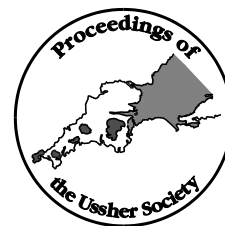


MINE WATER POLLUTION FROM KERNOW TO KWAZULU-NATAL: GEOCHEMICAL REMEDIAL OPTIONS AND THEIR SELECTION IN PRACTICE



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Pollution by mine drainage is a major problem in many parts of the world. The most frequent contaminants are Fe, Mn, Al and SO_4 , with locally important contributions by other metals / metalloids including (in order of decreasing frequency) Zn, Cu, As, Ni, Cd and Pb. Remedial options for such polluted drainage include monitored natural attenuation, physical intervention to minimise pollutant release, and active and passive water treatment technologies. Based on the assessment of the key hydrological and geochemical attributes of mine water discharges, a rational decision-making framework has now been developed for deciding which (or which combinations) of these options to implement in a specific case. Five case studies illustrate the application of this decision-making process in practice: Wheal Jane and South Crofty (Cornwall), Quaking Houses (Co Durham), Hlobane Colliery (South Africa) and Milluni Tin Mine (Bolivia). In many cases, particularly where the socio-environmental stakes are particularly high, the economic, political and ecological issues will prove even more challenging than the technical difficulties involved in implementing remedial interventions which will be robust in the long term. Hence truly 'holistic' mine water remediation is a multi-dimensional business, involving teamwork by a range of geoscientific, hydroecological and socio-economic specialists.

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INTRODUCTION

In virtually all corners of the world in which mining has been undertaken, a legacy of polluted drainage from old workings and spoil heaps has resulted (e.g. Banks *et al.*, 1997; Younger *et al.*, 2002). Nowhere has this been more evident in recent decades than in Cornwall (*Kernow*), which is likely the fabled *Cassiterides* of the ancients, and has indisputably been the heartland of British metalliferous mining from the Bronze Age onwards. Ironically, the intensity of later mining in Cornwall has resulted in a shortage there of reliably dated workings from the earliest periods of mining (~ 4000 years ago), compared with those which have been discovered in the less prodigious orefields of North Wales and south-western Ireland (e.g. O'Brien, 1996). Given the antiquity and intensity of historical mining in Cornwall, it is a good place to start in considering the range of pollution problems associated with mining. However, as Cornwall is devoid of coal deposits and sedimentary ores, it does not host examples of the full range of common mine water problems.

This paper draws upon examples from Cornwall and beyond to demonstrate: (i) the range of pollution problems associated with mine waters; (ii) the various hydrogeochemical remediation techniques recently developed for such waters; (iii) the rationale upon which their selection for application in practice ought to be based; (iv) five case studies (two of them from Cornwall) illustrating the application of this rationale. As such, this paper is a brief, annotated guide to the hydrogeochemical engineering of polluted mine waters as it is practised at the start of the Third Millennium.

MINE WATER POLLUTION – A BRIEF OVERVIEW

Terminology

In this paper the term "mine water pollution" is used to refer to all polluted drainage emanating from mine voids and/or mine waste depositories (i.e. spoil heaps, tailings dams etc). This is an all-embracing term, of which the better-known "acid mine drainage" is a sub-set: in referring to 'mine water pollution' rather than 'acid mine drainage', we are immediately acknowledging

that many of the most problematic mine drainage waters in the world are actually alkaline, or at least of neutral pH (e.g. Younger *et al.*, 2002). The paper further avoids using the term "acid rock drainage", since this misleadingly suggests that the pollution problems are ascribable to geology alone, rather than the stimulation of oxidative weathering of sulphide minerals wrought by mining activities including drilling, blasting and ventilation of workings.

Common mine water pollutants

The most common pollutant of mine waters is iron (Younger, 2000a), which commonly occurs at concentrations within a range from 3 to 200 mg l⁻¹ Fe (Banks *et al.*, 1997) with extreme concentrations reaching tens of thousands of mg l⁻¹ Fe (e.g. Nordstrom *et al.*, 2000), the latter particularly where evaporative concentration of mine waters has occurred. Even at relatively modest concentrations, a few mg l⁻¹ of dissolved iron can render waters unsuitable for supply purposes without extensive (and frequently expensive) prior treatment (e.g. Younger, 2000a). The forms of dissolved iron vary with pH: ferrous iron (Fe^{2+}) is commonly found in solution across a wide range of pH values (typically 3-9), whereas the ferric form (Fe^{3+}) is only appreciably soluble under low-pH conditions (pH < 4.5). At higher pH values, ferric iron rapidly becomes insoluble as its hydroxide ($\text{Fe}(\text{OH})_3$), which forms the red/orange 'ochre' precipitates which discolour the beds of streams impacted by mine water discharges. In more acid waters, the 'ochre' tends to be lighter in colour (often being yellow in appearance) and is mineralogically dominated by ferric hydroxysulphate minerals such as schwertmannite and/or jarosite. Ochre deposition causes marked ecological degradation (principally by prevention of photosynthesis where the benthos is smothered in precipitates; Jarvis and Younger, 1997).

Besides iron, four other contaminants are very commonly found in mine waters: (a) protons, i.e. hydrogen ions, H^+ , which are manifest in low pH. Nordstrom *et al.* (2000) document and discuss the extremes which pH can reach in mine waters. (b) Sulphate (SO_4^{2-}) which can cause gastric upsets where present in drinking waters at concentrations in excess of 250 mg l⁻¹ SO_4 (a

threshold very commonly exceeded by factors of 2 to 100 in polluted mine waters). (c) Manganese (Mn^{2+}), which is *not* ecotoxic, but can cause unsightly black staining of domestic plumbing and laundry where it enters public water supplies and becomes oxidised to Mn^{4+} and precipitated as its oxide MnO_2 . (d) Aluminium (Al^{3+}) which is strongly ecotoxic, but is relatively immobile as its hydroxide ($Al(OH)_3$) where the pH lies between 5 and 9. White precipitates of 'alum' (i.e. aluminium hydroxides and hydroxysulphates) frequently form where the pH of Al-rich acidic mine water locally increases. Alum frequently forms unsightly off-white foams on the surfaces of affected waters, and can also coat stream beds in much the same way as ochre, smothering the benthos in the process.

Less common mine water pollutants

The deposition of alum and ochre is not all bad news, however, since these substances act as powerful sorbents for most other ecotoxic metals, especially at circum-neutral pH (e.g. Dzombak and Morel, 1990; Younger *et al.*, 2002). Where these and other 'natural attenuation' processes are incomplete, a range of other substances are frequently implicated in the acute toxicity of many mine waters to aquatic fauna, and can also render the receiving waters unsuitable for public supply purposes. Chief amongst the less common (but locally extremely important) mine water pollutants are the following:

Zinc (Zn^{2+}). This is commonly present at concentrations of up to 50 mg l^{-1} in waters draining former Pb-Zn mines in Wales (e.g. Fuge *et al.*, 1994), the North Pennines (e.g. Nuttall and Younger, 1999) and the Isle of Man (author's own unpublished data), and also occurs at ecologically deleterious concentrations in drainage from some Cornish tin mines. Zn is highly ecotoxic, especially in soft waters, and is a known cause of faunal impoverishment in many rivers in metalliferous mining districts (e.g. Kelly, 1988).

Nickel (Ni^{2+}). This occurs at mg l^{-1} concentrations in certain South Wales coal mine waters (author's own unpublished data), albeit ecotoxic effects directly ascribable to Ni are not documented in the receiving waters, probably due to the overwhelming impacts of the extremely high iron concentrations in the same discharges.

Cadmium (Cd^{2+}). This metal is extremely ecotoxic (indeed it is widely regarded as xenobiotic), and is often mobilised by oxidative dissolution of sphalerite (in which it often occurs in solid solution). While most mobile in the more acidic waters (e.g. Fuge *et al.*, 1994), Cd^{2+} is also non-negligible in waters draining carbonate-hosted Pb-Zn orebodies, and in both settings no doubt exacerbates the toxic effects due to Zn^{2+} , which is typically present in the same waters at far higher concentrations.

Copper (Cu^{2+}). Logically enough, copper is a notable pollutant in waters draining former copper workings in Cornwall, Wales and Cumbria. Although not as mobile in natural waters as Zn^{2+} , copper is nevertheless a significant cause of ecological degradation where it occurs in receiving watercourses at concentrations above about 0.5 mg l^{-1} .

