Preface

These Guidelines have been prepared by the "ERMITE" (Environmental Regulation of Mine waters In The European Union; FP5 contract no. EVK1-CT-2000-00078; www.minewater.net/ermite) Consortium, a project of the European Commission's 5th Framework R&D. Formally, this document corresponds to ERMITE deliverable number D6. The Guidelines have been prepared in response to the need to evaluate and develop solutions to mine water management problems specifically at the catchment scale. Most previous guidelines for managing mine waters have been focused at much smaller scales, e.g. individual mine sites (for instance, the handbook on "Technical Management of Water in the Coal Mining Industry", NCB 1982) or individual effluent discharges from mine sites (such as the new guidelines on passive in situ remediation of acidic/metalliferous mine waters produced by the PIRAMID Consortium (www.piramid.org 2003). However, trends in regulatory policy in the water sector world-wide are towards more integrated, targeted and prioritised approaches to management of water quantity and quality at the catchment scale. This is evident, for instance, in the South African National Water Act of 1998 (Republic of South Africa 1998) and in the European Union's Water Framework Directive (WFD) of 2000. It is the latter which provides the particular context for this publication, and readers wishing to fully understand the provisions of the WFD are therefore encouraged to refer to the original directive (2000/60/EC; European Commission 2000).

The guidelines are intended to assist those involved with the implementation of catchment management strategies in understanding and dealing with the peculiarities of the effects of mining on the water environment. Chapter 1 sets the scene and provides a brief overview of the key issues associated with mining in the catchment management context. Those with substantial experience in the mining sector may well wish to skip this section. Chapter 2—4 provide specific advice on technical and managerial measures appropriate to dealing with mining issues within overall catchment management operations. Many of these measures will be recognised as existing good practice by those familiar with the mining sector, but amongst the more familiar recommendations are significant innovations developed during the execution of the ERMITE project, with the catchment-management perspective particularly in mind.

While the Guidelines necessarily include many recommendations aimed at "hydrologically-defensive mine planning" (see Section 2.3.2 in particular), realism demands that we accept that many former mining activities have not been pursued preceeded such a paradigm, so that measures to deal with post-closure problems must also be specified (Chapter 4), even though future implementation of the recommendations during the exploration and working (Chapter 2) and closure (Chapter 3) phases ought to minimise the need for post-closure interventions for mines yet to be worked.

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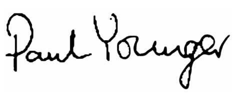
While these guidelines take into account the views of many stakeholders who participated in national and European level meetings as part of ERMITE, the opinions and recommendations expressed in this document are not to be taken as representing the official position of any of the organisations involved, nor of the European Commission.



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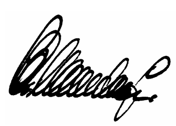

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Mining Impacts on the Fresh Water Environment: Technical and Managerial Guidelines for Catchment Scale Management

ERMITE-Consortium
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1 Mine Water Issues and Impacts

1.1 Introduction

Mining almost always impacts upon the natural water environment, and its effects may be manifest throughout the mine life cycle. The impacts can be beneficial. For instance, some mine waters are of good enough quality that they can be used for public supply (Banks et al. 1996). Beneficial impacts are noncontentious, and thus require little further discussion. On the other hand deleterious impacts, such as depletion of water resources by dewatering, or the pollution of surface watercourses by poor quality mine waters and mine waste leachates, demand careful scrutiny. Because some of these impacts can persist for centuries and even millennia after mine closure, routine approaches to the management of industrial discharges may not be wholly suited to regulation of the impacts of mining on the water environment. This document provides guidance on approaches to catchment scale water management which *are* appropriate to mined environments. However, first an overview of the ways in which mining affects the water environment is warranted, and this is presented in this first section of this report. In most of this introductory chapter, referencing of individual examples of the various impacts is deliberately minimised to economise on space; readers requiring a thorough bibliography on these topics should consult the recent synthesis of Younger et al. (2002) or Lottermoser (2003). Readers with broad experiences in the management of water issues in mining can skip this section and move on to Chapters 2 through 4.

1.2 How Mining Changes the Natural Water Environment

It is convenient to subdivide the potential impacts of mining on the water environment in order to discuss them with clarity. The subdivision followed below distinguishes between impacts associated with:

- mining *per se* (i.e. extractive activities),
- mineral processing and disposal of mine wastes,
- mine dewatering, and
- post-mining flooding and uncontrolled discharge of polluted waters.

However, it must be emphasised that this convenient subdivision is rarely so clear, on a real mine site, where all four categories may coexist to varying degrees. This considerably complicates the attribution of causes to observed water problems, underlining the importance of seeking advice from experienced specialists when devising management strategies.

Impacts due to mining *per se* – While a mine is operational the act of mining itself (i.e. the sinking of shafts or open pits and the excavation of overburden and ore) can have a significant impact on the natural water environment. This is because mining activities inevitably disrupt preexisting hydrological pathways within the host strata. While underground mining tends to have less conspicuous impacts on surface water features than an open pit surface mine has, all types of mining have the potential to directly disrupt groundwater flow (e.g. Booth 2002), which in turn can affect surface waters that are in hydraulic continuity with the affected groundwater systems. That having been said, it is possible to use so-called "supported" methods of underground mining (where pillars of intact rock are left in place to minimise subsidence) to reduce the disturbance to overlying groundwater systems that would be caused by "caving" methods of mining (e.g. longwall mining of coal or block caving of metalliferous ores), which result in wholesale fracturing of the superincumbent strata.

In the majority of cases, however, the impacts on the natural water environment arising from the act of mining itself tend to be relatively localised and limited when compared to other mining related impacts such as those associated with dewatering.

Mineral processing operations and seepage of contaminated leachate from waste rock piles and tailings dams – Waste products from both mining and mineral processing operations are often conveyed to and contained in large heaps (for coarse-grained discard) or in slurry impoundments known as "tailings dams" (for fine-grained mineral processing wastes).

The peculiar geotechnical properties of many fine-grained tailings, combined with the abundant presence of water, can lead in some situations to problems of dam instability, instances of which have led to several notorious pollution incidents in Europe and elsewhere over the past few decades. When tailings dam failures have been analysed, it is commonly found that the root cause of the failure is *not* poor design, but departures from the original design during the process of construction (UNEP 1996).

Seepage of contaminated leachate from waste rock piles and tailings dams is a significant cause of surface and groundwater pollution in many mining areas. This form of contamination can arise while the mine is operational and without remedial action, can persist long after site operations cease. In some cases, previously innocuous mine waste deposits have suddenly begun to generate acidic and/or metalliferous leachates many years after they have been revegetated and left unattended.

Mine dewatering – Dewatering is essential in all but the most limited of mining operations, both to secure access for miners and mining machinery to the mineral reserves, and to ensure the safety of personnel working in mine voids that adjoin naturally water bearing strata or old mine voids prone to flooding. Dewatering can be achieved by various means (see Section 2.4), but its impacts are due either to:

- disposal of the pumped water (especially if this is saline) and/or to
- depression of the water table around the dewatered zone (over areas which might vary, depending on the scale of the operation, from a few hundred metres to more than 2.500km^2; e.g. Younger et al. 2002).

The consequences of water table depression due to mine dewatering can include:

- Decreased flows in streams, wetlands, and lakes that are in hydraulic continuity with the affected groundwater body.
- Lowering of the water table in the vicinity of water supply or irrigation wells, leading at least to an increase in the pumping head (and therefore in pumping costs), if not to the complete drying up of wells.
- Land subsidence, either due to compaction of fine-grained sediments (especially silts and clays), or the collapse of voids in karstic terrains as buoyant support is withdrawn.
- Surface water or groundwater pollution, if the pumped mine water is of poor quality and is discharged to the natural environment without prior treatment.

Many of these impacts can be anticipated before they occur and mining companies will generally take steps to mitigate them. The effects of dewatering on water availability are discussed more fully in Section 1.3.

Post mining flooding of mined voids and discharge of untreated water – Although the cessation of active dewatering will often alleviate some of the impacts just discussed, the abandonment of mines can eventually lead to renewed impacts, following the recovery of groundwater levels (a process termed "rebound") to the natural base level of drainage. The process of rebound in underground mines commonly leads to a marked deterioration in the quality of mine water (Henton 1979; Wolkersdorfer 1996; Younger 1993, 1998b, 2000a, 2000b). In surface mines, water quality can deteriorate when backfilled materials are initially saturated after restoration. The flooding of open pit mines to form pit lakes can also cause water quality to deteriorate (Geller et al. 1998). The completion of mine water recovery usually results in the overspill of untreated waters after flooding is complete. This stage of the mining life cycle generally has a long lasting and significant impact upon the water environment. Mining induced changes in water quality are discussed more fully in Section 1.4.

Besides chemical changes, mine water rebound can also initiate physical changes in the mined system such as:

- subsidence (caused by the erosion of voids or support pillars by rapidly flowing water) and
- fault reactivation (whereby the increase in pore pressure in faulted strata can reduce the frictional resistance to movement in extensional fault planes).

Both of these processes can result in substantial property damage. Rising mine water can also cause drive the release of potentially hazardous methane (CH_4), radon, and oxygen deficient air (rich in CO_2) from abandoned underground workings.

Having listed the relationships of the various impacts of mining to the different phases of the mine life cycle, in the following two sections further details will be discussed relative to their impacts on *availability* and on *quality* of water.

1.3 Impacts of the Working and Dewatering of Mines on Water Availability

Sub water table mining affects water availability because dewatering mining areas invariably causes drawdown of the water table in the surrounding rock. The consequences of such drawdown are conceptually the same as those caused by pumping groundwater for other purposes (e.g. for public water supply) and can therefore generally be analysed in an analogous manner. A substantial literature exists on the impacts of public supply wells on ground and surface water resources. None of the more recent analyses have changed the basic principles as enunciated by Theis (1940), who clearly showed that any abstraction from groundwater will eventually be matched by some combination of the following three responses:

- a decrease in the volume of groundwater in natural storage
- an increase in the rate of groundwater recharge
- a decrease in the rate of natural groundwater discharge.

The peculiarity of mine dewatering systems lies in the deliberate maximisation of the first of those three responses above, which is principally manifested in a lowering of the water table. Lowering of the water table *per se* has a number of socio-environmental consequences, not all of which are bad. For instance, the drying out of previously waterlogged land can render it more useful for agriculture, forestry or economic development, though the eventual rebound must planned and accounted for. On the negative side, lowering of the water table can leave pre-existing abstraction wells high and dry. It can also lead to desiccation of ponds that previously occupied enclosed hollows. Where these were of ecological value, the localised impact can be serious.

