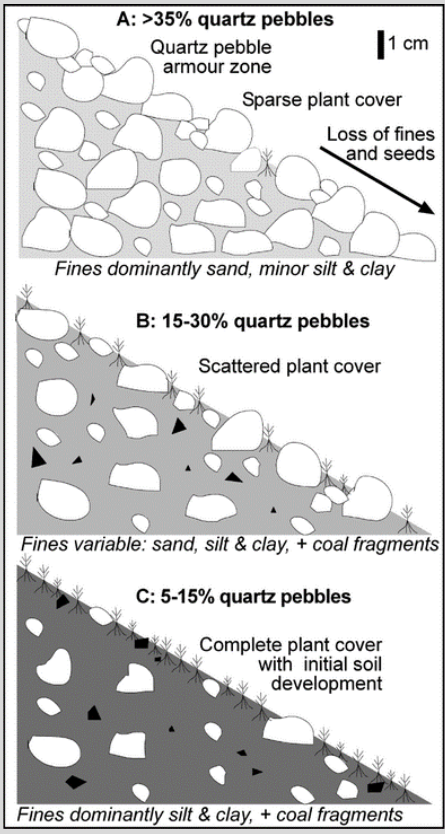

Figure C20. Plant cover and key characteristics of the three main root zones at Wangaloa.

Key findings relevant to mine rehabilitation:

- Monitoring plant survival and growth should include identification of site-specific limiting environmental factors by sampling 'good' and 'poor' performing areas.
- Nursery plants don't tolerate pH < 4; in such cases the root zones (20 cm depth) need to be effectively ameliorated and maintained over many years.
- Planting success should be measured over 5 to 10 years, as initial results may not reflect long-term outcomes.
- Unacceptable outcomes in planting mortality should trigger adaptive practices.

Key references:

Craw D, Rufaut CG, Hammit S, Clearwater SG, Smith CJM 2007. Geological controls on natural ecosystem recovery on mine waste in southern New Zealand. Environmental Geology 51: 1389–1400.

Rufaut CG, Craw D, Foley A 2015. Mitigation of acid mine drainage via a revegetation programme in a closed coal mine in southern New Zealand. Mine Water and the Environment 34: 464–477.

Todd AJ, Rufaut CG, Craw D, Begbie MA 2009. Indigenous plant species establishment during rehabilitation of an open cast coal mine, south-east Otago, New Zealand. New Zealand Journal of Forestry 39: 81–98.

Processes to achieve selected rehabilitation outcomes are described in the following sections: rehabilitation to pasture, plantation forestry, and native ecosystems. Many other post-mining land uses are possible; for example, cropping and horticulture (viticulture), residential housing, recreation (historical mine relics, mountain biking) and public amenity, but these are not covered in this document.

3.7.1 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 is 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 that 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 Case Study 9, p. 92 and section 5.7).

The post-mining topography controls rehabilitation options by influencing the media, depth, and drainage conditions in the rehabilitated root zone, and also the ability to access a site efficiently. High walls represent one extreme as the steep slopes prevent significant amendment of a rooting zone, which is typically skeletal: substantial root zones can only be developed on parts of benches or areas of backfill (i.e. buried high walls). Root zone depths may also be limited in tailings dam fill embankments and similar structures; in cases where deep root exploration is not desirable, or wind-throw exposes unsuitable sub-surface materials, trees may be excluded.

The roughness, or microtopography, of a steep surface is important: the rougher the surface, the greater the potential depth, coverage and accumulation of soil. Soil depth and stability limit the plant cover that can be established. Steep, smooth surfaces are a common issue that limits rehabilitation, especially on high walls and road cuttings: soils or hydroseeding mixes will not hold on a smooth surface, because they have few sheltered pockets in which seedlings can establish.

Overarching rehabilitation aims for plantation or agricultural land are often linked to 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 on productivity (drainage, erosion, fertility). However, this definition is not usually applicable to native ecosystems. It is useful to identify the 'things that people care about', and cross-check that long- and short-term success criteria are likely to address these things.

3.7.2 Identifying resources for rehabilitation

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

- *Areas that will not be disturbed* – these areas within the mine site are major sources of 'free' propagules that speed the diversity and recovery of natural ecosystems (and sometimes also weeds). Undisturbed areas may be especially valuable where they form peninsulas or 'islands', because such areas also provide shelter (modifying the climate). Limiting the mine footprint and keeping such undisturbed areas in the best health possible by weed and pest control should be the starting point of a rehabilitation programme.
- *Plants, mulches, soils and overburden*, which will be used to create new root zones. Also identify the resources that will devalue such material (acid-generating or weedy materials).
- *Materials for controlling and limiting erosion* – these include competent rock suitable for rip-rap used to create erosion-resistant drainage paths, mānuka and other vegetation (slash), and sods of direct transfer and wood (logs). An advantage of mānuka slash and direct transfer is the regeneration of native plants that deliver long-term erosion control (Case Study 10).
- *Plant and fungi/mycorrhizal propagules, including seeds and cuttings* – these include plants suitable for using as intact sods for direct transfer. Many native plants have mycorrhizal fungi that help them grow, and such fungi can increase water and nutrient uptake, and protect roots against pathogenic fungi. Plants may be selected that are difficult to propagate in nurseries and/or to create genetic 'reserves' that conserve a range of genetics from the site. Small plants growing in poorly drained and/or shallow soils are often candidates (e.g. *Celmisia* and sub-alpine *Dracophyllum* species).
- *Fauna, for relocation and potential reintroduction to rehabilitated areas* – these include invertebrates within direct-transfer sods (earthworms, wētā, and insects that break down leaf litter and are important food sources for other fauna). Permits issued under Wildlife Act provisions will require specific animals to be searched for and relocated, sometimes to areas where predator control is undertaken, and sometimes to available rehabilitation sites.
- *Material useful for creating habitat features for specific fauna of* interest – logs, boulders, tree stumps, individual plants (e.g. *Gahnia* for forest ringlet butterfly caterpillars).
- *Infrastructure* – such as posts, water troughs and buildings, and bridges.

Case Study 10: Mānuka slash is an effective revegetation method for difficult sites.

Slash or brush layering are informal terms for a revegetation technique known as *fascining*. Small- to medium-sized branches with ripe seed capsules or fruits are cut from parent plants. The branches (slash) are then laid in direct contact with the ground to maximise released seed fall. Multiple branches can be woven on top of each other (brush layering) to further increase fallen seed density and shelter. The leaves fall, creating a protective mulch for the germinating seeds. Seedlings developed from fascining can seemingly tolerate fairly hostile 'soil' conditions.

At the historic Wangaloa coal mine in south Otago, acute acid rock conditions developed over 0.3 ha of recontoured waste rock during retrospective rehabilitation that freshly exposed acidic overburden. The cause of a pH of 1–3 was oxidation of pyrite on and near the surface of a quartz rock conglomerate, which formed the bulk of the waste stack. Amendments, multiple plantings of nursery-raised seedlings, and hydroseeding all failed to establish an effective vegetation cover over 7 years.

In 2010 mānuka fascining (1–2 m long branches), cut from nearby shrubs, was placed over very hostile acid conditions (Figure C21). The slash was cut in late summer/early autumn, when seed pods were mature, and layered into eight 2 × 2 m plots. Wooden pegs and a 'lacing' of string secured the slash to the sloping, exposed and windy site. The plots received no further attention and were simply observed over time.

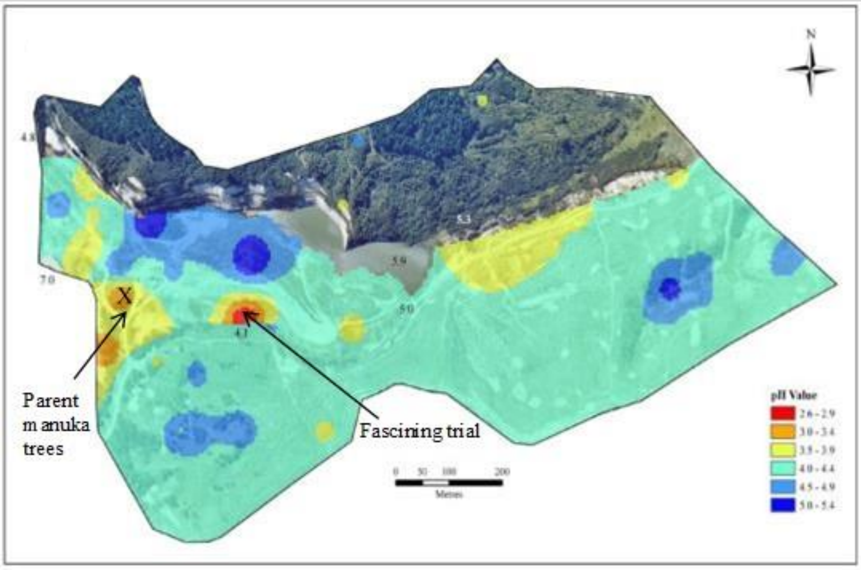

Figure C21. Average substrate pH at Wangaloa coal mine, and the location of the fascining trial and mānuka slash source.

Three years later, hundreds of young mānuka seedlings had reached c. 20cm height in the trial plots (Figure C22). Seedlings were aggregated in small depressions and rills in a mosaic with bare ground (Figure C22). In 2018 the seedlings have continued to grow and develop (e.g. flower), increasing in size and cover. Other native species (e.g. orchids) have been able to establish among the mānuka. The proportion of bare ground on the acid slope has been significantly reduced despite pH remaining around 2.5–3. At this site fascining has successfully 'matched' a local, native, nursery species to challenging conditions to support ongoing vegetation development.

Key findings relevant to mine rehabilitation:

- Key fascining practices underpinned success with mānuka (e.g. the slash had viable seeds, and was secured to the surface with netting and pegs).
- Highly acidic conditions and low nutrients removed grass and weed competition, which maximised mānuka establishment. Where soils are not so hostile, slash must be laid as soon as final surfaces are complete to give mānuka a head start.
- Choice of on-site parent plants as sources of slash may have helped seedling tolerance to the local, hostile conditions.
- Fascining supports the natural establishment of other native species. First, the slash creates stable, sheltered sites with higher humidity and lower temperature fluctuation (frost), which retain more water. As mānuka grows, the branches increase shelter, and roots bind and stabilise soils; mānuka leaves and fine roots also improve soil organic matter when they die (increasing nutrient supply and moisture storage).

Key references:

Rufaut CG, Craw D 2010. Geoecology of ecosystem recovery at an inactive coal mine site, New Zealand. Environmental Earth Sciences 60: 1425−1437.

Rufaut CG, Craw D, Foley A 2015. Mitigation of acid mine drainage via a revegetation programme in a closed coal mine in southern New Zealand. Mine Water and the Environment 34: 464−477.

Figure C22. Mānuka growth A) 3 years (2013) and B) 8 years (2018) after fascining.

3.8 Economics – mine bonds

The financial risk associated with mines in New Zealand is typically covered by a bond or bonds that comprise two components. One component is the 'performance' bond, which reflects the rehabilitation cost of a site. The other component reflects the 'post-closure' site management and maintenance costs. Both bond components may include a sub-component to cover costs arising from residual risks, although this is more commonly included in the post-closure component to cover long-term residual risks that will remain after the mine has been closed.

For clarity in the following discussion, two bonds are assumed: one to cover each component. However, it should be noted that the New Zealand mine bond approach usually requires an annual review of the closure and post-closure costs, and this regular review and update of the bond quantum means that a single bond covering both components is equally acceptable. The total value of the bond(s) held at any point in time varies directly according to the remaining closure obligations.

The intention of the performance bond is to provide sufficient funds for site closure and rehabilitation in the (unlikely, see above) event that a mining company is unable or unwilling to fulfil the conditions outlined in its resource consents. It includes costs to complete tasks such as:

- demolition and removal of structures
- site clean-up, including removal and disposal of contaminated soil
- stabilisation of earthworks and landforms
- rehabilitation, including re-contouring, spreading sub-soils and topsoil, revegetation, and weed control until closure conditions are met

- maintenance of roads
- environmental and geotechnical monitoring
- staff, administration and operating costs.

Other bond components can be added when specific consent conditions create obligations that are separate from those listed. For instance, if off-site habitat restoration is a consent condition, its cost can be included in the performance bond, or a separate bond can be posted to ensure this activity is implemented.

Some of these tasks extend over a period of years (e.g. monitoring and weed control). The performance bond covers costs over a period of aftercare up to the point in time when all of a site's closure criteria are achieved and the site can be considered fully rehabilitated. The cost of completing the tasks often reduces with time. Completed bulk earthworks, for instance, incur no further costs, and costs associated with revegetation and weed control decrease as new plantings become established.

A fully rehabilitated mine site that has met all of its closure criteria may still require some level of ongoing maintenance, management, and monitoring. If there is a need for such maintenance, the post-closure bond covers foreseeable ongoing costs for a period, and possibly forever, for such tasks as:

- ongoing treatment of mine drainage
- periodic repairs to low-permeability layers on landforms containing sulphidic waste/overburden
- regular clearing and periodic maintenance of drains
- geotechnical inspections of any large water-retaining structures
- periodic repair works following intense weather or seismic events (e.g. 20–50-year return period rainstorms, or earthquakes)
- groundwater and surface-water quality monitoring.

The post-closure bond is calculated for the most cost-effective way to achieve long-term environmental security. For water quality, for instance, the person calculating the bond decides, in discussion with relevant parties, whether to include the construction of an anoxic limestone drain, a mussel shell reactor, or a water treatment plant. This assumption can be updated as new information becomes available and would be reflected in the next re-evaluation of the bond.

The post-closure bond also typically includes a residual risk cost component that reflects risk events that could occur and would result in ongoing environmental impairment if no remedial action is taken. Residual risk events associated with closed mine sites usually fall into two main categories: geochemical risks and geotechnical risks. There may be several different risks under each category, and different sites may be subject to different types of risks. All risks need to be carefully assessed and quantified. Importantly, operational risks should never be included in a post-closure bond (and are rarely appropriate for inclusion in a performance bond risk cost component).

The residual risk cost component is derived using standard good-practice risk assessment methods, having regard to the likelihood of the occurrence of the risk events and the consequential costs. Because it applies to the period beyond the attainment of closure criteria, the likelihood of post-closure residual risk events and the residual risk itself should be small. As a result, the post-closure residual risk cost typically makes up a relatively minor portion of the total bond value.

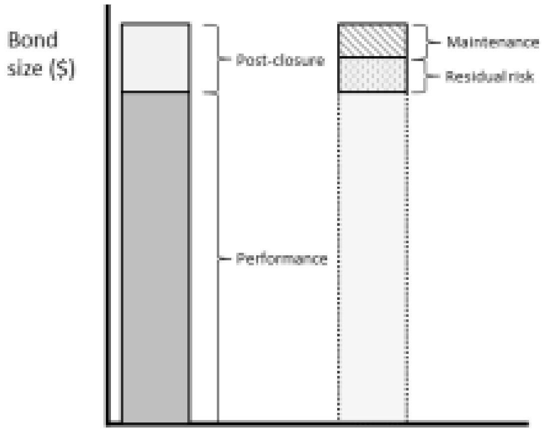

Figure 26. **Approximate common structure of mine bonds in New Zealand.**

Figure 26 shows that the performance and post-closure bonds are not of equal size. The ratio shown above is not fixed, however: the size of the performance bond relates directly to the area of disturbance, and maintenance and residual risk components are site-specific and partially determined by practice during operations.

While the magnitude of the performance bond is expected to fluctuate from year to year as new areas are disturbed, or as progressive rehabilitation matures, the post-closure bond is comparatively stable across the mine site's life. The starting point for the post-closure maintenance component remains constant (attainment of closure criteria). The residual risk element does not generally change, unless specific risks are reduced through changes in the closure plan or new risks are identified during the life of a mine. New technologies and changes in operational procedures, such as improved storage of waste rock, can reduce the expected costs of ongoing maintenance and thus the size of the maintenance element of this bond. Two such impacts are illustrated in the following sections.

The underpinning (conservative) assumption behind the New Zealand bonding approach is that the company that owns the mine site will fail within the next 12 months. The subsequent assumption is that all of the performance bond and the maintenance component of the post-closure bond will be spent on closing the site and maintaining it over time. As with all estimating exercises, there is uncertainty as to the magnitude of the closure cost and the post-closure maintenance costs. It is therefore reasonable for the bonds to include some appropriate level of conservatism, but it is equally important that the bonds not be overly conservative. Another conservative assumption typically applied to the estimate of the closure costs is that the bond will be called on at the point in time between annual bond reviews, when the area of disturbance is at its greatest.

Unlike the performance and maintenance components, the residual risk component has the additional uncertainty of whether any risk events will occur, and if so, which ones and when. Again, the risk cost component needs to be appropriately conservative without being overly so. One of the conservative assumptions applied to this component is that risk events can occur at the very start of the post-closure period.

The residual risk element of the post-closure bond covers a range of events, some of which may have extreme consequences but should be expected to occur only rarely. The magnitude of this component of the bond is determined by the range of possible events and rehabilitation actions, the cost of these actions, and their likelihoods. The risk is typically quantified, with Monte Carlo simulations used to provide the expected cost of remediation. In 10,000 Monte Carlo simulations of the closed mine a catastrophic event would occur, for instance, only 200 times. If there are many such events, only 50 simulations out of the set of 10,000 may include them occurring simultaneously; these outcomes would have a very high cost but a very low probability. Considering the probability distribution of various levels of financial risk, such events are only fully covered by the post-closure bond if levels of confidence close to 100% are demanded. Such a level of confidence is inappropriately conservative, so the residual risk cost component should not include the occurrence cost of such low-risk outcomes, even if the related costs are very high.

Both the performance and post-closure bonds apply discounting to costs that occur for more than 1 year or that occur after year one of the closure works. Discounting reflects the time value of money: one can put a small amount of money ($76.05) in a savings account earning 5% annually now, and compound interest ensures that $10,000 will be available in 100 years. Since the bond quantum accounts for ongoing costs into the future using discounting, the interest earned from the sum paid by the bondsman is sufficient to cover all future expenditures.

Selecting an appropriate, inflation-proof discounting rate means there is no need to account for future cost fluctuations relating to equipment hire, fuel, and labour costs: the bond is calculated to pay for rehabilitation that could start tomorrow and the appropriate discount rate addresses cost fluctuations in the future. For any given area of disturbance, the performance bond will fluctuate over time with market conditions, but this does not affect the exposure of regulators or other bond holders to rehabilitation risk. Progressive rehabilitation limits the exposed area and therefore the maximum size the performance bond can take.

Upon successful closure of the site, the performance bond is released. The mining company may choose to continue to maintain the post-closure bond or to settle a fund equal to the bond value on an appropriate entity, which inherits responsibility for the post-closure site maintenance and draws on the fund to cover those costs. Because the post-closure period may be long and even perpetual, the nature of the funded entity also needs to remain in existence over the long term. Where a mine site is on public land, the fund can be settled on the government department responsible for that land on the assumption that government will last in perpetuity. Where the site is on private land, the most secure and appropriate entity has been deemed to be a trust. The trust inherits ownership of the disturbed area of land and is charged with administering its ongoing care. Once the post-closure fund is settled, the post-closure bond can be released.

Note that current bonding practice does not consider the value of environmental damage *per se*: it covers the cost of preventing and mitigating such damage. While costs are unlikely to reflect the actual value of maintaining environmental quality, engineering and construction costs are easier to determine with some accuracy. By setting consent conditions in such a way that they reflect the (intangible) benefits communities expect from the post-closure landscape, bonds can bring these two perspectives (value vs cost) more into line.

4 Operations

4.1 Introduction

The operations phase of the mine environment life-cycle covers all activities associated with the extraction of coal, and extends from the development of the mine to activities leading to mine closure. As mining operations commence, the testing of the efficacy of different management options can be undertaken at different scales. A certain amount of ongoing monitoring is also required to be undertaken routinely to inform operational management to prevent unexpected negative changes in mine drainage chemistry and subsequent negative impacts on stream ecosystems. Technical studies might be required to determine the best way to manage waste rock or tailings, or deliver best revegetation outcomes.

The more detailed knowledge gained during ecosystem and overburden removal should be used to inform rehabilitation planning, particularly recoverable soil volumes and area of direct transfer, overburden strata suitability as root zone, and the efficacy of sediment control methods. Preliminary or 'pilot' rehabilitated areas may be restricted to access road batters and lower levels of permanent ex-pit dumps. Although they may be small, these are important sites to test and demonstrate site-specific sediment control (through prevention) and revegetation methods that will be scaled up over the larger site. Such trial sites provide a focal point for monitoring and discussions with regulators and stakeholders. Monitoring the volumes of recovered rehabilitation materials against the total stripped area, and the costs and performance of early revegetation, should inform calculations of accumulated rehabilitation liability.

This section outlines the different activities that occur during mine operations to minimise environmental impacts and when moving towards closure, as shown in Figure 30. Early operational activities, such as access road and ex-pit dumps construction, are also opportunities to demonstrate avoidance of effects by minimising the site footprint (including minimising edge effects), avoiding high-value sites, and conserving rehabilitation resources. Avoidance of effects is a key activity throughout the operational stage of a site. During the main phase of activity, the scale of mine operations may expand or contract as resource prices dictate, influencing the specific activities that are undertaken. Later-stage operational activities are focused on ensuring that mine management is heading towards closure (Figure 27).

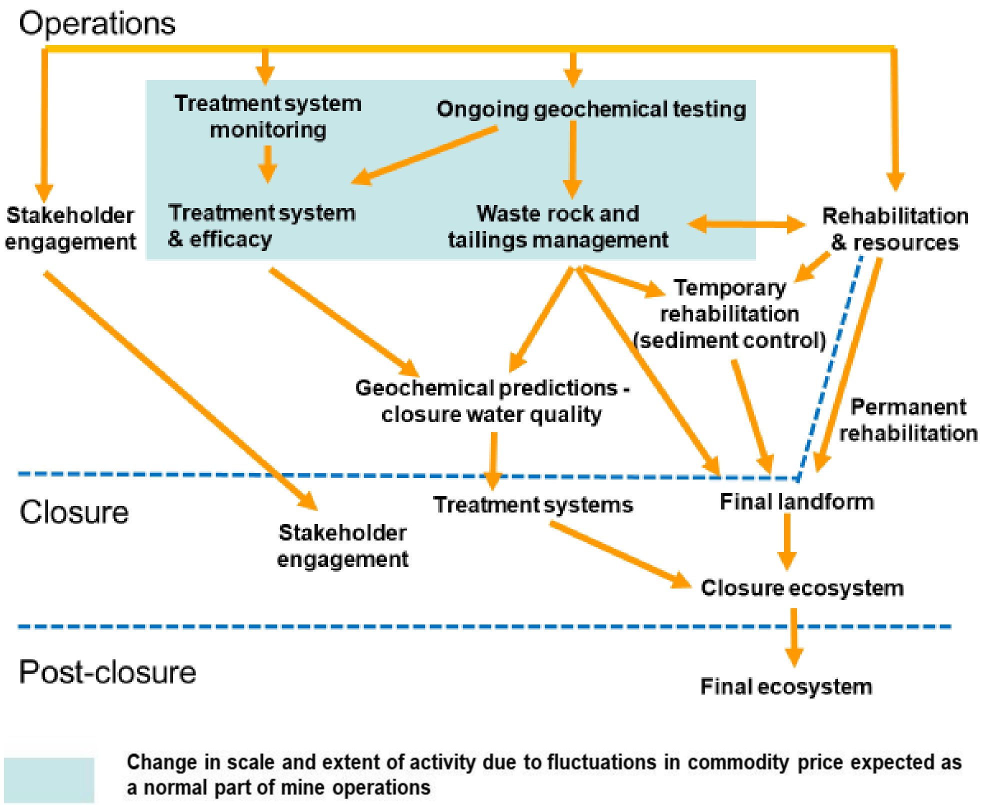

Figure 27. Overview of activities that occur during the operational stage of the mine life-cycle.

4.2 Engagement activities

Throughout mine operations there should be ongoing engagement with iwi and stakeholders. Initially this can help define short-term rehabilitation criteria. As mining proceeds, iwi and stakeholder input can contribute to defining greater specificity for the initially agreed general post-mining outcomes. Permit holders are required to report annually on mining operation progress, including the status of compliance with any consents. Consideration should be given to using this opportunity to update stakeholders on the status of environmental management, as all the relevant information should be to hand. This should include reporting on the progress of intermediate points of closure that have been developed with the ultimate landowners and/or custodians. As noted earlier, the language used will be important in determining the effectiveness of engagement, including whether 'what matters' has been understood and whether the conditions to deliver 'what matters' have been effective. Further guidance on reporting required by NZP&M is provided at http://www.nzpam.govt.nz/cms/permit-holders/reporting-requirements/annual-reports.

4.2.1 Iwi engagement

Under the *Minerals Programme*, Tier 1 permit holders are required to provide an annual iwi engagement report, and are encouraged to contact relevant iwi/hapū before submitting their report. Guidance for reporting on iwi engagement is also available (NZP&M 2016) and includes suggestions for what that engagement could consider; for example, how mining-associated activities could be used to develop economic and resourcing opportunities for Māori groups in the operation region.