Lead (Pb^{2+}). Although it is a significant ecotoxin, lead is considerably less mobile than zinc and copper, and hence is rarely reported as a principal pollutant of mine waters, even those emanating from lead mines. In a study of a working mine in a Pb-Zn-orebody in northern England, Younger (2000d) noticed that elevated lead concentrations were typical of waters being pumped from areas of active extraction, where fresh galena surfaces were being exposed daily by blasting. From observations in numerous abandoned mines in the North Pennine Orefield, Dunham (1990) concluded that galena surfaces tend to become coated with white films of cerussite ($PbCO_3$), which apparently hinder further oxidation. Nevertheless, recent surveys of the abandoned lead mines of Allendale (Northumberland)

suggest that Pb can be the principal contaminant in mine waters in certain (as yet poorly understood) circumstances (author's unpublished data).

Arsenic (As). This metalloid is occasionally an important accessory contaminant in mine waters, particularly where the orebodies contain arsenopyrite. For instance, concentrations as high as 350 mg l^{-1} total As are reported in acidic leachates emanating from old spoil heaps of a former Pb-Zn mine in Carnoulès, southern France (LeBlanc *et al.*, 2002). Arsenic is also the principal contaminant of concern in adit drainage waters flowing from abandoned mercury mines in northern Spain (Loredo *et al.*, 2002). Appreciable concentrations of dissolved arsenic are also occasionally reported from waters draining orebodies rich in iron oxides. This has been observed, *inter alia*, in the *Cuadrilatero Ferrifero* of Minas Gerais, Brasil, (haematite / magnetite orebodies), in old haematite mines in the vicinity of Egremont, west Cumbria, UK (author's own unpublished data), and in mine waters draining the haematised country rock of tin-bearing veins in the Roskear district of west Cornwall. In these geological environments, arsenic is presumably released to solution by desorption from the iron oxide mineral surfaces, likely prompted by the changes in Eh and pH conditions of the rock mass induced by mining activities. One peculiarity of the geochemical behaviour of arsenic is its tendency to be more mobile at circum-neutral pH than under acidic conditions. This arises because dissolved arsenic tends to be mobile as the oxyanion AsO_4^- , which is preferentially sorbed to mineral surfaces at low pH, but is displaced into solution by cations as pH rises.

It is perhaps worth briefly noting that there are many metals which tend not to behave as mine water contaminants, despite being present at high concentrations in the corresponding orebodies. Tin (Sn) for instance is not a problem metal in the waters draining tin mines in Cornwall, Malaysia, Brasil and Bolivia, due to its stability (even at very low pH) as the oxide cassiterite (SnO_2).

Salinity of mine waters

Leaving aside issues of metal contaminants and their ecotoxicity, the salinity of mine waters is often an even greater problem, at least in terms of the potability of receiving waters, especially in semi-arid/arid regions. The specific electrical conductance ('conductivity') of many mine waters exceeds $1800\text{ }\mu\text{S/cm}$, and occasionally exceeds $5000\text{ }\mu\text{S/cm}$. This unfavourably compares with a requirement for fresh drinking waters to have conductivities of less than $1500\text{ }\mu\text{S/cm}$. Hence even where the metals are removed from mine waters, their high degree of mineralisation usually persists (indeed, some treatment methods such as lime dosing actually *increase* the overall mineralisation of mine waters). Mine water discharge can thus serve to remove entire surface water catchments from the inventory of fresh water resources, making them unusable for public supply on account of their salinity even if all the toxic metals were to be removed. This is true, for instance, of the rivers Ore and Almond in central Scotland (Younger, 2001), of the River Gaunless in County Durham (Younger, 2000a), and of the Carnon River in Cornwall. While such problems can be tolerated in countries such as the UK, which have abundant alternative freshwater resources, they represent the principal problem associated with mine water pollution in many Mediterranean and southern hemisphere countries, and they continue to provoke a great deal of research into cheaper methods of desalination (e.g. Boshoff, 1999).

Mine water pollution source zones

One of the key characteristics of the global mining sector from the mid-20th Century onwards has been the dramatic expansion of surface mining, which currently accounts for around 80% of global mineral production. As surface mining necessarily involves wholesale excavation of overburden (in contrast to deep mining,

which delves beneath it) it is not surprising that more than 70% of all the material excavated in modern mining operations world-wide is waste, and that more than 99% of all mine waste rock is being generated by surface mines (Hartman, 1987). The preoccupation of active mining companies with managing such large volumes of waste rock, as well as the tailings produced by mineral processing operations, has led to the perception that mine wastes (overburden, inter-burden, tailings etc) are the principal source of polluted leachates associated with mines. Yet how true is this? Because the geometry of orebodies cannot be controlled, whereas the 'footprints' of mine waste repositories are deliberately minimised for economic reasons, abandoned mine voids (deep and surface) tend to underlie far greater surface areas than their corresponding waste depositories. Since the capture of precipitation is directly proportional to surface area, mine voids generally intercept much more natural water than mine waste depositories. For this reason, mine voids are generally far more important sources of mine water pollution (at least in volumetric terms) than are mine waste repositories. This deduction is borne out by a recent study of all of the mining districts of Scotland (which are nearly all abandoned now), which revealed the following breakdown of sources of polluted mine drainage, with respect to volumes of polluted waters emanating per unit time (Younger, 2001): (a) flooded mine voids, 98%, (b) old tailings deposits / waste rock piles, 2%. Of course this volumetric breakdown is not the entire story, for waste rock depositories, being generally shallower than mine voids, tend to be far more exposed to the Earth's atmosphere than are the deeper portions of flooded voids. This means that mine waste depositories "punch above their weight" in the degree to which they release the products of oxidative weathering of sulphide minerals: many spoil-heap leachates are acidic, whereas many deep mine discharges revert to being alkaline in the long-term. Thus in terms of the *loadings* of contaminants released by the two source zones, we obtain the following breakdown for the Scottish example (Younger, 2001): (a) flooded mine voids, 72%, (b) old tailings deposits / waste rock piles, 28%.

The above figures help to inject a sense of proportion into the current European debate over the desirability of a proposed Mine Waste Directive as a tool of environmental protection: if the voids are neglected as a source of contamination, the directive will miss the main issue.

But how important is mine water pollution in the overall scheme of things? In many mining areas, abandoned mines are now the main source of freshwater pollution, and even on a national basis in Scotland, for instance, mine water pollution is second only to sewage as a cause of freshwater degradation (Younger, 2001). This is an astonishing fact given that sewage production is distributed nationwide, whereas mine waters are naturally only a problem in geographically-restricted areas. There is as yet no Europe-wide compendium of the damage caused to surface waters by abandoned mine discharges. The most comprehensive national dataset currently available is that for the UK (NRA, 1994), the most recent (as yet unpublished) contents of which indicate that some 400 km of watercourse are currently degraded by abandoned coal mine discharges, with a further 200 km or so similarly contaminated by abandoned metal mine discharges. By extrapolation weighted by the distribution of coalfields and orefields in mainland Europe, it is likely that the equivalent figures for the whole of Europe will eventually prove to be on the order of 2000 to 3000 km of watercourses polluted by coal mine drainage, and 1000 to 1500 km polluted by metal mine discharges. This suggests that the total length of watercourses polluted by mine drainage in Europe may well prove to exceed 5000 km.

REMEDIAL OPTIONS FOR POLLUTED MINE WATERS

Principal options

Having established the nature and approximate extent of the mine water pollution problem, it is now appropriate to consider

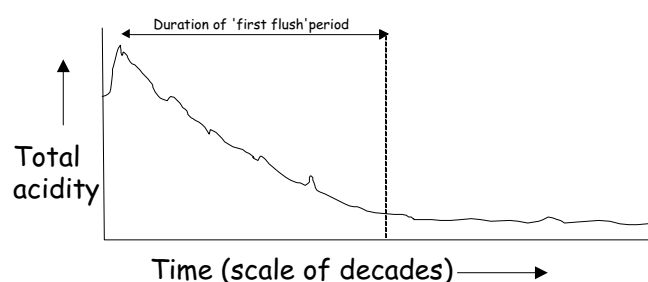


Figure 1. Typical temporal changes in mine water acidity in the years following first emergence of water from newly-flooded mine workings (adapted after Younger, 1997).

what remedial actions are feasible. There are three principal options (of which the latter two are not mutually exclusive): 1. Monitored natural attenuation, in which natural processes are deemed sufficient to deal with the contamination, and monitoring is put in place to ensure that this conclusion is (and remains) valid. 2. Prevention/minimisation of pollutant release processes. 3. Mine water treatment, by either active (chemical dosing etc) or passive (wetlands etc) means.