Lowering of the water table in areas where surface runoff was previously being generated above saturated ground can induce further recharge to the subsurface (albeit this may locally be at the expense of wetland habitats). Water can also be induced to enter the subsurface directly through the beds of nearby streams and rivers. For instance, dewatering of an open pit limestone mine in Poland induced varying amounts of inflow from the Vistula River, which accounted for as much as 80% of the total water make (Motyka and Postawa 2000). In extreme cases, dewatering can lead to streams drying up altogether for part of their course and/or for part of the year. Examples of the latter are particularly common in the mining districts of Mediterranean Europe.

A decrease in the natural discharge of groundwater from aquifers is probably the most common impact of mine dewatering. The drying up of springs in limestone aquifers subject to quarry dewatering is widely reported (e.g. Hobbs and Gunn 1998), although in karstic areas effects of dewatering can be extremely difficult to distinguish from natural waxing and waning of flows due to the inherently low groundwater storage capacities of fractured lime-stones. Probably more widespread, but usually far less conspicuous, are reductions in the rates of natural groundwater discharge to perennial streams. In many cases, such reductions will be masked by the disposal of dewatering pump effluents into the same streams.

Where the quality of the pumped water is good, the disposal of dewatering effluents can actually be a considerable benefit to surface water catchments, diluting, for instance, other polluted waters. Wetlands can also be supported by pumped discharges from mines.

1.4 Impacts of Mining on Water Quality

Impacts of mining on water quality has been the subject of many thorough reviews in recent years (most of which are in turn listed by Younger et al. 2002) and will therefore be dealt with here in an abbreviated manner. Mining can affect water quality in three principal ways:

- by liberation of sediment, loosened by excavation processes,
- by mobilising preexisting waters of poor quality (most commonly naturally saline waters) so that they artificially enter the freshwater environment, and
- by promoting the weathering of previously stable minerals, which release ecotoxic metals and other solutes, usually also contributing some degree to salinization of the waters with which they come into contact.

In most active mines, steps will be taken to limit the discharge of excessive sediment loads (suspended solids). However, incautious management of tailings, spoil heaps, and mineral stockpiles always has the potential to lead to runoff of water with excessive suspended solids to nearby streams. Where the sediment includes reactive particles (e.g. sulphide minerals, or organic matter) the suspended solids can contribute to deoxygenation of the water, with potentially serious consequences for aquatic fauna (see Section 1.5).

The pumping of saline mine waters is a common cause of fresh water degradation in many European mining districts (e.g. Gandy and Younger 2002, Wedewart 1995). Saline waters are associated with the mining of salt deposits, and the mining of other commodities contiguous to salt bearing strata, but also occur as trapped, ancient waters (possibly of marine origin in the distant past) within the deeper parts of most European coalfields. In the Upper Silesian Coalfield of Poland, disposal of saline mine waters has led to the degradation of freshwater resources in the Odra and Vistula Rivers. Some of these saline mine waters are also notably radioactive and their disposal into rivers can lead to increased environmental exposures to ionising radiation. In general, one would expect these deep-seated saline mine waters to be less prevalent post-closure than during mining. However, emerging evidence in northeastern England suggests that saline waters are persisting and intruding to shallower depths during mine flooding than had ever been anticipated, giving rise to particularly difficult management problems where the long-term degradation of shallow aquifers and surface waters must be prevented.

Before most minerals and coal are mined, they are relatively chemically stable in the air free subsurface environment. However, when mining commences, some minerals are especially vulnerable to weathering under the influence of atmospheric oxygen and water. Sulphide minerals such as sphalerite (ZnS), galena (PbS), chalcopyrite ($CuFeS_2$), millerite (NiS) and arsenopyrite (FeAsS), weather to release both sulphate (SO_4^{2-}) and their constituent metal ions (e.g. Zn^{2+}, Pb^{2+}, Cu^{2+}, Fe^{2+}, Ni^{2+}, and the metalloid As in the case of arsenic sulphide minerals into aqueous solution. A large proportion of these metals may well be trapped on or close to the source minerals by precipitation of secondary minerals. This is especially marked in the case of Pb^{2+}, for instance, which very often accumulates as cerrusite ($PbCO_3$) on the surfaces of weathering galena crystals. The extent of metal immobilisation by formation of secondary minerals can be assessed by comparing the molar concentrations of sulphate and the relevant metals in a given mine water. Sulphate can be present at molar concentrations ten or more times greater than those of the corresponding metals, in marked contrast to the 1:1 correspondence within the mineral formulae. Despite the entrapment of more than 90% of the metals as secondary minerals, the remaining fraction that escapes to solution is usually sufficient for them to exhibit toxic effects to fish and invertebrates in receiving rivers (see Section 1.5).

The oxidative weathering of the iron disulphide pyrite (FeS_2) and its close relative marcasite (same composition, different mineral structure) elevates acidity and decreases pH, in addition to mobilizing sulphate and iron. This, in turn, enhances the mobility of metals derived from the weathering of other sulphide minerals and promotes the dissolution of other metals previously held on mineral surface sorption-sites. Indeed, the acidity of waters derived from pyrite oxidation is often so marked that clay minerals are dissolved, releasing aluminium (Al^{3+}).

The role of secondary minerals in limiting metal release has already been discussed. However, the immobilisation of metals in this way may not be permanent. Seasonal fluctuations in the hydrology of mined voids can lead to subsequent dissolution of some of these minerals, especially rapidly dissolving phases such as hydroxysulphates. Even more extreme is the wholesale dissolution of hydroxysulphate minerals (and, to a lesser degree, hydroxides and carbonates) during the final flooding of a mined void following the cessation of mining. This process is responsible for the marked deterioration in water quality that often accompanies the flooding of mine workings. Younger (2000a) noted that this commonly leads to a tenfold increase in the concentrations of contaminants (particularly iron).

1.5 Impacts of Mining on Aquatic Ecosystems

1.5.1 Effects due to De-oxygenation and Increased Suspended Solids

An understanding of the effects of mining on aquatic ecosystems is a *sine qua non* for the attainment and sustenance of good ecological status in receiving watercourses, i.e. for compliance with the European Union Water Framework Directive. Despite the great importance of this topic, the last thorough review on this subject was published more than 15 years ago (Kelly 1988); hence this section goes into rather more detail than the preceding sections (which concerned topics well covered by recently published reviews).As noted above, many mine waters contain reactive metals and other components that consume oxygen. They ca have very strong impacts on the biota, often resulting in a complete loss of invertebrates and fish in affected reaches. Oxidation of ferrous iron (Fe^{2+}) to the ferric form (Fe^{3+}) is a particular problem in mining affected streams, for it gives rise to the precipitation of voluminous orange/red rusty coatings of ferric hydroxides/oxyhydroxides (generally termed “ochre”) on stream beds. A thick coating of ochre on a streambed can eliminate benthic algae and benthic invertebrates. Even where the ochre remains in suspension (or is re-suspended following erosion of benthic precipitates), nonreactive suspended solids can adversely affect aquatic biota as described below.

Where mine sites give rise to heavy loadings of suspended sediments in receiving watercourses, the increased turbidity decreases light penetration, which directly affects the primary producers in aquatic ecosystems, i.e. the periphyton and algae, by inhibition of photosynthesis. In doing so, it reduces the availability of food for the macroinvertebrate community, and for the fish population that feeds on them (MacDonald et al. 1991). In addition, indirect effects of increased turbidity include the disruption of mating and territorial behaviour patterns, which are highly dependent on visual cues and have strong effects on reproduction, abundance, and population size (Hodgson 1994).

Besides these indirect effects, elevated suspended sediment loads have a number of direct effects on fish, principally due to clogging of the gills (limiting the flow of oxygen-bearing water over them) and abrasion of the gill epithelium (i.e. damaging the sensitive tissues that transfer oxygen to the bloodstream). While these effects are only likely to be lethal to most adult fish where exposure to high sediment concentrations (> $1000 mg \cdot L^{-1}$) continues for several days (Hodgson 1994), much lower sediment concentrations (90—$100 mg \cdot L^{-1}$) acting over much shorter time periods have been found to reduce fish life expectancy and increase susceptibility to disease under laboratory conditions. In addition, the accumulation of sediments (including ochre precipitates) on biological surfaces such as gills, eggs or other tissues has frequently been reported to affect the survival, reproduction and behaviour of aquatic animals (e.g. suffocation of trout eggs, precipitates on the gills of *Ephemeropterans* and caddis larvae; Vuori 1995)

1.5.2 Effects of Low pH and Ecotoxic Metals

In general, a pH range of 5.0—9.0 is not directly lethal to fish and other aquatic invertebrates, albeit if pH is maintained below 6.5 for extended periods it can result in decreased reproduction and growth of fish and aquatic invertebrates (e.g. Ikuta and Kitamura 1995). In addition, unfavourable pH conditions tend to increase the toxicity of other common pollutants. For example, while 4mg·L^{-1}) of iron would not present a toxic effect at a pH of 5.5, as little as 0.9mg·L^{-1} of iron at a pH of 4.8 can cause fish to die.

The release of ecotoxic metals to the aquatic environment often results in more serious environmental consequences than those associated only with a lowering of pH. The particular hazards posed by the various metals released to the aquatic environment will depend on their:

- persistence in the various environmental compartments (water, suspended solids, and sediment),
- toxicity to specific aquatic organisms (which varies from species to species, and even between differently acclimatised members of the same species),
- bioaccumulation by these organisms,
- bioamplification along the trophic web, and
- indirect effects on the biota.

The persistence of polluting substances in a given environmental compartment (water, sediment, soil) increases the possibility that it will accumulate over time and that exposure will increase with additional inputs. Although metals do not undergo degradation, they are not necessarily bioavailable: changes in chemical speciation related to the interchange of metals between the sediment, water and soil compartments result in varying degrees of exposure to potentially toxic forms by the biota. The most relevant measure of exposure, persistence of bioavailable metals (Adams et al. 2000), depends chiefly on metal complexation, precipitation and mineralization. The system is further complicated in the aquatic environment by the multiphasic states in which the metallic species occur, with complex equilibria between metals in the sediment and the aqueous phases where soluble, colloidal and particulate forms are all potentially present (Coombs 1980; Cameron and Liss 1984).

