Chapter 36

Impacts of mining

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INTRODUCTION

New Zealand has a long history of mineral extraction, beginning with the discovery of gold in the Coromandel Peninsula in 1852 and Central Otago in the 1861, and coal on the West Coast in the 1860s. Coal was first mined in 1872 at the Albion mine, and soon after underground mines were extensively developed along the Stockton-Denniston Plateau, North Westland, and in the Reefton area, where coal was discovered in close proximity to gold. In Central Otago widespread alluvial gold mining began in the 1860s, while the Ohinemuri goldfield opened in the Coromandel in 1875. Most of the early activity occurred in the Karangahake area, but in 1878 the rich gold-bearing quartz at Martha Hill in Waihi was discovered. Placer mining and gold dredging operations became widespread in Central Otago and Westland in the 1870s and these areas continue to be reworked as both technology and the price of gold improve (Fig. 36.1).

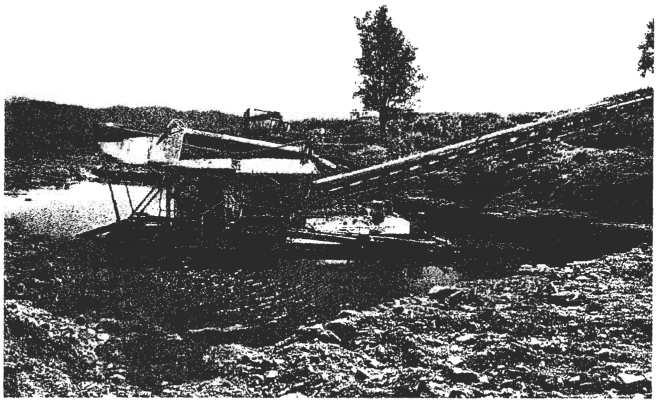

Figure 36.1 Placer mine gold operation in Carton Creek near Reefton. Photo: Jon Harding

Other minerals that have been mined in New Zealand include gravel, tin, copper and uranium.

Of the mineral extraction methods, underground mining is the most widespread in New Zealand and has been used extensively in the coal industry, particularly on the West Coast. Underground mines involve excavating either an inclined shaft (a decline or drift) or a horizontal shaft (an adit), followed by parallel shafts (bord-and-pillar or longwalls), which enable the maximum amount of a coal seam to be extracted. The introduction of hydro-mining (using high-pressure water blasted against the coal face) improved the efficiency of coal extraction, while exacerbating the effects of mine drainage.

Several large opencast mines have been developed, including the Waihi (extracting gold) and Stockton (coal) mines. Opencast mining involves the removal of surface topsoil and rock to expose shallow mineral seams. It creates considerable waste material or overburden, which must then be disposed of. Excavated soil and rock take up 50% more volume than the in-place material, hence back-filling of opencast mines still leaves substantial quantities of excess overburden (Fig. 36.2).

Placer or alluvial mining is the other major extraction technique used in New Zealand. Much of the current gold

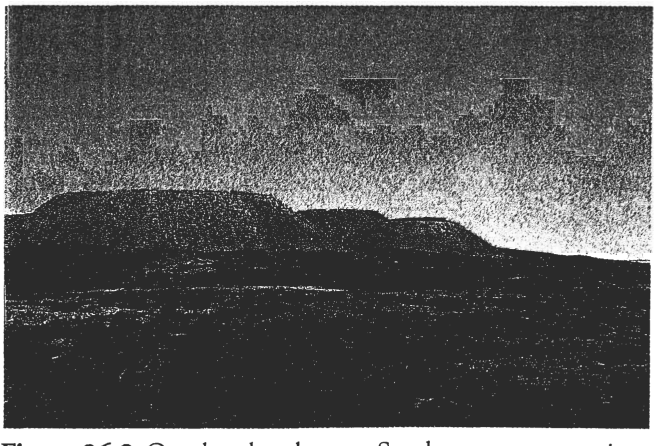

Figure 36.2 Overburden dumps, Stockton opencast mine. Photo: Jon Harding

mining in the South Island is alluvial mining. Gold fines eroded from seams in the mountains are washed naturally down to the valley floors. These heavy gold fines filter into the deep riverbed gravels, and alluvial mining operations excavate the riverbed, often down to bedrock, to sift out the gold.

Each of these three extraction methods can cause marked changes to mining landscapes, and consequently surface water and groundwaters associated with mines may be significantly affected by mine leachate, sediment and by mine operations.

In this chapter we discuss the effects of mining activities on the water quality, physical morphology and biotic communities of freshwater ecosystems. Increasing pressure for improved environmental management has resulted in a greater emphasis on the restoration and remediation of sites affected by current and historic mining activities, hence research is now beginning to focus on these problems.

Much of the mining activity in New Zealand is associated with running water ecosystems, and we will focus on these. No major lakes in New Zealand currently receive mine discharges.