Monitored natural attenuation

This will often be the most "sustainable" option overall in the long term, particularly for the less heavily polluted mine water discharges. Selection of this option will be based on two key factors: (i) the availability of dilution in the receiving watercourse, and (ii) the degree of natural 'self-cleansing' occurring in the flooded workings.

The availability of dilution is a site-specific factor, which can only be assessed on a case-by-case basis by means of hydrometric and water quality monitoring, possibly supported by mathematical modelling. The second factor, the process of 'self-cleansing', is a more generic geochemical phenomenon, which has been extensively studied in recent years (see especially Younger, 1997, 1998, 2000b). When workings are first left to flood, the water quality generally deteriorates markedly in comparison to that encountered during active mining and dewatering. Why should this be? During the working of a mine, the everyday procedures of drainage, ventilation and excavation promote the oxidative weathering of pyrite and other minerals which are only stable in the absence of air. The weathered remnants of these minerals are frequently to be found as white and yellow efflorescent crusts on certain strata within a working mine. When the workings are left to flood, these efflorescent salts dissolve rapidly, delivering high concentrations of sulphate, iron and other metals into solution, often also rendering the water very acidic. Subsequent to the initial overspill of the flooded workings to the surface or an adjoining aquifer, water quality improves. This is essentially because pyrite and the other oxygen-intolerant minerals barely dissolve at all where they remain below the water table, whereas benign minerals which counteract acidity (such as calcite) continue to dissolve anywhere, above or below the water table. Thus as fresh water continues to make its way to the flooded workings, it gradually displaces the highly contaminated water which developed in them during the initial flooding of the strata. In the long-term, therefore, significant pollutant release can only occur in strata which are permanently or temporarily (e.g. in summer and autumn only) above the water line in the workings. The overall effect of these processes is manifest in a gradual improvement in water quality over time, termed the 'first flush' (Younger, 1997), which usually follows an exponential pattern (Figure 1).

Prevention / minimisation of pollutant release processes

Strategies aimed at minimising the release of contaminants to mine waters must focus on breaking one or other of the following

pre-conditions for the oxidation of sulphide minerals: (i) the presence of oxygen, (ii) the availability of moisture, (iii) the presence and activity of iron- and sulphur-oxidising bacteria.

The access of both atmospheric oxygen and moisture to mine wastes in old spoil heaps / tailings dams can be severely limited by the installation of low-permeability covers. Adequate covers will generally include a coarse grained "capillary break" layer (to prevent upward migration of pore-waters in response to surface layer desiccation), overlain by a low permeability cap (typically 0.5 m or more of clay, compacted such that it retains a permeability no greater than 10^{-4} md^{-1}), usually with a final veneer of vegetated topsoil. While such covers have been successfully implemented for a range of surface depositories of mine waste (e.g. Gustafsson *et al.*, 1999), such a simple approach is rarely likely to be feasible in relation to extensive networks of deep mine voids. Selective diversion of surface waters away from known zones of infiltration to deep mine voids may be possible in some cases (see Younger, 2000c, for further discussion).

Besides denying them moisture and oxygen using covers, direct inhibition of the activities of iron- and sulphur-oxidising bacteria using bactericides has also been attempted on a number of occasions (for a brief review, see Younger *et al.*, 2002). Although such efforts have met with a certain degree of success in controlling the leaching of acidity from ore/coal stockpiles and waste rock heaps of modest areal extent, the beneficial effects rarely last any longer than 6 months, making this a very expensive option in practice. It is also not feasible to apply this approach to very large mine waste depositories nor to vast labyrinths of underground mine voids.

Mine water treatment technologies

These have been extensively documented, together with detailed guidelines for their design and implementation (Younger *et al.*, 2002) and therefore only a brief summary is given here.

Active treatment. This denotes the use of conventional wastewater treatment unit processes, which typically require ongoing inputs of electrical power and/or chemical reagents in a closely-controlled process (which usually demands frequent operator attention). The classic approach to active treatment of acidic and/or ferruginous mine drainage involves three steps:

(i) oxidation (usually by means of a simple cascade), which helps to convert soluble ferrous iron (Fe^{2+}) to far less soluble ferric iron (Fe^{3+}), as well as allowing pH to rise by venting excess CO_2 (where present) until equilibrium with the atmospheric CO_2 content is attained.

(ii) dosing with alkali (usually hydrated lime ($\text{Ca}(\text{OH})_2$), and less frequently caustic soda (NaOH)), both to raise the pH (thus lowering the solubility of most problematic metals) and to supply hydroxyl ions for the rapid precipitation of metal hydroxide solids.

(iii) accelerated sedimentation, usually by use of a clarifier or lamellar plate thickener, often aided by the addition of flocculants and/or coagulants. Current practice in the industry favours the re-circulation of an aliquot of iron hydroxide sludge into the influent of the sedimentation unit, which has been shown to favour the densification of the sludge overall. This practice is called the "high density sludge" process and it typically yields sludges with 25 – 30% solids by volume, as opposed to the 5% solids contents typically obtained without re-circulation.

For most purposes this time-honoured approach will suffice. However, where it is important that the treatment process yields a net reduction in mine water salinity, then alternative approaches will most likely be necessary. In some cases, it may prove worthwhile applying conventional desalination technology (i.e. flash distillation and reverse osmosis) to mine waters. However, these processes are extremely costly. In South Africa, innovative research is yielding new approaches to the desalination of mine waters which result in far cheaper processes. In one approach, termed 'biodesalination', co-treatment of acidic mine waters

with sewage from the adjoining cities is effectively "using one waste stream to cancel out the other", in a process which is extremely efficient at removing sulphates from the mine water (by means of bacterial sulphate reduction). Other technologies employing bacterial sulphate reduction are available as commercially-proven turn-key operations, and are particularly suitable where very low metal concentrations must be attained in active treatment plant effluents, and where one or more of the metals recovered from the water is valuable (e.g. zinc/copper). Another very promising desalination process of South African lineage is the SAVMIN™ process (Smit, 1999), which is essentially a variant of conventional alkali dosing and sedimentation in which a series of cyclical precipitation and sedimentation steps eventually lead to recovery of virtually all of the previously dissolved sulphate as potentially marketable gypsum. The key to achieving this is a step in which sulphate is removed down to residual concentrations of only a few mg l^{-1} by equilibration of the water with respect to the aluminium sulphate mineral ettringite ($3\text{CaO} \cdot 3\text{CaSO}_4 \cdot \text{Al}_2\text{O}_3 \cdot 31\text{H}_2\text{O}$), which is stable only in a narrow range of high pH (11.6 - 12.0).

Beyond these neutralisation and desalination technologies, there is a considerable range of alternative approaches to the active treatment of mine waters, many of them borrowed from the field of metallurgical processing, which all have potential as niche applications in cases in which recovery and re-use of metals is an economic possibility. These approaches include (Younger *et al.*, 2002): (1) sorption and ion exchange processes, (2) solvent extraction, (3) electrochemical extraction, (4) biochemical extraction techniques, (5) the barium sulphide process (in which sulphate is removed from mine water by precipitation of BaSO_4), (6) biological trickle filters. Few of these techniques are ever likely to enjoy widespread uptake in practice, on account of their costs and limited track record, and are therefore not considered further here.

Passive treatment. The term "passive treatment system" in the context of mine waters is defined by the European Commission's PIRAMID project as follows: "A water treatment system that utilises naturally available energy sources such as topographical gradient, microbial metabolic energy, photosynthesis and chemical energy and requires regular but infrequent maintenance to operate successfully over its design life". The working definition of "infrequent" in this context is currently around six-monthly. For further details, see <http://www.piramid.org>.

The development of passive treatment of mine waters to date is documented in detail by Younger *et al.* (2002). Types of passive system currently in use include: (i) aerobic, surface flow wetlands (reed-beds), (ii) compost wetlands with significant surface flow, (iii) mixed compost / limestone systems, with predominantly subsurface flow (so-called reducing and alkalinity producing systems (RAPS)), (iv) subsurface reactive barriers treating acidic, metalliferous ground waters, (v) closed-system limestone dissolution systems for zinc removal from alkaline waters, (vi) roughing filters for the aerobic treatment of ferruginous mine waters where there is no room for a surface wetland.