In general, metal ions are generally not persistent in the water column of natural water bodies (Adams et al. 2000). Instead, metals tend to associate preferentially with the sediment and suspended particulate material (Philips and Rainbow 1993; Martin 2000). A variety of laboratory, field mesoscosms and whole lake experiments have shown that total and soluble fractions of most metal ions decline rapidly following release in surface waters, showing an exponential decline with half-lives typically less than 30 days (except for Cd in saline waters, in which it forms a soluble chloride complex). Rates of decline increase with increasing particulate fraction, due to settling of particle-bound metals (Adams et al. 2000). Removal of soluble metals by the particulate fraction is continuous since additional particles enter the water column, through algal or local sediment inputs.

Persistence of metals in the water column (and in sediment pore waters; see below) currently provides the best known assessment of their potential toxicity, since aquatic hazard assessment procedures are based upon toxicity tests designed and carried out in surface waters. Toxicity of mono- and divalent metals is due predominantly to the free metal ion in solution. Hence, most toxicity studies use soluble metal salts under the assumption that metal ions are then completely dissolved and bioavailable. As solubilities of most metals depend on pH, dissolved oxygen, water hardness and other factors, these variables also influence metal toxicity. For example, toxicity increases with increasing temperature, and decreased oxygen content. In general, toxicity tests have been used to select concentrations deemed to be protective for most species in the environment, based on the response of the most sensitive species (typically planktonic invertebrates, such as *Daphnia magna* and fish; e.g. Warrington 1996, EPA 2001). Toxicity thresholds usually range from 0.1—1000μg·L^{-1} and show large variations between the various metals (e.g. U.S. EPA chronic water quality criterion ranges from 0.12μg·L^{-1} for Ag to 1000μg·L^{-1} for Al; Adams et al. 2000), thus discouraging the use of common concentration limits for all metals.

Because most polluted mine waters contain mixtures of different metals in solution, it is necessary to weight the effects of the various metals present in the mixture. The simplest models assume pure summation of the toxic effects of the different metals present; e.g. the toxic unit (TU) model relates the toxicity of each chemical present in the mixture to its toxicity, most often expressed as its 50% lethal concentration (LC50):

$$\text{Number of TU} = \frac{\text{Concentration}}{\text{LC50}} \qquad (1)$$

and adds their strengths to obtain a total number of toxic units (Vermeulen 1995). However, the toxicity of metal mixtures not only includes the simple summation of their individual effects. Significant interactions between different metals often result in an enhancement of their toxic effects (synergism), due for example, to physiological inhibition of metal excretion (as proposed for Cd effects on Zn accumulation by perch; Berninger and Pennanen 1995). In a few cases, metals can mitigate each others effects (antagonism), due for example, to competition between metals for common-sites of action or uptake (Coombs 1980) or, in the longer term, from physiological interactions during detoxification processes (e.g. formation of Fe granules during the detoxification of surplus metals; Vuori 1995). Out of 26 studies on the toxicity of metal mixtures reviewed by Vermeulen (1995), 13 (50%) showed synergistic effects, six (23%) were antagonistic and seven (27%) purely additive. Virtually all these studies utilised mixtures of Cu with one or more metals, typically Zn, Cd, or Hg. Synergistic effects of metals predominated in experiments carried out using fishes (eight out of twelve studies), typically involving increases in mortality. Synergistic and antagonistic effects were equally frequent on experiments carried out using invertebrates (four each out of eleven studies), typically involving effects on mortality, reproduction, and/or physiological activity.

Although the existence of interactions between toxic substances has been known for more than five decades (e.g. antagonistic effects of Pb on Cu were reported by Jones 1939), the joint toxicity of chemical mixtures released to the aquatic environment has received surprisingly little attention in the literature, perhaps due to the complexity and variability of the various effects. Nevertheless, some generalisations can now be reasonably made. Firstly, mixture toxicity depends on various environmental factors, such as water hardness, temperature and even time of exposure (Vermeulen 1995). Second, comparable mixtures of metals shows contrasting toxicity effects on different organisms. For example, Cu and Hg had synergistic effects on a copepod, additive effects on a brine shrimp and antagonistic effects on an amphipod (Vermeulen 1995). Such contrasts in interactive effects of metals probably reflect the interspecies variations in mechanisms of metal regulation or tolerance (see below). Until more research allows for further generalisation, the most precautionary approach to the characterisation of the toxicity of mixtures probably lies in the use of joint toxicity indices, particularly those based on the sum of toxic units present in the mixture (such as the Additivity and Toxicity Enhancement Indices, Marking 1977). In this way, the toxic unit concept discussed above allows for the assessment of joint toxicity not only by making summation of different magnitudes of concentrations, but also by offering an objective and comparable yardstick against which synergistic and antagonistic effects may be evaluated (Vermeulen 1995).

Bioavailability of metals in soils and sediments, particularly under conditions of high siltation, is another important aspect of site-specific risk assessment for metals (Adams et al. 2000). While bed sediment can be viewed as a “sink” for pollutants, it can also function as an important contributor of remobilised contaminants to the environment, in many cases long after the original discharges have ended (Miles and Harris 1971). Furthermore, metals immobilised in aquatic sediments and floodplain soils may contribute substantially to downstream metal concentrations (Martin 1997, 2000): a large percentage of eroded sediments are often stored in the drainage basin rather than removed immediately from it and, over time (years to centuries), they move through the basin during flood episodes (James 1989; Beach 1994; Philips 1997). Benthic organisms living in metals-contaminated sediments have two primary routes of exposure: direct exposure to metals in (surface and pore) water and the accumulation of metals via food supply (as particulate matter in the tissue of other organisms). The importance of these two routes is likely to vary in extent between individual contaminants, Al, Fe, Pb, Mn) tend to be found almost completely in the particulate-associated fraction, while more soluble metals such as As, Cd, Se are maintained in solution to a greater extent (Vermeulen 1995).

As for surface water, metal concentrations in pore water are often considered the best surrogate measure of metal bioavailability. Anoxic sediments may immobilise metals in the form of sulphide precipitates, minimising their bioavailability even in the presence of high concentrations on a sediment dry weight basis. For example, cationic metal activity and toxicity in the sediment pore water system is controlled by a key partitioning phase, the acid volatile sulphide (AVS) fraction (Ankley et al. 1996). AVS binds a number of cationic metals (such as Cd, Cu, Ni, Pb, Zn) forming insoluble sulphide complexes with minimal biological activity, as indicated by short-term laboratory studies, life cycle laboratory toxicity tests and field colonisation experiments using freshwater and marine sediments and organisms. For these metals, therefore, AVS concentrations can be combined with the molar concentrations and summed toxicity (in toxic units) to provide an assessment of sediment toxicity.

However, food may be at least as important as water as a pathway for metal uptake by aquatic invertebrates, particularly for predators but also for other feeding groups, such as herbivores, detritivores and filter feeders (Hynes 1963; Wang et al. 1996; Kiffney and Clements 2003). That food should be a significant pathway for metal uptake is hardly surprising, given that benthic primary producers and decomposers often accumulate and tolerate high concentrations of certain metals without suffering any deleterious effects (e.g. aquatic plants – Siebert et al. 1995; Sparling and Lowe 1998; Sánchez et al. 1998; aquatic fungi, Miersch et al. 1997 and refs. therein) and can be expected to transfer them to herbivorous and detritivorous invertebrates. These invertebrates often suffer deleterious effects from metals ingested in this manner (e.g. *Gammarux pulex* feeding on food containing Cd-contaminated fungi suffered higher Cd loads and increased mortality – Maltby and Booth 1991; Abel and Bärlocher 1984). In turn, these invertebrates may transfer the metals to higher trophic levels; for example, Woodward et al. (1994) found that diet was more important than water as a metal uptake route for rainbow trout, contributing significantly to reduced fish survival and growth.

Within a given organism, the contribution from ingested food (as compared to the dissolved phase) to metal uptake may vary also between metals. For example, a detailed study on the uptake, assimilation and excretion of six different metals by the marine filter feeder *Mytilus edulis* revealed that the contribution of particulate food to metal ingestion ranged from 96% in Se to 4—30% in Co, and it was governed by both, trace element partitioning coefficients for suspended solids and the assimilation efficiency of ingested trace elements (Wang et al. 1996).

Bioaccumulation varies greatly between the different metals, with bioaccumulation factors (BAF: concentration in animal tissue/concentration in the abiotic surrounding medium) differing by several orders of magnitude. For example, average BAFs of 1, 270, and 42,000 have been reported for Co, Pb, and Hg, respectively. Tolerance to accumulated metals may be achieved through methylation, the production of metal-binding proteins or their sequestration in intracellular granules, which may be excreted or stored (Coombs 1980, Adams et al. 2000). The mechanisms that organisms use to cope with potentially harmful metals have important implications for metal transfer to higher trophic levels, since they may determine their bioavailability to predators and/or detritivores (Kiffney and Clements 2003). For most metals in the aquatic environment, there is general agreement that consumers take up and accumulate them from their contaminated food, often to high concentrations (i.e. there is bioaccumulation). In some cases, such as As and Pb, bioaccumulation diminishes with increasing trophic level, showing weak transfer from zooplankton to fish (Chen and Folt 2000; Chen et al. 2000), perhaps due to regular loss of metals bound to the zooplankton's exoskeleton during moulting (Schäffer and Ratte 2000). However, biomagnification, which takes place when the transfer efficiency of a contaminant is greater than the biomass transfer efficiency, is rarely involved (see e.g. Kay 1984; Biddinger and Gloss 1984; Schäffer and Ratte 2000; Adams et al. 2000). Exceptions to this are methylated mercury (MeHg) and Zn (see e.g. Stemberg and Chen 1998; Chen et al. 2000), and perhaps Se (Biddinger and Gloss 1984). Biomagnification of other metals, such as Ca and Pb, may also take place in aquatic food webs that include "nonaquatic" (i.e. air breathing) components, such as piscivorous water birds and marine mammals (dolphins and seals; USAEWES 1985; Schäffer and Ratte 2000).