The opportunity to be involved in activities, particularly rehabilitation practices (such as plant propagation, planting, and maintenance) and environmental monitoring is often appreciated and can result in innovative approaches to rehabilitation that would otherwise be missed (see Case Study 11). Long-term maintenance, particularly pest plant and animal control, can support practical ongoing kaitiakitanga (stewardship) by tangata whenua, but may require training and long-term contracts. Notwithstanding potential restrictions on involvement due to safety considerations on operational mine sites, these opportunities can meet operational needs and may provide Māori with additional skill sets that can be used elsewhere.

Transparency around planned activities, fluctuating commodity prices and/or resource identification influencing those activities, will be a valuable contribution to developing a value-based relationship between iwi and the mining company. A challenging aspect of the ongoing relationship may be identifying or agreeing when the mine will move into closure, particularly in locations where this is of greatest interest to iwi, but also where there is potential for ongoing mining as further resource extraction opportunities are identified.

Engaging with local iwi to develop and facilitate a cultural awareness programme that all mining employees participate in can enable the cultural aspect to be considered in the context of day-to-day mining operations. This may initially be stimulated by a consent condition, but progress towards a genuine relationship will be reflected by a common desire between the mining company and iwi to deliver a meaningful programme that represents the interests of all tangata whenua.

Case Study 11: Integrating cultural values with rehabilitation practice, Tui mine

Remediation of the abandoned Tui mine, on the slopes of Mt Te Aroha, stabilised 100,000 m^3 of tailings in a dam within a steep-sided valley. The tailings dam, mine foundations and ore stockpiles were stripped, stabilised, sown in grass and replanted by May 2013. Remediation also aimed to 'address, as far as practicable, within the limitations of the project, the impacts of the Tui Mine on the taonga of the Te Aroha maunga for iwi' (Waikato Regional Council[24]). This case study contrasts the revegetation approach and results used in the ore stockpiles and mine foundations (Figure C24) with those used in 2015 to rehabilitate tailings and ore in a gully adjacent to the tailings dam, known as Area S (Figures C23, C25). The Area S works were planned after the Iwi Advisory Group was established, and when a Cultural Monitoring Plan had been developed (Anderson 2013). These informed the Area S works in an iterative process involving site visits before and during remediation works. Combined with the narrow footprint and shallow excavation, this resulted in a visibly different outcome, at least in the medium term.

Figure C23. Upper part of Area S showing diverse regeneration of native plants (five finger, cabbage tree, rangiora, māhoe, astelia, tree ferns), September 2018.

[24] The Tui mine remediation project, including many water quality and ecological reports: https://www.waikatoregion.govt.nz/Services/Regional-services/Waste-hazardous-substances-and-contaminated-sites/Tui-mine/

Figure C24. Conventional planted ore stockpile area at age 1, and then at age 4.5 years, when starting to blend with the adjacent forest.

The rehabilitation approach for Area S was designed to:

- promote natural succession by creating root zones that would allow seeds and transplants from the adjacent forest to establish (i.e. helping the maunga to heal itself)
- create a more natural topography, avoiding linear and obviously 'engineered' lines and vertical cuts or cleared areas; for example, the new rip-rap-lined water course was gently sinuous, and tree ferns were placed to break up linear features and prevent 'long views' down the stripped area (Figures C23, C25)
- limit importation of materials (no nursery plants or imported soils were used)
- recycle suitable plants (especially tree ferns), soils and logs by salvaging them ahead of stripping and placing them back into rehabilitation areas.

The approaches were supported by a cultural induction with all contractors. Iwi representatives, an ecologist, engineers and earthworks contractors worked together to get options that were both technically and culturally feasible, and to reduce risks to the contractor of 'non-standard' works and outcomes. This included:

- prioritising those plants to protect – a crane was used to avoid damage to plants screening the site, and high-value trees and ferns were physically identified to help protect them during earthworks
- understanding the cultural value of specific plants and how this affects their suitability for rehabilitating this site, which remains contaminated (e.g. using *Clematis paniculata* as a warning)
- post-rehabilitation weed control to remove specific weeds (pampas, butterfly bush)
- developing medium-term interventions that enhance kaitiakitanga (e.g. weed control, collection of road-fall tree ferns and fronds, and selective removal of weeds along roadsides to encourage native plants, not indiscriminate mowing, which promotes pampas).

Figure C25. Natural regeneration with transplanted tree ferns, kiekie cuttings, and natural regeneration at age 2 years, September 2018.

Key findings relevant to mine rehabilitation:

- Slower revegetation that uses natural seed sources, recycled vegetation and wood can be a viable alternative to planting where suitable erosion-resistant surfaces can be created, local seed sources are nearby, and weed control is strategic. A much more natural and broader range of native plants is developed in this way.

- Pre-works consultation between iwi, engineers and ecologists helped understand priorities and develop solutions. This continued through site works to give confidence to earthworks contractors to create 'messy, rough' outcomes, to conserve and recycle suitable plants and wood, to protect edges from unnecessary damage, and to place tree ferns to reduce visual impacts.
- Natural regeneration needs to be supported by funding/contracts allowing at least 3 years of selective weed (and pest) control, preferably by kaitiaki, and working with technical advisors where necessary.

Key references:

AECOM 2012. Tui Mine landscape Plan. Tui Mine remediation project. 2 August 2012. Prepared for Matamata Piako District Council. https://www.waikatoregion.govt.nz/services/regional-services/waste-hazardous-substances-and-contaminated-sites/tui-mine/

Anderson A 2013. Tui Mine Cultural Monitoring Plan 2012–2017. Tui Mine remediation project. Prepared for Tui Mine Iwi Advisory Group.

DOC (n.d.). Tui Mine remediation project. http://www.doc.govt.nz/our-work/tui-mine/

Environment Waikato 2011. Tui Mine remediation project: Project Plan Document (Phase 1).

Morrell W J 1997. An assessment of the revegetation potential of base metal tailings from the Tui mine, Te Aroha, New Zealand. Unpublished PhD thesis, Massey University.

Quickfall GP, Basheer G, Croucher B, Jenkins IR, Fellows DL, Wilson T 2013. Tui Mine tailings remediation project – geotechnical environmental and construction challenges. WASTEMINZ Conference. https://www.wasteminz.org.nz/wp-content/uploads/WasteMINZ-2013-Tui-Mine-Remediation.pdf

Sabti H, Hossain MM, Brooks RR, Stewart RB 2000. The current environmental impact of base metal mining at the Tui mine, Te Aroha, New Zealand. Journal of the Royal Society of New Zealand 30(2): 197–208.

4.2.2 Other stakeholders

Ongoing engagement with other stakeholders will depend on the interest expressed during and after consenting processes. As indicated above, the required annual reporting to NZP&M on mining operations provides an excellent opportunity to update interested stakeholders on environmental management progress.

4.3 Operational management to minimise environmental impacts

Ongoing monitoring should be routinely undertaken by the mining company to inform optimal operational management 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 of treatment system discharge, is required to confirm 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 if improvements are required. Monitoring will be required to inform the size of bonds (section 4.7)

If long-term mitigation relies on a minimum cover quality and depth, the actual and predicted availability of suitable materials for the cover need to recorded to allow for the development of alternative strategies if the volumes or quality of cover materials change. Similarly, if the final exposed area of high wall changes, such that geology and/or aspect is unlikely to support the planned vegetation outcomes, this needs to be identified as early as possible. A main reason for poor-quality rehabilitation outcomes is inadequate volume and quality of topsoil or not using a suitable rooting medium, and so monitoring the volume, quality, and accessibility of such resources, and the areas yet to be rehabilitated, is critical throughout the operational mining phase. Water quality and biological monitoring provide confirmation on a broader spatial scale that management and treatment systems are working effectively.

4.3.1 Monitoring rock geochemistry

Rock geochemistry should be monitored throughout the operation of the mine to identify rocks that may affect mine drainage chemistry and to determine appropriate ongoing management of mineralised rock and waste rock – particularly if rocks that are likely to generate adverse effects (e.g. PAF) are present. Monitoring of rock geochemistry through acid–base accounting (ABA) for water quality purposes can be conducted alongside sample collection for exploration and resource development purposes (Sinclair

2018). The specific requirements for effective monitoring of rock geochemistry in operational mines are difficult to generalise because they are site- and deposit-specific. An approach to maximising the value of analytical testing is provided by Olds et al. (2015) (see Case Study 5, p.63). This approach could mean that all or a selection of ABA tests are completed, or that field-portable tests such as paste pH, portable XRF, or 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 ongoing 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 or trace element 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. The factors that must be monitored will vary between sites and could include either bulk rock chemical factors such as acid-neutralising capacity (ANC) or maximum potential acidity (MPA), or the availability and reactivity of specific trace elements. In some cases the relationships that control mine drainage can be complex and inter-related. Results from testing may also reveal that the rocks are less acid-forming than expected based on geology, as shown by Case Study 12 below.

Case Study 12: Variability in acid–base accounting for the Brunner coal measures – it always pays to do the analysis

Brunner coal measures rocks typically produce acid mine drainage (Pope, Weber et al. 2010) with low pH and high Fe and Al, and elevated trace element concentrations, particularly Zn, Ni, and Mn, and sometimes Cd, Co, Cr, Cu, and Pb (Pope, Newman et al. 2010). The characteristics of mine drainage from these rocks have been described for many coal deposits in the Buller and Reefton areas (McCauley et al. 2010; Davies et al. 2011; Mackenzie et al. 2011; Olds et al. 2016).

The Te Kuha coal deposit lies at the southern tip of the Buller coalfield, about opposite the town of Westport. It has been mapped as Brunner coal measures (Nathan et al. 2002), and mining this deposit would have been expected to produce AMD with a character typical of other mines in these coal measures. Recently (2011/2012) resource development work has been completed at the Te Kuha deposit by Stevenson Mining Ltd. Petrographic analysis of the coal indicated that the lower half of the deposit occurs in the Paparoa coal measures, while the upper half of the deposit occurs in the Brunner coal measures (Dutton et al. 2013).

As part of the resource development process, a suite of samples were collected for ABA analysis. These analyses revealed that the Brunner coal measures samples from the Te Kuha deposit have a lower potential for acid formation than previously tested deposits (Stockton and Reefton) hosted in these rocks. The average maximum potential acidity for Brunner coal measures rocks collected at Stockton and Reefton is 24.7 kg/tH_2SO4, whereas samples from Te Kuha have an average MPA of 4.2 kg/tH_2SO4. In addition, the Paparoa coal measures at Te Kuha have excess acid-neutralising capacity (average ANC is 19.3 kg/t H_2SO4). This means the NAPP values for Te Kuha are much lower, on average, than Brunner coal measures samples from other mines.

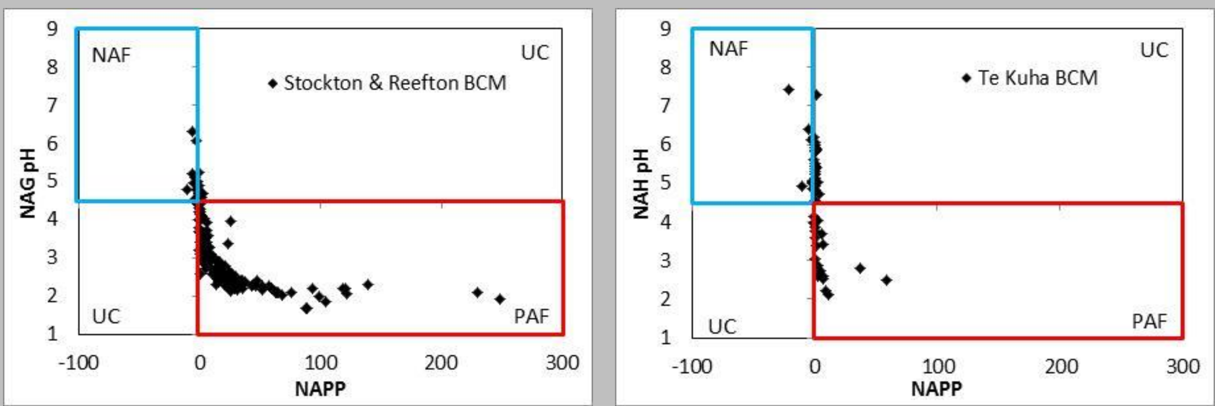

Figure C26. Acid–base accounting for typical Brunner coal measures (Stockton and Reefton) compared to Te Kuha Brunner coal measures.

These data indicate that there is an opportunity to manage mining (e.g. waste rock management, strategic resource extraction) of the Te Kuha deposit without making acid mine drainage.

Key findings relevant to mine management:

- Site-specific analyses are critical for optimal management.
- Strong regional trends in ABA could break down at the scale of a mine site.
- When combined with other analysis, the low MPA value for Brunner coal measures at this site indicates that AMD can be avoided and managed at Te Kuha.

Key references:

Davies H, Weber P, Lindsay P, Craw D, Pope J 2011. Characterisation of acid mine drainage in a high rainfall mountain environment, New Zealand. Science of the Total Environment 409: 2971–2980.

Dutton A, Newman J, Newman N, Pope J, Field A 2013. The Te Kuha sector, Buller Coalfield: a revised model. AusIMM Annual Branch Conference, Nelson.

Mackenzie A, Pope J, Weber P, Trumm D, Bell D 2011. Characterisation of Fanny Creek catchment acid mine drainage and optimal passive treatment remediation options. AusIMM New Zealand Branch Conference, Queenstown, pp. 281–292.

McCauley CA, O'Sullivan AD, Weber PA, Trumm D 2010. Variability of Stockton coal mine drainage chemistry and its treatment potential with biogeochemical reactors. New Zealand Journal of Geology and Geophysics 53: 211–226.

Nathan S, Rattenbury M, Suggate R 2002. Geology of the Greymouth area. Wellington, Institute of Geological and Nuclear Sciences.

Olds W, Weber P, Pope J, Pizey M 2016. Acid mine drainage analysis for the Reddale Coal Mine, Reefton, New Zealand. New Zealand Journal of Geology and Geophysics 59: 341–351.

Pope J, Newman N, Craw D, Trumm D, Rait R 2010. Factors that influence coal mine drainage chemistry, West Coast, South Island, New Zealand. New Zealand Journal of Geology and Geophysics 53: Special Edition – Mine Drainages, 115–128.

Pope J, Weber P, MacKenzie A, Newman N, Rait R 2010. Correlation of acid base accounting characteristics with the geology of commonly mined coal measures, West Coast and Southland, New Zealand. New Zealand Journal of Geology and Geophysics 53, Special Edition – Mine Drainages, 153–166.

4.3.2 Kinetic testing during operations

Once mine operations commence, larger-scale kinetic testing can be completed, and this might include trial waste rock dumps (WRDs) or tailings impoundments with alternative capping materials or neutralising agents added. As mining progresses, and WRDs and tailings storage facilities are constructed, kinetic test work could include lysimeters and instrumentation inside the WRD/storage facility (Pope, Weber et al. 2016b), 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 reasons to use simple kinetic tests at any time during mine life, depending on the duration of mining and the information required (Figure 28; Kerr, Druzbicka et al. 2015; Kerr, Pope et al. 2015).

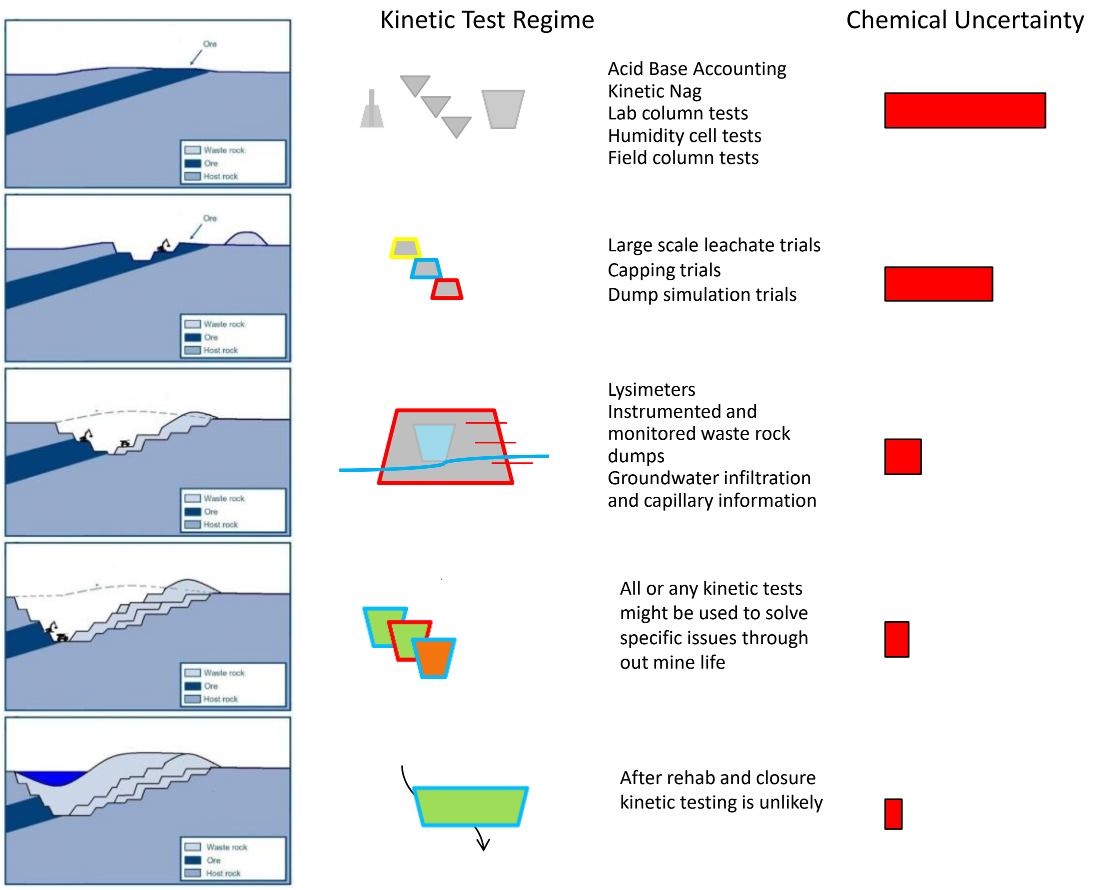

Figure 28. Schematic representation of the hierarchy of kinetic testing.

The value of kinetic testing is related to the certainty it provides for mine drainage chemistry, both 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 the 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 WRD)
- water ingress (water ingress can be substantially reduced by compaction in a real WRD compared with a column test)
- secondary minerals (secondary minerals can precipitate and might vary in their distribution in a real WRD compared with a column test).

At the point at which kinetic test work reflects monitoring of the geochemical conditions in full-scale WRDs, the results should accurately reflect mine conditions and the main source of uncertainty becomes forward projection of the mine plan through time. Some recent developments towards laboratory-scale kinetic testing that match field conditions have been made (Pearce & Scott 2015) and applied to PAF coal measures from New Zealand (Malloch et al. 2018).

In New Zealand, published kinetic test data from both field and laboratory column trials are available (e.g. Pope et al. 2011; Kerr et al. 2013; Pope & Weber 2013; Malloch et al. 2018). In addition, large-scale dump monitoring data are available from some coal mines (Weber et al. 2013). These data sets and papers provide generic information on the rates of acid and trace element release from coal and gold mines.

4.3.3 Leachate monitoring

Mine leachate comes from different sources that have different characteristics and implications, including high walls, waste rock, and mine adits. Often monitoring data are not collected in a specific enough manner to distinguish the characteristics of these different sources, and the chemistry of leachate from each source can be similar. However, it is important to understand these different sources of leachate during operations for predicting into the closure phases. High walls are likely to remain after mining and therefore will produce leachate into and beyond closure. Waste rock and tailings typically have a saturated and an unsaturated zone, can have low to high porosity and permeability, and might contain partial oxidation products that flush through during high rainfall events. Appropriate water collection systems should also be put in place at the base of the rock pile tailings storage facilities to characterise the quality of the leachate early in pile construction. Mine adits can produce a consistent flow rate driven by groundwater, and contain a chemistry limited mostly to Fe, sulphate, and low-pH water (Trumm et al. 2017).

High walls

Acid release from pit walls can be one of the most difficult aspects of a mine site to rehabilitate, especially if the closure plan does not include backfilling of the mine pit. In this circumstance, mine pit walls could continue to be a source of acid and trace elements long after mine closure. Rehabilitation of pit walls using grasses has been trialled at Golden Cross mine and with native vegetation at Martha mine, with some success, depending on the aspect and geochemistry of the wall. Further research of these techniques is required.

4.3.4 Overburden management

Overburden management during operations is critically dependent on appropriate monitoring and interpretation of rock geochemistry (section 4.3.1) to confirm the efficacy of WRD designs proposed during planning. In addition, it may be useful to undertake small-scale trials of proposed overburden management to determine its efficacy for minimising AMD.

During operations a key focus should be opportunities to prevent AMD during blasting and deposition of overburden material. During blasting, consideration should be given to:

- minimising the time that exposed PAF rock sits prior to blasting, mining and removal, and placement in the WRD, to reduce the time for oxidation to occur
- reducing the blast intensity to increase the size of PAF waste rock, thereby reducing the reactive surface area, or increasing the blast intensity in NAF waste rock to increase the surface area for neutralisation reactions to occur
- substituting stemming material (gravel pack placed in blast holes over explosives) with limestone, which can provide a source of ANC for the waste rock.

Deposition of waste rock should be undertaken in a way that prevents grain-size segregation, whereby the rock separates into coarser and finer-textured materials as waste rock rolls down a tip-head. This is a key element to minimising the availability of oxygen. Grain-size segregation is characterised by developing coarse rubble zones at the base of, or within, waste rock piles, and increases as the tip-head height increases (Figure 29). Grain-size segregation permits oxygen ingress by advection (Fala et al. 2003; Wilson 2008) and also promotes water ingress by permeability contrast. In a poorly constructed waste rock pile, advection accounts for the higher proportion of oxygen ingress. For example, Brown et al. (2014) indicated that c. 90% of oxygen ingress was by advection, and that diffusion accounts for 10%. Therefore, construction of WRDs to minimise grain-size segregation is likely to reduce oxygen ingress rates by approximately one order of magnitude.

Figure 29. Grain-size segregation at PAF coal mines, New Zealand.

Successful management of waste rock at epithermal mines occurred at Golden Cross mine, where rehabilitation has led to farm land being established over the old WRD (Figure 30). At this site a cap of NAF material up to about 2 m thick was placed over variably PAF waste rock. In addition, PAF material was re-deposited into the base of the open pit, where it remains water saturated. Water quality draining both the open pit and rehabilitated WRDs has achieved compliance for about the last 20 years. Successful management of waste rock also occurs at the operational Waihī mine, where waste is classified into seven types and is used for constructing the tailings storage facilities. These facilities are up to 20 years old and to date have successfully contained the tailings materials without a discharge that breaches consent conditions (Figure 30B).

Figure 30. **A) Rehabilitated waste rock (foreground) and open pit with PAF material placed in the base of the pit at Golden Cross mine. B) Rehabilitated tailings dam at Waihi showing rehabilitation to pasture and native plantings.**

Quality control / quality assurance

To ensure overburden is being managed appropriately, the following activities should be undertaken during operations:

- testing of blasted waste rock to confirm its geochemical classification and to validate the waste rock block model
- GPS tracking of excavators and trucks to ensure waste rock is delivered to the correct location
- testing of waste rock within the final constructed landform to ensure it is the correct material.

Suspended solids

In addition to considering AMD, consideration needs to be given to managing erosion and the potential to increase TSS in surface waters. Rehabilitation activities will be critical to minimise erosion, and these are discussed further in section 4.5.1. Techniques for minimising elevated TSS by managing surface water 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
- an 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, which can be used in low-water-flow zones to slow water velocities and catch particulates
- re-grading – reducing the slope length and angle to reduce the velocity and volume of runoff.

Auckland Council (2016), Environment Canterbury[25] and Basher al. (2016) also include useful information to manage erosion.

4.4 Water treatment systems

During the operational phases of mining, water quality predictions used to select treatment systems during mine planning can be confirmed through monitoring (see below), and selected treatment options tested. These tests can be conducted in the laboratory. Mine drainage can be synthesised in the laboratory to match the water chemistry that is predicted to occur during mining and at closure (see above). If active treatment is selected as an option, this test would involve a sequential titration with different neutralising chemicals, and analysis of dissolved metals and pH following each titration step. (See Trumm & Cavanagh 2006 for an example of this used at a New Zealand AMD site.) If passive treatment is selected, various treatment systems can be constructed in the laboratory and tested for effectiveness. Also, it may be feasible to establish small-scale field trials to determine the efficacy of treatment for the mining discharge (see Case Study 13; Trumm et al 2017).

Case Study 13: Trialling passive treatment options for Mn removal at an epithermal gold mine

[25] Erosion and Sediment Control Toolbox for Canterbury http://esccanterbury.co.nz/

Manganese exists in mine waters typically as Mn II, which can remain in solution over a wide pH range. Treatment systems that aim to remove Mn typically utilise the poor solubility of Mn IV and use oxidation techniques to precipitate Mn IV oxides. This oxidation is slow at circumneutral pH but can proceed rapidly at high pH. Other Mn minerals that may form in treatment systems where pH is raised and alkalinity and Ca are added include manganese hydroxides and manganese carbonates (e.g. rhodochrosite and kutnahorite).