Each of the above technologies is appropriate for a different kind of mine water, or for specific hydraulic circumstances. The degree to which each type of system can currently be considered to be "proven technology" corresponds to the order in which they are listed above. This ranking of confidence is reflected in uptake rates to date (Younger, 2000c).

One of the principal attractions of wetlands as treatment systems is the possibility of integrating them into the surrounding landscape, and achieving healthy connections with the existing eco-systems in the area. Integration of wetlands into a landscape at the level of aesthetics is readily attainable, as a number of recent projects illustrate (see the case studies later in this paper). Ecological integration is rather harder to achieve in practice, however, due to a number of factors including, (i) physical limitations on the areas available for treatment, (ii) the frequent insistence of regulators that treatment wetlands be surrounded

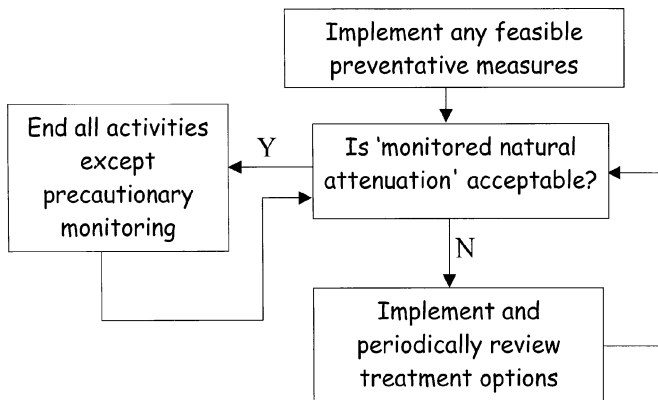


Figure 2. Flow-chart giving the basic decision-making logic for the selection of the most appropriate remedial option for a polluting mine water discharge. Note that this flow-chart proposed no final end-point for all activities, because precautionary monitoring (even if only by visual inspection) will nearly always be advisable.

by flood defence bunds, which preclude two way exchanges of water, solutes, sediments and plankton with adjoining rivers, (iii) engineering limitations, such as the need to allow freeboard at the perimeters of systems and the need to prevent erosion by extreme flows.

Designing a remedial strategy

In the light of the foregoing summaries of the nature of mine water pollution and the technologies available to combat it, it is now appropriate to summarise the logic which ought to underlie the rational selection and implementation of remedial options. Figure 2 summarises the coarse-scale logic, by means of which a choice is made between monitored natural attenuation on the one hand, and various treatment strategies on the other. From examination of Figure 2, it is at once apparent that reasonable steps to minimise long-term pollutant release should always be taken, though in many deep mine scenarios there will be relatively little that can be undertaken cost-effectively in this regard. It will also be seen that the flow-chart is essentially never-ending, in that a residual duty to retain precautionary monitoring (which may

be visual rather than chemical) will always remain. The reason for retaining this residual duty is that it is never likely to be possible to categorically state that a mine water which is currently alkaline in nature will *never* later revert to being acidic. The possibility of a 'toxic shock', in which this reversion occurs, has been observed in practice at a number of sites (Younger, 2000c) and predicted for others on the basis of the rates of weathering of sulphides and carbonates, and their relative proportions in the leaching rock mass (e.g. Strömberg and Banwart, 1994).

Figure 3 attempts to place the lower two-thirds of Figure 2 into a technical context, by relating the decision logic (for all activities except implementation of feasible preventative measures) to the flow magnitude and pollution intensity of the mine water in question. The axes of Figure 3 have been deliberately labelled in a qualitative manner to allow for the fact that the precise definitions of the terms 'high flow' and 'high contaminant concentrations' in a particular case will crucially depend on the characteristics of the catchment in question (in terms of climate and runoff patterns etc) such that these categories will be reached at far lower absolute values of flow and concentration in a semi-arid area than in a humid temperate area. The key lesson of Figure 3 is that choosing between active and passive treatment options is usually not a matter of simple personal preference: we simply do not have passive treatment technologies capable of coping with the most voluminous and most contaminated of mine water discharges, so that nothing less than intense active treatment is likely to suffice for such cases.

SELECTED CASE STUDIES OF REMEDIAL OPTION SELECTION

The manner in which the remedial design logic outlined above is implemented in practice is well illustrated by the following five case studies. 1. Wheal Jane (Cornwall), 2. South Crofty (Cornwall), 3. Quaking Houses (County Durham), 4. Hlobane Colliery, KwaZulu-Natal (South Africa), 5. Milluni Tin Mine (Bolivia). Figure 4 indicates in broad terms how these five case studies map onto the 'decision space' defined by Figure 3. From Figure 4 it is readily apparent that these case studies have been chosen such as to populate most parts of this decision space.

Case Study 1: Wheal Jane tin/zinc mine, Cornwall

Wheal Jane was the last of a large number of mines which once worked the tin and zinc bearing lodes of the Carnon Valley (between Truro and Falmouth, Cornwall). By dint of being connected underground to an extensive network of very old, long-abandoned mines, Wheal Jane has long borne an exceptionally heavy dewatering burden, with a pumping capacity reaching 200 ls⁻¹ during the most recent period of working (1980 to 1991; Davis and Battersby, 1985). Mining at Wheal Jane was latterly marginally economic, and it is widely rumoured that the withdrawal of governmental subsidies which previously helped to defray the costs of dewatering was a contributory factor in the decision to abandon the mine in March 1991. Since the abandoned workings of Wheal Jane flooded to river level eight months later (in November 1991) it has proved to be the single worst case of mine water pollution in the UK, at least in terms of flow rates and contaminant loadings. The sudden release of an estimated 50000 m³ of highly polluted water and sludge from an old adit on January 16th 1992 resulted in the development of a highly conspicuous orange plume of contaminated water which prograded through the Fal Estuary and into the western approaches of the English Channel (Bowen *et al.*, 1998). A flow of contaminated water ranging between 90 and 300 ls⁻¹ has continued ever since, prompting the expenditure of almost £20M on remedial measures over the following decade.

One of the paradoxes of the Wheal Jane case is that the discharges of mine water from Wheal Jane since November 1991 have caused relatively little biological damage. The reasons for this are twofold: (1) The immediate receiving watercourse (the

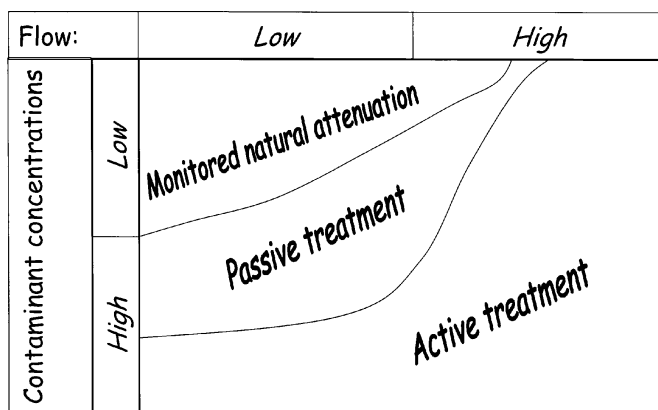


Figure 3. Suitability of monitored natural attenuation, passive treatment and active treatment options for remediation of polluted mine waters, expressed as functions of the magnitude of flow and intensity of contamination of mine water discharges. The qualitative descriptions "high" and "low" can and should be replaced by appropriate quantitative ranges for particular hydrological/climatological settings. For UK conditions, the split between low and high flows lies at around 5 ls⁻¹, and very high flows are on the order of 200 ls⁻¹. The split between low and high concentrations is not only geographically dependent, but also varies from species to species. For instance, in relation to total acidity, the low/high boundary will lie at around 50 mg/l as CaCO₃ equivalent, for Fe at around 20 mg/l, and for Zn around 2 mg/l.

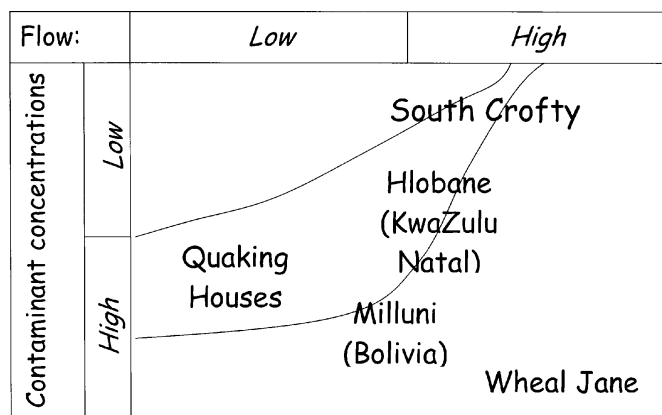


Figure 4. Conceptual locations of the five case studies discussed in the text in relation to the 'suitability fields' defined by Figure 3.