Whether biomagnification plays a role in the trophic transfer of metals remains a matter of debate in research circles. Hg provides an illustrative example of the difficulties faced in the analysis of bioamplification in aquatic food webs. While Hg contents has been observed to increase with increasing trophic level in a variety of food webs (e.g. Becker and Bigham 1995), there has been controversy over whether this simply reflects an increased bioaccumulation in longer-lived organisms (typically occurring at higher trophic levels) or whether it results from bioamplification. This issue has proven difficult to resolve, owing to uncertainties in the characterization of trophic structure in complex food webs (Atwell et al. 1998). The use of stable isotope techniques to elucidate food webs has indicated that, in both lacustrine and marine food webs, biomagnification of Hg *is* responsible for the high levels of MeHg in the upper links of the food chain (Atwell et al. 1998; Bowles et al. 2001; Power et al. 2002). In Lake Murray (Papua New Guinea), both MeHg concentrations and the proportion of total Hg present as MeHg increased with trophic level (from $0.015 \mu g\, g^{-1}$ to approx. $0.4 \mu g \cdot g^{-1}$ MeHg on average, representing < 1% and 94% of total Hg, for seston and piscivorous fish respectively; Bowles et al. 2001). Despite the lack of conspicuously elevated concentrations of inorganic Hg or MeHg in the lake water column or sediments, over 23% of the piscivorous fish had Hg concentrations exceeding the WHO recommended limit for human consumption ($0.5 \mu g \cdot g^{-1}$). A comparably strong biomagnification has been observed in North American lacustrine and arctic marine food webs (Kidd et al. 1995; Atwell et al. 1998; Power et al. 2002), with MeHg vs. $\delta^{15}N$ slopes typically between two and three (positive values indicate that the transfer efficiency of a contaminant is greater than the biomass transfer efficiency, i.e. that there is biomagnification, Atwell et al. 1998). Owing to the high bioamplification power of the plankton, fish feeding on the pelagic food web have higher MeHg contents than those feeding on benthos. However, food web structure affected bioamplification factors: in a survey of 38 USA lakes, Hg and Zn concentrations in fish were found to increase in consort with the length of the zooplankton chain, with decreasing food web connectivity (number of feeding links between species), and complexity (number of lateral links – Stemberger and Chen 1998). In eutrophic lakes, however, dilution of mercury in consumed algal cells during algal blooms may decrease mercury concentration in the upper trophic levels (Pickhardt et al. 2002).

It is also important to note that, although the regulatory framework for aquatic pollution has hitherto focused exclusively on concentrations rather than loads, the sessile elements of the biota (such as most benthic invertebrates) integrate the complete history of exposure to metal discharges and thus reflect long-term loadings more closely than instantaneous concentrations. Previous exposure to low concentrations of a metal can increase an individual's tolerance due to acclimation responses (e.g. 14 days of acclimation to 1.5 and 15μg Ag L^{-1} increased LC50 by approximately 30%, from 30—35 to 41—46μg Ag L^{-1} in the fish *Pimephales promelas* – Warrington 1996) while at the same time, bioaccumulation can result in the build up of toxic endogenous levels following long-term exposure to low (sublethal) concentrations. On the other hand, a number of studies on invertebrates and fishes have indicated that different individuals may show considerable variations in sensitivity to metal exposure (Hynes 1963), suggesting that long-term exposure to metal contamination may result in evolutionary responses (at population level). Many anecdotes exist of brown trout populations surviving in long polluted streams in former mining districts where dissolved Zn concentrations are perennially in excess of levels that would prove lethal to non-acclimated fish. Indeed, invertebrate populations chronically exposed to heavy metals often exhibit increased tolerance relative to unexposed populations (e.g. Moraitou-Apostolopoulou et al. 1979), arising from evolutionary responses to the selecting pressure posed by metal exposure (e.g. Klerks and Levinton 1992). However, the benefits of tolerance come with an energetic cost, making organisms more susceptible to novel stresses (such as low pH – Courtenay and Clements 2000); there is thus a trade-off between tolerance to current stressors and sensitivity to novel ones (Kiffney and Clements 2003).

Community effects of metal pollution typically include a reduction in the abundance and diversity of the biota (e.g. Kiffney and Clements 2003; Hynes 1963). Species able to grow on polluted areas are a subset of the local, unpolluted community, i.e. no special "pollution fauna" develops. Aquatic plants are more severely affected than algae (Hynes 1963), although the latter are often removed also due to smothering by loose sediments and metal incrustations (see above). At decreasing metal concentrations downstream, algae may build up to large numbers owing to the absence of grazers (which are typically more sensitive to metal toxicity). Most research on how contaminants interact with benthic invertebrate communities has focused on direct effects, such as changes in abundance of a particular species. In a given habitat, the species present generally show a spectrum of sensitivities to metals and other contaminants, which thus cause changes in the composition of the aquatic community. Some of these changes are fairly predictable, allowing for the development of biomonitoring methods: for example, stream metal contamination generally results in decreasing abundance of mayflies and increased relative abundances of chironomids (Kiffney and Clements 2003). More subtle changes involve indirect effects of metals in species interactions and community function, resulting from alterations in the physiology and/or behaviour of the various organisms. For example, metal exposure affected the territorial behaviour in hydropsychid larvae, relaxing the levels of interspecific competition (Vuori 1994), causing benthic invertebrates to be more susceptible to predation (Clements et al. 1989; Clements 1999; Kiffney 1996; Lefcort et al. 2000) and increasing the mortality rate of parasitized amphipods and snails (Brown and Pascoe 1989; Guth et al. 1977). However, these effects are likely to vary among populations, communities, and ecosystems in different geographical areas. For example, the effects of metals were greater on macroinvertebrates from small, high altitude streams compared with those from large, low altitude streams, probably due to differences in abiotic factors (such as water hardness, alkalinity, or water temperature) and in the abundance of sensitive species between localities (Kiffney and Clements 1996a). Knowledge of the factors responsible for this variation (such as the strong, negative relationship between body size and response to metals; Kiffney and Clements 1996b) can be used to adjust hazard/risk evaluations, local environmental quality criteria and the choice of potential remediation measures.

The release of metals to the aquatic environment may also affect a number of ecosystem functions, chiefly primary productivity, nutrient cycling, energy flow, and decomposition (Kiffney and Clements 2003; Vuori 1995). First, metals can affect primary production in aquatic ecosystems (McKnight 1981; Crossey and La Point 1988; but see Kettle and DeNoyelles 1986; Schindler 1987), which might have implications for secondary production. Second, certain metals such as iron can alter phosphorus dynamics in freshwater ecosystems, since iron hydroxides and oxides vigorously absorb and/or precipitate ferric phosphate (Vuori 1995). Thirdly, changes in the composition and abundance of invertebrates may result in reductions in leaf litter breakdown rates and thus in the rate of export of coarse organic material (Kiffney and Clements 2003). This effect is exacerbated by the fact that direct effects of metals on the abundance of detritivorous invertebrates are enhanced by the decreased feeding rates of those individuals that remain (e.g. Fe-induced decreases in food consumption by *Leptophlebia*, mayfly nymphs, and *Gammarus pulex* – Maltby and Crane 1994; Vuori 1995).

1.6 Social and Cultural Heritage Issues

Mining is one of the oldest and most important activities of humankind. Surface scratchings for useful minerals were no doubt made by the earliest hominids, and a number of extant sites in Europe attest to the fact that organised, underground mining for flint was being undertaken at several locations (e.g. Grimes Graves in Norfolk, UK and Spiennes in Belgium) during the Neolithic period, between about 3500 and 2000 BC (Holgate 1991). Subsequently, in the Bronze Age, between about 2000 and 600 BC, a major expansion of mining for copper and tin (i.e. the two components of bronze) took place at numerous locations in Europe, with international commerce becoming established on a relatively large scale to bring these two commodities together for refinement and casting (O'Brien 1996). It is still possible today to visit Bronze Age copper mines in North Wales (UK) and Schwaz (Austria) and a Bronze Age salt mine (Hallstatt-Dachstein Salzkammergut) in Austria. From these early but still impressive beginnings, mining grew to become one of the most important economic activities, providing the foundations for entire societies and yielding riches that still circulate in the world's financial markets many centuries after they were first won from the ground (e.g. Shepherd 1993; Lynch 2002).

While the earliest mines worked relatively inert geological materials, by the Bronze Age mining was already beginning to disturb sulphide deposits that were capable of giving rise to polluted mine drainage. Evidence from sediment cores obtained from the Odiel-Tinto estuary in south western Spain clearly indicate the advent of water pollution due to mining activities during the Middle Bronze Age (Ruiz et al. 1998). Episodes of increased sedimentation attest to the clearing of native woodlands to provide timber for smelting and other purposes associated with mineral production. By the Classical period, between about 600 BC and 300 AD, smelting of ores in the vicinity of the Odiel and Tinto rivers was conducted on such a large scale (first by the Phoenicians, then later by the Romans) that associated atmospheric pollution has left its trace in ice cores obtained from the Greenland ice sheet (Rosman et al. 1997). Furthermore, cores recovered from a peat bog at Morvan (France) yielded lead anomalies of anthropogenic origin, which together with dating information provide clear evidence for substantial Bronze age mining and Iron age smelting records (Monna et al. 2003). Extensive mine drainage adits were subsequently driven by the Romans in various parts of Europe, commencing a process of irreversible alteration of natural groundwater flows that has continued to the present-day. Documentary evidence of water pollution associated with *mineral processing* operations is provided in the classic book *De Re Metallica* (Agricola 1555), which documents the first golden age of metalliferous mining in Germany during the 16th Century; both the accomplishments of this German golden age, and no doubt the knowledge about problems of water pollution, were exported to most other European countries within the following decades. One of the earliest accounts of polluted drainage from a *mine void* (as opposed to the mineral washery effluents mentioned by Agricola) are listed in a legal deposition filed in northern England in 1620, which complained of the "unwholesome, cankered and infectious" water flowing from the world's first industrial scale underground coal mines (Younger 2004), which had been developed in the area over the preceding few years.

The history of mining has thus long had associations with both economic prosperity and environmental degradation. Indeed, in countries with long mining histories, the legacies of ancient environmental changes wrought by mining are often very significant (even if they are frequently considered "natural" by many present-day local residents). For instance, where adits were the principal means of dewatering, drawdowns of water levels are usually permanent, so that many individual springs will never flow again, no matter how much time elapses after mine closure.

The nature of the various impacts of mining activities on water availability and water quality has already been discussed. In terms of the social and cultural repercussions of these impacts, the following aspects are especially notable (from a cost-benefit analyses of options for mine water management, Younger and Harbourne 1995):

- Increased water treatment costs where water sources are affected by increases in Mn, Fe, SO_4 and (to a lesser extent) other solutes derived from insufficiently treated mine water discharges into streams or aquifers.
- The additional costs associated with "avoidance measures" such as relocating a water intake to avoid the polluted reach of a river, or arranging for increased reservoir discharges of clean water to

dilute the polluted river water to acceptable standards.