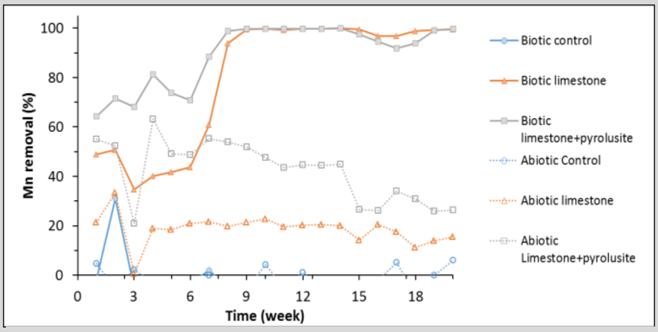

Figure C27. Manganese removal over time in different biotic and abiotic treatment systems.

Laboratory-based trials have compared the relative importance of biological activity and reactivity of Mn with Mn oxide surfaces relative to pH amendment alone in limestone leaching bed (LLB) treatment systems. Biological activity was the dominant process allowing optimum removal of Mn, together with the trace metals Ni and Zn. The Ni and Zn removal was coupled with Mn removal, and was attributed to adsorption onto Mn oxide surfaces. Over 99% of all three trace elements were able to be removed using biotic LLBs (Figure C27).

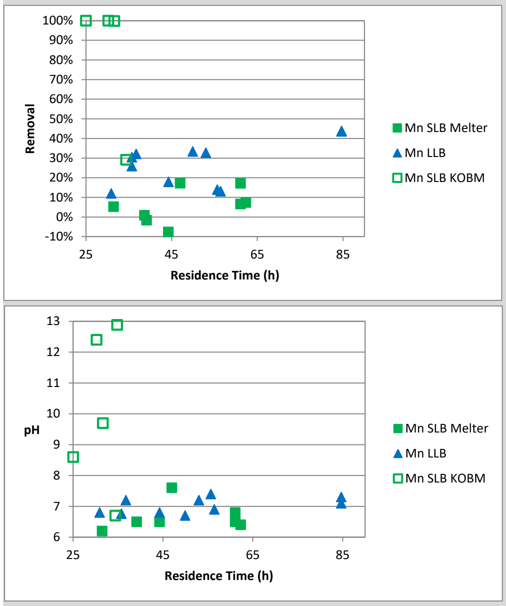

Figure C28. Change in Mn removal (upper) and pH (lower) with residence time in an LLB and SLB with two types of steel slag.

A field-scale trial using the LLB was tested alongside slag-leaching beds as a possible long-term Mn removal option. Steel slag contains CaO, the dissolution of which can achieve high pH in a steel-slag-leaching bed (SLB), resulting in precipitation of Mn minerals. A drainage with c. 100 mg/L Mn and elevated concentrations of Fe, Ni, Zn and Co was treated with a vertical flow reactor (VFR) to allow oxidative removal of Fe from solution, followed by one of two parallel systems: an SLB (using melter slag from iron making) and an LLB. After 7 months, a second type of steel slag with greater CaO content was trialled (KOBM slag from steel making).

The SLB with melter slag removed up to 17% of the Mn, 64% of the Ni, 96% of the Zn, and 70% of the Co (Figure 28). However, the highest pH attained was only 7.6. The KOBM SLB raised the pH to between 8.6 to 12.9 and removed >99% of the Mn, Ni, Zn, and Co. Correlation with pH suggests that removal of Mn in the SLB is likely through precipitation due to low solubility at higher pH. The LLB removed up to 44% of the Mn, 40% of the Ni, 60% of the Zn, and 78% of the Co. Removal of Mn in the LLB improved with time and was likely to be through microbial and autocatalytic oxidation on MnO2.

Key implications of this work relevant to mine management:

- Passive treatment should be considered even if Mn concentrations are very high.
- Different types of steel slag may perform differently.
- Limestone can be effective at removing very high Mn and can improve with time.
- VFRs can remove Fe from neutral drainage prior to treatment for trace elements.

Key reference:

Trumm D, Christenson H, Pope J, Watson K, Mason K, Squire R, McDonald G, Mazzetti A 2017. Passive treatment of Fe and Mn using vertical flow reactors, limestone leaching beds, and slag leaching beds, Waihi Gold, AusIMM New Zealand Branch Conference, Christchurch. Pp. 334–343.

As noted in section 3.6.6, management activities (e.g. overburden management, rehabilitation activities) will be insufficient to completely mitigate suspended sediment, and treatment (typically through the use of settling ponds will be required (Figure 31).

Figure 31. **Sediment control pond at Waihī mine. Water quality is continuously monitored, and when water chemistry (pH, EC and turbidity) is appropriate, this pond can be discharged directly. At other times it reports to the water treatment plant.**

4.4.1 Monitoring of treatment systems

Active treatment systems

For an active treatment system, both inlet and outlet water should be monitored regularly. If inlet water chemistry or flow rate changes, then changes can be made to the system immediately to ensure 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
- any other metals or metalloids of concern in the mine drainage
- dissolved oxygen.

The 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.

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 the 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 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
- 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 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:

- the ability of the system to continuously treat to acceptable limits
- the sensitivity of the receiving environment to changes in water chemistry
- the level of treatment by the system
- the risk of failure of the treatment system
- site access, and the ability of the 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 should also include monitoring of the inlet water for sulphate, and the outlet water for sulphides, dissolved organic carbon, total nitrogen, nitrate-N + nitrite-N, total phosphorus, and total biochemical oxygen demand.

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. Preferential pathways may develop due to compaction of treatment media or a build-up of precipitates that can clog passageways, reducing permeability and treatment effectiveness. Should this occur, rapid flushing of the system may be required to remove precipitates and restore permeability.

For systems relying on sulphate reduction, operation and maintenance are primarily focused on ensuring that reducing conditions are maintained. Colder winter conditions, a substantial drop in sulphate concentrations, or a drop in hydraulic residence time can all effect sulphate reduction and must be monitored. If these conditions cannot be controlled, organic supplements to assist microbial activity should be considered. A study is underway by CMER to determine how biological sulphate reduction is affected by the addition of a waste product from biodiesel production. Preliminary results show that sulphate reduction is increased by at least 10 to 20 times (Christenson et al. 2017, 2018).

Suspended solids treatment

Effective maintenance of sediment traps and ponds is necessary. Such sediment may be suitable for recycling and reuse for plant growth media. Where treatment systems to reduce high suspended solid loads are in place 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.

4.5 Terrestrial rehabilitation

Rehabilitation at the operational phase will be largely focused on avoiding impacts, salvaging resources for later rehabilitation, and maximising the direct use of soils, rocks and vegetation stripped in front of mining or access roads/drill tracks. Environmental impacts can be avoided by minimising the mine footprint, and this also avoids creating areas that need later rehabilitation. There are usually choices for the location of overburden dumps, access roads, and other ex-pit infrastructure: their location should minimise effects on the highest-value areas, whether these are ecosystems, landscapes or culturally important sites.

Effects are also minimised by ensuring that buffering and protection of adjacent or residual unmined areas are effective (see Case Study 16, p. 117). This is likely to include ongoing monitoring to ensure 'out of bound' areas are not inadvertently degraded (e.g. by vehicles going 'off road' or dumping loads). A focus on avoiding impacts will also prioritise erosion control and minimising the volume of TSS generated, as this reduces the cost of treating dirty water. If soils are used as a cover material, reducing TSS also conserves this valuable material where it contributes to plant growth. The main strategies used to reduce the generation and movement of TSS are discussed in this section.

Rehabilitation during the main operational phase will also focus on keeping track of the volumes and qualities of rehabilitation resources, along with any issues that may inhibit rehabilitation outcomes, such as changes in overburden chemistry or final topography, slow plant growth rates, or the impacts of pest plants and animals. The creation of mine schedules that allow direct use of rehabilitation resources without interim storage (i.e. taking soils and plants straight from stripped areas to final landforms) should be a high priority during the operations phase of mining because it has many benefits: the costs of double handling are avoided, the quality of rehabilitated materials is higher, and outcomes are usually therefore better, leading to shorter closure timeframes, and reduced bonds and environmental liabilities. Direct use of a resource can be the only way to re-establish some native plants, animals, and ecosystems through the use of 'direct transfer' or 'ecosystem translocation' of plants and soils in large sods (see Case Study 14 below).

Case Study 14: Direct transfer of forest: ugly but worthwhile on the West Coast

Direct transfer is a method of rehabilitation used mainly on the West Coast of the South Island by alluvial gold and coal miners to establish native forest and wetland ecosystems. Alluvial gold miners often use front-end loaders or face shovels to transport up to half a metre depth of intact soils, together with plants, as 1 to 3 m^2 sods from in front of mining to areas ready for rehabilitation. Trees up to 5 m tall are shifted, but this generally has a high proportion of dieback; consistently low dieback can be achieved with plants under about 2 m height, and where rooting depth is shallow enough so that the whole root mass is removed intact.

At this lowland alluvial site, direct transfer of cutover shrubland and low forest had been started on a variety of slopes to at least 20 degrees (Figure C29). The miner had used direct transfer at a previous site and had developed innovations based on that experience:

A thin layer of dredge fines (salvaged from inlet to sediment ponds) was placed over recontoured gravels to form a soft (but trafficable) base for direct transfer that minimised air gaps forming under sods (air gaps can dry out roots).

Taller trees were not felled before stripping (a saving). Toppling of these taller trees was minimised by removing large sods (two per truck) and placing these into pre-prepared depressions/holes as self-supporting clusters, and then placing soil around the edges.

The direct transfer was effective at preventing sediment entering surface waters. Miners enjoyed the challenge of direct transfer and instant result, especially when trees remained upright.

Figure C29 Effective, young direct transfer. Note the high proportion of intact leaf litter, many upright saplings, and soil placed around and between sods to help protect them from drying out and keep them upright.

Key findings relevant to mine rehabilitation:

- Direct transfer is an effective rehabilitation method for native ecosystems because it conserves soils, seedlings, fungi and insects. Direct transfer is also useful to buffer native ecosystems, including protecting them from erosion.

- Results may look ‘messy’ in the short term (1 to 4 years), as plants die, lots of dead wood is exposed, and surfaces are rough, but this ‘mess’ means there are lots of protected microsites in which seedlings can establish, insects can inhabit, and birds can perch.
- Die-back is minimised where drainage is similar in the source and destination sites, the whole root zone is shifted, and soil is placed around sods (and/or sods are placed in depressions) to reduce drying of edges.

Key references:

Department of Conservation 2010. Revegetation of alluvial gold mines: a prescription for the West Coast Tai Poutini. http://www.doc.govt.nz/Documents/about-doc/concessions-and-permits/mineral-exploration/revegetation-alluvial-gold-mines.pdf

Simcock R, Ross C 2017. Chapter 18. Mine rehabilitation in New Zealand: overview and case studies. In: Bolan NS, Kirkham MB, Ok YS eds. Spoil to soil: mine site rehabilitation and revegetation. CRC Press. Pp. 334–357.

Processes to achieve selected rehabilitation outcomes are described in section 5.7 and include rehabilitation to pasture, plantation forestry and native ecosystems. Other land uses are possible, such as cropping and horticulture (viticulture), residential housing, recreation (historical mine relics, mountain biking), and public amenity, but these are not covered in this document.

Maintenance and monitoring are integral components of successful rehabilitation. Most New Zealand mines do not have adequate records of what rehabilitation methods were applied and when, or how they have developed over time. This monitoring can help to inform closure criteria (see Case Study 15).

Case Study 15: The value of rehabilitation monitoring

Few mines regularly collect quantitative data on the performance of rehabilitated areas, yet such data can provide important insights and provide regulators with confidence that native succession trajectories are resilient and predictable. Monitoring of permanent plots at the Globe Progress mine demonstrates the value of long-term monitoring: the information has helped inform closure criteria tailored to the site and supported bond release.

At Globe Progress, the performance of rehabilitated areas is tracked using two methods. One permanently marked 10 × 10 m plot is established in every 1 to 2 ha of rehabilitation planted that year. The plots are mapped and given a unique identifier. All the planted seedlings in each plot are counted, identified (by species), and their height measured. This provides a baseline against which performance is assessed. All the planted seedlings are then remeasured, along with groundcover and adventive seedlings. Together the data will help refine and validate closure criteria of native vegetation dominance, minimum percentage cover, and establishment of adventive seedlings. The number of self-established (adventive) seedlings is increasing over time, and non-native species are declining (Figure C30).

Permanent photo-points were also established for each main rehabilitation area (Figure 31). These have been useful to show overall vegetation development and variation, but as the trees grow tall it is harder to retake some photos; the increasing cost-effectiveness of drones could be valuable for photo-point monitoring.

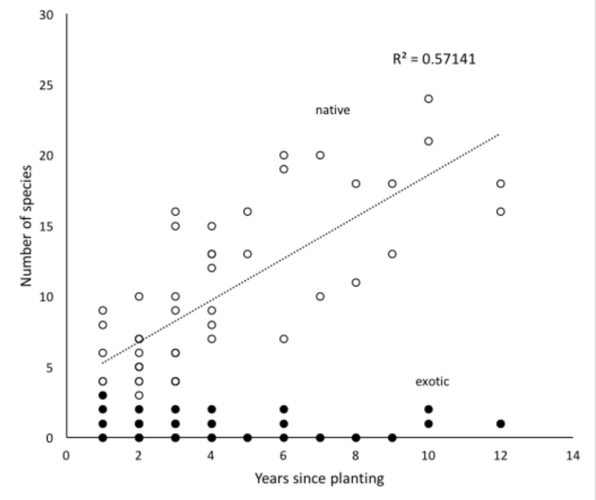

Figure C31. Number of native species (open circles) and non-native species (closed circles) in permanent monitoring plots (Prof. David Norton, University of Canterbury, pers. comm., 2018).

Figure C31. Devil's Overburden backfill slope planted with beech and mānuka trees, 2017.

Key findings relevant to rehabilitation:

- Photo-points are useful to give an overview of rehabilitation progress and the spatial variation of rehabilitation across a landscape. They should be taken at the same time of year, annually, to avoid the effects of seasonal changes in groundcovers, especially grasses and lotus.
- Photo-points using drones may help avoid photo-sites being 'grown out', and could enable rapid assessment of features such as native canopy dominance.
- Permanent plots provide the best in-depth data to inform key closure criteria such as seedling establishment (indicating succession processes) and plant cover. Plant growth rate and survival data are also important and require baseline measures recorded at planting, or plant establishment, against which to compare.

Key references:

Norton DA, Creedy S, Keir D 2013. Substrate modification for enhanced native forest restoration, Reefton. Ecological Management and Restoration 14(2): 147−150.

PAF rock and AMD are hostile to plant growth and prevent rehabilitation where they are severe, which means 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 acid-generating areas 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 relatively limited, but they include:

- revegetation by hydro-mulching or hydro-seeding with pioneer species such as mosses, lichens, fragments of herbs and tolerant low-growing shrubs – ensuring plants above the high wall are entirely species that are desirable (and exclude weeds) and are close to the edge (rather than leaving a cleared strip) helps generate a 'seed rain' to naturally revegetate the high wall over time, in the same way road cuttings throughout the country have revegetated
- holding plant growth media onto the high wall using anchored geotextiles, engineered cages or wood slash, although wood slash is only useful for the lower 1 to 2 m of individual batters as it can be dangerous if unstable
- screening by growing vines, shrubs and trees in suitable plant growth media up from benches, and growing vines or draping plants or groundcovers down from benches (an outstanding species is pōhuehue, *Muehlenbeckia complexa*).

Surface preparation has a substantial impact on potential outcomes, as do slope and aspect. The rougher the surface of the high wall and the lower the moisture stress, the more likely it is that vegetation can establish. This is why south-facing aspects generally achieve a higher and taller cover than north-facing aspects, and why rough surfaces achieve a better cover than smooth surfaces. The height and width of benches and batters can also be selected to optimise rehabilitation outcomes: varying the width and height of benches may help achieve a more natural landform and create areas where plants are better able to establish a dense cover (gentler and shorter batters). When selecting areas for such treatments, match favourable slopes with the most favourable rock (weathered and fractured) and aspect. In England, quarry high walls have been deliberately blasted to create rock screes and talus slopes, which creates a variety of habitat niches that favour rehabilitation in parts of the slope and helps to integrate landforms into a landscape.

A practical and economical 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 such rock may be able to be revegetated by hydroseeding with lime in the mixture.

4.5.1 Erosion and sediment control

Six main strategies are used to reduce the generation and movement of TSS. All should be adopted.

- Keep clean water clean using cut-off drains to separate 'mine water' from clean water

- Minimise the area that generates sediment, and the duration this area is exposed, by 'treating' areas as soon as possible so the surfaces are resistant to erosion. Minimise the area of 'pre-strip', as bare soils are usually high-sediment-yield areas, and establish vegetation and/or mulch or slash on areas being rehabilitated as soon as possible after soils are spread. Use road sheeting that has a low fines component and high competency, and/or refresh such sheeting regularly in ex-pit areas. Delay stripping until soil conditions are drier, so less damage occurs (the earthworks season for soil/vegetation stripping is shorter than for overburden stripping – often October to March for soil stripping).
- Control slope length and slope steepness to minimise erosion potential: steep slopes and long slopes have much higher erosion risk than short, gentle slopes.
- Minimise catchment sizes in areas vulnerable to erosion – large catchments allow water to concentrate. Creating many small catchments and detaining sediment close to source in many small devices will help reduce sediment losses and preserve the capacity of larger devices.
- Create a surface and subsurface with adequate roughness – this roughness will differ depending on the post-mining land use. An easy way to achieve roughness is stripping plants and wood with soil, and even rocks. Another method is to cover loose soils with 'slash' (felled trees and stumps) (Figure 32). Many mines create surfaces that would perform better with greater attention paid to accentuating roughness: roughness of overburden backfills allows soils to bind and key-in, and roots to strengthen the interface. A smooth interface is much more vulnerable to slumping and tunnelling, especially where the permeability of the subsurface layer is much lower than that of the overlying soils, because water will move laterally across the top of the subsurface layer.
- Design adequate sediment capturing and treatment volumes, with suitable access to allow desilting. This includes along roads, where road sediment is likely to contain weeds, so sediment needs to be disposed of in areas where weeds can be controlled.

Figure 32. Slash spread across backfilled slopes reduces the risk of erosion.

Receiving water standards for TSS vary depending on the receiving environment. Many West Coast streams have naturally very low levels of TSS, even during storm events. Water quality requirements in such sites must focus on prevention of sediment at source. Some regions reference erosion and sediment control guidelines; for example, 'TP90' (Technical Publication 90, Auckland Regional Council 1999), which has been updated by GD05 (Guideline Document 2016/005, Leersnyder et al. 2016) or the Environment Canterbury equivalent[26]).

Mine sites usually have a greater range of materials suitable for creating surfaces resistant to erosion than typical earthworks sites and can use them in atypical ways. For example, vegetation on roadsides and in urban earthworks is typically stripped and chipped, whereas on mine sites such vegetation is more valuable. Examples include:

- large logs (which provide greater erosion control and surface roughness, as long as they are relatively short, and/or are placed to avoid concentrating runoff)
- intact soil/vegetation sods (direct transfer), which can be placed as instant living filters to protect sensitive areas such as water courses, and at the tops of slopes (i.e. preventing water flowing over the edges of backfills)

[26] http://ecan.govt.nz/publications/General/FullErosionandSedimentControlGuideline.pdf

- mixed soil and plant slash, particularly for areas being rehabilitated to native ecosystems.

Mines generally also have access to aggregate and boulders, both of which can be useful to temporarily or permanently 'lock down' slopes. They are particularly useful for stabilising short, very steep fill slopes along roads and cut-off drains. Mines are generally less likely to use straw or hay mulching (with the exception of mines in farmland), or erosion control fabrics. Both straw and hay are likely sources of weeds in native ecosystems, although hay can be useful to rehabilitate agricultural land, and it is very effective when 'crimped' into the surface by the tracks of a bulldozer.

Guidance is provided elsewhere on methods to create stable stream channels and floodplains of adequate capacity (this includes ensuring flood flows can spread out so that stream energy is reduced at high flows) and designing stream pools/riffles/meanders and flood plains that can support the expected flow energy. Temporary surface stabilisation may be used until plants are mature enough to lock down surfaces.

4.5.2 Resources for rehabilitation

Figure 34 provides an overview of the general process to identify and salvage materials for rehabilitation. It is also important to identify where resources will be stored if they are not used immediately during progressive mine rehabilitation. The value of rehabilitation resources is maximised by using 'no-stockpiling' techniques, whereby materials are taken directly to areas ready for final rehabilitation, such as in direct transfer. Mine scheduling needs to allow time and access to the vegetation ahead of bulk stripping, and mine planning needs to allow for designated stockpile space, where materials are kept separate from overburden, and access to these stockpiles is retained so that they can be used.

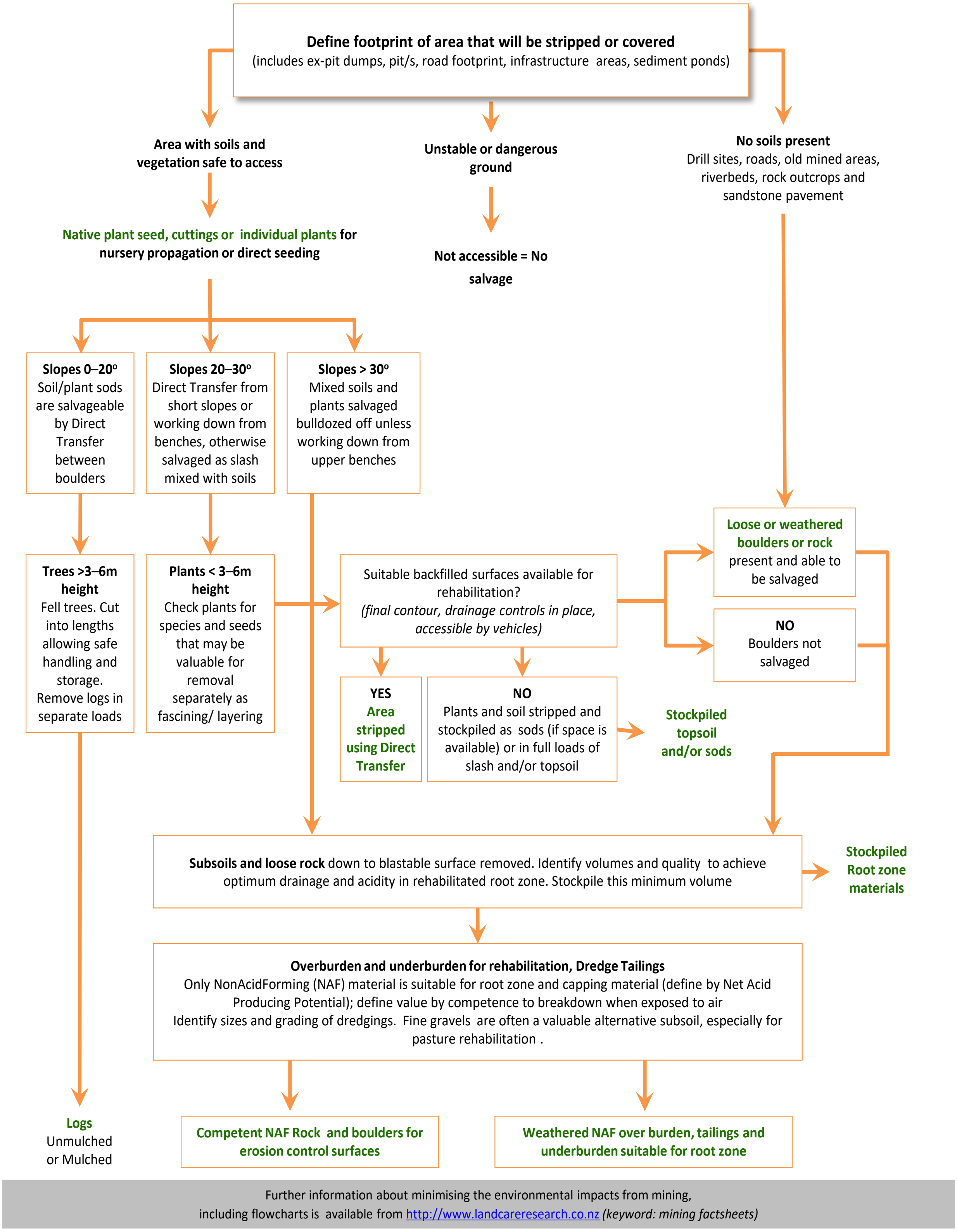

Figure 33. **Overview of the typical steps to identify and salvage materials for rehabilitation. The height of trees that need to be felled and the steepness of slopes that are accessible for different methods is indicative only, as it depends on the machinery and access.** Note: green text highlights the resources for rehabilitation.

4.5.3 Monitoring land rehabilitation

Day-to-day monitoring of rehabilitation progress must be undertaken by the mining company to fulfil mine safety requirements. This monitoring includes assessing opportunities to minimise the mine footprint, keeping track of the resources that can be used for rehabilitation, and ensuring PAF materials don't get placed in the root zone. Where rehabilitation of the final landform has been undertaken, the monitoring described in section 4.6.3 is best used to determine rehabilitation performance.

Mine footprint minimisation

Opportunities to minimise the footprint of a mine and its infrastructure need to be continually assessed as each new drill track, cut-off drain, and ex-pit feature (e.g. overburden dump, sediment pond) is designed and constructed. Information from the first rehabilitated areas needs to inform design (e.g. the high wall or backfill slope lengths and angles with different roughnesses and erosion-control covers). Steeper faces, if stable, generally help reduce the disturbed footprint. During mine opening, the edges of stripped or filled areas should have been treated to minimise edge effects. These edges should be monitored for their condition and the results used to further reduce impacts in subsequent areas being cleared. Footprint minimisation should include monitoring of ‘out of bound’ areas to ensure these are not inadvertently degraded; for example, by vehicles going off road or dumping loads.