Carnon River) was already so badly affected by discharges from other mines upstream (most notably the County Adit) that there were no fish, and precious-few benthic invertebrates present to be affected by the additional input of mine waters. Since it is impossible to kill something which is absent, the outflows from Wheal Jane did not produce a demonstrable biological impact in the Carnon River. Indeed the only faunal mortalities reliably attributed to the Wheal Jane discharge relate to heavy metal poisoning of a few swans in Restronguet Creek downstream. (2) The Fal Estuary, into which the Carnon River drains via the tidal Restronguet Creek, is a zone of marine shelf waters of very high conservation value. In particular the estuary hosts substantial colonies of coral-like maerl, and has generally diverse and healthy faunal and floral communities. One might, therefore, expect the release of large quantities of acidic, metal-laden waters into the Fal Estuary to cause severe ecological degradation. In reality, density stratification of the estuary seems to have prevented most of the mine water contaminants from affecting the benthos, and the sheer volume of the estuarine water column served to assimilate the potentially toxic inputs from Wheal Jane by means of dilution and dispersion. Hence, despite concerted searches for biological impacts in the estuary, it appears to have survived this potentially perilous episode unscathed. In other words, Wheal Jane is a very bad discharge, but thankfully the Fal Estuary is more-or-less capable of 'swallowing it whole'.

So what have been the drivers for the £20M clean-up of Wheal Jane? A number of obligations incumbent upon the UK government under international environmental legislation (especially the North Sea Accords and the Dangerous Substances Directives) provide powerful motives for attempts to restore the Carnon River to internationally-acceptable environmental quality standards (EQS). However, after six years of detailed monitoring and modelling (by teams of consultants employed by Knight Piésold and Partners Ltd, and W S Atkins Ltd) it became apparent that EQS are not feasibly attainable for the Carnon River. Indeed it was found that even if the entire flow of the Carnon River were pumped from the channel at its tidal limit (Devoran Weir) and treated to levels commensurate with EQS, the river would once again be in breach of EQS within a few hundred metres downstream, due to the ongoing release of toxic metals from the vast quantities of historic mine tailings which line the channel (according to unpublished monitoring and modelling results of the Environment Agency). Given the infeasibility of removing these many millions of tonnes of tailings without causing even greater environmental damage, compliance with conventional EQS is impossible, and the principal remaining driver remains public outcry at the discoloration of the Fal Estuary.

The inferred history of discoloration of the Fal Estuary is worthy of some consideration. Metal mining began in the Carnon Valley as long ago as 2000 BC and peaked in the Victorian era, before widespread mine closures effectively brought extraction in the valley to a halt in 1913. There is no doubt that, in the period

1913 - 1969, Wheal Jane and the other mines in the Carnon Valley will have been flooded up to river level and thus discharging large fluxes of acidic, metal-rich water to the river. However, there are no records of any environmental outcry over discoloration of the estuary in this period. Between 1969 and 1978, Wheal Jane and Mount Wellington mines experienced a period of renewed working, before the excessive costs of dewatering led to a further period of closure commencing in April 1978 (Davis and Battersby, 1985). For twelve years the mine lay abandoned, and will once more have been discharging acid waters to the Carnon River, though still apparently without provoking any public outcry. It is only since the most recent closure of Wheal Jane in 1991 that public outcry has necessitated remedial interventions. Why should this be so? Changing public perceptions of environmental issues are certainly one issue (Younger and Robins, 2002). However, it is important to note that the 1992 outburst was significantly different from the advent of the earlier events of uncontrolled discharge. While earlier periods of polluted drainage from disused workings at Wheal Jane (1913 - 1969, and 1978 - 1980) will certainly have led to discoloration of the Carnon River (and also, from time to time, of Restronguet Creek) the major outburst from the Nangiles Adit on 16th January 1992 was probably the first plume ever to discolour the Fal Estuary. It was certainly the first pollution plume to do so in the era of television, international coverage by which not only brought the incident to the attention of a much wider public than had previously been aware of the mine water problems of the area, but also ensured that the event was noticed by the European Commission officers in Brussels who held responsibility for implementation of the Dangerous Substances Directives (A. Kroll, *personal communication*, 1999).

Earlier published accounts of the 16-1-1992 outburst have invoked failure of a man-made plug in the portal of the Nangiles Adit (e.g. NRA, 1994; Hamilton *et al.*, 1994; Banks *et al.*, 1997; Bowen *et al.*, 1998), supporting the widely-held notion that the outburst was an unfortunate accident. The reality is rather more complex. The roots of the incident lie in legislative developments in the late 1980s, particularly in the implementation of the Water Act 1989 which (*inter alia*) resulted in the creation of the National Rivers Authority (NRA) as the nation's guarantor of environmental quality. Under the 1989 Act, the NRA was granted powers (unavailable to its predecessor organisations, the Water Authorities) to actively intervene in emergency mode in order to prevent pollution occurring, and to recover its costs thereafter. A decision to do just this was taken in November 1991, when Wheal Jane first decanted with poor quality water. It is important to note that the November 1991 discharge, though of very poor quality, did not give rise to discoloration in the Fal Estuary. With the benefit of hindsight it is possible to identify a number of problems with exercise of the NRA's new-found preventative powers in this case, one of which is legal and the remainder technical. The legal problem was that, ever since 1963, UK legislation had exonerated mine owners/operators (chief amongst whom was of course HM Government itself in the guise of the National Coal Board) from culpability for water pollution attributable to 'permitting water to flow from abandoned mines'. This exonerating clause was finally removed with effect from 1-1-2000, a date conveniently falling a few years after HM Government had ceased to be a mine owner. The existence of this 'get-out-of-jail-free card' meant that the mining company at Wheal Jane could not be held responsible for pollution arising from mine water discharge from the mine: hence, whatever the NRA did in its emergency intervention, it would have to bear the liabilities and costs itself. Before this fine point of law was fully appreciated by all concerned, however, more pressing technical issues had emerged. Difficulties of access precluded the installation of suitable emergency treatment facilities at the point of natural decant from the mine (i.e. a drainage level named "Jane's Adit"). While the mining company had installed an S-shaped lagoon downstream of Jane's Adit portal, this was completely ineffective in improving the quality of drainage emanating from the mine. A decision was therefore taken by the

	pH	Determinand (mg l ⁻¹)					
		Iron	Zinc	Manganese	Copper	Nickel	Cadmium
Influent	3.8	159	44	5	0.4	0.4	0.056
Effluent	9	1.3	0.4	0.6	<0.01	<0.1	<0.001

Table 1. Wheal Jane active mine water treatment plant (Unipure HDS plant). Average pH change and total metals removal performance, 2000 - 2001. (Data courtesy of Dr R. Coulton of Unipure Europe Ltd).

NRA to seal the adit portal, thus creating a sump from which water could be pumped via one of the many mine shafts onto the higher ground near the pithead buildings, where room for treatment facilities was available. On that site, the mine water could be dosed with lime and settled prior to discharge to the river. Serious maintenance problems began to emerge with these *ad hoc* pump-and-treat facilities around Christmas 1991. As availability of spare parts and service personnel became scarce or prohibitively expensive over the Christmas period, a decision was taken to suspend pumping (with Jane's Adit still plugged) and allow the water to accumulate in the workings until the New Year, at which time it was intended that improved pumping and dosing facilities would be commissioned. It was assumed that there would be sufficient storage volume in the shallow workings to accumulate the total water make over the holiday period, but by 16th January, when pumping had still not re-commenced, the now-infamous outrush of 50000 m³ of ochreous water occurred from the hitherto-neglected Nangiles Adit. Although rumours of the previous existence of a plug in the portal of the Nangiles Adit then began to enjoy wide circulation (e.g. NRA, 1994; Hamilton *et al.*, 1994; Banks *et al.*, 1997; Bowen *et al.*, 1998), subsequent underground exploration revealed no trace of any such plug; rather, there is strong evidence that the mine water had simply been impounded behind a pile of roof-fall debris (Gatley *et al.*, 1998). It appears that rapid erosion of flow paths through the debris pile occurred once sufficient head of mine water had accumulated behind the pile.