HYDROLOGICAL EFFECTS

Many early mining activities relied heavily on a continual supply of water, either to assist with excavation (as in hydro-mining) or to transport mined material to the surface. In Central Otago, for example, hundreds of kilometres of shallow canals were constructed to transport water from foothill streams to gold-mining digs. Initially, this water was used for washing and sluicing gold fines. Later hydro-mining was developed, in which high-pressure water jets were used to carve soil and gravels from the hillside. In addition, the manipulation of flow conditions is common in placer mining, where flow is temporarily diverted to expose streambeds to be re-worked for alluvial gold. The long-term hydrological impacts of these flow diversions have not been well studied. Prior to and during mining the riverbed is usually cleared of vegetation and riverbed gravels are turned over. In at least one instance on the West Coast, channel diversion lowered the water table, causing the diverted river to dry in late summer (Harding and Greenwood 2003). Placer mining may also involve diversion and storage of water in dredging and sediment-settling ponds, which may also have significant short-term effects on the water table.

In underground mining, the diversion of surface water and groundwaters is also common. However, the long-term effects of altered hydrological regimes are not well documented.

WATER QUALITY

Water chemistry associated with mining activities has been the focus of much investigation. Surface water and groundwaters associated with mines are often affected by acidification, the presence of toxic metals, and sedimentation. Discharges emanating from coal mines can cause the most severe problems. Many coal seams, such as those in the Brunner Coal measures in North Westland, contain pyrite, which has high levels of sulphur. When pyrite is exposed to water and oxygen, several well documented reactions can occur (Singer and Strumm 1970), resulting in the acidification of mine waters.

1) $FeS_2 + 7/2O_2 + H_2O = Fe^{2+} + 2SO_4^{2+} + 2H^+$
2) $Fe^2 + {}^1/_4\, O_2 + H^+ = Fe^{3+} + 1/2H_2O$
3) $Fe^{3+} + 3H_2O = Fe(OH)_2 + 3H^+$
4) $FeS_2 + 14Fe^{3+} + H_2O = 12Fe^{2+} + 2SO_4^{2+} + 16H^+$

Thus the exposure of pyrites to oxygen is a crucial step in the generation of acid mine runoff (sulphuric acid), whereas in natural undisturbed coal formations, where exposure to oxygen is uncommon, acid generation is rare. Acid mine drainage is frequently cited as one of the most important environmental side effects of coal mining.

In environmental monitoring of acid mine drainage, acidity is usually assessed by measuring pH. However, pH is not a true measure of acidity, nor is it necessarily an accurate indicator of the extent of acid mine generation occurring in a system (Kelly 1988). More precisely, pH is a measure of the concentration or activity of hydrogen ions in a solution. In acid generation, the crucial factor is the *availability* of hydrogen ions to neutralise bases, hence, the determining factor controlling acidity is the excess of hydrogen ions over other ions. Thus "total acidity" (measured as $CaCO_3$) is a more accurate measurement of acidity than pH. However, at low acidity there is often a correlation between total acidity and pH, and realistically pH is easy to measure with standard field meters (Fig. 36.3). Above about pH 7 there is rarely any acidity.

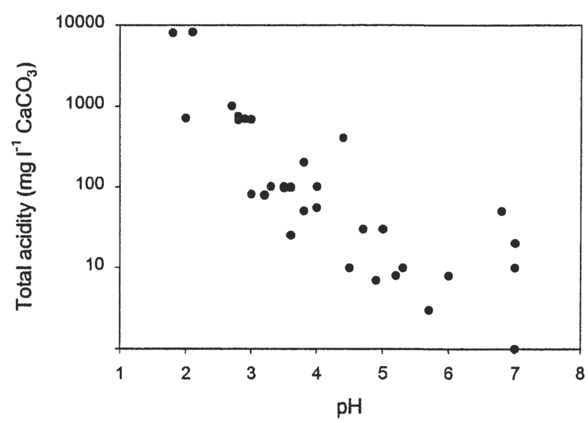

Figure 36.3 Relationship between pH and total acidity (after Kelly 1988).

Acidification can have severe effects on freshwater biota, however from a chemical perspective one of the most important side effects may be the reduction in the bicarbonate buffering capacity of the water. Once pH falls below about 4.2, carbonate and bicarbonate are converted to carbonic acid. The upshot of this is twofold. Firstly, stream water loses its capacity to buffer changes in pH, so that additional acid mine effluents entering a stream may decrease the pH markedly. Secondly, the loss of bicarbonate will have a negative impact on the many photosynthetic organisms that require inorganic carbon. Algae and bryophytes living at pH <4.2 need to be able to process free carbon dioxide in the absence of bicarbonate. Furthermore, even after acid mine inputs have ceased, it may take a stream or river many decades to recover from a loss of buffering capacity.

Lowering of pH can also have other chemical effects. Acidification tends to increase the rate of precipitation of silt and clays, and thus may increase sedimentation on the riverbed. However, depending on the pH, this may also have the positive side effect of reducing the turbidity of the water. Furthermore, increased acidification increases the decomposition of minerals, including feldspars and carbonates, resulting in the release of metals such as aluminium, and the release of silica. The release of silica may stimulate the growth of acidophilic algae, particularly diatoms, whereas aluminium may have several negative impacts. As the sources of acidity in mine drainages are strong mineral acids, acidification in these systems is usually associated with high conductivity (Fig. 36.4).