- Where dewatering (temporary or permanent) affects the availability of groundwater, the costs of deepening wells, lowering pumps and increasing pumping heads, or of relocating wells to unaffected areas.
- The loss of fisheries stocks in rivers affected by mine waters, the value of which may be of great economic importance locally.
- The harm done to tourism and recreational activities (and thus to the incomes associated with these) by unsightly ochre staining of watercourses that formerly might have formed the centrepiece of a beautiful landscape, or a particular viewpoint for a historic monument, etc.
- The harm done to both religious sensibilities and the economic activities associated with pilgrimages etc. where mining and/or dewatering disturb the local hydrology such that they cause the drying up of springs that formed the foci for religious shrines. This happened, for instance, in the case of the Shrine of the Blessed Virgin Mary at Holywell, North Wales, where under-drainage of the local limestone aquifers by the driving of the Milwr drainage adit led directly to a cessation of flow from the eponymous holy well in the early 1930s. This calamity for the shrine led to a dramatic decline in pilgrimages, damaging the local economy.
- Problems of ground subsidence can be exacerbated by the erosion of old workings by inflowing mine waters, often giving rise to localised surface collapses in districts where mine subsidence had been considered a thing of the past.

Rebounding mine waters can give rise to a suite of socio-cultural problems (many of which are similar to those associated with rising groundwater levels in urban areas worldwide), such as:

- flooding of basements and tunnels constructed after dewatering had already drawn the water table down to greater depths,
- attack of sulphate rich mine waters on concrete structures formed from ordinary portland cements,
- flooding of low-lying areas of ground (damaging agricultural land, for instance),
- enhanced release of hazardous mine gases (see Section 1.1), driven ahead of the rising water table. These gases can accumulate in confined spaces near the surface, where they may give rise to risks of explosions (in the case of methane) or asphyxiation (in the case of CO_2-rich gases).

Looking on the positive side, mining-induced fracturing has been known to beneficially increase the yields of some water wells in overlying aquifers (e.g. Booth 2002). Furthermore, provided a mine water discharge is not causing major environmental problems, the presence of conspicuously coloured water flowing from an old mine provides a vivid reminder of the former mining activity in villages that may have lost all other evidence of the activities of our forebears. Indeed, some abandoned mine discharges have been developed as mineral water spas, to the distinct economic benefit of local residents. Plans are even under development in northern England to use abandoned mine waters as the feedstock for brewing ale, on the grounds that the chemistry of the mine water is ideal for this purpose.

It is increasingly appreciated that the heritage conservation issues can be an important constraint on proposed mining developments, especially in Europe where a very rich history of ancient mining has left many old mining features that are now protected monuments. Concerns are currently being raised, for instance, in relation to a planned extension of the Roşia Montană gold mine in Romania, which (if approved) would make this the largest open pit mine in Europe. In addition to heritage issues (the destruction of Roman mining remains), relocation of settled communities is also an important issue in that case. Increasingly, it is coming to pass that plans for the remediation of polluted mine waters are also subject to the same constraints: there may be significant, valid local objections to the construction of mine water treatment plants if this entails any disruption to local activities or quality of life. Furthermore, if the ideal treatment location for mine water compromises the conservation of an officially designated ancient monument, then it may prove difficult to reach a consensus on how to strike the most appropriate balance between environmental protection and heritage conservation. Issues of this nature have already arisen, for example, in the Rio Tinto district of Spain, in the North Pennine Orefield of England, and in the Avoca mining district of Ireland. Resolution of these types of conflicts demands painstaking work with representatives of all relevant stakeholders.

Finally, it is eminently possible for unanticipated, “natural” changes in the hydrology of mining districts to give rise to deleterious effects on historic monuments. For instance, the National Coal Mining Museum for England uses underground galleries of the former Caphouse Colliery in Yorkshire. These galleries were selected for the development of the museum on the grounds that they lie above the shallowest levels to which mine water was anticipated to rise. However, several years after the Museum was opened, mine water levels in its vicinity began to rise, presumably due to collapse of a major drainage route via a flooded roadway (at substantial depth and no longer accessible) that formerly took the water away to a pumping station some kilometres to the southeast. In order to safeguard this important national museum from loss through inundation, a pump-and-treat facility was installed at a shaft close to the main museum site. This intervention has now added a further dimension to the museum, in that the mine water management facilities are now being developed as a further visitor attraction and educational facility operated by the museum.

1.7 Managing the Impacts of Mining on the Water Environment: the Catchment Perspective

The management of mining impacts on the water environment in a particular catchment entails two general types of task:

- the formulation of integrated water management plans and action programmes, which in Europe must now be in line with the new European Union Water Framework Directive (WFD), bearing in mind that these plans and programmes may well include more than one surface water catchments, and associated groundwaters,
- the evaluation of environmental permit applications in relation to either new mining operations (and associated mine waste management activities) and/or redevelopment or remediation of abandoned mine sites that are adversely affecting water quantity, water quality, and freshwater ecology.

Previous mitigation of the environmental impacts from mining has most commonly focused on source regulation, i.e. on pollutant release processes and their resultant emissions, with remedial measures being mainly considered and applied at the mine sites themselves. The WFD, however, focuses on meeting water quality standards in various water environments, surface water as well as groundwater, on the welfare of aquatic ecosystems, and on the sustainable development and cost-effective management of all water bodies. With such a perspective, measures for mine water pollution abatement may instead (or in addition) be applied downstream of mine sites, for instance close to important compliance boundaries, thus considerably widening the range of different possible measure allocations within a catchment to produce the same or similar water quality/quantity effects in a given

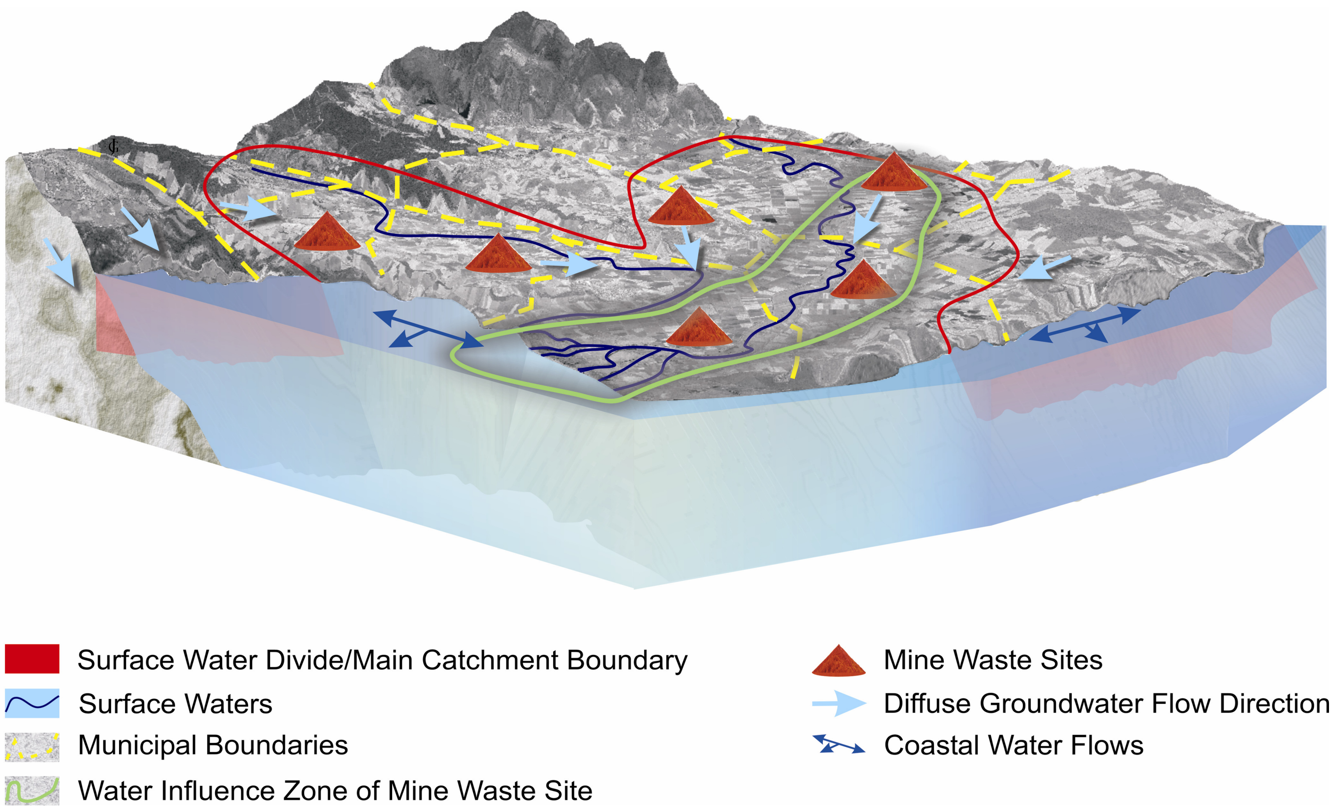

Figure 1. Schematic illustration of a surface water catchment showing that the catchment may cross administrative-political (municipal) boundaries and indicating that ground and coastal water flow paths may cross the boundaries of a surface water catchment. In the figure, mine waste sites are indicated by waste rock piles, with the possible downstream water influence zone of one mine waste site being indicated by the green line; the water influence zone may cross administrative-political boundaries and water catchment boundaries, surface, ground and coastal water movements are all considered

downstream water environment. A practical implication of this perspective shift, from sources to targeted water recipients, is that mine water problems must now be scientifically assessed and quantified, not only with regard to processes and emissions within the sources (mine wastes) themselves, but also in terms of the downstream transport and attenuation of water pollutants along the different water pathways and environments within the long-term water influence zones of these sources (Figure 1). The widened range of possible measure allocation possibilities within a catchment requires a new decision framework for making rational choices on where to actually allocate the resources, i.e. choosing remediation options among different possible mine waste sites that may affect the same stream and/or among different possible locations and measures for mine water pollution abatement further downstream in a catchment.

In the following three chapters of these guidelines, we describe different possible measures for mine water pollution abatement, at and/or downstream of mine waste sites, throughout the life cycle of a mining operation. In Chapter 5, we propose a concrete decision framework for rational choice among such different possible abatement measures. Appendix III provides a complementary description and quantification of specific economic decision rules, which constitute an important part of the proposed general decision framework, and a specific case study of cost-efficient measure allocation for mine water pollution abatement at the catchment scale, using these rules.