The NRA acted rapidly to retrieve the situation by engaging mining specialists Knight Piésold to implement temporary pump-and-treat works (which in the end operated for some eight years) while a long-term management strategy for the mine water was developed. In autumn 2000, the EA (as successor to the NRA) finally delivered a long-term treatment solution for the Wheal Jane site. Some of the hydrological and geochemical assessments upon which the selection of the best option was based have already been mentioned. In tandem with these, evaluations of a full range of treatment technologies was undertaken. These evaluations bore fruit in a substantial increase in UK expertise in mine water treatment technologies.

Early in the search for a long-term treatment strategy, the NRA undertook to evaluate the possibility of developing a wetland-based passive treatment solution for the Wheal Jane site. In 1992, passive treatment was still in its infancy world-wide. However, it was already evident that the technology was unlikely ever to be suited to the treatment of discharges with very high contaminant loadings (Cohen and Staub, 1992), such as that at Wheal Jane, which to this day remains in a category of its own amongst the problematic mine waters of north-west Europe. However, the quixotic search for a passive treatment solution for the Wheal Jane site caught the imagination of EA public relations officers, who repeatedly used images of reeds on publicity material explaining the Wheal Jane project to a curious public.

The Wheal Jane Pilot Passive Treatment Plant (PPTP) has generated large volumes of interesting data. However, with the benefit of hindsight, it is now clear that the PPTP suffered from an unfortunate design flaw: it was essentially configured "back-to-front", inasmuch as acid-generating aerobic processes were sited upstream from the crucial alkalinity-generating anaerobic processes. This contrasts sharply with slightly more recent passive treatment designs, such as that at Quaking Houses discussed below. This design flaw has limited the ability of the PPTP to yield useful information in relation to the unique Wheal Jane discharge itself, let alone in relation to other, less extreme, mine water discharge sites around the UK and elsewhere.

While the PPTP investigations continued to bask in the lime-light, the search for a more suitable (albeit less glamorous) long-term treatment solution continued in parallel. A temporary treatment plant was operated from 1992-2000, based on conventional lime dosing, flocculation, and sedimentation technology. Active treatment was finally confirmed as the only feasible option for long-term prevention of intolerably frequent discoloration of the 'Carrick Roads' area of the Fal Estuary. In essence, without active treatment at Wheal Jane, the Carrick Roads would be perceptibly discoloured two to three times per year on average, whereas treatment might reduce the frequency of this event to once or twice per decade. After 8 long years of investigation, therefore, the long-term treatment solution selected for Wheal Jane is a state-of-the-art active treatment plant. The works commissioned in the autumn of 2000 comprise a high-density sludge alkali dosing plant, designed and commissioned by Unipure Environmental. This plant is equipped to treat up to 350 ls⁻¹ of mine water, and is achieving the major improvements in water quality indicated in Table 1. During its first full winter of operation, the Unipure plant successfully treated a total of 4.4 Mm³ of water, removing a total of about 1000 tonnes of metal which would otherwise have been discharged to the Carnon River.

Before concluding the Wheal Jane case study, it is worthwhile considering the exhortation in Figure 2 to "implement any feasible preventative measures". Quite a lot of effort was put into identifying what might be done in this regard at and around the Wheal Jane site, principally in relation to minimising water ingress to the old workings. Most options considered (e.g. sealing places where old stopes daylight, impermeabilising streambeds to prevent leakage to workings), had either already been implemented during mining, were prohibitively expensive or were incompatible with conservation requirements and/or other socio-environmental goals. In the end, only one real preventative option was identified: preventing occasional underground overspill of water from the County Adit system to the western parts of the Wheal Jane lode system. This task was completed in 1998 by means of repairing underground diversion walls.

Case Study 2: South Crofty tin mine, Cornwall

By 1997, the closure of South Crofty tin mine (near Redruth, Cornwall) had begun to seem a likely prospect. At that time, the search for a long-term solution for the Wheal Jane site was still very much an ongoing concern for the EA. Given the relative proximity of the two mines, and the fact that they were both owned by the same mining company, it was natural that the EA should be concerned least Crofty should prove to be another "Wheal Jane in waiting". A series of detailed investigations were therefore launched, which very soon revealed that several grounds for optimism existed. For instance: (1) The most recently-mined lodes at Crofty contain hardly any sulphide minerals, whereas those worked up to 1991 in Wheal Jane mine had an average sulphide content of around 15 % by weight (much of which was present as pyrite). (2) While 'red water' was conspicuous in the underground workings at South Crofty, this appeared to be largely due to liberation of suspended Fe₂O₃ from the extensively haematized country rock in which the Crofty lodes are developed, rather than from oxidation of dissolved Fe²⁺ (which was the cause of red waters at Wheal Jane). (3) Mine water quality during the working of South Crofty was relatively good: waters pumped up from the deepest workings were subjected only to informal treatment (basic settlement of suspended solids in an old stope at the 195 fathom level) before being discharged into the Red River. The mine waters were in compliance with the existing EA discharge consent for the site albeit Zn and Cu concentrations were slightly in excess of levels which would have been called for by European regulations which had been brought in after the granting of the existing discharge consent. The latter point has considerable importance for current discussions over the possible re-opening of South Crofty, as the old discharge consent was revoked after the mine closed, and the EA would be obliged by international law to impose a more strict discharge consent

(complying with the EU Dangerous Substances Directive) on any new mine operator. (4) Detailed surveying by the geologists and surveyors of South Crofty plc identified that the lowest-lying feasible decant point from the modern deep workings after completion of flooding would be to the Dolcoath Deep Adit, rather than directly to the ground surface. Water would migrate from the flooded 20th Century workings of South Crofty via a single deep roadway connection to the North Roskear Shaft, up which it would rise (in effect acting as a giant inverted siphon) to flow onwards to the Dolcoath Deep Adit via a short cross-cut (Figure 5). Some 50 to 80 ls⁻¹ of mine water emanating from the long-abandoned workings of Dolcoath mine to the south were already flowing through the Dolcoath Deep Adit, and it was reasoned that these would provide some dilution for water arising from the modern deep workings, thus muting the potential impact of any 'first flush' on the Red River. The principal source of concern over this identified decant route related to the cross-cut from the North Roskear Shaft to the Dolcoath Deep Adit, which had been partly back-filled with waste rock during adit maintenance operations in the 1950s. To avoid the risk of mine water building up (as it had at Wheal Jane) and bursting out in a spectacular manner, the backfill material was mined out of the roadway at the expense of the EA before South Crofty was finally abandoned. A 'new mine' ('Roskear Mine') was opened for this purpose, so that the last underground mining of the modern era in Cornwall was undertaken for environmental protection purposes rather than for ore recovery. (5) Exploration of the Deep Adit itself by staff of South Crofty plc revealed that the water decanting to the Dolcoath Deep Adit via North Roskear Shaft would further be constrained to pass through an accidentally-constructed "built-in clarifier", which should serve to efficiently remove any excess suspended solids from the water. The 'clarifier' consists of two shafts and a "u-bend" of flooded old workings through which the entire flow of the Dolcoath Deep Adit passes (except at times of very high flow rates). Thus water falls into the Doctor's Shaft, and after about four hours (determined by tracer tests) it resurges a few hundred metres further down-adit via the Daylight Shaft. In its passage through this loop of old workings, the water tends to lose all suspended sediment which it was carrying as it arrived at the Doctor's Shaft. The process of settling is analogous to that of a conventional upflow clarifier (common at most water treatment works). This bolstered confidence in the likelihood that suspended sediment would not be a significant cause of degradation of the Red River following the completion of flooding of the deep voids of South Crofty mine.

With these grounds for optimism in place, the grounds for adopting a 'monitored natural attenuation' approach to site management (see Figures 2 and 3) were established. After the cessation of dewatering in March 1998, water level rise ('rebound') in the workings was monitored in various shafts piercing the system (New Cooks, Roskear, Taylor's, Robinsons). In the later stages, the arrival of water in the lowermost reaches of the Tuckingmill Decline was also observed. Modelling of the rebound process (Adams and Younger, 2002) provided clues as to the likely flux of water which would eventually decant to the Deep Adit, with the impression arising that this would probably be equivalent to around 40-50% of the previous pumped dewatering rate.