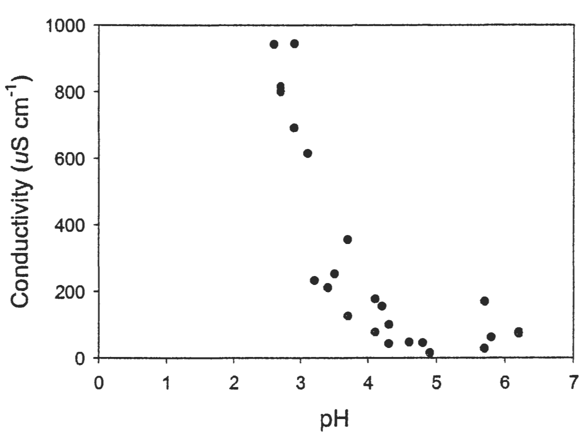

Figure 36.4 Relationship between stream pH and conductivity in 24 South Island streams (after Winterbourn *et al.* 2000).

Mine discharges may markedly reduce the pH of receiving waters, and in New Zealand adits with a pH <2.9 are not uncommon. A range of metals may also be associated with coal formations, including aluminium, arsenic, copper, iron, lead, nickel and zinc (Fig. 36.5). High

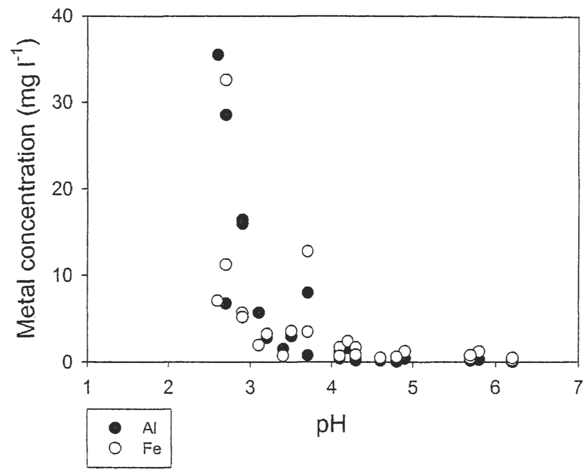

Figure 36.5 – Relationship between pH, and total reactive aluminium and dissolved iron, in 24 South Island streams (after Winterbourn *et al.* 2000).

acidity enhances the solubility of these metals, however as pH increases the chemical nature of the metals is affected. At very low pH, e.g., <3, any metals present remain in solution, however, as pH increases flocs of these metals may form, and if the metals are present in high concentrations precipitates form on the streambed. In particular, the yellow or ochre precipitates of iron (Fe^{3+}) ("yellowboy") are commonly found in streams where iron is associated with exposed coal. This precipitate is composed primarily of iron hydroxide ($Fe(OH_3)$), but may also include a complex of hydrates.

The critical pH value for iron precipitate formation ranges from about 3.5–4.3 (Table 36.1). Below this, Fe^{3+} remains dissolved in stream water and although it is toxic to some biota at high concentrations in this state, it does not cause sedimentation problems. Iron hydroxide precipitate can cause significant modification of the streambed by cementing substrata, clogging interstitial spaces, and covering biota. The substrate "armouring"

Table 36.1 Minimum pH values for precipitation of metal ions as hydroxides (Kelly 1988; Niyogi *et al.* 1999)

Metal	Minimum pH-hydroxide
Sn	4.2
Fe^{3+}	3.5–4.3
Al	4.9–5.4
Pb^{2+}	6.3
Cu^{2+}	7.2
Zn	8.4
Ni	9.3
Fe^{2+}	9.5
Cd	9.7
Mn^{2+}	10.6

effect caused by precipitate reduces refugia for invertebrates and fish, making them more susceptible to floods and disturbance and increasing the potential for predation and competition. Other metals precipitate in streamwater at differing pHs (e.g., aluminium), and several critical pH values are shown in Table 36.1. When acid inputs to a stream are stopped, these precipitates may re-dissolve and the substrate may recover in as little as six months (Niyogi *et al.* 1999). As we discuss later, Fe, Al, and other metals, both dissolved and as precipitates, may be directly toxic to freshwater biota.

Aluminium hydroxide precipitate, seen as a white coating on stones, has been widely documented in North America, whereas in New Zealand it has rarely been reported. However, dissolved Al is a significant problem in many catchments. On the West Coast of South Island naturally acidic brown-water streams are common; this acidity is produced by natural tannins (humic and fulvic acids) derived from decomposing vegetation in temperate rainforests. These naturally low pH waters (down to about pH 4) may have relatively high levels of dissolved Al, which is less toxic than in acid mine drainage waters because the Al is bound with organic carbon (Collier *et al.* 1990; Stenzel and Herrmann 1990).