2 Minimising Impacts of Mining during the Exploration and Working Phases

2.1 Rational Design of Boreholes

Boreholes used in mining operations can be classified according to their intended purpose, as being for exploration, monitoring, or production boreholes.

Exploration boreholes serve various purposes, including:

- mineral resource identification, characterisation, classification, and delimitation,
- determination of the geological structure which hosts the mineral resource,
- host and waste rock characterisation,
- determination of groundwater conditions, and
- sources of specific data to assist in mine planning and design.

The particular design of a given borehole will be governed by which of the above purposes it is intended to serve, and its required longevity. Borehole design and the density of the borehole network further depend on the complexity of the local geology and hydrogeology and on the value of the mineral resource being investigated (e.g. a greater number of boreholes may be justified for coal than for limestone). Exploration boreholes need only be short life structures in most cases. Even where they are used to test the hydraulic conductivities and pore water pressures of different rocks and layers (e.g. to provide information for pit/dewatering design), unless they are to be retained for longer-term hydrogeological monitoring (see below) they will have a finite life of a few months to a few years. Once exploration boreholes have yielded the data they were designed to obtain, they must be completely backfilled with grout for their full length before they are finally abandoned. This is for reasons of mine safety (not leaving a potential pathway for a catastrophic inrush of water within the area to be mined) and environmental considerations (i.e. avoiding passive interconnections of different aquifers to minimise future inflows of clean water to mine workings where they may become contaminated and to make sure that there are no pathways open for migration of polluted mine waters to shallower aquifers in the post-closure period). Nonetheless, it should be kept in mind that exploration boreholes can be used as monitoring boreholes if this is planned for, saving drilling costs.

Monitoring boreholes – Here we consider only boreholes used for groundwater monitoring, though gas monitoring and stress monitoring boreholes are also commonly used in mining applications. Boreholes drilled for long-term observations of time-dependent groundwater level fluctuations in the water bearing strata surrounding the mined area must be designed to function throughout the planned observation period. Experience shows that in mining environments, the casing and/or well screens of groundwater observation boreholes may become corroded within the planned lifetime of the mining operation. To avoid this, careful initial design is needed, utilising the same design standards as are used in the water industry.

In general there is a tendency in the mining sector to deploy too few groundwater monitoring boreholes (notable exceptions tend to be in the case of surface mines working aquiferous rocks, such as limestones). To improve the ability of the mining sector to handle hydrogeological issues throughout the entire mine life cycle, careful consideration should be given to designing multipurpose boreholes, which both provide exploration data *and* provide facilities for long-term groundwater monitoring. While such boreholes may well be individually more expensive than single-purpose boreholes, the overall cost of the borehole network per unit of data obtained is likely to be substantially less. Correct design and quality assurance are a prerequisite for the development of an effective borehole network, but no matter how well the boreholes are designed, it is always necessary to grout redundant boreholes before final abandonment.

Production boreholes can serve at least four purposes in mining:

- dewatering or depressurisation of the mine workings or of adjacent rocks
- production of minerals extracted from the highly mineralised groundwater
- metals or minerals production by *in situ* leaching of the ore body
- *in situ* coal gasification.

As far as the design is considered, the least demanding of the above are the normal groundwater pumping wells. Yet, even here, well construction should:

- allow for rock column contraction and land subsidence due to the effects of aquifer depressurisation,
- allow for the progressive removal of well casing by standard rock grinding machinery (in the case of open pits with fibrocement casing),
- allow for well connection to mine galleries (in the case of dewatering through underground galleries),
- allow for mine subsidence, without impairing their dewatering function; in the case of wells draining into an underground mine, the upper part should be grouted prior to mine subsidence,
- be carried out so as to avoid or minimise corrosion of well screens and casings during the well's planned lifetime, and,
- should allow for the complete back-sealing of the borehole with grout at the end of the well's planned lifetime, for the same mine safety and environmental protection reasons outlined previously.

Boreholes or wells related to the extraction of minerals from highly mineralised groundwaters are essentially a historic rarity. They were, for instance, used to produce boron salts from the Lardarello water and steam generating geothermal field in Tuscany, Italy. In some cases, these boreholes may be considered not to fall within the ambit of conventional mining. Nevertheless, the related well design and backfilling requirements are much stricter than that associated with conventional mining due to the extreme water temperatures and enhanced corrosion potential.

Design of boreholes or wells linked to the extraction of metals or minerals by *in situ* leaching is very demanding, since they must sustain corrosive fluids and withstand eventual rock subsidence. Two basically different cases must be mentioned in this context: salt leaching and metal leaching. In the case of salt leaching, uncontrolled leaching has to be absolutely avoided to maintain aproper well and well field design. Cases of uncontrolled salt leaching due to improper mine, well and well field design have led not only to the collapse of mine structures and wells, but also to important incidences of surface collapse and damage (e.g. the case of the city of Tuzla, Bosnia). In the case of metal leaching, no records of surface collapse have so far been recorded. Nevertheless, uncontrolled leaching may lead to local and regional aquifer damage and may also potentially impair surface water resources (e.g. in the case of uranium mining in the former Eastern Germany and Czechoslovakia, Merkel et al. 2002). For environmental reasons, these wells also must be properly back-grouted after completion of their planned use.

In situ coal gasification at present remains a technology of tomorrow, so as yet little can be said of production boreholes associated with this technology. Due to high temperatures and corrosion, they are technologically analogous to hydrocarbon or geothermal production wells, due to roof caving and subsequent subsidence, they will have to be back-grouted for environmental reasons, just as other production boreholes are today.

2.2 Rational Design of Mine Access Structures

By their very nature, mine access structures serve one or more of the following purposes: personnel access, in-mine and out-mine haulage of materials, water drainage, and/or (in the case of underground mines only) ventilation. Access features are vital for the functioning of an underground mine and are therefore well maintained.

Mine access structures vary according to the type of mining technology employed. In surface mining or quarrying, mine access features are just simple surface roads or tracks (if we exclude relatively rare cases with ports). In underground mining, the nature, variety or complexity of access features (shafts, inclines, and adits) reflect local topography and the relative depth of the resources to be mined. Shafts are vertical structures whereas inclines are disposed at some angle between the vertical and horizontal, as their name suggests. Adits are essentially horizontal structures. When serving as mine access features, shafts, inclines, and adits must all be well-built and well-supported structures, effectively designed for permanence. Generally, adits are less likely to constitute permanent mine structures than shafts.

Minimisation of water ingress to mine workings via shafts and adits requires thorough sealing of these structures during sinking, wherever they pass through aquifer horizons. The installation of tight seals to hold back hydraulic heads of many hundreds of metres has a long pedigree in mining (Younger 2004, Figure 6). At the present time, there are two main approaches to successfully sinking and sealing a shaft or inclined adit through an aquifer horizon:

- ground freezing, in which coolants are circulated in closed loops using drilled boreholes surrounding the column of ground through which the sinking will proceed. When the native groundwater freezes, it can be excavated like any other rock, and an impermeable lining can be installed, before thawing is finally allowed.
- pressure grouting, in which a radiating cluster of boreholes is drilled ahead of the sinking shaft, and grout injected under a pre-specified pressure (in excess of hydrostatic pressure) to render all water bearing fractures around the shaft area effectively impermeable.

Given that mine access features can represent long-term flow pathways for mine waters after closure, there exists an obvious case for treating them similarly to exploration boreholes after they have finished their productive life, i.e. to backfill and grout them. This approach has been standard practice in the mining industry for many decades. However, viewed from the perspective of *catchment management*, there are a number of compelling reasons for considering alternative treatment of redundant mine shafts, as will be discussed in Section 3.2.4 below.

2.3 Mining Techniques Designed to Minimise Impacts on the Water Environment.

2.3.1 General Principles

When considering working methods that are consistent with minimisation of water ingress and disturbance, it is important to distinguish between surface and underground mine applications. Although the principles used are similar in both cases, the details of the techniques differ markedly. Furthermore, it is also important to define the time scale over which the intended preventative measures are expected to be effective. Obviously, measures and actions that are appropriate for extractive operations may not have significant benefits for the post-closure phase of the mine life cycle. One example would be the design of rapid retreat longwall faces, which move so rapidly that induced feeders of water enter the mine in the worked-out area of a mine panel (i.e. into the goaf) rather than being encountered at the working face. While this is convenient for face workers, the water does still add to the overall water make of the mine.

A number of approaches to achieve minimisation of water ingress can be applied during the various stages of planning, development, operation, and closure of a mine. These approaches may be classified conceptually as:

- defensive (or "evasive") mine design, i.e. a mine design that reduces the possibilities of water ingress (e.g. design of caving operations so that the zone of net extension above the working does not impinge on an overlying aquifer),
- passive protection, i.e. the use of unmined pillars of rock (which may include potentially payable mineral) as *in situ* barriers ("protective pillars") to groundwater inflow,
- active protection; i.e. depressurisation of the water-bearing strata surrounding the mine, to minimise the head-gradient towards the mine workings,
- combined passive and active protection; i.e. the combined use of water inflow protective pillars and depressurisation of the water bearing strata beyond the pillars
- compartmentalisation of mine structures/areas; i.e. combined use of hydrogeologic structure and (in the case of underground mines) of artificial structures (bulkheads etc.) to compartmentalise the mine with respect to water ingress safety, and the zoned application of other (passive and active) protective measures to further enhance this safety.

It should be noted that these approaches were originally developed from the twin perspectives of safety of mine personnel-, and minimisation of the costs associated with removal of water from a mine. The quantities of water entering mine workings that may be deemed insignificant (or at least tolerable) from these two perspectives may actually be very significant in terms of catchment water management. Hence, in particularly sensitive situations, even greater precautions might be warranted than are described in detail below. Of course, increasing the stringency of such measures may well affect the viability of potential mine developments. Where this is so, the full economic value of the proposed mining operation must be assessed within a holistic framework, including the non-use value of the mineral left *in situ* (e.g. its value as a support system for water resource systems) as well as the more traditional valuation of mineral worth, in terms of the market price of the mineral in relation to its production cost. To date, the sophistication of socioeconomic analysis methods for such problems is not very high. Questions arise such as: would it be appropriate, in the interests of equity, for a mineral rights owner to be reimbursed for leaving their potentially mineable reserves in place to meet socially desirable objectives, such as sustenance of present water resource systems? Before this and similar questions can be answered, substantial innovation in political theory and practice will no doubt be needed.