The process of rebound in South Crofty yielded a few surprises. The first of these was an apparent 'explosion' during the early stages of rebound in July 1998, which is thought to have been caused by entrapment of gas in a cupola formed by upward termination of a large stope (Younger and LaPierre, 2000). It is believed that trapped gas came to be subject to such a high pressure by the rising water that it eventually fractured the stope crown, giving rise to seismic waves and a loud report widely heard at the surface. The second surprise related to the late stages of rebound, in an inaccessible interval of ground between the shallowest modern workings (around the 195 fathom level) and the Deep Adit (see Adams and Younger, 2002). There was a distinct lack of old mine plans for this interval, albeit accessible

copper workings just above them (e.g. those of Wheal Susan) are well-mapped and reliably recorded as having been excavated in the 1830s. So did the lack of plans mean that this interval had never been worked, or simply that any mine plans for the interval had been lost over the years? The consensus locally was that the latter was more likely to be true. If this was so, one would expect the water level in the workings to continue to rise fairly slowly when traversing this interval, as old voids gradually fill up with water. In the event, the record of water level rise showed that the interval *was* devoid of significant volumes of old workings, for the rate of water rise increased considerably in this interval, at a rate far in excess of any increase that could be explained by the wet weather at the time (Figure 6). Thus, while decant occurred as predicted via the north Roskear Shaft to the Dolcoath Deep Adit, the timing of this event (in November 2000) was several months earlier than had previously been anticipated.

Water quality monitoring during rebound was also undertaken in the New Cooks and Roskear shafts (shaft fittings precluded sampling in Robinsons and Taylors shafts). On the whole, the chemical properties of the shaft samples revealed only modest deterioration in mine water quality, thus vindicating the adoption of a monitored natural attenuation approach to the management of the South Crofty system.

Since the rising waters of the modern South Crofty workings decanted to the Dolcoath Deep Adit as predicted, via the North Roskear Shaft and the newly-cleared cross-cut, water quality changes have occurred. The lack of any other insets to the North Roskear Shaft has meant that all water flowing from the newly-flooded workings has to pass to depths of more than 400 m before flowing upwards and out of the shaft. On the one hand, this forced deep circulation ensures that full advantage is taken of the supply of buffering minerals (which includes many silicates; Younger *et al.*, 2002) present in the country rock before the water sees the light of day. However, the peculiar inverted siphon which the South Crofty flow system comprises (Figure 5) is giving rise to a curious "reverse first flush", in which the waters from the top of the water column in the main body of modern workings will be the last to discharge to the Dolcoath Deep Adit. Recent monitoring of these shallow waters has shown that they are of poorer quality than the mixture currently flowing from the North Roskear Shaft and the water at depth within the main body of modern workings. The shallowest waters sampled in the New Cooks Kitchen Shaft show unmistakable signs of sulphide oxidation (a process conspicuous by its absence in the deepest, modern workings), with a pH around 5.5 and up to 150 mg l⁻¹ Fe. Whether the unavoidable circulation of these waters to great depths prior to their emergence via the North Roskear Shaft will be sufficient to sustain the current 'monitored natural attenuation' remains to be seen at the time of writing (September 2002). To judge from the model of first flush duration proposed by Younger (2000b), it will be at least April

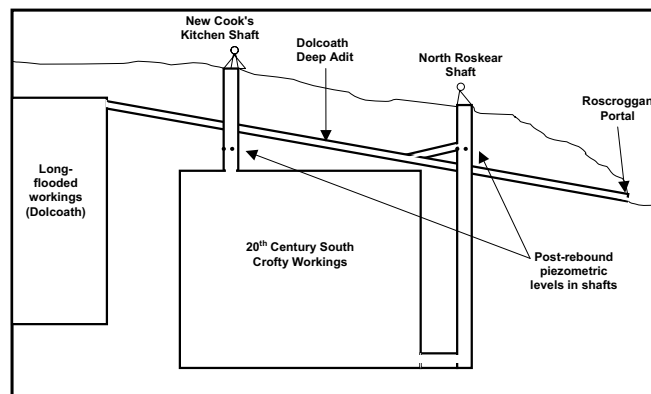


Figure 5. Schematic S-N cross-section through the workings of South Crofty tin mine, to explain why decant from the deep workings occurred only via a cross-cut from the North Roskear Shaft to the Dolcoath Deep adit, which was already carrying a substantial flow (up to 80 ls⁻¹) sourced from the long-abandoned workings of Dolcoath Mine to the south.

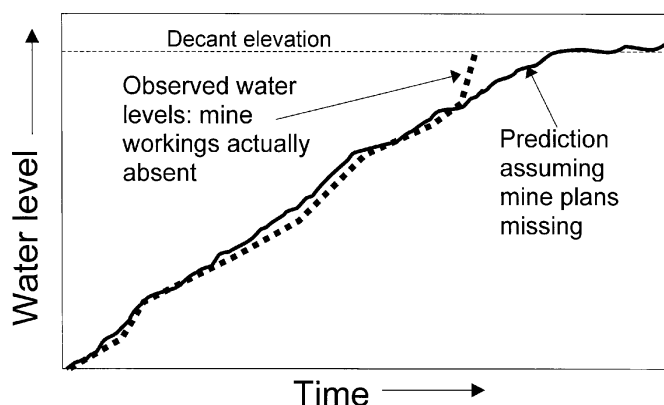


Figure 6. Schematic sketch graph illustrating the surprise acceleration of rebound in the later stages of flooding of South Crofty mine. For the actual data and further discussion, see Adams and Younger (2002).

2011 before the water quality emerging from the South Crofty workings will have stabilised to its long-term character.

Case Study 3: Quaking Houses mine water wetland, County Durham, UK

Quaking Houses is a small, former coal mining village in northwest Durham. In the mid-1980s, some years after the closure of the last working colliery in the district, the local stream (the Stanley Burn) which flows through public open space and a park below the village became visibly polluted with orange and white precipitates, and was found to have a very low pH (around 4). The pollution problems started after a cutting for a new road pierced an old colliery tip (pertaining to the Morrison North and Morrison Busty pits, which had closed in the early 1970s) and triggered oxidation and leaching of pyrite present in the mine spoil. Leachate from the tip got into the road drainage system and caused severe pollution of the Stanley Burn. Up to March 1994, local residents had undertaken a campaign of letter writing, in an attempt to persuade the obvious public bodies (NRA, local councils) to address the unsightly degradation of the Stanley Burn. Given the status of the law on mine drainage at that time, negative responses were all that the residents ever received.

An alternative strategy to letter-writing was therefore launched in collaboration with the University of Newcastle. Drawing upon US experiences, it was considered that a simple wetland system with a thick compost substrate might be a suitable remedial technology for the Quaking Houses discharge. The principle of this form of treatment is that bacteria living in the compost will consume sulphate (which is always present at high concentrations in mine waters), neutralising acidity in the process and trapping ecotoxic metals in the substrate as sulphide minerals. Local residents persuaded the NRA to fund a feasibility study of this possibility.

Work began on December 27th 1994, with a programme of sampling and laboratory testing of possible reactive substrates for a compost wetland. Village residents visited the University laboratories to see the tests in progress and discuss experimental findings, and by early February 1995 all were agreed that installation of a pilot-scale wetland should go ahead. So it was that on 20th February 1995, in the school half-term holiday, a gang of enthusiastic volunteers from the village worked with University staff to build a pilot wetland by hand, into which we diverted around 5% of the flow emanating from the Morrison Busty site. The wetland was fitted with a 0.3 m thick substrate of composted horse manure from the Quaking Houses village stables.

After 18 months of monitoring, this pilot wetland yielded an average acidity removal rate some 50% greater than that reported from similar systems in the USA (Younger *et al.*, 1997). These encouraging findings were used to persuade potential sponsors to finance the construction of a full-scale wetland treatment

system (Jarvis and Younger, 1999). Construction commenced in August 1997, and the system was officially opened in November the same year. The full-scale wetland comprises two compost ponds (each about 200 m² in area) followed by a small aeration pond and rush beds. The substrate in the compost ponds is about 0.5 m thick, and comprises a mixture of horse manure and straw, cow manure and straw (BSE-free, from the University's farms), and some composted municipal waste. Monitoring of the full-scale wetland over its first 27 months of operation (Table 2) revealed mean removal rates for iron, aluminium and acidity in line with target values, so that the Stanley Burn has now been thoroughly restored as a healthy stream ecosystem. The wetland itself has acquired impressive stands of reeds and rushes and is now a public amenity in its own right, which won the UK Conservation Award for 1998.

	pH	Determinand (mg l ⁻¹)			
		Iron	Zinc	Manganese	Aluminium
Influent	6.0	5.3	1.4	3.6	6.2
Effluent	6.6	2.4	1.1	3.0	2.4
¹ mg l ⁻¹ as CaCO ₃ equivalent					
					Acidity ¹
					52
					25

Table 2. Quaking Houses compost treatment wetland for mine spoil leachate. Average influent and effluent concentrations of selected parameters (November 1997 - January 2001) for the compost-based units in the treatment wetland system. (Note that overall contaminant removal also takes place in subsequent aerobic units, the effects of which are not included in these figures).