Different metals have highly variable degrees of toxicity on stream biota, depending on localised conditions. Several workers have shown that Cu can be 13 times more toxic than Zn; however when these two metals occur together the cumulative toxicity can be far greater (Gray 1998). Similarly, numerous studies have shown that dissolved Al is more toxic than dissolved Fe at comparable concentrations. Winterbourn *et al.* (2000) reported concentrations of Fe that were three times greater than Al in the tissues of insects from 24 West Coast Streams. Fe can be photo-reduced by sunlight, and therefore open streams are liable to have Fe at less toxic levels (Niyogi *et al.* 1999). Not surprisingly, seasonal variations in discharges from mines alter the concentration and toxicity of receiving waters. Thus, a stream may be toxic at one time of the year and significantly less toxic at other times. Furthermore, heavy metals are transported in the water column both as dissolved free metal ions and complexes, and as metals bound to suspended sediments. In the Waihou River, Waikato, Cu, Mn and Zn were mainly in dissolved form in river water, whereas Fe was predominantly bound to suspended acid-soluble particulates in neutral pH waters (7.0–7.9). However, in the Waihou River estuary, almost all Fe and Mn had flocculated out, so high sediment metal concentrations occurred there (Webster 1995).

So far we have primarily discussed conditions associated with coal mining. However, alluvial mining may also release minerals associated with buried riverbed material and cause significant turbidity in receiving waters. The successful use of a cyanide process for extracting precious metals was crucial to the further development of gold mining. However, the extraction process also results in other toxic materials (e.g., arsenic, antimony) being released. For example, X-ray diffraction analysis of sediments showed quartz, magnetite, hypersthene, ilmenite, anglesite, hornblende and cummingtonite were present in sediments in the Karangahake region of the Waihou River (Sabti *et al.* 2000). Waste products from cyanide-based metal extraction include toxic sludge. In 1895, following sustained pressure from mining companies, the Government declared (by Proclamation) the Ohinemuri and Waihou Rivers in the Coromandel as sludge channels (Watton 1995). This allowed the discharge of an estimated 250,000 tonnes per annum of mine tailings into the Ohinemuri River.

Placer gold-mining frequently results in significant increases in suspended sediment levels downstream. For example, turbidity has been shown to increase from 2.4 to >100 NTUs (Nephelometric Turbidity Units) in West Coast streams above and below placer operations (Davies-Colley *et al.* 1992). Increased turbidity reduced light penetration by as much as 400% and affected both benthic algal biomass by reducing photosynthesis and benthic invertebrate densities by degrading food quality (Quinn *et al.* 1992).

MODIFICATION TO THE PHYSICAL HABITAT

Gold dredging produces large amounts of sifted gravels, which historically, were deposited as tailings. Numerous West Coast and Otago rivers (e.g., the Grey, and Taramakau) have extensive areas of modified riverbeds. These tailings form convoluted ridge and valley hummocks along the river banks, frequently creating small artificial ponds, reducing or preventing riparian re-vegetation along rivers, altering the river channel morphology, and exposing minerals to weathering and erosion.

In contrast, opencast mining results in different, though no less obvious, modification of the landscape. Overburden dumps where excess pit spoil is piled can form new hills within the mine terrain (Fig. 36.2). These overburden dumps frequently act as sources of leachate and sediment, which enter surface waters associated with the mine. Capping and remediation of overburden dumps is an emerging research challenge.

Sedimentation of waterways can be a major problem associated with excavation and roading at mine sites, as suspended sediment can smother algae, benthic invertebrates and the substrate, reducing substrate heterogeneity (Fig. 36.6).

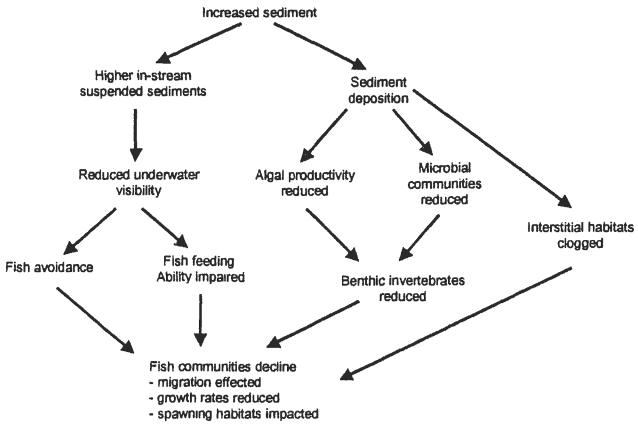

Figure 36.6 Model of the pathways by which suspended sediment might affect components of the stream ecosystem (after Rowe and Dean 1998).

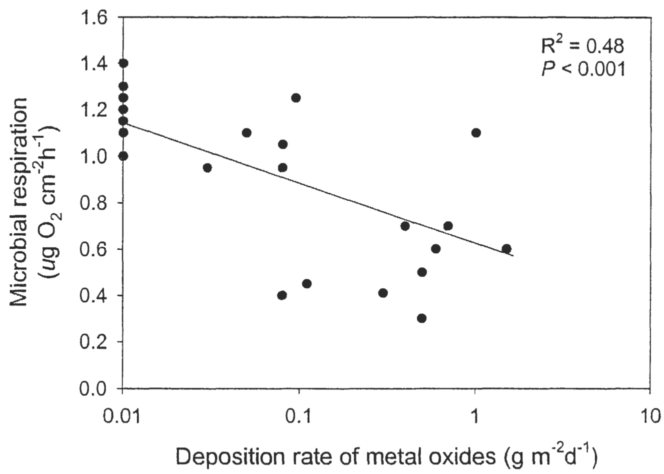

Figure 36.7 Microbial respiration and deposition rate of metal oxides (after Niyogi *et al.* 2001).