2.3.2 Specific Measures for Underground Mines

Measures and actions related to minimisation of water ingress in the case of underground mining can be classified according to their intended objectives:

- Prevention of roof collapse to minimise fracturing of overlying strata, which might lead to the development of groundwater inflow pathways. This is generally achieved by proper sizing of pillars, by the careful design and implementation of roof support (including roof grouting and creation of additional support pillars using cemented backfill). With adequate design, experience shows that it is possible to work voids as close as 45m to an overlying water body without inducing substantial inflows (albeit in practice a minimum approach of 60m would be recommended in most jurisdictions).
- Design of caving operations so that the patterns of overlying stratal disturbance do not give rise to continuous hydraulic connections from some overlying aquifer to the mine workings. Extensive experiences of undersea and sub-aquifer longwall coal mining in several countries have yielded quantitative models for the design of caving operations. One of the most widely used models of this kind is that developed by the former National Coal Board in the UK, which stipulated the prior simulation of proposed longwall faces to calculate the maximum tensile strain (MTS) that their collapse would induce on the base of an overlying aquifer (or the sea bed, where no aquifer is present). It was deemed that where the MTS at the base of the overlying water body remained less than $10 mm \cdot m^{-1}$, major inflows from that water body were unlikely to be induced. Given the general similarities in lithology between major UK coalfields, routine application of this approach soon led to the conclusion that it was possible to define a minimum depth of cover (105m) from the base of a water body to a zone of longwall workings, which should ensure protection against excessive water ingress (for further discussion, see Younger et al. 2002). It should be noted that more conservative criteria (i.e. MTS $< 10 mm \cdot m^{-1}$; minimum cover > 105m) might well be appropriate where the object is to avoid disturbing natural hydrological systems, as opposed to ensuring adequate working conditions within underground mines (given that "adequate working conditions" often include tolerance of non-negligible quantities of water).
- Localised prevention of water inflow into workings, by enhancement of roof and floor supports and/or pressure grouting of the surrounding rock mass, and in particular any identifiable fracture planes that yield water to the mine.
- Impoundment of mine water in redundant workings to prevent inflow to the active workings. This is achieved by means of dams of various types (see Section 3.2.2ff), supplemented by hydraulic doors and bulkheads, usually accompanied by localised grouting of the surrounding rock mass.

2.3.3 Specific Measures for Surface Mines

Measures applicable to surface mines are in general less elaborate than those for underground mines, simply because the open-air nature of the operation greatly reduces the risks of entrapment and drowning of mine workers in the event of sudden inrushes of large volumes of water. Indeed, apart from fulfilling the requirement for a reasonably dry working environment, the principal motivation in traditional approaches to managing water ingress to surface mines has been to depressurise the pit walls in order to improve their geotechnical stability (see Younger et al. 2002 for further discussion). However, minimisation of water ingress is often not very high on the list of priorities at surface mining operations since depressurisation can often be achieved by deliberately *enhancing* the inflow of water into the open pit, for instance by gravity drainage using galleries or horizontal boreholes drilled below the highwall (e.g. McKelvey et al. 2002). The water can then be removed using sump pumps.

Nevertheless, a number of approaches to minimisation of water ingress have been developed for surface mining applications, including:

- Defensive mine design, which would typically include the design of pit walls and the staging of cut-and-fill operations, so that the area of the active open pit is kept to a minimum at all times, clearly it is essential to avoid the open cuts intersecting any known groundwater flow pathways.
- Passive protection, which is achieved by leaving barriers of unmined country rock (or even mineral resource) between the mined zone and known aquifers. The dimensions of appropriate passive protection barriers must be designed on a case-by-case basis, but are seldom likely to be less than around 50m wide, and may well reach 100m or more.
- Active protection by means of strategic depressurization of the surrounding water bearing strata, either using drainage galleries or dewatering using boreholes (see Section 2.4).
- Combined passive and active protection, i.e. the couples use of depressurisation techniques *and* passive protection barriers (perhaps locally augmented with injected grout curtains).

Where surface mining operations involve continuous backfilling with broken overburden (as in opencast coal mining), certain additional measures for minimisation of long-term water ingress may be appropriate:

- Minimisation of water inflow into the backfilled rock mass by limiting direct infiltration to the backfill by the use of encapsulation techniques (see Section 3.2), reducing lateral groundwater inflows (e.g. lining the final flanks of the loose wall with clay, etc.), and eliminating water inflows from nearby streams. The latter has been routinely undertaken in the past by two approaches: either diversion of watercourses (see also Section 3.2) or lining or grouting of streambeds. Given present-day perceptions of the importance of hyporheic ecosystems, these kinds of solutions will require much more cautious design and implementation if they are to retain a place in the armoury of management options in the future.
- Minimisation of backfill permeability, by designing tipping to avoid accidental sorting of clasts (which tends to result in the development of preferential flowpaths within spoil) and/or by ensuring an adequate mixing-in of fine-grained materials in order to reduce the mean sizes of pores in the backfill.
- Precautions against future water quality deterioration, which can involve overburden rock segregation through selective mining and selective re-disposal, and the deliberate introduction of buffering reagents (such as fly ash-based grouts in acid water environments). With regard to the latter, it is essential not to confuse this activity, which amounts to conditioning of mine wastes, with the superficially similar practice of co-disposal of two separate waste streams, which would be problematic in the light of the European Union Waste Framework Directive.

2.3.4 Specific Measures for Remote, in situ Methods of Mining

With the exception of salt extraction by means of brine wells, most other remote *in situ* methods of mining (or *in situ* mineral exploitation) are in their relatively infancy. However, these techniques are almost certainly going to become more widely used as time goes on. Measures aimed at minimisation of water ingress/hydrological disturbance associated with *in situ* leaching operations can be classified according to their objectives, as follows:

- prevention of uncontrolled leaching, which demands both careful design and robust construction of leaching wells, and their rapid and effective back-grouting at the end of their useful lives,
- prevention of roof and pillar collapse in brine well extraction systems (to avoid inducement of inflows from aquifers), which demands careful design of well-fields and modelling of the development of the associated salt leaching chambers,
- prevention of roof and pillar collapse induced by *in situ* coal gasification, which demands careful design and installation of injection/recovery wells, and vigilant development of underground burn zones.

2.4 Mine Dewatering

2.4.1 Dewatering and the Catchment Context

Given the importance of its impacts on the water environment, it is surprising how sparse the literature on mine dewatering is. Although an account of the hydrogeological principles of dewatering has recently been published (Younger et al. 2002), few comprehensive texts on the topic have appeared since the 1950s (Kegel 1950, Sinclair 1958). This relative sparcity of literature might well reflect the fact that dewatering installations in mines are generally the responsibility of electrical engineers, who have little interest in the hydrogeological context that gives rise to the need for pumping in the first place. Of course it is also true that this is a relatively mature field of endeavour, in which major changes in technology are rare, so that there is little new to write about in most cases. However, the catchment context of dewatering has received such little attention to date that it can be considered a relative novelty.

As with any activity that involves direct interference with the natural hydrological conditions in a catchment, the key to successful implementation of dewatering is to consider what impact it will have on the general aim of attaining and sustaining "good ecological status" in the surface water courses of the catchment. The degree of analysis warranted by a given mining development is a matter of scale. Many modest, short-lived dewatering operations will not be of sufficient magnitude to make any lasting impact on the quantity and quality of waters flowing in sensitive surface watercourses. On the other hand, as the scale of the mining operation grows, so usually does the scale of the dewatering operation, and as Figure 2 indicates, there are few large mines in which the mass of water removed from the mine per unit time (the "water make") does not exceed the mass of run-of-mine production. Indeed, a ratio of water production to mineral production in excess of 100:1 is by no means unheard of (Figure 2). With water makes of this order of magnitude, it is highly unlikely that the dewatering operation will *not* be affecting the status of surface watercourses in the catchment, and both careful predevelopment predictions and monitoring/mitigation of effects during the working life of the mine will be warranted.

Drawdown or the water table by dewatering operations and disposal of dewatering effluents can both impact catchment status.

To assess the former, the usual suite of techniques for investigating the hydrochemical and ecological status of receiving watercourses (see Section 1.4 and Appendix I) are required. To assess the latter, the required methodologies are hydrogeological (e.g. Morton and Niekerk 1994). Recommendations in relation to the characterisation and mitigation of these impacts are given in the following sections.

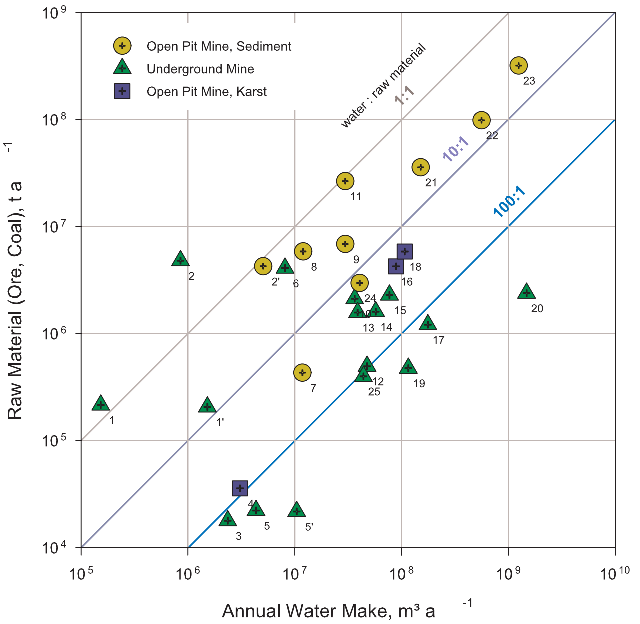

Figure 2. Selected worldwide ratio of water make and raw material production; modified and data added after Fernández-Rubio and Fernández Lorca 1993; 1 and 1' Agnew (Australia, Ni), 2 and 2' Lock (Australia, coal), 3 and 4 Burra (Australia, Cu), 5 and 5' Beaconsfield (Tasmania, Au), 6 German Creek (Australia, coal), 7 Scottish National Coal Board (UK, coal), 8 Bowmans (Australia, lignite), 9 Thorez (Poland, coal), 10 Picher (USA, Pb, Zn), 11 Morwell (Australia, lignite), 12 Fejér Co. (Hungary, bauxite) 13 Chingola (Zambia, Cu), 14 Dorog (Hungary, coal), 15 Tatabánya (Hungary, coal), 16 Pine Point (Canada, Pb, Zn), 17 Konkola (Zambia, Cu), 18 Kingston (Australia, lignite), 19 Bakony (Hungary, bauxite), 20 Kailuna Hebei (China, coal), 21 Atabasca (Canada, tarsand), 22 Rheinbraun (Germany, coal), 23 Lausitz (Germany, coal) 24 China (coal), 25 Vazante (Brazil, Zn)