Case Study 16: Practices to minimise edge effects in the Brunner coal measures

Edge effects are changes that occur along the edge or boundary of different habitats. In mining, clearing plants/soils, shifting surface waters and ground waters, and new landforms create new, often distinctive edge effects. Edge effects vary depending on the ecosystem and action. An edge cleared into forest can be expected to change the temperature, humidity, light, and wind effects at least 50 m from the cleared boundary. These changes mean some plants and animals die, especially those adapted to sheltered, humid conditions.

A cascade of impact can be created over time, especially where adjacent trees die back or are destabilised. Edges are also vulnerable to weed invasion, especially where bare soils are created. Edge effects mean mine access roads, cut-off drains, drill tracks, and other areas with small cleared footprints can have large environmental effects. Edge effects are exacerbated where the new edge acts as a barrier to movement, in a similar way that a single fish passage block from a road culvert or chemical barrier can stop movement in streams.

Edge effects should therefore be minimised. The benefits of minimising edge effects include a reduction in the effective mine footprint, a reduction in the cost and risk of weeds, and an increase in regeneration sources from adjacent areas to rehabilitated areas.

Three main methods can be used to minimise edge effects.

- Locate the disturbance to avoid ecosystems and species that are most vulnerable to edge effects (small-stature and dense plants are usually much more resilient than tall forest). This is very effective for drill sites in the Brunner coal measures because the ecosystems are a complex mosaic ranging from forest to herbfield, so moving a site 10 to 50 m can change the ecosystem.
- Constrain the area of disturbance and prevent spilling of soil and slash into adjacent undisturbed sites. Figure C32 shows boulders, sediment fences and erosion ‘logs’ placed to retain soils on the uphill mine side and to protect the adjacent alpine herbfield. Real logs and stumps can also be effective retainers in the medium term.
- Quickly stabilise and revegetate edges to help recreate a dense buffer. A variety of methods have been used on the Brunner coal measures: the most effective is direct transfer of intact sods, shown in Figures C33. Figure 34 also shows a range of other methods, such as erosion blankets, and planting and stimulation of mosses through fertilisation.

Figure C32. A range of soil-retaining and buffer-creating practices to protect adjacent ecosystems along a steep ridgeline, Stockton Mine. A: Barrier of large boulders; B: planted seedlings; C: Erosion ‘log’; D: Sediment/silt fence. Slow-release fertiliser was used to stimulate cover of mosses (which were already present, but sparse).

Figure C33. Construction of the Cypress Mine haul road. The steep road batters are covered with sods of tussock, mānuka and wire-rush. These immediately provide sediment control and buffering.

Key findings relevant to mine rehabilitation:

- Edge effects can greatly increase the area affected by mining, especially in forested, sub-alpine and native ecosystems.
- Minimising edges is important to reduce avoidable impacts, and it can also benefit on-site rehabilitation.
- Minimise edge effects by stopping soil and rock moving from stripped areas into undisturbed areas. Use bunds, large rocks and logs, and direct transfer sods, and cover bare areas with erosion netting or mats or slash/branches until plants establish.

Figure C34. Sods of mānuka and boulders placed to buffer adjacent habitat of *Powelliphanta augustus*, Stockton mine.

Tracking rehabilitation resources

As mining continues, the quality, volume and location of rehabilitation resources need to be monitored and audited. The quality of rehabilitation resources may improve over time; for example, seedlings may regenerate in stockpiles, and these may be useful to salvage prior to bulk stockpile use. It is also common for weeds to germinate in stockpiles, and the identification and killing of these before they set seed (and create a seed bank) will help retain the quality of the soil resource and reduce future costs. For example, if problem weeds (say gorse) are only found in one part of the stockpile, then placing this contaminated soil in an easily accessible area, away from watercourses, instead of spreading it across a wide area, reduces the costs of weed control. The identification of site-specific weeds and preventing the establishment and spread of new weeds is particularly important for relatively weed-free areas (see Case Study 17 below).

When considering if a plant is likely to be a weed of native ecosystems, ask the following questions – if any answers are no, then exclude and/or control the plant:

- Can disturbance be excluded (disturbance is likely to favour the weed regenerating)?
- Is the weed intolerant of shade as a seedling (i.e. it won't re-establish under canopy)?
- Will the native canopy over-top the mature weed canopy?
- Has the site low drought stress (including aspects, location, substrates)?

Weed control is arguably more important if rehabilitation is using adventive (natural) seeding from remnant adjacent ecosystems (or patches of direct transfer); for example, where a low rate of browntop grass is seeded onto a cut batter, or just mānuka and beech are planted over a mine site.

Case Study 17: Weed management at Strongman: proactive management pays dividends

Weeds are plants that are unwanted, often because they smother and displace productive or desirable plants, whether pasture on farmland or native seedlings of forests or wetlands. Many small West Coast sites rapidly establish a gorse, broom and/or Himalayan honeysuckle cover. Gorse and broom have a very long-lived seed bank and are nitrogen fixers; fleshy-fruited weeds such as blackberry, barberry and Himalayan honeysuckle are brought in by birds, while weeds such as pampas blow in on the wind. Most of these weeds grow faster than native species, and many increase the risk of fire and future disturbance. Gorse is not a substitute for native species. Gorse-covered areas are more vulnerable to fire, have lower native plant diversity, are more vulnerable to bird-dispersed woody weeds such as barberry, and probably lead to different native forest outcomes. Where weeds are sparse, preventing their establishment saves time and money, and reduces rehabilitation risk.

Figure C35. Planted overburden dump after about 5 years, Strongman open-cast, showing adjacent forest, which supplied wind-blown fern and kāmahi, which enriched rehabilitated areas.

Strongman open-cast mine had few weeds. Preventing establishment of new weeds was a key strategy to speed native plant regeneration to native forest post-mining. In the absence of weeds, 'free' natural regeneration occurred from nearby forest into the rehabilitated areas, which had a soil cover and were sown with about 3 kg/ha browntop grass. Plant numbers over 10 cm tall increased from 4,700/ha at planting to over 11,000 after 3 to 6 years and over 50,000 in 7 to 10 years. This shortened the recovery time, reduced risk, and increased native plant diversity. Regeneration included kāmahi (an important canopy tree that was not planted), ferns, and many seedlings of planted nursery seedlings such as koromiko, mānuka and toetoe (Figure C35).

Figure C36. The weed tree, silver birch, growing from planted hebe, Strongman 2004.

Three actions were used to prevent weeds establishing. First, the main pathways of weeds were identified. These were, in order:

- road gravels (gorse, broom, and others) and vehicles
- road-side weeds on adjacent public land (pampas, cotoneaster, foxglove, fleabane)
- trial erosion control groundcovers (lotus)
- historical dumping (mimosa, agapanthus, tiger lilies, Himalayan balsam)
- 'hitch-hikers' in nursery-seedlings (e.g. silver birch, Figure C36).

Second, annual surveys of high-risk areas such as roadsides, gravel stockpiles, sediment ponds, and rehabilitated areas were done. During the survey, isolated weeds were killed immediately using a 'cut and paste' technique, and any flowers or seeding parts were bagged and disposed of where they could not spread. GPS coordinates were captured to allow efficient follow-up. Larger infestations were included in routine spraying of roadsides. These proactive practices prevented a build-up of weeds that would threaten closure criteria and increase fire risk, and allowed native plants to self-establish.

Finally, a high cover of native species was established over 4 to 5 years, which removed favourable sites for most of the weed species.

Key findings relevant to mine rehabilitation:

- Weeds are a common reason for rehabilitation failure for all land uses.
- At Strongman, preventing weed competition combined with favourable surfaces allowed a continual influx of desirable native seedlings to establish over 10 years.

- Detection and early treatment of weeds was cost-effective. This included annual surveillance, and prompt weed control to prevent weed spread. The access road needed regular sweeps.
- The species of weeds that need controlling varies from site to site.

Key references:

Simcock R, Ross C 2017. Chapter 18. Mine rehabilitation in New Zealand: overview and case studies. In: Bolan NS, Kirkham MB, Ok YS eds. Spoil to soil: mine site rehabilitation and revegetation. CRC Press. Pp. 334–357.

Williams P. 2011. Secondary succession through non-native dicotyledonous woody plants in New Zealand. http://www.science.canterbury.ac.nz/nzns/issues/vol36-2011/williams.pdf

Wooten DM, McAlpine KG 2013. Predicting native plant succession through woody weeds in New Zealand. DOC Research and Development Series 336. https://www.doc.govt.nz/Documents/science-and-technical/drds336entire.pdf

New resources may include favourable overburdens, or new opportunities to salvage materials due to new machinery, especially if machinery has greater trafficability. However, it is not unusual for soil or slash stockpiles to be 'lost' (i.e. covered with overburden or rendered inaccessible), and therefore unusable. An annual stocktake of rehabilitation resources helps adapt the mine plan and to take remedial action. For example, if topsoils are likely to be in short supply, they can be 'manufactured' from wood chip and overburden, given enough lead time (see Case Study 20, p. 149). At the very least, the minimum required topsoil should be used so that what is present is used wisely.

Identifying potential rehabilitation problems: acid (PAF) in root zones

The early identification of issues that may inhibit rehabilitation outcomes is important and is enabled by monitoring. Common issues are much-steeper-than expected landforms, and impacts of pest plants and animals. The early identification of pest plants allows much more cost-effective action: an old saying is, 'One year's seeding means 7 years' weeding'. In the case of weeds with long-lived seed banks (50-plus years), such as gorse, broom or *Juncus squarrosus*, or weeds that spread rapidly, such as Asiatic knotweed or pampas, failure to achieve early weed control, especially of a new weed, can mean decades of costs.

A critical issue is ensuring PAF is not present in the root zone. This can occur with unexpected changes in overburden chemistry, especially where they influence the predictability of PAF identification. In some cases, a change in rehabilitation method can mitigate mild to moderate PAF conditions; for example, by using surface amendments (an organic material such as mulch or biowaste), changing the plant species used to more tolerant plants (e.g. mānuka), or changing the method of revegetation from planting nursery-raised seedlings to seeding and natural regeneration (see Case Study 18 below).

Case Study 18: Free revegetation and the importance of seed islands

Nature doesn't tolerate a vacuum: sooner or later bare ground is naturally colonised by vegetation. The rate and composition of colonisation depends on the proximity and dispersal modes (bird, wind) of surrounding seed sources. At highly disturbed sites, it also depends on the characteristics of the receiving substrate and landforms.

The Wangaloa coal mine in south Otago is a legacy example; the miner walked away when the economic benefits ran out in the 1920s to 1940s. For 40–60 years waste rock lay *in situ* after the last dumped truck loads. Patches of native, secondary forest that were adjacent to two-thirds of the mine site (252 ha) provided a source of kānuka, mānuka, kāmahi, five-finger, and *Coprosma* species. When retrospective rehabilitation was planned in 2002, parts of the oldest waste rock near the site perimeter had 2–3m tall mānuka and kānuka trees. Large pine trees and much gorse had also established. Large-scale bulldozing was used to remove (and bury) the pines and gorse, but some of the bigger patches of mānuka and kānuka (c. 3–5 m diameter) were retained during earthworks. In hindsight that is where a technical approach should have stopped for two-thirds of the waste rock area, because clearing the gorse and pines inadvertently opened up a colonisation window for light-loving mānuka and kānuka at the eastern and western ends of the overburden footprint, where no loess was present and soil conditions were hostile (see Case Study 8). Instead, tens of thousands of native nursery plants were planted, then replanted in these areas of mass nursery seedling mortality. The potential of the waste rock to support natural successional processes was not recognised in rehabilitation plans.

Figure C37. Naturally colonising mānuka and kānuka are encroaching on overburden showing poor planting success.

Mānuka and kānuka rapidly expanded from the patches of pre-rehabilitation kānuka and mānuka scrub; 10 years after the major revegetation works, seedlings that had originally naturally established outside the patches were up to 4m⁺ tall. Their growth rates, and moderate density (average four stems per square metre) smothered most of the stunted and struggling nursery-raised plantings (Figures C37 and C38). Within 7 years, mosses, lichens, ferns, and other native tree and shrub species were also establishing in the series of increasing mānuka and kānuka interlocking islands (Rufaut & Craw 2010). In particular, kāmahi has colonised; this is a very long-lived forest canopy species, so it is of great value and importance to long-term outcomes.

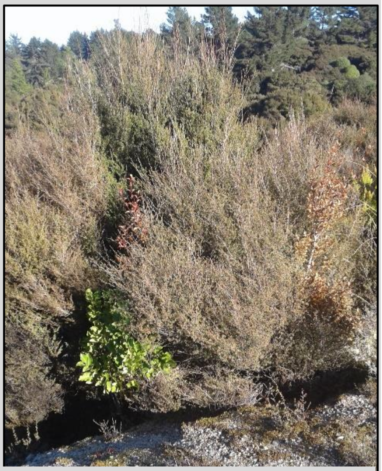

Figure C38. Planted broadleaf overgrown by naturally established kānuka and mānuka.

One of the main reasons naturally colonised mānuka and kānuka have been so successful is that the geochemistry of raw overburden (with low fertility and high acidity) has supressed the establishment and growth of tall, rank grasses and flat-weeds (Rufaut et. al. 2015). This inadvertent retention of bare ground during rehabilitation (because soil amendment treatments were localised and ineffectual) has initiated natural forest succession processes favouring native species (Figure C39.

Figure C39. Natural forest succession under mānuka and kānuka on coal-rich overburden.

Key findings relevant to mine rehabilitation:

- The natural succession potential should be identified for native species on all final surfaces.
- Natural succession can be encouraged/assisted by creating suitable (low N, P and inhospitable pH) stable surfaces with high light levels to minimise failure risks.
- Retention of within-site ‘islands’ and ‘peninsulas’ of native plants (no matter how small) promotes natural succession and diversity.

- Natural succession is cost-effective but also ecologically effective for promoting native cover and diversity when weed threats are low or can be managed. However, without trials it can be a less certain outcome, and so is perceived as higher risk.

Key references:

Rufaut CG, Craw D 2010. Geoecology of ecosystem recovery at an inactive coal mine site, New Zealand. Environmental Earth Sciences 60: 1425−1437.

Rufaut CG, Craw D, Foley A 2015. Mitigation of acid mine drainage via a revegetation programme in a closed coal mine in southern New Zealand. Mine Water and the Environment 34: 464−477.

4.6 External/compliance monitoring

4.6.1 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 3.3.2) 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 final downstream water quality, and include the following important considerations.

- Samples of undiluted mine drainage or leachate from tailings impoundments should be collected as close as possible to the source of the seep or the entrance to an adit.
- Downstream environment monitoring points should always be 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 measuring physicochemical parameters across the stream from the sample collection point. If there is no change in physicochemical properties across the stream, and they are consistent over time, then the sampling point is acceptable.
- Monitoring points 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 included in areas where mine development will not occur. Ideally, these 'control site' monitoring points are ones that have been used during baseline surveys (section 3.3). 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 data sets from multiple locations.
- Changes in the partitioning of trace elements between dissolved species and suspended particulate material will occur with changes in water chemistry.
- 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 (section 3.3).

What to monitor

There are many alternative monitoring strategies that can be used 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 aim is to minimise the number of analytical parameters to reduce costs and provide maximum useful information.

An example of a three-tier monitoring system, with rationale for sample types

- Tier 1: The minimum useful analytical suite is turbidity, pH, electrical conductivity (EC), and flow, and potentially specific trace elements of particular concern (e.g. where concentrations are expected to be close to consented limits). This approach assumes that pH and EC are suitable proxies to identify variations in stream chemistry caused by changes in flow volume or to the 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 for 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, and while dissolved species are more bioavailable and therefore more toxic, contaminated suspended particulate matter can accumulate in bed sediment, leading to longer-term contamination of the waterway. For example, many trace elements adsorb to suspended particulate material but desorb if chemical conditions change to favour their release. The relationship between suspended particulates and dissolved species is an active area of research and is likely to be site specific. Conditions specified in resource consent documents should take into account 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 or 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 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 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 costs decrease and the availability of portable analytical capability improves.

4.6.2 Aquatic biological monitoring

Ongoing monitoring should be undertaken to detect impacts occurring during and after active mining. Sampling should include both control/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. These might be 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, because it may take some time for fauna to re-colonise habitats.

4.6.3 Terrestrial rehabilitation

Often only small areas of land are rehabilitated during the operational phase of mining. Rehabilitation monitoring during mine operations generally focuses on the performance of these early areas as guides to the outcomes of later, larger-scale rehabilitation. This is combined with an assessment of other factors that influence the cost and risk of rehabilitation, including the efficacy of 'on land' short-term sediment control, and actual and predicted rehabilitation resources (their quality and quantity). These factors should be identified at mine planning, and updated with information gained during mine operations and with each annual mine plan.

These data are likely to influence the bond that is set if the mine has an annual review. Even the most basic bonds at small mines are usually linked to the area that has been stripped or impacted, with some dispensation for the area that has been rehabilitated.

Monitoring of terrestrial rehabilitation during mining will typically involve assessing the following indicators of success.

- *Implementation of avoidance, the first step in the mitigation hierarchy:* assess the edges of the mine site. There should be no unconfined side casting or unnecessary damage to edges, and permanent edges should be 'sharp' and buffered. Have techniques to reduce the footprint been adopted? Has moveable mine infrastructure been located on the lowest-value ecosystems (e.g. sediment ponds, roads and stockpiles)?
- *Records of compliance with wildlife permits, particularly for salvage and translocation*: the areas where any wildlife is placed need to be recorded to avoid future disturbance and to allow suitable management (e.g. pest control).
- *Recovery and suitable stockpiling of rehabilitation resources*: the volume and quality of topsoil (usually mixed with plants) and root zone layers are usually priority resources. Logs and specific competent rocks are also often important resources, as are intact plant-soil sods. The stockpiled resources should have an adequate volume to cover the stripped site footprint, excluding areas that will be permanently flooded, permanent roads/ races, or other agreed areas. It is common to have a soil deficit, but a soil deficit increases the risk of inadequate vegetation outcomes. The presence of pest plants in stockpiles also generally increases the risk of achieving low-maintenance rehabilitation.
- *Use of direct transfer of soil and plant resources:* if a mine is conditional on a minimum area and/or type of direct transfer, the mine schedule must allow for salvage and placement of this material by suitable machinery. Sometimes 'direct transfer' material is temporarily stored, and in this case the condition of the material will be assessed.
- *Identification and suitable management of resources hostile to rehabilitation, particularly any acid-generating material*: the combination of a deficit of NAF materials and acid-generating material should ring alarms, as should the presence of any areas of acid-generating rock that are unable to be accessed, covered or backfilled, such as PAF in high walls that are permanent (i.e. not planned for backfill with NAF overburden).
- *Topography of final surfaces and volume of suitable materials available (and accessible) to create the final surface indicated in the consented concept plan*: bulk earthworks are costly, as is reshaping of dumps, particularly if materials need to be transported uphill to cover high walls or fill pits, or to construct ridgelines.
- *Adequate sediment control practices on final surfaces*: there should be little evidence of rill erosion, and sediment should be captured within the site in concave surfaces or on benches (if present – check drainage channels are not scoured or blocked). Sediment control on temporary surfaces is also useful.
- *Revegetated areas on final surfaces*: plant species, survival, condition and growth are usually indicators of adequacy of root zone depth and quality, but check for evidence of animal browse. Plant species differ in their palatability, so browsed plants can indicate stock access or pest animal pressure. Plants also differ in their tolerance of exposure, seen as tip die-off or leaf shredding in flax and cabbage tree, or collar-rock for nursery-grown plants that are poorly anchored. Excavating the root zone around healthy and unhealthy plants can be particularly informative.
- *Review rehabilitated areas against closure conditions:* is it likely that rehabilitation will comply with closure, or are remedial actions required? For example, will a minimum specified density of trees survive? Will a maximum weed cover be reached? Does the final landform blend with adjacent landscape visually? Can the soil depths being used be maintained across the whole site? For example, does the depth of topsoil allow a farmer to cultivate without mixing in subsoil? Is the aggregate placed on races of suitable size to avoid laming animals? Are fences stock proof and located to allow grazing access with agreed stock water reticulation?

Ex-pit and access roads are important areas to assess because they include the first to be revegetated. The stability and revegetation of cut-and-fill batters and pest plant colonisation and management (particularly any flowering or seeding weeds) should be assessed. Roads in native vegetation should be able to demonstrate the use of direct transfer, techniques to avoid high-value ecosystems, and ways runoff/sediment movement has been managed. Growth rates on road-fill batters are commonly faster than on bulk earth-worked areas because slopes on road batters are typically short, sites are more sheltered/protected, and soil has not been stockpiled, or has been stockpiled for only short periods, so natural plant regeneration occurs.

A field-based tool (score card) to assess rehabilitation progress towards mine 'closure' has been developed to assist mining companies ensure that rehabilitation closure objectives are met in a cost-effective manner. At closure, a mine site should typically have reached a condition at which it can be returned to a landowner, and the majority of any rehabilitation bonds are returned. Sites should nearly always be geotechnically stable, with minimal surface erosion. Where vegetation is desired[27], a suitable root zone and topography that are capable of sustaining the agreed plants or ecosystems should be in place. This plant cover may, however, need specific ongoing maintenance to reach or maintain its productivity and/or develop into the agreed long-term post-closure condition. For many areas, including pasture, plantation or native ecosystems, this may be similar to reference or undisturbed ecosystems.

Although the score card focuses on closure, it is designed to help identify the positive and negative impacts of actions taken during and prior to revegetation. The objectives in developing the score card were:

- to cross-reference closure criteria (commonly used in resource consents and access arrangements) and long-term success criteria
- to promote the mitigation hierarchy (i.e. avoid – remedy – mitigate under the RMA) because damage avoided/minimised is nearly always preferred to rehabilitation, as long as adequate space is available to achieve water treatment and stockpiling of resources, such as soil for rehabilitation
- to be useful for small to medium sites and suitable for use by people with general ecological /land-use knowledge, not specialists (large sites will often have a scale of impact that justifies a more complex approach developed by in-house or consultant specialists)
- to be able to be applied over several hours using readily available information
- to be applicable throughout the mining life-cycle, not just near closure, to indicate future likely risks and success, which means it includes four stages of rehabilitation: landforms, root zone/surfaces, initial vegetation, and sustained plant cover.

Two additional criteria are included for native ecosystems:

- to enable actions and issues that would adversely impact reaching the desired closure conditions to be identified early, to be assessed in terms of risk and impact, and to be remedied where necessary with specified 'management controls' – a challenge was to integrate with cultural values and mātauranga Māori
- to cross-reference international standards, such as the Society for Ecological Restoration score card (six classes of attributes: controlling threats, physical conditions, species composition, community structure, ecosystem function, and external changes).

Using the score card

Using the score card involves six steps (A to F, Table 5). Each step has a range of criteria that are common to most mine sites. Experience has shown these criteria are useful indicators of rehabilitation performance. Each criterion is assigned a low, medium, high or extreme environmental risk ranking, based on the likelihood (probability) of it occurring, and the severity of the consequence. This approach aligns with New Zealand and Australian Health and Safety risk assessment, which is familiar to nearly all mine sites and many workplaces.

Table 5. The six steps in the proposed rehabilitation score card

Step	Criterion
A. Define rehabilitation objectives	Identify reference site, if relevant. Define closure criteria (short-term criteria for bond release). Cross-reference specific consent conditions.
B. Assess 'avoidance'	Assess mine edges, tracks and cut-off drains, watercourses, the management of high-value and/or vulnerable areas.
C. Assess criteria that underpin rehabilitation success at all sites	1. Safe stable **topography** that creates suitable drainage conditions and erosion controls for identified land use/species. 2. Chemically and physically favourable surrogate **root zone.** 3. Dense, erosion-resistant and weed-resistant **plant cover.** 4. **Sustainable biomass and soil recovery.**

[27] In some cases vegetation is not required to be established across a site. For example, alluvial gold mines may create farm infrastructure such as stand-off pads and races, and some coal mines create areas covered with a high proportion of boulders and rock, sometime for erosion control or to create ecosystems that naturally have a high proportion of rock cover.

	5. Actual and potential **pest plant and animal** pressures.
D. Assess additional criteria for native ecosystems	6. **Natural regeneration** indicators. 7. Replacement of wood/logs and **habitat features.** 8. **Connectivity** across the site and landscape.
E. Assess non-ecological closure requirements	Checklist of structural requirements (e.g. removal of plant, roads, filling of sediment ponds, fencing, historical feature reinstatement).
F. Assess potential for changes	Checklist of indicators of changed risk profile.

Step A confirms the reference or baseline ecosystem – whether pasture or native forest or wetland – that has been agreed as the long-term outcome. For pasture this may be the closure condition, and this can be achieved in the short term. It is really useful if a reference ecosystem is physically present near the site. For example, at Waihī's Martha gold mine, a key rehabilitation standard is maintaining pasture productivity (as kilograms of dry matter per hectare) equivalent to that of an unmined area. Having a 'real' reference system helps physically assess rehabilitated areas (see the fact sheet 'Planning and agreeing rehabilitation to pasture',[28] which has a flow chart that covers five paddock-scale and four farm-scale criteria). In some cases agricultural productivity is improved, as in some alluvial gold mines in poorly drained, low-fertility pasture.