BIOTIC RESPONSES

Bacteria and fungi

Alterations in water chemistry, particularly increased acidity and the release of metals, can have a profound effect on microbial communities. Experiments on decomposition rates of leaves in coal mining streams have generally shown greatly reduced microbial activity, although Harbrow (2001) found highly variable breakdown rates in six West Coast streams. Winterbourn *et al.* (1985) observed a lack of fungi on stones in a stream with a natural pH 4.3, while Hildrew *et al.* (1984) noted impaired cellulolytic decomposition in streams with pH <5.6. The mechanisms that lead to a reduction in microbes at low pH are unclear, but there are at least two. Firstly, high acidity inhibits microbial enzymes. For example, the activity of pectin-degrading enzymes such as pectin lyase, which is involved in softening plant tissue in the early stages of leaf breakdown, is lower at low pHs (Suberkropp and Klug 1981). Secondly, reduced litter breakdown rates have been linked to metal oxide deposition, and Niyogi *et al.* (2001) found that microbial respiration decreased with increasing rates of deposition of metal oxides on leaf litter (Fig. 36.7). Thus, the precipitates themselves may either smother microbial complexes or be directly toxic to them.

Consequently, in lakes and rivers affected by mining and not subject to flushing flows from floods, coarse particulate organic matter may accumulate on the bed because natural decomposition is suppressed. Reduction in microbial processing of allochthonous inputs may also significantly reduce the availability of dissolved and fine organic matter within the system. These interrelationships, when coupled with a reduction in the numbers of invertebrate shredders due to toxic conditions, can combine to reduce overall allochthonous breakdown in mine drainage streams (Fig. 36.8).

The importance of metal oxide deposition has been confirmed in post-remediation studies in which litter breakdown rates failed to recover even after water quality improved. In some instances the presence of metal oxides may have continued to limit microbial activity. Poor or depauperate food resources frequently have been cited as additional confounding factors in the recovery of benthic invertebrate communities in mine drainage streams after remediation, however this has not been well studied in New Zealand.

The presence of high concentrations of dissolved metals has little effect on some species, particularly iron bacteria that occur in some systems affected by mining. Winterbourn *et al.* (1985) found the iron bacterium *Leptothrix* sp. dominated a stream with total iron concentrations of 6 g m^{-3}, and was eaten by a chironomid that lived on it. Little is known about the ability of other organisms to use these food resources.

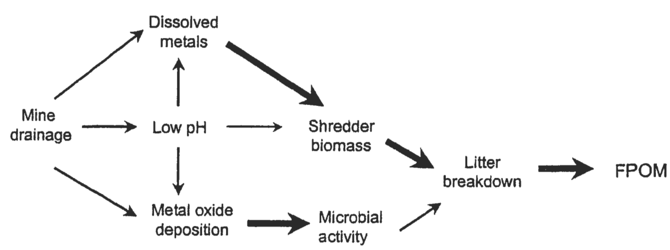

Figure 36.8 Model of the effects of mine drainage on leaf litter breakdown and the production of fine particulate organic matter (FPOM). The thickness of the arrow indicates the strength of the effect (modified from Niyogi *et al.* 2001).

Aquatic plants

Vascular plants are usually absent from acidic streams, but bryophytes and periphyton can be locally very abundant. Some algae are acidophilic and can occur in high densities in acidic sites with stable flows. Where acidic algae proliferate, communities are frequently dominated by a few species, with high biomass. Acid-tolerant algae, such as *Ulothrix* sp., have been shown to account for 99% of the algal biomass at mine discharges, with other filamentous algae such as *Microspora*, and *Tribonema* common (Niyogi *et al.* 1999; Winterbourn *et al.* 2000) (Fig. 36.9).

Figure 36.9 Algae and iron hydroxide precipitates at Sullivans West adit, Denniston Plateau, North Westland
Photo: Jon Harding

Like microbial communities, algae may proliferate at sites with low pH, stable flows and low metal oxide deposition (e.g., mine adits), but as soon as metal deposition rates increase (e.g., when pH rises above 3.5–4.3) algae can survive only if they are able to grow faster than the rate at which oxides smother their surfaces. Experimental manipulations of pH have shown that algal biomass can increase until Al precipitation occurs (at about pH 4.9) and periphyton growth is inhibited.