As the wetland matures, the small amounts of maintenance which it requires are undertaken by members of QHET and University staff. To date, no more than about £300 per year (mainly in terms of equivalent costs for voluntary efforts) has been expended on the maintenance of the Quaking Houses system; this is vanishingly small when compared with the running costs of at least £10000 per annum which a conventional treatment system would have required.

In terms of 'implementing any feasible preventative measures', once the wetland was in place the local council was persuaded to install a 0.5 m cap of clay on the part of the spoil heap which borehole investigations had revealed was the most acid-generating. This has resulted in a decline in infiltration which is manifest in a marked improvement in the quality of water entering the wetland treatment system (Table 3).

	pH	Determinand (mg l ⁻¹)			
		Iron	Zinc	Aluminium	Acidity ¹
Before cap installation	4.5	33.5	2.0	21.3	185
After cap installation	6.0	5.3	1.4	6.2	52
¹ mg l ⁻¹ as CaCO ₃ equivalent					

Table 3. Comparison of average concentrations of selected parameters in spoil leachates at the Quaking Houses site before and after the installation of a 0.5 m clay cap on an acid-generating 'hotspot' area of the Morrison colliery spoil heap.

Case Study 4: Hlobane Colliery, KwaZulu-Natal, South Africa

Hlobane Colliery underlies the mountain of the same name in the uplands of KwaZulu-Natal, and it has been in production for more than 100 years. A detailed history of the colliery is given by Edgecombe (1998). The effects of superimposed underground workings in up to three seams (Alfred, Gus and Dundas) has caused extensive fracturing of the ground surface of the overlying mountain (Figure 7), which is capped by a competent dolerite sill, inducing major changes in the hydrology of the mountain so that all surface runoff on the summit plateau is intercepted by large



Figure 7. One of several major fractures in the summit plateau of Hlobane Mountain, KwaZulu-Natal, South Africa, caused by fracturing of a dolerite sill overlying superimposed underground workings in three coal seams (the Alfred, Gus and Dundas seams) worked from Hlobane Colliery. In this particular case, the large crack corresponds to the rib-edge of zones of total extraction in all three seams. William Pulles of PHD Inc bravely provides the scale.

fractures and transmitted to the mine workings below. Now that production at Hlobane is nearly finished, closure of the mine is planned. However, before this can take place, the South African government must first approve a closure/after-care plan to secure water resources downstream in the long term. Since downstream water resources are used for drinking supply, irrigation and to support endangered species in major wildlife reserves, the stakes for this after-care plan are particularly high. Hattingh *et al.* (2002) have prepared a thorough account of the search for the best long-term solutions for the site. This is a major exercise involving data collection, conceptual and mathematical modelling (of both flows and geochemistry, within the mine and receiving watercourses), identifying realistic simulation scenarios for future conditions and design of appropriate measures for intervention and / or monitoring. Only a brief summary is presented here.

Returning to the logic of Figure 2, it is first apposite to ask whether some preventative measures are feasible. The obvious task in this regard is bridging of the fractures on the mountain top, though adequate design for long-term stability is clearly going to be challenging. Diversion of mountain plateau runoff to form a new waterfall off the mesa edge upstream of the major fractures might be a more sustainable long-term option, albeit more expensive in the short term. However, whilst much of the mountain top runoff can be diverted away from the workings in this manner, it is clear from hydrogeological analysis that residual

discharges of considerable magnitude will remain. Water currently draining from the workings is currently alkaline with relatively modest iron concentrations. However, some discharges from nearby abandoned mines in the same sequence of seams are already markedly acidic, so that some uncertainty besets planning for the longer term.

Even if the Hlobane discharges become acidic in the longer-term, passive treatment technologies are available to deal with low pH and elevated metals concentrations. In this catchment, however, it may ultimately prove necessary to achieve a significant lowering of overall salinity, which with present proven technology would require active treatment. In parallel with the detailed hydrogeological characterisation of the Hlobane system, therefore, the responsible consultants (PHD Inc) are engaged in field research into the feasibility of passive 'biodesalination', using a large pilot plant on the adjoining Vryheid Coronation Colliery site. Results to date are encouraging, though significant challenges remain to be overcome in relation to long-term stability/economics of the technology. Given the vast scale of the South African mining industry, and the existence of highly polluted drainage in both the larger coalfields and in the gold fields of the Rand, the trail-blazing study at Hlobane is of far more than local importance in a country where the socio-economic and environmental stakes raised by long-term pollution from abandoned mines are very high indeed.

Case Study 5: Abandoned tin/silver mine, Milluni, Bolivia

Mine water pollution is seriously degrading water resources in Bolivia, the poorest country in South America (e.g. Pescod and Younger, 1999; Banks *et al.*, 2002). Indeed, even the main water supply reservoir of the capital city, La Paz, is severely polluted by perennial drainage from abandoned tin and silver mines in the Cordillera Real of the Andes (Pescod and Younger, 1999). Some of this extremely polluted water is used without treatment by impoverished local residents, for certain domestic and agricultural purposes (e.g. as a water source for herds of llamas). Although the city waterworks efficiently remove iron from the water, unpublished data show that toxic elements such as cadmium have occasionally entered the public water supply in La Paz at concentrations well above the limits specified by the World Health Organisation.

The mine from which the pollution emerges, Mina Milluni, was closed abruptly in 1985 as a consequence of the same global slump in tin prices which led to the final closure of Geevor mine in Cornwall. The former mining company at Milluni has neither the financial resources nor the legal responsibility to remediate the polluted drainage. It currently seems that the problem will continue indefinitely unless voluntary remedial action is taken by some other organisation. For a problem of this magnitude, an active treatment solution would be the most logical (Figure 4). However, the capital and revenue funding for a high-tech treatment plant is unlikely to be forthcoming. The question therefore arises as to whether limited passive treatment would be preferable to zero treatment. Drawing inspiration from UK experiences, local engineers undertook preliminary experiments to assess the feasibility of treating the acid water using bacterial sulphate reduction and limestone dissolution. The problems of implementing this technology at Milluni are considerable: lying at about 4400 m above sea level, nocturnal freezing is the norm for much of the year. Furthermore, supplies of compost materials similar to those used to date in the UK (e.g. horse and/or cow manure, with composted tree bark mulch) are not readily available in the Bolivian Andes. However, llama droppings are abundant, though it hardly needs stating that no previous data exist on their performance in acid mine drainage remediation systems. In a series of tanks, therefore, limestone gravel was buried beneath 0.5 m layers of llama droppings and a continuous flow of acid drainage was directed through the tanks from 30th June to 28th November 2000 (which is during the coldest period of the year). The findings from these experiments were extremely encouraging.

As hoped, the llama droppings did promote the activity of sulphate-reducing bacteria (sulphate reduction averaged around 16%), and the average pH of the water rose from 3.2 to 6.3 on passing through the four tanks. At the time of writing, funding is being sought to support further work with local artisanal miners and other residents to develop larger bioreactors in the field at this and other sites, with the long-term aim of empowering these poverty-stricken communities of indigenous people to reclaim and sustain the water sources which are currently denied to them by the legacy of large scale mining.

CONCLUSIONS

This paper has outlined the basic logic involved in deciding when and how to implement various remedial options for polluted mine drainage. While it is clear that a rational decision-making framework now exists, it is also true that significant challenges remain. These are particularly evident where funding is initially insufficient (e.g. Quaking Houses, Milluni) and/or where the socio-environmental stakes are particularly high (e.g. South Africa). Even in districts with wet climates and thus relatively high availability of dilution (such as Cornwall), the challenges of achieving sustainable long-term solutions are considerable. It is hopefully clear from the foregoing examples that truly 'holistic' mine water remediation is a multi-dimensional business, involving teamwork by consortia of geoscientific, hydroecological and socio-economic specialists. These wider issues are the focus of a current EU R&D project 'ERMITE' (Environmental Regulation of Mine Waters in the EU); see 'acknowledgements' for details. While significant advances have been made in understanding mine water pollution phenomena over the last few decades, the attainment of effective multi-disciplinary engagement will continue to demand major collaborative efforts far into the future.

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