Unlike for pasture, the reference condition for a native ecosystem is highly unlikely to be met at mine closure, because revegetation usually needs to start by establishing a narrow range of native 'seral' species that can establish into the exposed conditions and bare soils of a bare mine site. Also, native shrubland and forests may take decades to centuries to develop the height and structural complexity present before mining. Hence the practical guide *Revegetation of Alluvial Gold Mines A prescription for the West Coast Tai Poutini* (DOC 2010) indicates that success (closure) is achieved when 'a core of healthy fast-growing native shrubs and trees are established, that can be left without further human assistance to aid development of the site to a complete indigenous plant cover'. Further, the native seedlings will 'exhibit positive growth ... have foliage of a normal healthy cover ... and not be suppressed by weeds', but natural regeneration through (non-native) weeds is accepted. However, it is still very useful to identify the ultimate reference or baseline site, as this helps to identify the soils and landscape conditions that underpin the desired long-term native ecosystems.

Step B is to assess the extent to which avoidance of impacts is achieved. The assessment focuses on site margins, drains, access roads, drill tracks, sediment ponds and watercourses. The aim is to reinforce practices that avoid the need for, expense of, and risk to rehabilitation, and prioritise protection of high-value or highly vulnerable ecosystems or landscapes. Currently four features are assessed in this criterion:

- impact along mine edges, whereby the condition and extent of buffer zones are assessed – these are areas that are not covered/stripped but are impacted (dewatering, wind causing die-back, weed impacts) but may need active rehabilitation
- impact of tracks and cut-off drains, taking into account practices are used to limit their impacts
- impact on watercourses, usually assessed by comparing upstream and downstream of the mine site and its discharges
- impact on unmined, high-ecological value / highly ecological vulnerable areas (e.g. evidence of physical protection).

Step C. Research and practice at mines rehabilitated to pasture over 30 years has identified four factors or criteria underpinning success:

- creating a safe, stable topography that strikes a balance between gradients steep enough to ensure adequate drainage yet gentle (and short) enough to resist erosion (e.g. Connolly et al. 1981; Gregg et al. 2003)
- creating a chemically and physically favourable surrogate soil profile, which is usually topsoil over suitable overburden (Ross & Widdowson 1985; Mew & Ross 1991)
- rapidly establishing a dense vegetative cover to protect surfaces from erosion
- maintaining pasture biomass that rebuilds organic matter and minimises weed competition by managing fertilisers, soil acidity, and grazing pressure.

Assessment of these four factors forms the core of the score card for pastoral rehabilitation (Step C in Table 5). At many pasture sites these closure conditions can be achieved in less than 5 years given an adequate maintenance regime.

Within each of the criteria the proposed score card identifies conditions that equate to a high probability of favourable, or unfavourable, outcomes. For example, assessment of the volume/depth and quality of salvaged topsoil is prioritised, because in most cases re-constructed soil profiles that use salvaged topsoil have the highest productivity in the short and medium term, the greatest resilience to variation in climate variation and maintenance, and require the least ongoing inputs. In most cases, mixed

[28] https://www.landcareresearch.co.nz/__data/assets/pdf_file/0005/76892/FS6-Pasture-Farm-Rehab.pdf

topsoil/subsoils and fine-textured soils on gentle topography are most vulnerable to compaction, structural and biological degradation, and poor outcomes.

An assessment of the pest plant and animal pressures and the vulnerability of the site to these pest plants and animals is important for most post-mining land uses. The pest plant criterion is important for pastoral areas; the pest animal criterion is only relevant for some pastures (e.g. West Coast sites affected by pasture-eating beetles), but is critical for many native ecosystems that are vulnerable to browsing hares, goats, possums, and deer, even at canopy closure. Browse can reduce the survival and growth of palatable native species.

The adverse impacts of browsing animals were measured by comparing fenced and unfenced treatments at Giles Creek, where broadleaf (*Griselinia littoralis*) and karamū (*Coprosma robusta*) were severely browsed by deer (Langer et al. 1999), and monitoring in permanent transects at Strongman mine, where tutu (*Coriaria arborea*) was eliminated from most monitored transects sites over 6 years. At Tui mine, karamū, wineberry (*Aristotelia serrata*), and koromiko (*Hebe stricta* syn *Veronica stricta*) were removed over 3 years by goats. At Stockton mine, growth of *Coprosma propinqua* and broad-leafed tussock (*Chionochloa conspicua*) adjacent to fenced plots was suppressed by hares. Mew et al. (1997) also reported native seedlings browsed by stock where fences were ineffective in West Coast mine sites.

Step D is only applied to native ecosystems. The interim or closure conditions for native ecosystem closure criteria are the sixth to eighth criteria in the rehabilitation score card (Step C in Table 5). Research on rehabilitation to native ecosystems is more recent than that for pasture, being 'virtually non-existent' before 1990 (Gregg et al. 1998). The four main factors underlying success for native ecosystems are similar to those for pasture (e.g. Langer et al. 1999; Rufaut & Craw 2010; Norton et al. 2013; Simcock & Ross 2014). However, the optimal root zone and drainage properties for native ecosystems may differ markedly from those for pasture. Some native ecosystems require impeded drainage, high acidity, and low chemical fertility – high fertility usually increases competition with non-native species, enhances palatability for introduced pests, and negatively impacts symbiotic mycorrhizae. Whereas uniformity of plant growth within paddocks or plantations is highly desirable (and this is underpinned by uniform slopes and root zones), this is not necessarily the case for native ecosystems, where variation helps underpin diversity and resilience. Native forest ecosystems also benefit from the return of wood/logs and slash, unlike pasture sites, and habitat features for specific native animals may be required to enhance re-colonisation and re-establish connectivity across the rehabilitated site.

The sixth factor is 'indicators of natural native regeneration'. This is primarily recorded as seedlings self-establishing within rehabilitated areas, and can be predicted by the density of favourable surface microsites, wood/log or boulder density, and the plant species present in the seedbank, both in any planted areas and in adjacent areas.

Step E is to assess non-ecological closure criteria. For grazed pasture sites this would commonly include reticulated water, fencing (to a specified standard and density), and quality of races or stand-off areas. For ecological areas it may include recovery and treatment of mining heritage, provision of tracks, interpretative signs, and/or carparking.

The rehabilitation score card is intended to be useful for all sites, but small and medium sites in particular, as these mines do not usually have specialist rehabilitation staff, and their compliance with revegetation requirements may be assessed by staff for whom mining is a small component of their work. In this case, there may be little information to underpin the adaptive management required to improve outcomes, so actions that may adversely impact reaching the desired closure conditions may not be detected early enough. Such actions may be driven by short-term financial indicators that incentivise cost controls at the expense of sub-optimal or more expensive medium- to long-term rehabilitation outcomes. Cost control actions that have these unintended consequences include: mining low overburden ratio areas / high grades; delaying rehabilitation; reducing overburden contouring and any backfills that require double handling; and adopting short hauls, which can lead to poor placement of soils and overburden, reducing the blending of acid and non-acid material in backfill, or delaying the covering of acid-generating material. Less valuable (lower grade) areas may be left behind. All these actions may slow the rehabilitation and closing of areas.

Step F is a checklist of factors that influence the likelihood that planned rehabilitation will meet closure criteria. Again, each point on the checklist is rated according to the likelihood and magnitude of risk to rehabilitation success. The factors include:

a. change in the life of the mine (especially if this is decreased), or new areas opened up (creating opportunities), or re-mining of areas (which may impact rehabilitated areas and outcomes, as root zone quality usually decreases with re-handling)

b. changes in mine plan direction, mining process or processing (e.g. tailings change), scale (new infrastructure) or rate

c. change in equipment or the method of stripping root zones or overburden (especially if the site contains acid-producing rocks that must be separately managed)

d. change in the rehabilitation plan – methods/process, scale or pace of rehabilitation, including changes in root zones, landform contours, sediment/erosion control strategies (e.g. use of hydroseeding instead of slash cover), change in suppliers of plants/planting contractors, change in plant species or suppliers of plants

e. non-compliances or design changes that indicate flaws in construction, operation or assumptions (e.g. ex-pit overburden stability, tailings properties)

f. change in land use or operation in areas adjacent to the mine (e.g. clearance or drainage for pasture establishment, reducing connectivity and native propagules, felling of plantations creating a sudden increase in weed pressure)

g. evidence of adaptive management in rehabilitation – adaptive management reduces risk, especially if it is based on monitoring by the mining company and the use of control sites in which standard practice is used, alongside areas where alternative management is applied and recorded.

Each criterion is assigned a low, medium, high or extreme environmental risk ranking based on the likelihood (probability) of it occurring and the severity of the consequence (Table 6). This approach aligns with New Zealand and Australian Health and Safety risk assessment, which is familiar to nearly all mine sites and many workplaces.

Table 6. **Matrix for assessing and scoring each rehabilitation criterion, from low to extreme. Refer to Tables 7 and 8 for descriptions of the different likelihood and consequence categories**

			Consequence		
Likelihood	Insignificant	Minor	Moderate	Major	Catastrophic
Rare	Low	Low	Low	Medium	Medium
Unlikely	Low	Medium	Medium	High	High
Possible	Low	Medium	High	High	Extreme
Likely	Medium	High	High	Extreme	Extreme
Almost Certain	Medium	High	Extreme	Extreme	Extreme

Table 7 provides descriptions of the likelihood categories, with examples, while Table 8 provides a description of the environmental consequences categories, with human health equivalents given to help link to the well-known health and safety assessment. As an impact increases from 'insignificant' to 'catastrophic' the following would be expected:

- increasing resources needed to mitigate environmental impact
- increasing time for recovery and/or reduced ability to fully mitigate impact
- increased severity and/or area impacted.

Table 7. **Categories of likelihood used to assess rehabilitation, with descriptors and examples**

Likelihood	Description	Example
Rare	Highly unlikely but may occur under exceptional circumstances, and less frequently than once every 11+ years	Erosion and scouring of newly topsoiled slopes caused by 1 in 20-year storm
Unlikely	Not expected, but could occur at some stage; has occurred at this or other sites once every 5 to 10 years	Large-scale death of a key plant species due to unusual drought, or frost
Possible	Could occur and has occurred in the past between once every year to once every 5 years (or relate to area disturbed / hours worked)	Dumping or vehicle trafficking on rehabilitated areas
Likely	More frequently than annually; will probably occur in most circumstances	Sediment controls needing maintenance
Almost certain	Continuous or several times a year. There is a history of it happening.	Plant mortality >5% in some areas Weed and pest incursion

Table 8. **Categories of severity of environmental impact used to assess rehabilitation, with descriptors and examples**

Severity	Description
Insignificant	Near miss. On-site controls are adequate. No remediation is required. (Equivalent to a person having an incident with no injury, or a minor injury not requiring first aid treatment)
Minor	Incident. On-site controls are adequate but additional resources are required to remediate. Small area (e.g. inadequate root zones or sheltered sites in <20% of an area, resulting in slow plant growth / localised mortality, limited pest browse remediated by enhanced pest control programme)

	(Equivalent to a first aid injury of a person, no lost time, on-site treatment)
Moderate	Remediation is required and involves substantial resources and/or a longer timeframe (e.g. landslide or water treatment failure discharges sediment to stream/river; significant boundary control failure requiring remedial earthworks and rehabilitation) (Equivalent to a lost-time injury and days off work with off-site medical attention)
Major	Examples: unstable landforms created that require substantial recontouring; more than 20% of site has deficit of favourable root zone material in site with low growth rates or unfavourable overburden; high wall failure with no access to remediate; part of a heritage structure damaged or covered; establishment and seeding of a wind-dispersed weed with long-lived seed bank that can smother native plants and is shade tolerant (Equivalent to a long-term illness or serious injury requiring hospitalisation)
Catastrophic	Serious and/or large on-site-area or downstream impacts cannot be remediated in the medium term due to inadequate physical or financial resources, lack of access, or irreplaceability of ecosystem/species. Examples: inadequate cover material for PAF rock or failure of acidic rock structure activating ARD into surface waterways, tailings dam failure (Equivalent to a fatality or permanent effect on a person's quality of life)

A site is assessed using the likelihood and consequence descriptions (Tables 7 & 8) to yield the risk for the site (Table 6). A low-risk outcome is one where management controls and mitigation actions may not be needed. A medium-risk outcome is one for which mitigation would be needed. A high-risk outcome would require controls and/or planned and budgeted mitigation to reduce risk, and this would be independently reviewed within the mining company. An extreme-risk outcome would also have controls/mitigation in place to reduce risk. If controls were not in place, extreme risk would be expected to trigger additional monitoring or reporting to allow timely intervention and review of the adequacy/efficacy of measures by an independent technical expert.

The risk likelihood/probability and severity of consequences deliver a 'raw' ranking. The basis for applying a low, medium, high or extreme ranking is provided in supporting photos, descriptions and case studies. A second residual or 'mitigated' ranking is then given, which is a ranking after specific management or controls are applied. The specific controls should be recorded, and these should be useful to capture, prioritise, and monitor actions that are important to achieve successful rehabilitation. For example, tracks to drill holes almost always cause adverse environmental effects ('almost certain'), and these would have minor to moderate consequences if sediment controls were in place and rehabilitation effective. However, if the impacted ecosystems were rare and unable to be rehabilitated (e.g. alpine herbfields once the plants and soils are destroyed), and the tracks were outside the Mining Permit or Concession boundary, the drill track may be considered major.

4.7 Economics – bond review

As part of annual reporting processes for mine operations the bond may also be reviewed. Ongoing monitoring can inform the size of bonds, as bonds are usually linked to the 'raw' footprint, or area impacted by the mining operation. They may also be linked to the volume of acid-generating rocks that are excavated but not yet mitigated (through, for example, capping completed landforms). Factors such as progressive rehabilitation limit the exposed area and therefore the maximum size the performance bond can take. This can provide financial incentives for undertaking rehabilitation. New technologies and approaches may also change projected future costs associated with the post-closure component of the bond. However, as noted earlier, this is typically a smaller component of the total bond.

5 Closure

5.1 Introduction

In the context of this guide the closure phase of the mine is when resource extraction ceases and rehabilitation activities are working towards the agreed post-mining environment. Under a staged mine plan, some areas of the mine may be in closure while other areas are still operational. Special attention should be given to areas containing lower-grade ore that might be re-mined if the commodity price increases. A mine is considered to be fully in the closure phase when the mining permit has been surrendered. The focus of the closure phase is on ensuring that treatment and rehabilitation activities are on track to meet agreed post-mining outcomes.

5.2 Engagement

During the closure stage, ongoing engagement is required to identify more specific rehabilitation outcomes and to demonstrate that rehabilitation and treatment activities are tracking towards the agreed closure criteria.

5.2.1 Iwi

The measure of effective engagement with iwi through the life of the mine will often be reflected by the level of interest and desire to be involved in post-mine rehabilitation activities, including shaping the outcomes as kaitiaki. Where they are interested in being directly involved in rehabilitation activities there may be the opportunity for complementary indicator and monitoring approaches to be developed by iwi/hapū alongside other stakeholder groups, whereby ongoing monitoring and data collection can be used to guide, assess, and measure progress towards rehabilitation goals and activities. This may be particularly advantageous to kaitiaki groups, landowners and/or custodians who take on or assume a greater management or guardianship role after area closure.

5.2.2 Other stakeholders

Other stakeholders will also be interested to know the progress of closure activities, and in potentially participating in ongoing monitoring and rehabilitation activities.

5.3 Geochemical studies

Additional geochemical information might be required as closure approaches to demonstrate that the closure strategy will produce the water, overburden, pit wall, and tailings outcomes that are desired. If data on the geochemical stability of waste within the management regime adopted have been collected in a proactive manner throughout operations, then the burden of demonstrating that the closure strategy will be successful will be minimal. However, if testing of geochemical stability and reactivity of waste rock or tailings under the management regime adopted has not been completed, then additional work might be required to demonstrate that the closure strategy is currently viable and is unlikely to fail in future.

The types of data that should be collected or included to demonstrate geochemical stability of waste materials include:

- long-term mine drainage seep monitoring from waste rock dumps that have had the remediation strategy applied and are representative of the mine site
- long-term monitoring of underdrainage networks or seepage at tailings dams, which is predictable and can be mitigated/managed/treated if necessary
- studies of pit wall geochemistry to enable runoff or pit lake modelling, as required
- studies that confirm engineered landforms (dams, caps or covers, dumps, pit walls, cells to isolate reactive material, etc.) are stable and will not fail in a manner that will negatively impact the geochemical stability of the mine site
- studies that demonstrate suspended sediment will not be generated at the site in a manner that causes impacts in future.

The key factor in many of these studies is establishing a geochemical trend over time that indicates water quality leaving the site is matching predictions used to determine the closure strategy. The data set will need to be collected in such a manner that seasonal or operational variations can be removed, and the sites selected will need to be representative of the closure strategy. Many of the sample types, analytical methods and examples of these studies are provided in previous sections.

The more proactive that data acquisition has been during operations, the less difficult it will be to demonstrate that the closure plan will meet desired geochemical outcomes for the site.

At epithermal gold mines, major concerns are likely to relate to future concentrations of trace elements in waterways of groundwater downstream or down gradient of the mine. There are two contrasting closed epithermal gold mine sites where lessons for future mine closure can be obtained: Golden Cross mine site, closed in the late 1990s (Goldstone & MacGillivray 2002), and Tui mine, abandoned in the 1970s and rehabilitated between 2007 and 2011 (Giles et al. 2010; Basheer 2012).

At Golden Cross, surface rehabilitation has been relatively successful, with pasture established over the rehabilitated waste rock dumps/pit wall and a wetland/pond established over the tailings storage facility. However, water from underground working is currently pumped and discharged through a water treatment plant before discharge to the environment. Manganese and iron concentrations in the underground water would exceed consent conditions if this water was directly discharged (Trumm & Pope 2015).

At Tui mine, site abandonment without rehabilitation led to tailings runoff and discharges from underground workings that caused elevated concentrations of trace elements, including Zn, Cu, Pb, and Cd in the streams surrounding the site (Hickey & Clements 1998). Rehabilitation of tailings and underground workings led to decreases in the concentrations of metals in the streams at the Tui mine site (Fairgray et al. 2016), although concentrations remain elevated. Elevated metal concentrations continue to discharge from underground mine workings, and stream sediment continues to contain elevated concentrations of metals (Fairgray et al. 2017).

Partial closure of the Golden Cross site in the 1990s demonstrates several things for future mine site closure.

- Surface rehabilitation techniques can be successfully applied to epithermal gold mine sites to establish pasture.
- With appropriate bonding in place, mining companies will complete outstanding duties at the site after operations cease.
- Some aspects of closure require long-term management, including stability above underground workings

5.4 Treatment

Ideally, operational management and prevention strategies adopted during the early stages of mining and used during the operational phase have resulted in the discharge of water from the mine site that meets water quality targets and does not require treatment. However, if water treatment has been used during operations, it is possible that continued water treatment may be necessary as the mine moves into the closure stage. Regardless, the water quality at closure is likely to be different to that during mining.

As mentioned above in section 5.3, seep monitoring, tailings underdrain monitoring, pit wall geochemistry, and geochemical modelling can help to predict water quality at closure. Several different sources of water may need to be addressed, including that from remaining high walls, pit lakes, underground tunnels, unmineralised and mineralised waste rock overburden stacks, and tailings dams and impoundments. These may show markedly different water chemistries, and so may have different treatment solutions.

Prior to a review of treatment options during closure, the anticipated various chemistries that might be produced during closure should be compared to the current discharge consent(s) for the mine. In addition, future discharge consent conditions may change, and if these can be anticipated at mine closure, water treatment should aim to meet these water quality criteria. For epithermal gold mines, typical water quality issues relate to elevated acidity, trace elements, sulphate, and TSS.

Once water quality targets during closure have been determined, and anticipated water quality and flow rates from all the various sources at the mine have been modelled, treatment options during closure can be reviewed and selected. Ideally, this will have been done in the earliest stages before mining even began at the site, and this review is only meant to update the decisions made early on with more recent data from the mine site and from resource consent requirements. The following steps can be used in this process.

- Review the effectiveness of treatment systems used during the operational phase to help determine optimal treatment options during closure. Water treatment during mining operations typically utilises active treatment systems and technologies, whereas at mine closure it is preferable to use passive treatment systems (Skousen et al. 2000; Ziemkiewicz et al. 2003; Skousen et al. 2017).
- Complete an analysis to determine if various water discharge points should be combined or treated separately. If possible, it is usually better to combine water sources if the same treatment solution will be needed, as the operation and management of fewer systems is preferable.
- Review typical treatment solutions for AMD (see section 3.6.5)
- Once treatment options have been identified, complete small-scale trials on-site to determine optimal operational parameters and system effectiveness before investing in a full-scale system (see Appendix F6 in Cavanagh et al. 2015). Sometimes these trials are preceded by laboratory experiments.

After treatment systems have been constructed, they should be monitored and maintained, as explained in the 'Operations' chapter section 4.4.1**Error! Reference source not found.**. The quality of the water discharging from the mine site will improve with time, and eventually the treatment system will no longer be required and any financial bonds that have been placed against the mine site can be released.

At the Golden Cross mine, active treatment was selected and is currently employed to treat discharge that is pumped from underground workings and that contains elevated concentrations of Fe and Mn. Passive treatment trials have been completed (Trumm & Pope 2015), but these systems have not been adopted by the mining company. At the Tui mine, passive water treatment trials are in progress.

5.5 Performance monitoring

Performance monitoring is a key aspect of closure to ensure that closure plans remain appropriate and are in agreement with expectations about waste rock dump performance, water quality effects, revegetation, etc. Performance monitoring should be linked to key resource consent criteria for the project. Deviation from the expected outcomes should be flagged for action and adjustments made according to any adaptive management procedure.

5.6 Aquatic ecosystems

Aquatic ecological monitoring during closure should be focused on determining that the waterways are recovering to a condition similar to that found in reference sites. The number of sites to be monitored can be reduced, but should still include sufficient sites to detect changes. Typically (depending on the mine footprint and number of waterways) this would not be less than three sites within the closed mine and two to three reference sites. The frequency of monitoring might be reduced to annual sampling, assuming the aquatic community is recovering. If this is not happening, then more intensive sampling may be required to determine why the community is not recovering. In turn, this may require additional action to facilitate recovery.

Failure of an aquatic community to recover to reference site conditions may be due to any number of reasons, but typically these include:

- water quality conditions have not been improved sufficiently to support a healthy freshwater community
- pulses of contaminants are still being discharged into the waterway
- despite rehabilitation the waterway still has poor habitat, which might include a lack of good habitat for species that spawn or lay their eggs
- the waterway may be isolated from other waterbodies so that new species cannot easily colonise the rehabilitated waterway.

5.7 Terrestrial rehabilitation

5.7.1 Introduction

In a staged mine plan some areas of a mine will be undergoing permanent rehabilitation to meet agreed post-mining outcomes, although the whole site may not be in closure. Early rehabilitation is vital to demonstrate the on-site capability and allow site-specific techniques to be developed through adaptive management. These planned outcomes will include short-term and closure criteria. These are established with input from landowners, 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).

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, such as cropping and horticulture (viticulture), residential housing, recreation (historical mine relics, mountain biking) and public amenity, but they are not covered in this document. Maintenance and monitoring are integral components of successful rehabilitation.

5.7.2 Undertaking rehabilitation to pasture systems

Pasture is a common rehabilitation interim or final outcome (Figure 34). 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 needs to develop a dense sward with deep root systems, supported by high post-grazing (residual) pasture dry mass and high tolerance to pugging.

Figure 34. 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 on 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 because they provide evidence that guides adaptive management. Adaptive management means that the 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 for planning and agreeing the outcomes from rehabilitation to pasture is to ensure that landowners (where they are not the mining company) or regulatory authorities have a clear understanding of what is achievable and anticipated to be achieved.

Several steps are required to achieve successful rehabilitation to pasture Figure 355 and 36). 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 (and greater where heavier stock classes are likely), 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 nitrogen and phosphorus fertiliser applications, soil pH between 5.5 and 7, and/or organic amendments are usually needed to establish vigorous, dense pastures.

Inadequate (late or lax) management of weeds is a common cause of failure, particularly where gorse or broom 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 fertiliser is 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 35. 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

6. Establish pasture, re-vegetate

- Stone pick or rake, remove metal or concrete and other stock and vehicle hazards as necessary
- Decide if a pre-sowing spray of weeds is needed
- Apply fertiliser (lime/gypsum, phosphate, nitrogen, etc.)
- Minimise compaction of topsoil by combining fertilising, harrowing, sowing operations during drier periods
- Minimise erosion by cultivating with the contour
- Best results (germination and growth) are in spring and autumn
- See native vegetation establishment flowchart for TREE and SHRUB establishment

7. Monitor pasture outcome

- After first storm and/or 3–6 weeks after planting
- For each rehabilitation area

Check and Control Weeds

- Identify source of weeds and why weeds are dominant
- Decide weed control method (including grazing

Check Pasture Vigour

- Will pasture resist grazing?
- If grass is yellow check soil drainage, then nitrogen deficiency by spot fertilising

Check Pasture Cover

- Successful if >90% cover after three months
- Find causes of poor cover (weather, soil, erosion, pests) and do remedial work

Check Erosion*

- Check land surface , watercourses , cut-off drains
- Identify cause of erosion, then do remedial work
- Amend rehabilitation method (slope, surface roughness, timing) unless erosion is caused by an unusual rainfall

Decide if the area is ready for grazing or topping

8. Prepare for grazing or topping/mowing

- Is the area stock-proof (permanent or temporary fences in place) and watercourses excluded from grazing?
- Can stock get to and from the area to be grazed without impacting mining?
- Is the ground surface safe for stock or mowing (no wire, concrete, etc.)? Is an adequate water supply present (not a unfenced stream or drain)?
- Are wet areas present that are likely to be pugged (damaged)?
- Walkover with grazing manager/ farmer and confirm objectives: short duration with light stock under dry conditions leaving adequate pasture height

Grazing / Topping
Record the duration, and weather (e.g. rain)

9. Monitoring after grazing

- Pasture health and density (was pasture pulled out by roots, is residual pasture adequate for rapid regrowth or has it been over-grazed with bare soil evident?). Grass/legume balance (high legume cover indicates nitrogen stress)
- Weed species and cover (was grazing effective in controlling weeds?)
- Erosion or pugging, damage or runoff from grazed areas to watercourses
- Check fences remain stockproof

Feedback with farmer/ manager

- Did grazing go to plan?
- Review access, fencing, stock health, pasture health
- Does grazing intensity, timing or duration need to be altered?