Where the substratum is relatively stable, bryophytes (mosses and liverworts) occur in naturally and mining-affected low-pH streams. The liverworts *Lophocolea*, *Jungermannia* and *Riccardia*, and mosses such as *Blinda* and *Sphagnum*, have been recorded in low pH streams in New Zealand (Winterbourn *et al.* 2000). Some bryophytes accumulate metals and, in particular, Fe. Winterbourn *et al.* (2000) reported concentrations of Fe 10 times higher than Al in bryophyte and algal tissue in West Coast streams. Concentrations of metals can reach high levels; Englemann and McDiffett (1996) reported concentrations of 17.3 and 9.1 μg/g dry mass of Fe in two species of bryophytes. Fe^{3+} is practically insoluble, and moderately toxic to plants. It competes with other substances for binding sites on cell membranes and is then taken into the cell where it accumulates (Englemann and McDiffett 1996). It has been suggested, however, that Fe can actually decrease the toxic effect of other metals by competing for binding sites, and by the partial sequestration of other metals by Fe colloids. Al has also been shown to accumulate in bryophytes, however Al absorption in moss usually peaks in tissue at intermediate pHs, i.e., 5.2–5.8 (Englemann and McDiffett 1996; Winterbourn *et al.* 2000). The reason for this is probably that the solubility of Al increases markedly below pH 5, where the more toxic Al^{3+} is more prevalent. Hence the higher concentration of Al in plants at intermediate pH may be a result of bioaccumulation during periods of lower pH, and poor solubility as pH rises. The low accumulation of Al at low pH may be due to the domination of Al^{3+}, which has been shown to cause mucilage production and root necrosis in duckweed (Crowder 1991). Where they are able to survive, higher-order aquatic plants can also accumulate metals. Sabti *et al.* (2000) found that aquatic macrophytes (*Egeria densa*) in the Ohinemuri and Waitekauri Rivers contained gold (302 and 672 $\mu g\ kg^{-1}$ dried matter respectively), and suggested that plants may be useful as bioindicators of heavy metal contamination.

Benthic invertebrate communities

The impacts of mine drainage on stream invertebrates are almost entirely negative and have been termed "acidaemia" by Kelly (1988). The effects range from acute and direct toxicity caused by combinations of low pH and/or the presence of toxic metals, to indirect effects in areas where food resources are limited (by reducing organic matter processing and algal growth), to altered in-stream habitat (by armouring and clogging of the riverbed substrata).

In naturally acidic brown-water streams (pH 4.3–5.7) benthic invertebrate communities often have fewer species, lower densities and altered community composition compared to communities in similar-sized natural streams (Collier and Winterbourn 1987). Collier and Winterbourn (1987) suggested that the depauperate state of naturally acidic brown-water streams was probably a result of changes in the food supply, particularly a reduction in periphyton biomass (dominated by diatoms such as *Eunotia* and *Fragilaria* spp.). By comparison, streams affected by acid mine drainage may be almost devoid of species (Fig. 36.10), and where organisms are present their densities may be as low as a few animals per square metre.

Naturally acidic waters are not uncommon on the West Coast of New Zealand, and it is apparent that some benthic invertebrate species are well adapted to the conditions found in low pH waters (Winterbourn and McDiffett 1996). Naturally acid brown-water streams in South Westland typically have communities dominated by the common leptophlebiid mayfly *Deleatidium*, Chironomidae, the elmid beetle *Hydora*, and the stonefly

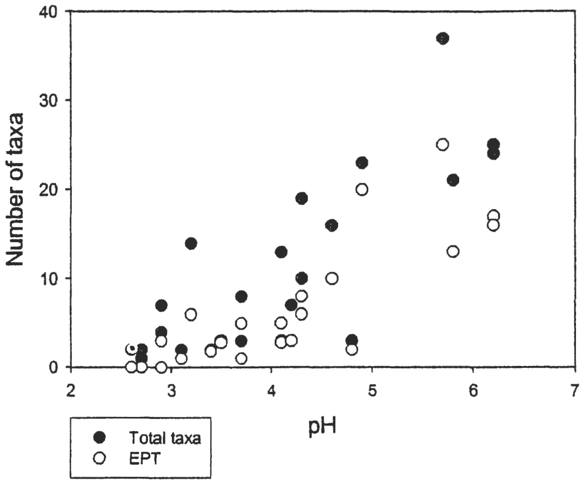

Figure 36.10 Relationship between total taxa and pH in 23 streams in South Island (after Winterbourn *et al.* 2000).

Zelandobius. However, in streams contaminated by acid mine drainage in North Westland (pH 2.6–4.2), a range of taxa were recorded in low densities, including the mayflies *Deleatidium* and *Austroclima sepia*, the stonefly *Spaniocercoides philpotti*, the caddisflies *Psilochorema* and *Oxyethira albiceps*, scirtid beetles and dipterans, including *Chironomus zealandicus* and *Eukiefferiella* (Boothroyd 2002; Harding 2002). Sites with pH <3 in the Denniston and Stockton Plateaus were dominated by chironomids and scirtids, but included two caddisflies—*Pseudoeconesus* and *Kokiria miharo* (Winterbourn 1998). Although Crustacea are frequently considered intolerant of acidity, Anthony (1999) found the amphipod *Paraleptamphopus* below pH 4, and the crayfish *Paranephrops* occur in naturally low pH waters (Collier *et al.* 1990).

Exposure to low pH and elevated metal concentrations may cause a number of physiological stresses to benthic invertebrates. Numerous studies have shown osmoregulation in many species is impaired by the disruption of acid base and ion balances, nitrogen excretion and respiration (Sutcliffe and Hildrew 1989). Low pH also causes a reduction in sodium uptake, detrimental to sodium-sensitive species such as crayfish (Haines 1981). Similarly, Cl, Ca and K can all be lowered in high-acidity waters (Rowe *et al.* 1989), while Bell (1971) showed that low pH could be lethal at critical junctures of the life cycles of stream insects. Thus, trichoptera adults had only 50% successful emergence at pH 4, while mayflies showed 50% adult emergence failure at pH 5.9. However, a number of New Zealand mayflies and caddis are clearly capable of successful emergence at low pH, although this may differ between natural and mine waters. The dominance of benthic communities by Chironomidae at low pH may relate to the ability of these taxa to tolerate ion imbalance (Forsyth 1983; Boothroyd 2002).

Low pH may also affect some species by making them more susceptible to disease, infection and parasitism. Leuven *et al.* (1986) found that amphibian eggs were unable to develop at pH <3.5 due to infection from pathogenic fungi, while the crayfish *Orconectes virilis* suffered egg mortality from protozoan parasites when the pH dropped below 5.6 (Schindler and Turner 1982). A high frequency of infections (visible sores) has been noted in benthic invertebrates from streams receiving discharge from abandoned gold mines near Reefton. This suggests that toxins in these mine waters may weaken benthic organisms, making them susceptible to disease and infection.

Toxicity of metals can be a major problem in many mining-affected waters. Toxocological studies have shown insects, crustacea and fish are frequently susceptible to toxic metals. Hickey and Clements (1998) noted that net-spinning hydropsychid caddis and orthoclad chironomids dominated at sites high in metals in the Coromandel Peninsula. Analysis of taxonomic richness data from North Westland shows that relatively few species are found in waters with high concentrations of dissolved Al and Fe (Fig. 36.11).

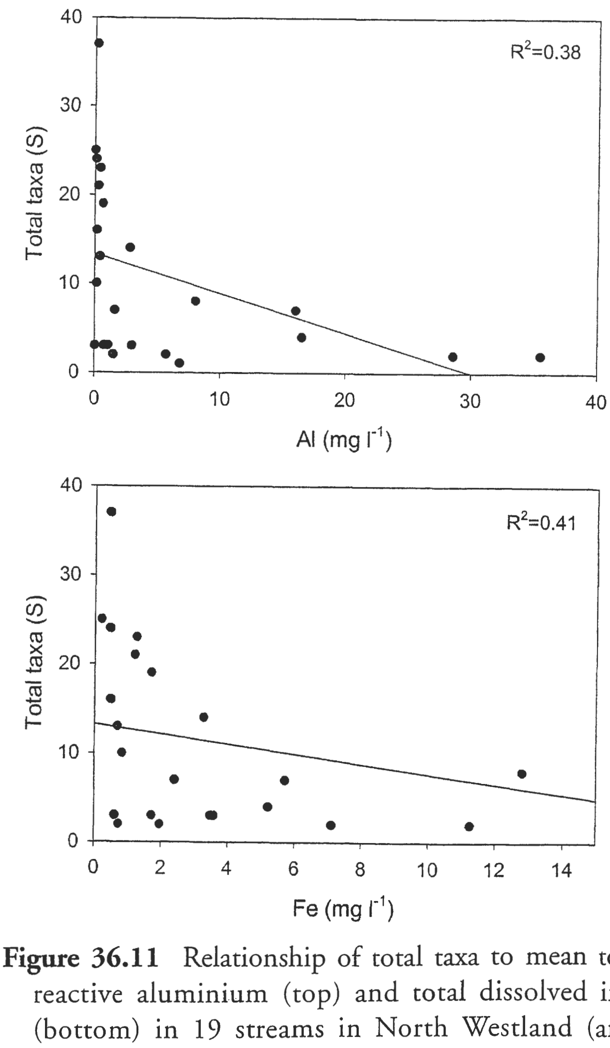

Figure 36.11 Relationship of total taxa to mean total reactive aluminium (top) and total dissolved iron (bottom) in 19 streams in North Westland (after Harbrow 2001).

Fish communities

Most fish species are negatively affected by acidification and mine leachate. These toxic effects may be acute—causing death, or chronic—resulting in impaired health (e.g., mucous secretion on gills impairing gas exchange), or physiological—from stress that reduces fish condition. Accumulation of heavy metals within the flesh of fish may have long-term toxic effects (Table 36.2). Furthermore, the presence of mine discharges may create a chemical barrier to diadromous species, reducing or preventing their migration to and from the sea. Several diadromous native fish have been recorded in low pH waters (Main 1988) and have been shown to be able to detect pH gradients and display pH preferences. In particular, short-finned eels (*Anguilla australis*) and two whitebait species, koaro (*Galaxias brevipinnis*) and banded kokopu (*Galaxias fasciatus*), prefer waters <6.5, whereas other species avoided low pHs in laboratory trials (West *et al.* 1997). These pH preferences were stronger in adults than juveniles, and probably account for the presence of galaxiids in naturally brown-water streams on the West Coast. The freshwater shrimp *Paratya curvirostris* seemed unable to detect pH changes, and had high mortality in high pH waters. These findings are consistent with Collier *et al.* (1990), who reported widespread tolerance of low pH in naturally acidic streams in Westland, with 9 out of 14 native fish species reported in pH <5, and 7 species at pH <4.5. Particularly low pH-tolerant species were inanga (*Galaxias maculatus*), giant kokopu (*G. argentus*), long- and short-finned eels and banded kokopu. The fish fauna in mine-contaminated streams have not been well studied. Historical anecdotal evidence from local residents in the Ngakawau River, north Westland, indicate that prior to mining crayfish and koaro were abundant in this system, however neither species have been recorded post-mining.