10. Maintain and complete rehabilitation

- Post-grazing fertilisation, weed and pest control as needed
- Annual maintenance fertiliser as needed (phosphate, potassium, nitrogen, trace elements as needed)
- Complete permanent fencing, races, water troughs, shelter, etc. as agreed
- If soil fertility levels are part of agreement, soil test and fertilise as indicated by results

11. Closure assessment for hand-over to landowner and Bond Release *(part/whole)*

- Confirm agreed outcomes with land manager/owner and administrator

Figure 36. Implementing rehabilitation to pasture.

5.7.3 Production forestry

Mine rehabilitation to exotic forest (*Pinus radiata*, *Cupressus lusitanica* or *Eucalyptus*) 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 criteria 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.

Successful rehabilitation to plantation forestry (Figure 37 and 38**Figure** 37) 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, although pines can establish where the rooting depth is 0.5 m, particularly with spot mounding and adequate slope
- slopes are greater than 1:20, which helps ensure adequate drainage.

Figure 37. Rehabilitated coal mine with plantation forest, Waikato coal mine.

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) – Forest Research conducted trials on a range of leguminous ground cover species that helped provide adequate nutrition on mined tailings, with suitable species influenced by pH and droughtiness of the 'soil'
- the 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; ripping or subsoiling overburden helps prevent perching of water tables and increase rooting depth
- slopes are shallower than 1:20, which 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 grow and manage a small area of trees to maturity that is out of synch with the larger forest. 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. In such cases the construction of permanent ponds and large, gravelled areas may be valuable.

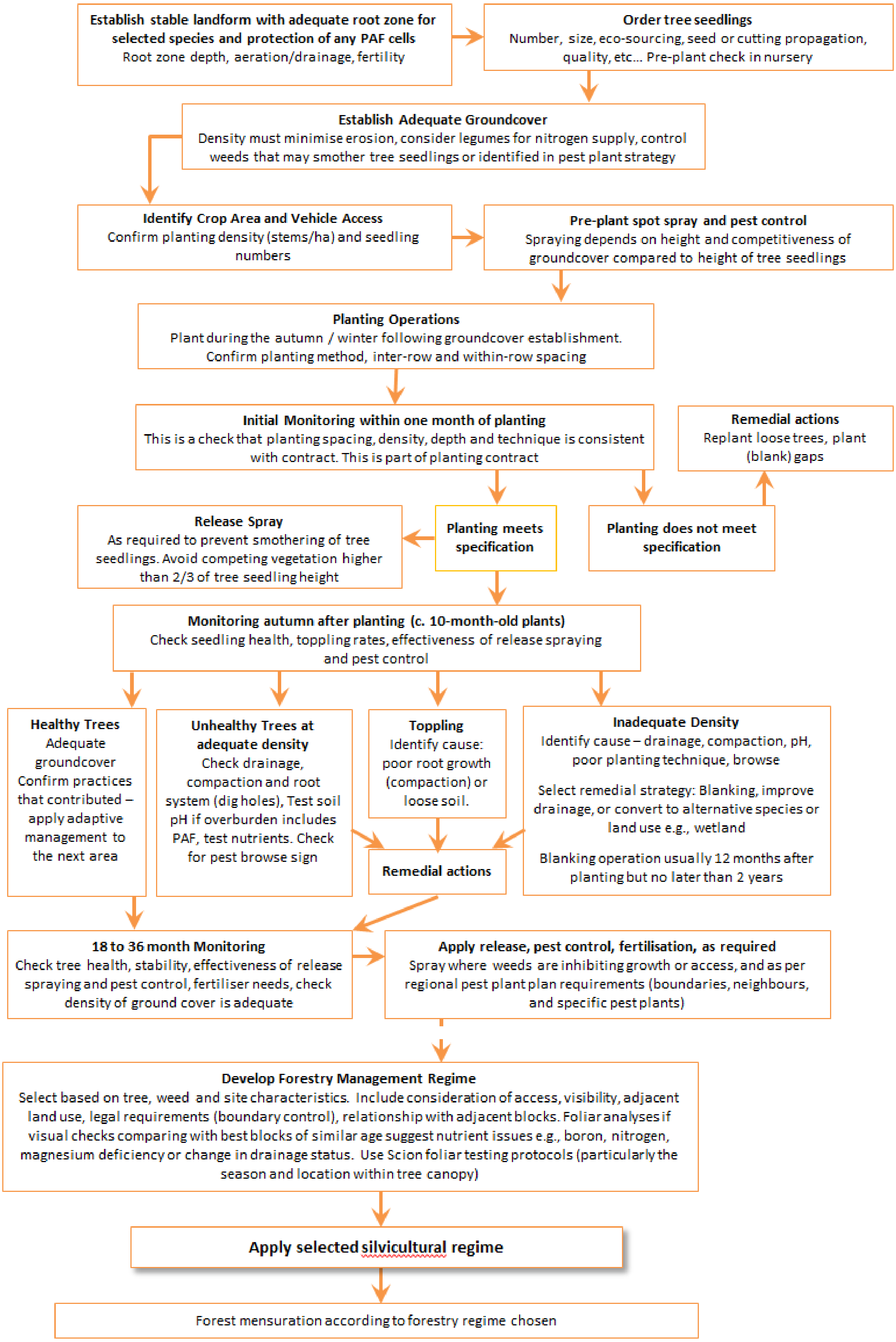

Figure 38. Process for rehabilitating mined areas to production forestry.

5.7.4 Rehabilitation to native ecosystems

Rehabilitation to 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 the development of similar ecosystems to those occurring pre-mining, it is important to understand the soils, landforms, and drainage that underpin the 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, whereby 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 are seen when mine planning maximises the quantity and quality of plants[29], seeds, soils 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 are 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 at building up new soil structure and organic matter. However, pasture can supress native plant growth and natural regeneration, particularly of herbs, and encourage grazing by deer, goats or other mammals. Lotus, particularly, smothers 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. Consequently, introduced plants tend to dominate many lowland mine sites, at least in the short to medium term.

Figure 39. The impact of soil conditions. Five-year-old mānuka and kānuka planted at the same time at the same site but with a 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, and many

[29] 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/)

species having mycorrhizal (fungal) associations that help plants access nutrients (including beech, mānuka and kānuka). 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, including:

- 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 mycorrhizae 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/or undulating) surfaces, enhanced with rocks or boulders, logs, and forest 'slash' (felled branches), assists regeneration by creating many protected and relatively stable, humid microsites. This approach is particularly important in the absence of planting seedlings or direct transfer.

The following flow chart (Figure 40) identifies the 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

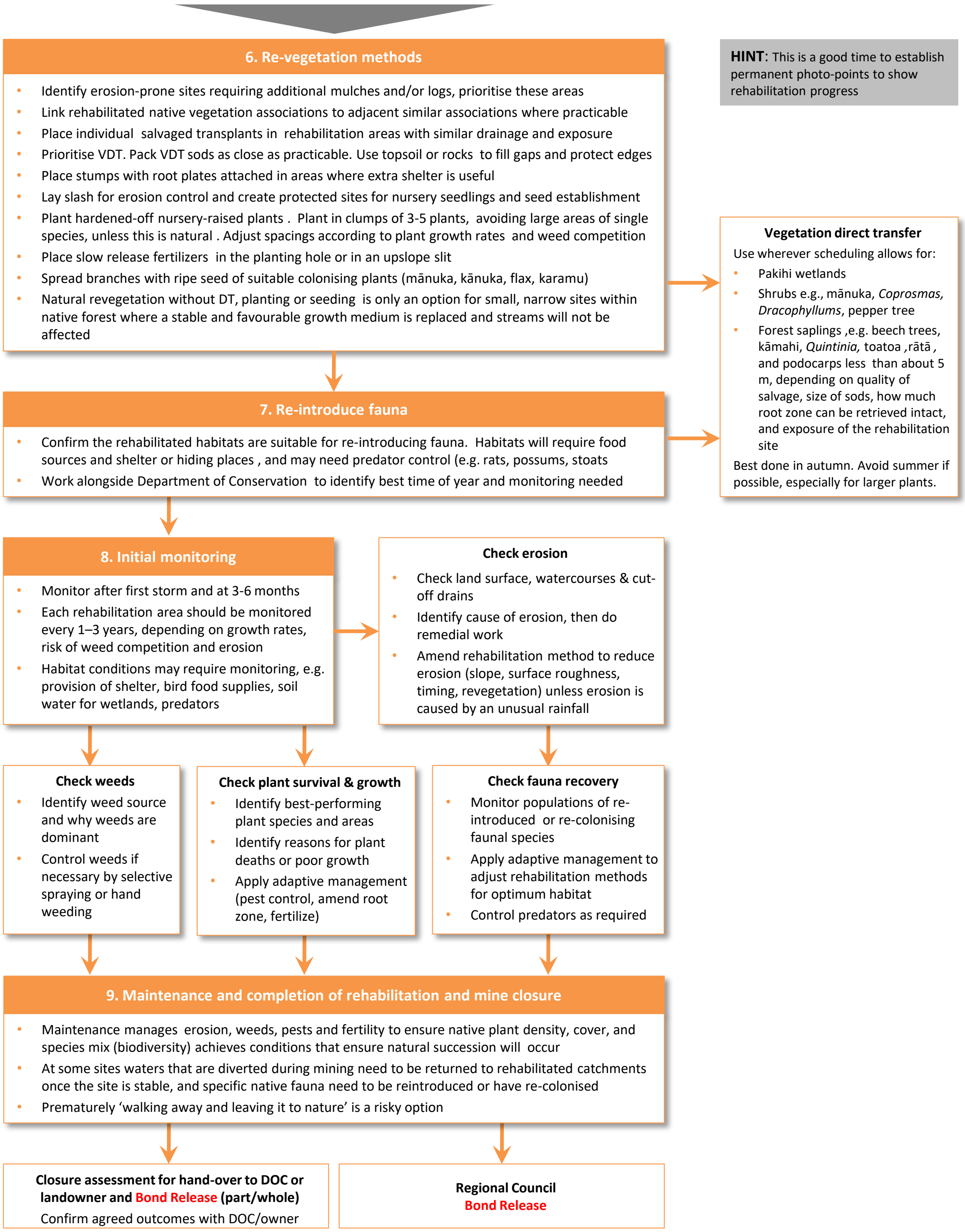

Figure 40. **Processes for undertaking rehabilitation to native ecosystems.**

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, die-back 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 the 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 41), 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 beech and podocarp) 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 the boundaries of rehabilitated areas.

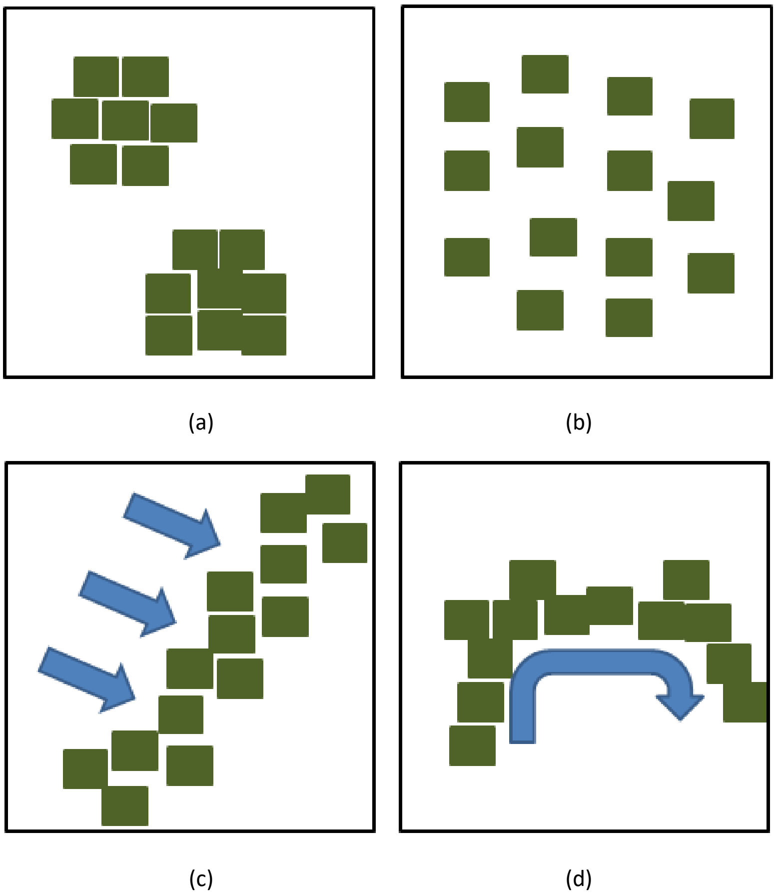

Figure 41. Direct transfer patterns used to achieve alternative goals: (a) to create islands as sources of plant and insect diversity; (b) to establish a target density of trees; (c) for erosion control (arrows show the direction of surface flow); and (d) to protect the edge, and shade, a pond or small watercourse.

Key factors for successful direct transfer are:

- transfer to wet environments not subjected to significant droughts
- transfer in autumn rather than summer/spring, while avoiding days when it is raining
- use engineer-constructed suitable landforms, with particular regard to 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 use a thin (20–30 cm) layer of salvage soil added 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 (e.g. 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, stand vegetation upright, 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 42. Direct transfer of lowland regenerating shrubland with rimu, kahikatea and other native trees, West Coast.

Selecting plant species

Where planting is to be undertaken, naturally local and locally sourced native plant species (e.g. '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. mānuka or kānuka) 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 (karamū, koromiko, flaxes and toetoe), especially where competition from pasture or weeds is likely. This means forest revegetation may use plants that have a 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 the sods.

Selecting plant or direct transfer sod density

The density of seedlings or sods planted depends on factors listed in Table 9, including the growth rate of plants relative to competing weeds, the extent of natural regeneration, and the purpose of planting.

Table 9. 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 until 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 the 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 may 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 such that ongoing maintenance is minimal. A high planting density usually speeds development of native plant cover, with a shorter time to closure than a low planting density.

Predicting the growth rates of 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 10. Each factor can be manipulated to improve rehabilitation outcomes.

Table 10. 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.	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 of 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 more susceptible to transplant shock.
High genetic potential – plants like mānuka, kānuka 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. *Phytopthera*).

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 11).

Table 11. Factors influencing the 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: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. mānuka, kānuka, hebe, toetoe)	High	Low, most species require sheltered forest floor
Contribution of birds to seed dispersal	High, perching and roosting sites present	Low
Weed species present that are likely to smother native plants in short term	Absent or sparse	Present and numerous

Factor affecting regeneration	High probability of fast regeneration	Low probability of regeneration, slow regeneration
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[1] or pest control) required to achieve acceptable outcomes	Low	High
Likelihood of disturbance, particularly from fire	Low	High

[1]*Releasing* is weed control undertaken post-planting, by hand or by chemical means. It is typically most effective if done while weeds are less than half to two-thirds the height of the plant we want, and timed to avoid summer drought (when bare soils may dry out quickly) and winter frosts.[30]

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 depends on the size of individual pieces, and 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 a 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 (resist 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) reduces 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 the 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 the 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 it on a rough surface and/or on a surface with freshly spread soils or rooting media. When seed-bearing branches or mulch are 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 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 necessary. 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 be established during spreading operations where logs could move onto roads or work areas. Bunds and/or boulders can also be used to restrict the 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 the 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.

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30 https://www.weedbusters.org.nz/weed-information/controlling-weeds/ and herbicides https://www.doc.govt.nz/about-us/science-publications/conservation-publications/protecting-and-restoring-our-natural-heritage-a-practical-guide/common-herbicides-used-to-control-weeds/

Case Study 19: Making the most of forest slash

Forest slash is the woody plant material that is felled and/or removed before soil and overburden stripping begins. Slash can include tree stumps, and may be mixed with surface soils. It is a rehabilitation resource, but can be difficult to manage, and to retrieve from steep slopes.

The re-use of slash in rehabilitation areas is a key criterion used to indicate the likely success of mine rehabilitation to native forest (Mew & Ross 1997). When sufficient slash is placed over replaced soils or bare overburden, it protects surfaces from erosion, provides sheltered and stable sites in which seedlings can establish, and protects young plants from browsing animals (Figure C41). When fresh slash is used (without stockpiling), native seedlings often resprout or survive, providing a source of 'free' native plants.

Slash that includes long (>2–4m) wood is often difficult to handle without specialist equipment. A solution is to cut trees into lengths that can be safely removed, transported and spread by machinery on site. It is tempting to place slash in a deep layers at the beginning of mining, when there are few places to store slash and small areas for rehabilitation. However, deep slash is difficult (and dangerous) to plant into, may be unstable, and can slow revegetation. A solution is to create windrows of slash, or 5 to 10 m diameter slash islands. These areas will tend to support many ferns (Figure C40), as the combination of shelter and wood favours these plants. Such slash piles are also valuable to create diverse habitats and refuges, both in forests and in wetlands. However, ensure slash is not placed in areas where it can be washed away in floods or slip into watercourses.

Figure C40. Relatively thick slash has regenerated with dense ferns and the occasional tree seedling. Mine closure criteria should encourage such areas, and not penalise them as 'unplanted'**.**

Figure C41. Left: Sparse slash of beech tree stumps interplanted with mānuka. Right: Slash in the foreground can be planted amongst and is effective at controlling erosion, while the pile in the background forms a habitat feature that cannot be planted.

Key findings relevant to rehabilitation:

- Slash is an important rehabilitation resource for forest rehabilitation. Fresh slash that contains root plates usually has some plant regeneration.
- A moderate cover of slash is effective at minimising erosion and conserving soils. A layer of slash should be used to protect topsoil stockpiles. Slash containing seeds (e.g. mānuka, karamū, hebe, flax) can be used as a seed source (Simcock et al. 2005, pp. 44−45).
- Slash can be used to protect plants from hares and deer.
- Deep slash can prevent planting and create safety (slip, trip and fall) hazards. Mortality of planted seedlings is increased because there is not enough soil.
- Slash piles or windrows help long-term habitat variation and site resilience against adverse weather (drought, waterlogging). Decomposing slash creates refuges and foraging places for birds and insects, and germination sites for specific plants.

Key references:

Mew G, Ross CW 1997. Part 1. Past performance of a range of mining sites. Science for Conservation 54: 7–32. https://www.doc.govt.nz/globalassets/documents/science-and-technical/sfc054.pdf

Anon. Minimising environmental impacts from mining. Fact Sheet 8. Native vegetation. https://www.landcareresearch.co.nz/__data/assets/pdf_file/0007/76894/FS8-Native-Vegetation.pdf

Simcock RC, Meurk CD, Smale MC 2005. Maintaining and revegetating roadsides: Handbook for road controlling authorities and contractors. Manaaki Whenua – Landcare Research. http://icm.landcareresearch.co.nz/knowledgebase/publications/documents/Road_reveg_handbook_31_Aug_2005.pdf

Rehabilitation trials

Rehabilitation trials may be useful to determine the principal issues limiting revegetation and the best plant species to grow, particularly in challenging environments (see Case Study 20).

Case Study 20: The value of rehabilitation trials

Most larger and longer-term mines establish rehabilitation trials in the first or second year of mine opening, and sometimes prior to mine development, to support resource consent applications and closure criteria. Trials typically confirm the value of different mine overburdens and soils as growth media, often alongside a variety of amendments such as fertilisers, and for the dominant plant species.

Figure C42. 'Devil's Overburden backfill slope planted with beech and mānuka trees flanked by beech forest. The mulch trial is in the centre of the photo (where tree growth is sparser).

The Globe Progress mine, near Reefton, established trials in the early stages of mine opening to compare the survival and growth of native beech trees in different root zones (2005) and alternatives to topsoil (2007). These trials informed three closure objectives: a suitable substrate and landforms for forest development, a canopy dominated by native tree species, and natural regeneration occurring. The 'Devils Root Zone' trial showed the importance of conserving soils adequate for growth and survival of beech (Figure C43; Norton et al. 2013). After 4 years, beech survival in overburden plots was 10%, compared with 40 to 50% for plots with soil or mixed soil. The topsoil, despite very low available macronutrients (phosphate and nitrogen) and high acidity, was a superior growth material to which native plants are adapted.

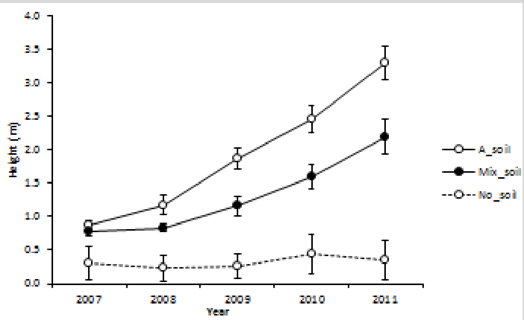

Figure C43. Beech tree height growth (m) over 5 years in topsoil (A_soil), mixed topsoil and overburden (Mix_soil) and overburden (No_soil), and key root zone chemistry (table at right).

Trial: Devils Root zone	pH	Olsen P	P retention	Total C	Total N
Topsoil	4.5	3	3	4.4	0.19
Topsoil overburden mixed	4.8	3	10	2.2	0.09
Overburden, nil soil	6.3	8	11	2.3	0.18

The Devils Root Zone trial showed mixing overburden with topsoil produced a degraded root zone; it was not useful to 'bulk up' or 'extend' topsoils. Wood chip was investigated, both as an alternative amendment and on its own. Mānuka and beech were planted directly into 100% woodchip, 1:1 wood chip:soil, or 100% soil (the latter being the 'control'). At age 4, wood chips appeared a poor medium for plant growth. However, after 10 years the wood chip had formed a humus-rich soil that supported adventive seedlings, indicating that given enough lead time, wood chip is a useful amendment, either mixed with soil or as a surface mulch.

Key findings relevant to rehabilitation trials:

- Native plants can grow well in soils considered infertile based on farming measures.
- Measure results over at least 4 years, as outcomes can change with soil development, weathering, plant interactions and adventive seedling establishment.
- Place trials in areas that will not be disturbed. Use replication in trials, ideally in different areas of the site in case changes in mine plans/road widening remove plots.
- Ensure results from small trials are ‘scalable – the trials should have similar slopes and ideally should use similar machinery.

Key references:

Norton DA, Creedy S, Keir D 2013. Substrate modification for enhanced native forest restoration, Reefton. Ecological Management and Restoration 14(2): 147−150.

5.7.5 Monitoring terrestrial 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 data 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 at 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 and constructed waterways and lakes/tarns can be useful at large mine sites to measure the stability of the rehabilitated landscape components.

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 the 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 useful for photographing areas before, during and after rehabilitation. Aerial photography using drones and LIDAR can be a useful tool 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 the 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 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

Monitoring of rehabilitation indicators for high-productivity pasture on a larger scale would typically include measures of seasonal and annual biomass against baseline productivity for an area identified prior to mining. However, this is relatively labour intensive and requires stock exclusion cages to be established. A reasonable indication can be gained from comparing the following features:

- pasture colour and growth across a grazed paddock, comparing urine/dung spots: if high initial nitrogen levels drop sharply, ryegrass will become less competitive and clover will dominate (given adequate phosphorus and pH > 5.5); variations in growth can indicate seepage points (taller growth in summer) or areas of waterlogging (yellower grasses in winter)
- topsoil depth and pasture-rooting depth: plants with deep root systems are more resilient
- soil chemical testing, which is useful to indicate capital requirements for phosphate (using the Olsen P and phosphate retention tests), liming (through pH measures), and build-up of organic matter (indicating resilience and nitrogen supply)
- pasture composition: wetness is typically indicated by growth of dock, rushes and buttercups; *Poa annua* and flatweeds tend to dominate in areas with bare patches, or that are overgrazed, seasonally dry or compacted/pugged
- surface microtopography – soft, wet soils are more vulnerable to animal pugging; minimal surface stoniness can be a key requirement because it can affect animal health (lameness).

Areas rehabilitated to pasture will also require monitoring of infrastructure (e.g. fences, gateways, under-drainage, access races, stand-off areas and water supplies) and may involve measures specific to the success of shelter belts and riparian areas (e.g. stock exclusion and native planting).