As mentioned previously, the presence of high concentrations of suspended sediments derived from mining (e.g., from placer mining, roading) can affect fish communities. Behavioural experiments by Boubee *et al.* (1997) on native juvenile migratory species have shown that banded kokopu can be sensitive to turbidities >17 NTUs, whereas koaro and inanga are less sensitive and avoid only much more turbid silt-laden waters (>70 and 240 NTUs, respectively). In contrast, short-finned and long-finned eels, and red-finned bullies were not effected by turbidities > 1000 NTUs (Boubee *et al.* 1997). A possible explanation for these preferences was offered by Rowe and Dean (1998), who demonstrated that feeding rates of banded kokopu and inanga declined significantly as turbidity increases. Banded kokopu feeding rates were 40% lower at 20 NTUs than in clear water, whereas koaro feeding rates were not affected, even at the highest turbidity.

Table 36.2 Mean copper concentrations in muscle tissue of rainbow trout from the Waitekauri River, Golden Cross (All data stated as mg/kg wet weight).

Year	Upstream	Immediately downstream	5 km downstream	Control site
1993	-	1.39 ± 0.09	1.22 ± 0.29	0.62 ± 0.08
1994	-	1.15 ± 0.14	0.76 ± 0.07	0.64 ± 0.04
1996	0.31 ± 0.06	0.32 ± 0.08	0.41 ± 0.15	0.29 ± 0.04

Ecosystem interactions

As we have seen, mining activities can have a profound effect on water chemistry, physical conditions and the biota of freshwater systems. The loss or reduction of bacteria and shredding insects means that the processing of organic matter is significantly reduced in mining-affected streams. Similarly, impacts on algal communities, which form the food base of many New Zealand stream food webs (especially outside forests), will have a cascade effect on invertebrate grazers, predators and fish communities. Figure 36.12 summarises the major chemical, physical and biological responses from the effects of mining.

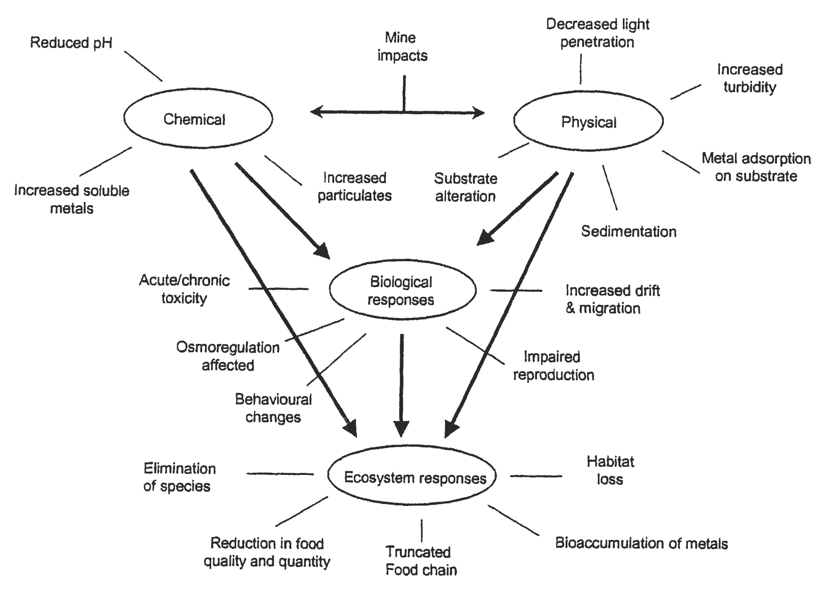

Figure 36.12 Model of the primary ecosystem factors influenced by mine drainage inputs (modified from Gray 1997)

Management and restoration strategies

The chemical and geological complexity of the effects of mine drainage presents significant challenges to the effective management of mining landscapes. The nature of the industry (i.e., large-scale extraction of minerals from the ground) means that environmental impacts are a byproduct of the process, and moderating these impacts has become the focus of research. Management strategies centre on reducing toxic inputs, remediation of effects, restoration and monitoring. While a number of remediation techniques have been developed internationally, the success of many techniques—artificial wetlands, limestone dosing, dilution and buffering, and bacterial reduction of leachate—have been highly variable, and have not been adequately tested in New Zealand.

SUMMARY

Considerable knowledge exists about the chemical processes involved in mine drainage, and its effects on water chemistry, algae and benthic communities have been well documented. The complexity of these relationships continues to present significant challenges to researchers, and those charged with the task of remediating mine impacts.

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