Plantation forestry monitoring

Monitoring of rehabilitation indicators for plantation forests on a larger scale would typically include records of the specific clone of seedlings used, as this can influence forest value. The following features may be expected to be recorded or available:

- tree density and evenness (an aerial photograph is an excellent tool to monitor this across a stand once the trees are 3 to 5 years old)
- tree form and health (a sweeping 'hockey stick' base and toppling are indicative of poor drainage or inadequate rooting depth); tree form affects the value of a crop, although plantations may be considered to have only re-establishment value until relatively mature, and tree health may be quantified by productivity (basal area volume calculated from tree diameter, height and density) and by measuring pine needle nutrition
- pruning history, particularly the timing, height and targeted maximum knotty core diameter of each lift.

Features of value to assess site stability during the early stages of development before tree canopy closure include initial ground cover for erosion control, and weeds present, particularly weeds that have the ability to supress tree growth, or that have a long-lived seed bank so may require resources to control once the site is harvested.

It may also be important to record site-wide, post-closure constraints that are needed to ensure ongoing site safety and stability. In some cases these may need to be noted on LIM reports or as conditions that are triggered when a site is thinned, harvested or cultivated to allow a subsequent crop of trees to establish.

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 is 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 causes 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.

5.8 Economics – bond release

In the context of this guide, a mine is considered to be closed when regulators have agreed that closure conditions have been met and the bond is released. At this point the mining company has no further responsibility for the mine site. If there are foreseeable ongoing costs, such as those that would arise from ongoing treatment of mine drainage or regularly cleaning out drains, these are included in the post-closure bond. For these costs, bond conditions dictate that, upon successful closure of the site and release of the bond, the share of the bond that covers the costs that are expected to continue in perpetuity is relinquished to a trust (or similar entity) that is charged with administering this ongoing care. Unless a number of extreme events occur simultaneously, the interest earned from the trust's capital reserve should cover all its future expenditures.

6 Post-closure

6.1 Introduction

For the purposes of this guide, post-closure is defined as the point when regulators have agreed that closure conditions have been reached and the bond is released. At this stage the mine company no longer has any responsibility for the site. Depending on the mine, ongoing treatment, monitoring, and maintenance may be required. The costs associated with this ongoing work may be covered by a trust that has been established for that purpose and would be a consideration by the regulators in releasing the bond. The post-closure phase should solely involve monitoring to confirm that any water treatment activities are operating appropriately, and the rehabilitation outcomes are on track towards their desired final state.

6.2 Water treatment

Water treatment will often cease during the closure stage as long as any remaining discharge from the mine site meets discharge quality requirement. However, there are times when discharge may still need to be treated post-closure. According to the definition of post-closure, any financial bonds that had been put against the mine site have been released at the end of the closure stage and the mine company no longer has any responsibility for the mine site or discharge from the site. Local regulatory agencies (such as regional councils), however, should continue to monitor any discharge from the site to ensure that receiving water bodies are not impacted by the discharge. If ongoing treatment is required, a trust can be established to manage the treatment system.

These costs are likely to include:

- operating costs
- capital replacement costs
- maintenance costs for the treatment system
- maintenance of incidental services (roads, power, communications, etc.)
- maintenance for low-probability events (e.g. storm events greater than the design criteria for the system) where damage results
- sludge disposal
- monitoring
- analysis and reporting of monitoring data to regulatory authorities
- compliance review costs (regulatory and any third-party subject matter expert review as required)
- health and safety review and compliance auditing
- technology review and cost/benefit analysis.

6.3 Ecosystem recovery, maintenance of productivity and structures

Initial biological monitoring of aquatic ecosystems may be necessary to ensure the recovery objectives are being achieved. Specific medium- and/or long-term constraints may be required to ensure rehabilitated land can continue to recover. Some mines that are rehabilitated to native ecosystems establish funds and groups to achieve pest plant and animal control post-closure. The period may be calculated during consenting as part of compensation for productivity of native ecosystems; for example, the time needed for a rehabilitated area to develop specific fruiting trees for native birds, or leaf litter development for native invertebrates.

Some mines that are rehabilitated to pasture systems may require limitations on cultivation frequency or depth, or crop type, or the weight/timing of grazing stock until organic matter levels and topsoil depths have sufficiently recovered. Such limitations are most likely on rehabilitated areas with a topsoil deficit compared with pre-mining conditions. In such cases, higher-maintenance phosphorous fertiliser applications may also be required to sustain pasture growth compared with unmined areas.

Ongoing monitoring and maintenance may be required for landforms that have areas where deep-rooted vegetation is not wanted. This includes some fill embankments of tailings ponds where all, or some, tree species may be removed in perpetuity to minimise the risks of exposing buried layers (through tree toppling), or tree roots invading and increasing the permeability of capping layers. In this case, the post-closure constraints needed to ensure ongoing site safety and stability may include maintaining the site in grazed pasture. Such conditions need to be noted on LIM reports or long-term consent conditions that are triggered if management changes are desired; for example, if a production forest is thinned, harvested or cultivated to allow a subsequent crop of trees to establish (e.g. ripped or spot mounded), or a new road is constructed through the area, or buildings are proposed, or it is proposed that flows to a constructed watercourse be increased. These cases may be similar to constraints placed on old landfill sites.

Ongoing monitoring and maintenance are also likely where built structures that protect human or stock safety can degrade over time; for example, fenced adits or building foundations with sudden drops that may be obscured with vegetation growth. For this reason, it is generally better to design closure to avoid structures that require such ongoing maintenance by, for example, filling adit entrances or back-filling sudden drops.

7 REFERENCES

ACSMP (Australian Centre for Sustainable Mining Practices) 2011. A guide to leading practice sustainable development in mining. https://www.im4dc.org/wp-content/uploads/2014/01/A-guide-to-leading-practice-sustainable-development-in-mining.pdf

Ahern CR, McElnea AE, Sullivan LA 2004. Acid sulphate soils laboratory methods guidelines. Queensland Department of Natural Resources, Mines and Energy.

ANZECC & ARMCANZ (Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand) 2000. Australian and New Zealand guidelines for fresh and marine water quality. Canberra, Australia, ANZECC/ARMCANZ.

Arnold DE 1991. Diversion wells: a low-cost approach to treatment of acid mine drainage. Proceedings, 12th West Virginia Surface Mine Drainage Task Force Symposium, 3−4 April 1991, Morgantown, WV, USA.

Auckland Council 2016. Erosion and sediment control guide for land disturbing activities in the Auckland region. Guideline document 2016/005. http://www.aucklanddesignmanual.co.nz/project-type/infrastructure/technical-guidance/erosionsedimentcontrol

Auckland Regional Council 1999. Technical publication 90 [updated by Guideline document 2016/005, Auckland Council 2016].

AusIMM 2012. Australasian code for reporting of exploration results, mineral resources and ore reserves. AusIMM Joint Ore Reserves Committee.

Basheer G 2012. Rehabilitation of tailings at the former Tui Mine. In: Proceedings AusIMM New Zealand Branch Conference Rotorua, p. 25.

Basher L, Moores J, McLean G 2016. Erosion and sediment control practices in NZ: information gaps. Landcare Research Contract Report LC2629. https://www.mfe.govt.nz/publications/fresh-water/erosion-and-sediment-control-practices-new-zealand-information-gaps

Berger BR, Bethke PM 1985. Geology and geochemistry of epithermal systems. In: Robertson JM ed. Reviews in Economic Geology. Society of Economic Geologists, p. 298.

Bigham JM 1994. Mineralogy of ochre deposits formed by sulfide oxidation. In: Jambor JL, Blowes DW eds. The environmental geochemistry of sulfide mine-wastes: short course handbook, Vol. 22. Mineralogical Association of Canada. Pp. 103–132.

Black C, Ziemkiewicz P, Skousen J 1999. Construction of a limestone leach bed and preliminary water quality results in Beaver Creek. Proceedings, 20th West Virginia Surface Mine Drainage Task Force Symposium, Morgantown, WV, USA.

Boubée JAT, Dean TL, West DW, Barrier RFG 1997. Avoidance of suspended sediment by the juvenile migratory stage of six New Zealand native fish species. New Zealand Journal of Marine and Freshwater Research 31: 61–69.

Brathwaite R, Torckler L, Jones P 2006. The Martha Hill epithermal Au-Ag deposit, Waihi: geology and mining history. In: Christie T, Brathwaite R eds. Geology and exploration of New Zealand mineral deposits. Burwood, AusIMM, Australia Pp. 171−179.

Brown PL, Logsdon MJ, Vinton B, Schofield I, Payne K. 2014. Detailed characterisation of the waste rock dumps at the Kennecott Utah Copper Bingham Canyon Mine – Optionality for Closure. In: Miller H, Preuss L eds. Proceedings of the Eighth Australian Workshop on Acid and Metalliferous Drainage. Adelaide May 2014; pp 1-12.

Brown KL, Simmons SF 2003. Precious metals in high temperature geothermal systems. Geothermics 32: 619−625.

Bunder JM 2015. Water quality and macroinvertebrate community structure of mining lakes and ponds across New Zealand. Unpublished MSc thesis, University of Auckland.

Burdon F, Harding JS, McIntosh AR 2013. Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. Ecological Applications 23: 1036−1047.

Castendyk DN, Mauk JL, Webster JG 2005. A mineral quantification method for wall rocks at open pit mines, and application to the Martha Au-Ag mine, Waihi, New Zealand. Applied Geochemistry 20: 135−156.

Castendyk DN, Webster-Brown JG 2007a. Sensitivity analyses in pit lake prediction, Martha Mine, New Zealand; 1, Relationship between turnover and input water density. Chemical Geology 244: 42−55.

Castendyk DN, Webster-Brown JG 2007b. Sensitivity analyses in pit lake prediction, Martha Mine, New Zealand; 2, Geochemistry, water-rock reactions, and surface adsorption. Chemical Geology 244: 56−73.

Castendyk D, Webster-Brown J 2010. Effects of mine expansion on geochemical predictions of pit lake water quality: an example from Martha Mine, Waihi, New Zealand. New Zealand Journal of Geology and Geophysics 53: 143−151.

Cavanagh J, Pope J, Simcock R, Harding J, Trumm D, Craw D, Weber P, Webster-Brown J, Eppink F, Simon K 2018. Mine Environment Life-cycle Guide: mesothermal (orogenic) gold mines. Landcare Research and CRL Energy Ltd.

Cavanagh JE, Hogsden KL, Harding JS 2014. Effects of suspended sediment on freshwater fish. Landcare Research Technical Report LC1986. West Coast Regional Council Envirolink Advice Grant: 1445-WCRC129.

Cavanagh JE, Pope J, Harding JS, Trumm D, Craw D, Rait R, Greig H, Niyogi D, Buxton R, Champeau O, Clemens A 2010. A framework for predicting and managing water quality impacts of mining on stream ecosystems: a user's guide. Report for Ministry of Science and Innovation by Landcare Research (LC09190/114), CRL (Report 10-41100), University of Canterbury and University of Otago.

Cavanagh JE, Pope J, Harding JS, Trumm D, Craw D, Simcock R, Ross C 2015. New Zealand Minerals Sector Environmental Framework. Landcare Research Contract Report LC2033. Landcare Research and CRL Energy Ltd.

Christenson H, Weber P, Pope J, Newman N, Olds W, Trumm D 2017. Enhancing mine drainage treatment by sulphate reducing bacteria using nutrient additives. Proceedings 13th International Mine Water Association Congress – Mine Water and the Circular Economy, Lappeenranta. Pp. 1196−1203.

Christenson H, Weber P, Pope J, Newman N, Olds W, Trumm D 2018. Enhanced passive treatment systems: addition of nutrients to bioreactors. AusIMM New Zealand Branch Annual Meeting, Tauranga, AusIMM NZ Ltd. Pp. 97−104.

Christie A, Simpson M, Brathwaite R, Mauk J, Simmons S 2007. Epithermal Au-Ag and related deposits of the Hauraki Goldfield, Coromandel Volcanic Zone, New Zealand. Economic Geology 102: 785−816.

Connolly LP, Wein VS, Denize PR 1981. Land rehabilitation practice at Waipipi Ironsands Ltd. Paper presented at the Australian Institute of Mining and Metallurgy New Zealand Branch Annual Conference, Thames, 19 August – 21 September 1981.

Cravotta CA, Means BP, Arthur W, McKenzie RM, Parkhurst DL 2015. AMDTreat 5.0+ with PHREEQC titration module to compute caustic chemical quantity, effluent quality, and sludge volume. Mine, Water, and the Environment 34: 136−152.

Cravotta CA, Trahan MK 1999. Limestone drains to increase pH and remove dissolved metals from acidic mine drainage. Applied Geochemistry 14: 581−606.

Craw D 2001. Tectonic controls on gold deposits and their environmental impact, New Zealand. Journal of Geochemical Exploration 73: 43−56.

Craw D, Chappell DA 2000. Metal redistribution in historic mine wastes, Coromandel Peninsula, New Zealand. New Zealand Journal of Geology and Geophysics 43: 187−198.

Craw D, Mulliner T, Haffert L, Paulsen H-K, Peake B, Pope J 2008. Stratigraphic controls on water quality at coal mines in southern New Zealand. New Zealand Journal of Geology and Geophysics 51: 59−72.

Crowe A, Hay J 2004. Effects of fine sediment on river biota. Cawthron Report No. 951.

Davies H, Weber P, Lindsay P, Craw D, Pope J 2011. Characterisation of acid mine drainage in a high rainfall mountain environment, New Zealand. Science of the Total Environment 409: 2971−2980.

DITR (Department of Industry, Tourism and Resources) 2007. Leading practice sustainable development program for the mining industry: managing acid and metalliferous drainage. Canberra, ACT, DTIR.

DOC 2010. Revegetation of Alluvial Gold Mines A prescription for the West Coast Tai Poutini. Department of Conservation, Hokitika.

Druzbicka J, Craw D 2011. Experimental turbidity evaluation of Central Otago paleoplacer gold deposits. AusIMM New Zealand Branch Annual Conference, Queenstown, pp. 93–103.

Druzbicka J, Craw D 2015. Metalloid attenuation and runoff waters at an historic orogenic gold mine, New Zealand. Mine Water and the Environment, v. 34, Special Issue: Mine Water Research in New Zealand, pp. 417−429.

EIANZ (Environment Institute of Australia and New Zealand Inc) 2018. Ecological Impact Assessment (EcIA) EIANZ guidelines for use in New Zealand: terrestrial and freshwater ecosystems, 2nd edition. https://www.eianz.org/resources/publications/2018---ecological-impact-assessment-guidelines-for-new-zealand-2nd-edition

Eppink F, Trumm D, Weber P, Pope J, Cavanagh J 2017. A robust cost-effectiveness assessment of passive AMD treatment. AusIMM NZ Branch Conference, Christchurch, 10–13 September.

Fala O, Aubertin M, Molson J, Bussière B, Wilson GW, Chapuis R, Martin V 2003. Numerical Modelling of Unsaturated Flow in Uniform and Heterogeneous Waste Rock Piles. In "Proceedings of the Sixth International Conference on Acid Rock Drainage", Cairns, Australia, 14-17 July (Eds. T Farrell and G Taylor), pp. 895 - 902. Australian Institute of Mining and Metallurgy, Carlton, Victoria.

Fairgray M, Webster-Brown J 2017. Release of toxic trace elements from stream sediment at Tui Mine, Te Aroha, New Zealand. In: Proceedings AusIMM New Zealand Branch Conference, Christchurch. Pp. 392−397.

Fairgray M, Webster-Brown J, Harding J, Waters S 2016. Geochemical modelling of metal toxicity in the Tui Mine catchment, Te Aroha, NZ. AusIMM New Zealand Branch Conference, Wellington. Pp. 100−108.

Giles E, Jenkins I, Williams S, Kirk A, Fellows D, Press R 2010, Tui mine remediation detailed design report. Underground mine, access road, waste rock stack remediation works. URS Ltd.

Goldstone A, MacGillivray R 2002. Closure of the Golden Cross Mine. In: Proceedings AusIMM New Zealand Branch Conference, Auckland,pp. 29−34.

Gray DP, Harding JS 2012. Acid Mine Drainage Index (AMDI): a benthic invertebrate biotic index for assessing coal mining impacts in New Zealand streams. New Zealand Journal of Marine and Freshwater Research 46(3): 335−352. DOI: 10.1080/00288330.2012.663764

Gregg PEH, Stewart RB, Brodie KM, Stewart F 2003. From field trials to reality: rehabilitation of the tailings storage embankment at the Waihi Gold Mine. In *Proceedings of the workshop Environmental Management using soil-plant systems*. Occasional Report No.16 Fertiliser and Lime Research Centre,141-148. Massey University, Palmerston North, New Zealand.

Gregg PEH, Stewart RB, Ross CW 1998. Land reclamation practices and research in New Zealand. In: *Land reclamation: achieving sustainable benefits,* ed. Fox, Moore and McIntosh, 365-372. Blakerna Rotterdam

Haffert L, Craw D 2008. Mineralogical controls on environmental mobility of arsenic from historic mine processing residues, New Zealand. Applied Geochemistry 23: 1467−1483.

Haffert L, Craw D 2009. Field quantification and characterization of extreme arsenic concentrations at a historic mine processing site, Waiuta, New Zealand. New Zealand Journal of Geology and Geophysics 52: 261−272.

Hammarstrom JM, Sibrellb PL, Belkina HE 2003. Characterization of limestone reacted with acid-mine drainage in a pulsed limestone bed treatment system at the Friendship Hill National Historical Site, PA, USA.

Harding JS, Clapcott JE, Quinn JM, Hayes JW, Joy MK, Storey RG, Greig HS, Hay J, James T, Beech MA, Ozane R, Meredith AS, Boothroyd IKG 2009. Stream habitat assessment protocols for wadeable rivers and streams of New Zealand. University of Canterbury, Christchurch, New Zealand.

Hedin RS, Watzlaf GR 1994. The effects of anoxic limestone drains on mine water chemistry. Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, USA. Pp. 185−194.

Henley RW, Truesdell AH, Barton Jr PB, Whitney JA 1984. Fluid-mineral equilibria in hydrothermal systems. In: Robertson JM ed. Reviews in economic geology. Society of Economic Geologists.

Hickey C, Clements W 1998. Effects of heavy metals on benthic macroinvertebrate communities in New Zealand Streams Environmental Toxicology and Chemistry 17(11): 2338−2346.

Hilton T 2005. Low pH - iron oxidation. 26th Annual West Virginia Surface Mine Drainage Task Force Symposium, 20 April 2005. http://wvmdtaskforce.com/proceedings/05/Hilton.pdf.

INAP (International Network for Acid Prevention) 2009. Global acid rock drainage guide (GARD guide). http://www.gardguide.com/

Kepler DA, McCleary EC 1994. Successive alkalinity producing systems (VFW) for the treatment of acidic mine drainage. Proceedings of the International Land Reclamation and Mine Drainage Conference, Pittsburgh, PA, pp. 195–204.

Kerr G, Pope J, Trumm D, Craw D 2013. Experimental mobilisation and secondary mineralisation of antimony and arsenic from gold ore, Reefton New Zealand, 26th International Applied Geochemistry Symposium: Rotorua, The Association of Applied Geochemists, p. 46.

Kerr G, Druzbicka J, Lilly K, Craw D 2015, Jarosite solid solution associated with arsenic-rich mine waters, Macraes Mine, New Zealand. Mine Water and the Environment, v. 34, Special Issue: Mine Water Research in New Zealand, pp. 364−374.

Kerr G, Pope J, Trumm D, Craw D 2015. Experimental metalloid mobilisation from an orogenic gold deposit, New Zealand. Mine Water and Environment, v. 34, Special Issue: Mine Water Research in New Zealand, pp. 404−416.

Kolling M, Schuring J 1994. Pyrite weathering in coal mine tailings. Adelaide, Australia, Water Down Under.

Langer L, David MR, Ross C 1999. Rehabilitation of lowland indigenous forest after mining in Westland. Science for Conservation 117. Wellington, Department of Conservation.

Leathwick J, Morgan J, Wilson G, Rutledge D, McLeod M, Johnston K 2003. Land environments of New Zealand. Auckland, David Bateman Ltd. https://www.landcareresearch.co.nz/resources/maps-satellites/lenz/products/land-environments-of-new-zealand-technical-guide

Leersnyder H, Bunting K, Parsonson M, Stewart C 2016. Erosion and sediment control guide for land disturbing activities in the Auckland region. Auckland Council Guideline Document GD2016/005. Prepared by Beca Ltd and SouthernSkies Environmental for Auckland Council. Available at

http://www.aucklandcouncil.govt.nz/EN/planspoliciesprojects/plansstrategies/unitaryplan/Documents/Section32report/Appendices/Appendix%203.11.1.pdf

Malloch K, Petrov P, Weber P 2018. Mine drainage management – replicating dump physiochemical conditions in laboratory columns. AusIMM New Zealand Branch Annual Conference, Tauranga, AusIMM NZ Ltd. Pp. 199−210.

Mattes AG, Higgins JP, Gould WD 2007. Passive biologically based anaerobic treatment systems for the removal of metals: an overview of current research with examples. Copper Symposium, Cu2007 Conference, CIM Conference of Metallurgists, August 2007, Toronto.

Mauk J, Hall C, Chesley J, Barra F 2011. Punctuated evolution of a large epithermal province: the Hauraki Goldfield. New Zealand Economic Geology 106: 921−943.

Mayer KU, Blowes DW, Frind EO 2003. Advances in reactive-transport modelling of contaminant release and attenuation from mine-waste deposits. In: Jambor JL, Blowes DW, Ritchie AIM eds. Environmental aspects of mine wastes. Mineralogical Association of Canada Short Course Series Vol. 31. Pp. 283−302. Available at: www.mineralogicalassociation.ca.

McEwen WM.1987 (Editor). Ecological regions and districts of New Zealand. Biological Resource Centre Publication No. 5: part 3. Wellington, Department of Conservation.

Means B, McKenzie B, Hilton T 2003. A computer-based model for estimating mine drainage treatment costs. 24th Annual West Virginia Surface Mine Drainage Task Force Symposium, Morgantown, WV, USA. Also available at: http://wvmdtaskforce.com/proceedings/03/Means03.pdf. Computer program available free at: http://amdtreat.osmre.gov/.

MEND (Mine Environment Neutralising Drainage) 2001. The MEND manual volume 4. Natural Resources Canada.

MEND (Mine Environment Neutralising Drainage) 2009. Prediction manual for drainage chemistry from sulphidic geological materials. MEND report 1.20.1. Natural Resources Canada.

MEND (Mine Environment Neutralising Drainage) 2012. Cold regions cover system design technical guidance document. MEND report 1.61.5c. Natural Resources Canada.

Mew G, Ross CW 1991. Beech forests after mining. Terra Nova 4: 52–53.

MfE (Ministry for the Environment) 2015. An everyday guide to the RMA: applying for a resource consent. Publication number ME 1182. http://www.mfe.govt.nz/publications/rma/everyday-guide-rma-applying-resource-consent

Miller S 1987. Golden Cross Mining Project technical report series: geochemistry and leaching of solid wastes. Cyprus Minerals New Zealand Ltd.

Mills JE 1996. The North Branch of the Potomac River: results of two years of lime dosing. 17th Annual West Virginia Surface Mine Drainage Task Force Symposium, 2−3 April 1996, Morgantown, WV, USA. Also available at: http://wvmdtaskforce.com/proceedings/96/96MIL/96MIL.htm.

Morin KA, Hutt NM 2009. Mine-water leaching of nitrogen species from explosive residues. GeoHalifax 1549−1553.

Myers 2011.

National Research Council (US) 2002. Coal waste impoundments: risks, responses and alternatives. Washington, DC, National Academy Press.

Nieman TJ, Merkin ZR 2000. Premining planning for postmining land use: applying principles of comprehensive planning and landscape architecture to reclamation. In: Barnhisel RI, Darmody RG, Daniels, WL eds. Reclamation of drastically disturbed lands. Monograph 41. Madison, WI, American Society of Agronomy. Pp. 667–686.

Nordstrom DK, Alpers CN 1999. Geochemistry of acid mine waters. In: Plumlee G, Logsdon MJ eds. Reviews in economic geology Volume 6A: Environmental geochemistry of mineral deposits, Part A, Processes, techniques and health issues. Littleton, CO, Society of Economic Geologists. Pp. 133−160.

Norton DA, Creedy S, Keir D 2013. Substrate modification for enhanced native forest restoration, Reefton. Ecological Management and Restoration 14(2): 147−150.

NZP&M (New Zealand Petroleum and Minerals) 2013. Minerals Programme. Wellington, New Zealand Petroleum and Minerals. https://www.nzpam.govt.nz/assets/Uploads/our-industry/rules-regulations/minerals-programme-2013.pdf

NZP&M (New Zealand Petroleum and Minerals) 2016. Permit holder guidance for iwi engagement reporting. Wellington, New Zealand Petroleum and Minerals. https://www.nzpam.govt.nz/doing-business/maori-waitangi/permit-holder-engagement/

O'Sullivan A 2005. Passive treatment technologies for managing metal mine wastes: lessons learnt from global applications. In: Moore TA, Black A, Centeno JA, Harding JS, Trumm DA eds. Metal contaminants in New Zealand, sources, treatments, and effects on ecology and human health. Christchurch, Resolutionz Press. Pp. 279–300.

Olds W, Bird B, Pearce J, Sinclair E, Orr M, Weber P 2015. Geochemical classification of waste rock using process flow diagrams. AusIMM New Zealand Branch Conference, Dunedin. Pp. 307−318.

Olds W, Weber P, Pope J, Pizey M 2016. Acid mine drainage analysis for the Reddale Coal Mine, Reefton, New Zealand. New Zealand Journal of Geology and Geophysics 59: 341–351.

Osterkamp WR, Joseph WL 2000. Climatic and hydrologic factors associated with reclamation. In: Barnhisel RI, Darmody RG, Daniels WL eds. Reclamation of drastically disturbed lands. Monograph 41. Madison, WI, American Society of Agronomy. Pp. 193−216.

PCE (Parliamentary Commissioner for the Environment) 2010. Making difficult decisions: mining the conservation estate. http://www.pce.parliament.nz/media/1301/making-difficult-decisions.pdf

PDEP (Pennsylvania Department of Environmental Protection) 2001.The Science of Acid Mine Drainage and Passive Treatment, Pennsylvania Department of Environmental Protection.

Pearce DW, RK Turner 1990. The economics of natural resources and the environment. Baltimore, MD, Johns Hopkins University Press.

Pearce S, Scott PW 2015. Waste rock dump geochemical evolution: matching lab data, models and predictions with reality. 10th International Conference on Acid Rock Drainage & IMWA Annual Conference, Santiago, Chile. Pp. 1−11.

Peck P, Sinding K 2009. Financial assurance and mine closure: stakeholder expectations and effects on operating decisions. Resources Policy 34(4): 227−233.

Peterson DE, Kindley MJ 1993. Contaminated water management for the Golden Cross Mine: the changing world of mining. Proceedings of the 27th Annual Conference, AusiMM, Wellington. Pp. 277−288.

Plumlee GS, Logsdon MJ 1999a. The environmental geochemistry of mineral deposits. Part A: Processes, techniques and health issues. Reviews in Economic Geology, Vol. 6A. Chelsea, UK, Society of Economic Geologists.

Plumlee GS, Logsdon MJ 1999b. The environmental geochemistry of mineral deposits. Part B: Case studies and research topics. Reviews in Economic Geology, Vol. 6B. Chelsea, UK, Society of Economic Geologists.

Pope J, Christenson H, Newman N, Gordon K 2016a. Acid release decay curve for Brunner Coal Measures. AusIMM New Zealand Branch Conference, Wellington, pp. 342−348.

Pope J, Rait R, Newman N, Hay S, Rogers M, McCracken L 2011. Geochemical studies of waste rock at the proposed Escarpment open cast mine, Denniston Plateau, West Coast. AusIMM New Zealand Branch Annual Conference, Queenstown, pp. 369−380.

Pope J, Trumm D 2014. New Zealand coal acid mine drainage: mineral control on acidity and downstream chemical evolution. 12th IMWA Congress Interdisciplinary Response to Mine Water Challenges, IMWA, Xuzhou, p. 5.

Pope J, Trumm D 2015. Controls on Zn concentrations in acidic and neutral mine drainage from New Zealand's bituminous coal and epithermal mineral deposits. Mine Water and the Environment, v. 34, Special Issue: Mine Water Research in New Zealand, pp. 455−463.

Pope J, Weber P 2013. Interpretation of column leach characteristics of Brunner Coal Measures for mine drainage management. AusIMM New Zealand Branch Annual Conference, Nelson, pp. 377−385.

Pope J, Weber P, MacKenzie A, Newman N, Rait R 2010. Correlation of acid base accounting characteristics with the geology of commonly mined coal measures, West Coast and Southland, New Zealand. New Zealand Journal of Geology and Geophysics 53: 153−166.

Pope J, Weber P, Olds W 2016b. Control of acid mine drainage by managing oxygen ingress into waste rock dumps at bituminous coal mines in New Zealand. In: Drebenstedt C, Paul M eds. International Leipzig, Mine Water Association. Pp. 368−376.

Quinn JM, Davies-Colley RJ, Hickey CW, Vickers ML, Ryan PA 1992. Effects of clay discharges on streams. 2. Benthic invertebrates. Hydrobiologia 248: 235−247.

Radcliffe JE 1974. Seasonal distribution of pasture production in New Zealand. II. Southland. New Zealand Journal of Experimental Agriculture 2(4): 341−348.

Radcliffe JE, Cossens GG 1974. Seasonal distribution of pasture production in New Zealand III. Central Otago. New Zealand Journal of Experimental Agriculture 2(4): 349−358.

Rajaram V, Glazer A, Coghlan G 2001. Methodology for estimating the costs of treatment of mine drainage. 17th International Mining Congress and Exhibition of Turkey-IMCET. Also available at:

http://www.maden.org.tr/resimler/ekler/ee1bc7fa5da061b_ek.pdf. Complete worksheets are included in: US Department of the Interior, Office of Surface Mining (OSM) 2000. Methodology for estimating the costs of treatment of mine drainage. Prepared by Tetra Tech EM Inc. under Contract No. 143868-CT99-12063.

Reid D, Quinn J 2011. Preliminary information for developing sediment guidelines for streams of the West Coast, New Zealand. Hamilton. Prepared for West Coast Regional Council. NIWA Client Report HAM2011-012.

Roberts AHC, Thomson NA 1984. Seasonal distribution of pasture production in New Zealand. XVIII. South Taranaki. New Zealand Journal of Experimental Agriculture 12(2): 83−92. https://doi.org/10.1080/03015521.1984.10421416

Ross CW, Widdowson JP 1985. Restoration research after opencast mining in Southland. New Zealand Soil News 33(5): 163−169.

Rowe DK, Dean TL 1998. Effects of turbidity on the feeding ability of the juvenile migrant stage of six New Zealand freshwater fish species. New Zealand Journal of Marine and Freshwater Research 32: 21–29.

Ruckstuhl K, Gale K, Carter L, Ellison E, Flack S, Russell K 2017. Ka Runaka expectations for oil and gas companies in East Otago. Kai Tahu ki Otago Ltd.

Rufaut CG, Craw D 2010. Geoecology of ecosystem recovery at an inactive coal mine site, New Zealand. *Environmental Earth Sciences* 60: 1425-1437.

Ryan PA 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25: 207–221.

Sawyer J, Stanley R 2012. Criteria for the identification of significant ecological areas in Auckland. Auckland, Auckland Council.

Simcock R, Ross C 2014. Guidelines for mine rehabilitation in Westland. Envirolink Advice grant: 937-WCRC83. Landcare Research. Prepared for West Coast Regional Council.
http://www.wcrc.govt.nz/Documents/Environmental%20Management/Guidelines%20for%20mine%20rehabilitation%20in%20Westland.pdf

Simmons J, Ziemkiewicz P, Black DC 2002. Use of steel slag leach beds for the treatment of acid mine drainage. Mine Water and the Environment 21: 91−99.

Simpson M, Strimc-Palinkas S, Mauk J, Bodnar R 2016. Fluid inclusion chemistry of adularia-sericite epithermal Au-Ag deposits of the Southern Hauraki Goldfield, New Zealand. Economic Geology 101: 763−786.

Sinclair E 2018. Acid base accounting and modelling the acid generating potential of waste rock at Canterbury Coal Mine, Malvern Hills. In Proceedings AusIMM New Zealand Branch Annual Conference, Tauranga, p. 295.

Singer PC, Stumm W 1970. Acid mine drainage: the rate determining step. Science 167: 1121−1123.

Skousen J, Lilly R, Hilton T 1993. Special chemicals for treating acid mine drainage. Green Lands 23(3): 34–41.

Skousen J, Sextone A, Ziemkiewicz PF 2000. Acid mine drainage control and treatment. In: Barnhisel RI, Darmody RG, Daniels WL eds. Reclamation of drastically disturbed lands. Monograph 41. Madison, WI, American Society of Agronomy. Pp. 131−168.

Skousen J, Ziemkiewicz P 2005. Performance of 116 passive treatment systems for acid mine drainage. Proceedings 2005 National Meeting of the American Society for Mining and Reclamation, 19−23 June 2005, Breckenridge, CO, pp. 1100−1133.

Skousen J, Zipper CE, Rose A, Ziemkiewicz PF, Nairn R, McDonald LM, Kleinmann RL 2017. Review of passive systems for acid mine drainage treatment. Mine, Water, and the Environment 36: 133−153.

Standards New Zealand 1998a. Water quality sampling – Part 1. Guidance for the design of sampling programmes, sampling techniques and the preservation and handling of samples. Australian/New Zealand sample 5667.1. Wellington, Standards New Zealand.

Standards New Zealand 1998b. Water quality sampling – Part 2. Guidance on sampling groundwaters. Australian/New Zealand sample 5667.11. Wellington, Standards New Zealand.

Stark J, Boothroyd IGK, Harding JS, Maxted JR, Scarsbrook MR 2001. Protocols for sampling macroinvertebrates in wadeable streams. New Zealand Macroinvertebrate Working Group Report No. 1 (SMF Project No. 5103). Prepared for the Ministry for the Environment, Wellington.

Stumm W, Morgan JJ 1996. Aquatic chemistry: chemical equilibria and rates in natural waters. 3rd edn. Wiley-Interscience.

Terrence JT, Black JP 2000. Topographic reconstruction: the theory and practice. In: Barnhisel RI, Darmody RG, Daniels WL eds. Reclamation of drastically disturbed lands. Monograph 41. Madison, WI, American Society of Agronomy. Pp. 41−76.

Trumm D 2010. Selection of active and passive treatment systems for AMD: flow charts for New Zealand conditions. New Zealand Journal of Geology and Geophysics 53: 195−210.

Trumm D, Black A, Cavanagh J, Harding J, de Joux A, Moore TA, O'Halloran K 2003. Developing assessment methods and remediation protocols for New Zealand sites impacted by Acid Mine Drainage (AMD). Sixth International Conference on Acid Rock Drainage, 12−18 July, Cairns, Queensland, Australia, pp. 223−232.

Trumm D, Black A, Gordon K, Cavanagh J, de Joux A 2005. Acid mine drainage assessment and remediation at an abandoned West Coast coal mine. In: Moore TA, Black A, Centeno JA, Harding JS, Trumm DA eds. Metal contaminants in New Zealand, sources, treatments, and effects on ecology and human health. Christchurch, Resolutionz Press. Pp. 317−342.

Trumm D, Cavanagh JE 2006. Investigation of remediation of acid-mine-impacted waters at Cannel Creek. Landcare Research Contract Report LC0506/169, CRL Report 06-41101. Prepared for West Coast Regional Council. Available at: http://www.wcrc.govt.nz/Resources/Documents/Publications/Environmental Management/BellvueMineRemediation.pdf.

Trumm D, Christenson H, Pope J, Watson K, Mason K, Squire R, McDonald G, Mazzetti A 2017. Passive treatment of Fe and Mn using vertical flow reactors, limestone leaching beds, and slag leaching beds, Waihi Gold, AusIMM New Zealand Branch Conference, Christchurch. Pp. 334−343.

Trumm D, Christenson H, Pope J, Watson K, Mason K, Squire R, McDonald G, Mazzetti A 2018. Treatment of high Mn concentrations at Waihi Gold Mine, New Zealand by two methods: A limestone leaching bed and a slag leaching bed. Editors: Wolkersdorfer C, Sartz L, Weber A, Burgess J, Tremblay G. In: Proceedings of the 11th International Conference on Acid Rock Drainage, Pretoria, South Africa, 10-14 September 2018.

Trumm D, Pope J 2015. Passive treatment of neutral mine drainage at a metal mine in New Zealand using an oxidising system and a slag leaching bed. Mine Water and the Environment, v. 34, Special Issue: Mine Water Research in New Zealand, pp. 442−454.

Trumm D, Pope J, West R, Weber P 2016. Bellevue Mine – downstream geochemistry and proposed treatment, AusIMM New Zealand Branch Conference, Wellington, p. 419–429.

Trumm D, Pope J, West R, Weber P 2017. Downstream geochemistry and proposed water treatment − Bellevue Mine AMD, New Zealand, 13th International Mine Water Association Congress − Mine Water and the Circular Economy, Lappeenranta, Lappeenranta University of Technology, pp. 580−587.

Trumm D, Watts M, Pope J, Lindsay P 2008. Using pilot trials to test geochemical treatment of acid mine drainage on Stockton Plateau. New Zealand Journal of Geology and Geophysics 51: 175−186.

US Environmental Protection Agency (USEPA) 2000. Wastewater treatment technologies. In: Development document for the centralized waste treatment point source category. Available at: http://www.epa.gov/guide/cwt/final/develop/

US Environmental Protection Agency (USEPA) 2004. Primer for municipal wastewater treatment systems. Office of wastewater management. EPA 832-R-04-001. Available at: http://www.epa.gov/owm/primer.pdf

Waters JC, Santomartino S, Cramer M, Murphy N, Taylor JR 2003. Acid rock drainage treatment technologies: identifying appropriate solutions. Proceedings Sixth International Conference on Acid Rock Drainage (ICARD), 12−18 July 2003, Cairns, Australia, pp. 831−843.

Watzlaf GR, Schroeder KT, Kairies CL 2000. Long-term performance of anoxic limestone drains. Mine Water and the Environment 19: 98−110.

Weber P, Olds W, Crombie F, Thomas D, Pope J, Pizey M 2013. Acid mine drainage investigations at the Reddale Coal Mine, Reefton. AusIMM New Zealand Branch Annual Conference, Nelson, pp. 535−545.

Weber P, Scott P, O'Kane M 2014. Derivation of acidity loads based on oxygen flux and net percolation: implicatons for closure. AusIMM New Zealand Annual Branch Conference, Hamilton, p. 605.

Williams DJ 1990. Coal mine tailings disposal alternatives. International Coal Engineering Conference, Sydney, Australia.

Williams DJ 1991. Developments in coal mine tailings disposal and rehabilitation. Queensland Coal Symposium, Brisbane, Australia.

Wilson GW 2008. Why are we still battling ARD? In "Proceedings of the Sixth Australian Workshop on Acid and Metalliferous Drainage", Burnie, Tasmania. 15-18 April 2008. (Eds LC Bell, BMD Barrie, B Baddock, and RW MacLean) pp. 101 - 112 (ACMER: Brisbane).

Younger PL, Banwart SA, Hedin RS 2002. Mine water: hydrology, pollution, treatment. Springer Netherlands.

Ziemkiewicz PF, Skousen JG, Brant DL, Sterner PL, Lovett RJ 1997. Acid mine drainage treatment with armoured limestone in open limestone channels. Journal of Environmental Quality 26: 1017−1024.

Ziemkiewicz PF, Skousen JG, Lovett RJ 1994. Open limestone channels for treating AMD: a new look at an old idea. Green Lands 24(4): 31−38.

Ziemkiewicz PF, Skousen JG, Simmons JS 2003. Long-term performance and cost benefit analysis of passive acid mine drainage treatment systems installed in the Appalachian coal region of the Eastern United States. Proceedings Sixth International Conference on Acid Rock Drainage (ICARD), 12−18 July 2003, Cairns, Australia, pp. 855−862.

Zipper C, Jage C 2001. Passive treatment of acid-mine drainage with vertical-flow systems. Publication 460-133. VA, USA, Virginia Polytechnic Institute and State University, Virginia Cooperative Extension.

Zurbuch PE 1996. Early results from calcium carbonate neutralization of two West Virginia rivers acidified by mine drainage. 17th Annual West Virginia Surface Mine Drainage Task Force Symposium, 2−3 April 1996, Morgantown, WV.

Appendix 1. Ecosystem Services Review approach

A full list of ecosystem services typically considered in the early stages of engagement is provided in the table below. Through facilitated workshops, the ecosystem services of importance to the stakeholders are identified, which may include ones not considered by the mining company. Thus, this approach can support thinking about the requirements for a post-closure landscape and the receiving environments during the design of a mine plan. Ideally, the post-closure landscape provides the same combination and quality of ecosystem services as it did before the start of mining. Where this is not (fully) possible, knowing which ecosystem services are most strongly diminished during mining helps target avoidance and mitigation plans.

Table A1. List of ecosystem services to consider in structured engagement with stakeholders.

ECOSYSTEM SERVICES DEPENDENCE AND IMPACT QUESTIONNAIRE

Site										
Assessment scope:	Company operations									
Scope details:		***Company DEPENDENCE on ecosystem services***			***Community DEPENDENCE on ecosystem services***		***Mining IMPACT on ecosystem services***			
		1. Does this ecosystem service serve as an input or does it enable/ enhance conditions for successful company performance?	**2. Does this ecosystem service have cost-effective substitutes?**	**Applicability to NZ Mining**	**3. Is this ecosystem service relevant to the community?**	**Applicability to NZ mining**	**4. Do construction and operations affect the quantity or quality of this ecosystem service?**	**5. Are mining's impacts positive or negative?**	**6. Can negative impacts be avoided, mitigated, or restored?**	**Comments or supporting information**
Ecosystem services	**Definitions**	If 'no' skip to next ecosystem service			If 'no' skip to the next ecosystem service		If 'no' skip to the next ecosystem service	Positive: The company increases the quantity or quality of this ecosystem service Negative: The company decreases the quantity or quality of this ecosystem service		

Provisioning													
Food	Crops	Cultivated plants or agricultural produce harvested by people for human or animal consumption as food. Examples: grains, vegetables, fruit				Likely not applicable			If (part of) the intended mine site is currently used for crops				
	Livestock	Animals raised for domestic or commercial consumption or use. Examples: sheep, pigs, deer, cattle				Likely not applicable			If (part of) the intended mine site is currently used for livestock				
	Capture fisheries	Wild fish captured through non-farming methods. Examples: local fish				Likely not applicable			If the intended mine site affects downstream water quality during or after operations				
	Aquaculture	Fish, shellfish, and/or plants that are bred and reared in ponds, enclosures, and other forms of fresh- or salt-water confinement for purposes of harvesting. Examples: shrimp, oysters, salmon				Shellfish can be used for AMD treatment			If the intended mine site affects downstream water quality during or after operations				

	Wild foods	Edible plant and animal species gathered or captured in the wild. Examples: fruit and nuts, fungi, mahinga kai				Likely not applicable			If (part of) the intended mine site is in area used for hunting or gathering					
Biological raw materials	Timber and other wood fibre	Products made from trees harvested from natural forest ecosystems, plantations, or non-forested lands. Examples: industrial roundwood, wood pulp, paper				Likely not applicable			If the intended mine site is in woodlands that currently provide wood for products sold in markets					
	Fibres and resins	Non-wood and non-fuel fibres and resins extracted from the natural environment. Examples: flax, twine and rope				Likely not applicable			If the intended mine site is in woodlands that currently provide, e.g. flax					
	Animal skins	Processed skins of cattle, deer, pig, snakes, sting rays, or other animals. Examples: leather, rawhide				Likely not applicable.			If (part of) the intended mine site is currently used for livestock					
	Sand	Sand formed from coral and shells. Examples: White sand from coral				Likely not applicable.			Likely not applicable					

	Ornamental resources	Ecosystem-derived products that serve aesthetic purposes. Examples: wild flowers, feathers, pounamu				Likely not applicable			Potentially applicable (if the intended mine site (partially) overlaps with an area where ornamental goods are collected – this can be stakeholder specific)					
	Biomass fuel	Biological material derived from living or recently living organisms – both plant and animal –that serves as a source of energy. Examples: fuelwood, charcoal, grain for ethanol production, dung				Likely not applicable			If (part of) the intended mine site is in woodlands that currently provide fuel wood					
	Freshwater	Inland bodies of water, groundwater, rainwater, and surface waters for household, industrial, and agricultural uses. Examples: freshwater for drinking, cleaning, cooling, industrial processes, electricity generation, or				Applicable (see also water regulation and purification)			If the intended mine site redirects water away from community usage (agriculture, recreation, …)					

		mode of transportation												
	Genetic resources	Genes and genetic information used for animal breeding, plant improvement, and biotechnology. Example: genes used to increase crop resistance to disease				Likely not applicable			Likely not applicable					
	Biochemicals, natural medicines, and pharmaceuticals	Medicines, biocides, food additives, and other biological materials derived from ecosystems for commercial or domestic use. Examples: mānuka honey				Likely not applicable			If (part of) the intended mine site is in areas where the community gathers medicinal resources					
Regulating														

	Maintenance of air quality	Influence ecosystems have on air quality by emitting chemicals to the atmosphere (i.e. serving as a 'source') or extracting chemicals from the atmosphere (i.e. serving as a 'sink'). Examples: lakes serve as a sink for industrial emissions of sulphur compounds; tree and shrub leaves trap air pollutants from roadways				If dust affects operations			If dust or other airborne contamination is generated during construction, operations or rehabilitation					
Regulation of climate	Global climate regulation	Influence ecosystems have on the global climate by emitting greenhouse gases or aerosols to the atmosphere or by absorbing greenhouse gases or aerosols from the atmosphere. Examples: forests capture and store carbon dioxide; cattle and rice paddies emit methane				Likely not applicable			Not applicable under RMA.					

	Regional/local climate regulation	Influence ecosystems have on local or regional temperature, precipitation, and other climatic factors. Example: forests can impact regional rainfall levels				Likely not applicable			Potentially applicable if large landscape elements disappear or are created (e.g. mountaintops, forests, wetlands)					
	Regulation of water timing and flows	Influence ecosystems have on the timing and magnitude of water runoff, flooding, and aquifer recharge, particularly in terms of the water storage potential of the ecosystem or landscape. Examples: permeable soil facilitates aquifer recharge; river floodplains and wetlands retain water, which can decrease flooding during runoff peaks, reducing the need for engineered flood control infrastructure				If predictable, non-extreme water flows matter to processing			Potentially applicable during operations (compacted soils contributing to more extreme floods)					

Erosion control	Role ecosystems play in retaining and replenishing soil and sand deposits Examples: vegetation such as grass and trees prevents soil loss due to wind and rain; forests on slopes hold soil in place, thereby preventing landslides				Likely not applicable			Potentially applicable during operations					
Water purification and waste treatment	Role ecosystems play in the filtration and decomposition of organic wastes and pollutants in water; assimilation and detoxification of compounds through soil and subsoil processes. Examples: wetlands remove harmful pollutants from water by trapping metals and organic materials; soil microbes degrade organic waste, rendering it less harmful				If water quality matters to processing			If (part of) the intended mine site affects wetlands that currently provide this service					

Disease mitigation	Influence ecosystems have on the incidence and abundance of human pathogens. Example: intact forests reduce the occurrence of standing water—a breeding area for mosquitoes—and thereby can reduce the prevalence of malaria				Likely not applicable			Likely not applicable					
Maintenance of soil quality	Role ecosystems play in sustaining soil's biological activity, diversity and productivity; in regulating and partitioning water and solute flow; and in storing and recycling nutrients and gases. Example: some organisms aid in decomposition of organic matter, increasing soil nutrient levels; some organisms aerate soil, improve soil chemistry, and increase moisture retention; animal				Likely not applicable			Potentially applicable during operations					

	waste fertilises soil												
Pest mitigation	Influence ecosystems have on the prevalence of crop and livestock pests and diseases. Example: predators from nearby forests – such as bats, toads, snakes – consume crop pests				Likely not applicable			If the disturbance promotes weeds and pests beyond the mine site or pests remain after its closure					
Pollination	Role ecosystems play in transferring pollen from male to female flower parts. Example: bees from nearby forests pollinate crops				Likely not applicable			Likely not applicable (requires specific land use/cover patterns in relation to pollinator presence and the mine)					

	Natural hazard mitigation	Capacity for ecosystems to reduce the damage caused by natural disasters such as hurricanes and to maintain natural fire frequency and intensity Examples: biological decomposition processes reduce potential fuel for wildfires				Likely applicable in relation to tailings dams and storage facilities for hazardous materials			Potentially applicable if large landscape elements disappear or are created (e.g., mountaintops, forests, wetlands)					
Cultural														
	Recreation and ecotourism	Recreational pleasure people derive from natural or cultivated ecosystems. Examples: hiking, camping, bird watching, scuba diving, off-road courses in ex-mine sites				Likely not applicable			Potentially applicable					
	Ethical and spiritual values	Spiritual, religious, aesthetic, intrinsic, 'existence', or other values people attach to ecosystems, landscapes, or species. Examples: spiritual fulfilment derived from sacred lands				Likely not applicable			Likely applicable (mahinga kai, use of fibres as part of heritage, mauri of the area)					

		and rivers; people's desire to protect endangered species and rare habitats												
	Educational and inspirational values	Information derived from ecosystems used for intellectual development, culture, art, design, and innovation. Examples: the structure of tree leaves has inspired technological improvements in solar power cells; school field trips to nature preserves help to teach scientific concepts and research skills				Likely not applicable			Potentially applicable					
Supporting														

	Habitat	Natural or semi-natural spaces that maintain species populations and protect the capacity of ecological communities to recover from disturbances Examples: native plant communities often provide pollinators with food and structure for reproduction; rivers and estuaries provide nurseries for fish reproduction and juvenile development; land natural areas and biological corridors allow animals to survive forest fires and other disturbances				Likely not applicable			Applicable, particularly when (part of) the intended mine site is on land with sensitive and/or rare biota					
Other services identified by company														
	Fossil or mineral resources	materials for plant-based species												

Note 1: Table 1 in *The Corporate Ecosystem Services Review* provides a list and definitions of the supporting services for those companies that want to consider them during the ESR.

Note 2: Biodiversity – the variability among living organisms within species and populations, between species, and between ecosystems – is not listed as an ecosystem service because it is not an ecosystem service in itself but rather provides the foundation for ecosystem services. If you add ‘biodiversity’ as another service, recognise that many of your company's dependences and impacts on biodiversity will materialise through one or more of the ecosystem services listed above.