2.4.2 *Discharge of Dewatering Effluents*

As was noted in Section 1.4, by no means are all mine waters a source of pollution. Indeed, there are numerous mine waters that can be used for purposes of irrigation or municipal supply (e.g. Banks et al. 1997). Depending on the geological setting, hydrogeological conditions and the nature of the extractive process, it may well be possible to discharge mine waters to the river environment following only minimal settlement to remove suspended solids. For instance in the lignite mining areas of western Germany, several million cubic metres of pumped groundwater is suitable for discharge to surface waters without any further treatment (Regionaler Planungsverband Westsachsen 1998; Landesumweltamt Nordrhein-Westfalen 2002). Nevertheless, it is always prudent to investigate issues of compatibility between high-quality dewatering effluents and receiving watercourses to ensure that:

- the quantity of water discharged to the river does not change the flooding regime,
- the increased flow in the channel due to the additional flow does not give rise to velocities that are sufficient to cause erosion of previously stable beds and banks (which has implications both for property protection and in-stream ecology),
- the mixing of the dewatering effluents with the existing water in the river does not alter the geochemical balance of the watercourse in such a manner that dissolution or desorption of previously immobilised contaminants occurs.

Examination of these three factors constitutes the minimum "table of contents" for a rational mine drainage plan.

In hydrogeological settings where water of poor quality is intercepted by the dewatering system, the mine drainage plan must address the design, implementation and monitoring of treatment. In most European Union jurisdictions, this requirement is already required in the mining licence or in industrial effluent control permits. Appendix II provides a summary overview of the most common types of water treatment technologies applicable to mine waters. Up-to-date guidance on the relatively new class of techniques known as "passive treatment" can be found in the recently published engineering guidelines of the European Commission's 5th Framework R&D project PIRAMID (PIRAMID Consortium 2003; available for free downloading from www.piramid.org). After treatment, the drainage water can be discharged into surface streams or lakes. Solid residues arising from the treatment processes may (in order of preference) be processed for use as by-products, used along with coarse discard as backfill for mine voids, disposed *in situ* (at least over the lifetime of a passive treatment installation), or removed from the site for disposal elsewhere.

Although these four alternatives are listed in order of preference, it is important to note that the first of these has only rarely been achieved. One example is the mine water treatment plant at the abandoned Mecsekérc uranium mine (near Pécs, Hungary), where uranium is extracted from the mine water in the form of yellow cake, for later sale to nuclear power generation companies in the European Union (Gombkötö et al. 2002). However, in this case, as in many others, the price received from sale of this by-product does not come close to covering the cost of its production. In the case of Mecsekérc, recovery and sale of uranium is only continued because the political decision has been made that this is preferable to disposal of radioactive sludge in a landfill. At the time of writing, a number of initiatives are under way in the USA and the UK to identify options for recovery of ferric hydroxides from mine water treatment plants, for end-uses including the production of pigments (Hedin 2003) and as a reagent for removing phosphate from sewage and/or agricultural runoff (Heal et al. 2003). Current indications are that these options, if not especially profitable, should at least be able to cover the costs of their implementation, thus turning a waste disposal problem into a recycling operation.

Besides these end-of-pipe approaches to controlling dewatering effluent quality, it is also eminently possible to alter the manner in which dewatering is implemented in order to minimise contamination of waters in the first place. The principle is to intercept water entering the mine workings at the first convenient point, and to pump it out of the workings immediately instead of allowing it to flow all the way down to the lowest sump in the mine, by which point it will almost always have deteriorated in quality. In some surface mining operations, it has proved possible to achieve complete dewatering of the mined void by pumping well networks outside of the working void (i.e. *external dewatering*; see Table 1), so that groundwater never enters the void at all. This will usually be more difficult to achieve in the case of deep underground mining operations. However, depending on the geometry of the mined system and the distribution of pyritic strata within it, it may in some cases be possible to improve water quality prior to discharge by deliberately *increasing* the distance over which water must flow before reaching the dewatering pumps. This is done, for instance, by Deutsche Steinkohle at Bergwerk Ost/Haus Aden Shaft 2 water pumping station in the Ruhr coalfield of Germany, where ferruginous water is diverted through 7km of ventilated roadways that are no longer being used for any other purpose; enough iron precipitates from the water by oxidation and hydrolysis (accumulating as ochre on the roadway floors) that the final pumped effluent can be discharged to the River Emscher without further treatment.

While most of the problems associated with disposal of dewatering effluents relate to water quality issues, there is also an important category of problems related to water *quantity*. For instance, where the drawdowns caused by a mine dewatering operation lead to local drying up of springs or stretches of riverbed, for the decision may be made to strategically locate the point of discharge for the pumped dewatering effluents to provide “compensation flows” to the affected surface water systems. Of course if the discharged effluents tend to re-infiltrate through the streambed, a certain degree of recirculation of water will be established. This can be avoided to some extent by prior lining of the channel bed, albeit this needs to be handled sensitively in order to avoid long-term impacts on the ecology and post-mining hydrology of the watercourse. In some cases, it has proved possible to avoid negative interactions between rivers and mine dewatering systems by relocating the sensitive reaches of the river. Such large scale measures are only likely to be justified for the richest of mineral resources. Recent cases of this type have shown the feasibility of nurturing the new reach of the river using part of the flow from the existing channel so that colonisation by invertebrates and fish is complete before the remainder of the flow is diverted into the new channel. Such operations typically take several years to accomplish.

2.4.3 Drawdown Management

There are a number of feasible strategies for attaining adequate dewatering of a working mine, all of which result in drawdown of the water table within and beyond the working mine void. The various approaches can be classified as shown in Table 1. It is important to note that in many mining operations it will be advisable or even essential to use one or more of these approaches in tandem.

Although this section is concerned primarily with dewatering management during the working phase of a mine, it is important to note here that one of the three common dewatering approaches, namely adit dewatering, is essentially irreversible, resulting in permanent drawdown of the water table to a new base level of drainage. This ensures that mined voids above the adit level will remain largely unsaturated forever after (with possible negative implications for water quality, especially in sulphide-rich strata). Hence adit dewatering should be used only where this will not be detrimental to achieving and retaining good status in the rivers of the catchment(s) affected.

Because mine dewatering causes drawdown of the natural water table within and beyond the mine site, it is often suspected of giving rise to widespread impacts on water resources, ground stability and ecology (see Section 1.2). Nevertheless, there are innumerable cases in which drawdown due to mine dewatering has had no such negative impacts. However, in order to be able to demonstrate this in any particular case, extensive monitoring of groundwater levels in the area surrounding the proposed area to be dewatered is necessary. As previously argued (Section 2.1), there are distinct advantages to ensuring that mineral exploration drill holes are designed so that they can be retained for use as groundwater monitoring wells, so that the evolution of the water table before, during, and after dewatering operations can be accurately recorded (Plotnikov et al. 1989).

Table 2 provides guidance on the optimum spacing of groundwater monitoring boreholes around an area of proposed dewatering. The guidance is based on the well-known premise that drawdown usually declines with distance from the centre of pumping in an exponential manner, so that linear radial spacing of monitoring wells is much less likely to yield useful data than logarithmic spacing.

Nonetheless, the exact drawdown curve depends on factors such as hydraulic permeability of the host rocks, water make and the type of flow involved: turbulent, non turbulent. To understand Table 2, it is important to note that it envisages one or more lines of groundwater monitoring wells installed along lines radiating away from the edge of the mined area. The spacing of these wells along each of the lines is specified in Table 2 in terms of the distance from the edge of the workings (e.g. from the highwall in a surface mine) to the innermost well, the intermediate wells and the outermost well. In some cases few intermediate wells will be needed to develop an accurate profile for the drawdown caused by dewatering, whereas in other settings several intermediate wells will be needed. Clearly these recommendations will require customisation in every specific case, to take account of the local vagaries of geology (e.g. outcrop patterns and locations of faults, especially where these act as preferential flow pathways for groundwater), land access and other

Table 1. The three principal methods of mine dewatering (after Younger et al. 2002)

Method of dewatering	Outline of method	Range of conditions where this approach can be recommended
Sump dewatering	Roads and ditches within the mine are routed and graded to deliver all water to one or more centralised sumps, whence it is pumped from the mine.	• if the mine is above the water table, or • if the mine is below the water table, but the strata are of low permeability, or • the mine is an open pit operation in chemically inert rocks, or • the mine is small and/or isolated.
Adit dewatering	A drainage adit with a gradient of > 1:500 is driven from a portal in a valley beneath the area to be mined, and all mine drainage is routed to the adit via roadways, shafts, pipe-work etc.	• if the maximum depth of working is above the minimum geographical elevation to which an adit can economically be constructed, and • given that adit dewatering results in permanent drawdown of the water table in the mining area, it should only be used following careful evaluation of likely long-term implications.
External dewatering	Boreholes and/or shafts in the aquifers or old workings outside the mine site are used to pump water, either to prevent water from entering the mine by gravity flow, or to lower the water table below the areas of active mining.	• if the mine is an underground mine is surrounded by highly permeable aquifers and/or large volumes of flooded old workings, or • if the mine works pyritic or otherwise highly reactive strata that may cause a deterioration in water quality if water is allowed to enter the workings, or • the mine is very deep, but external dewatering wells can intercept the water make at much shallower depth.

factors. However, they provide a useful starting point for determining the scale of monitoring likely to be required for successful characterisation of the effects of a dewatering operation of a given magnitude.

One issue that frequently hinders earnest attempts to implement adequate monitoring of dewatering impacts is the mismatch between the demands of the authorities for distributed groundwater monitoring and the lack of any rights on the part of the mine operator to install monitoring wells in appropriate locations. It is recommended that the powers of public bodies be mobilised to ensure that an adequate numbers of monitoring wells be installed in logical locations, albeit with the costs being borne by the mine operator.

One of the principal reasons why drawdown of the water table can give rise to negative environmental impacts is that most groundwater flow systems give rise to surface discharges of water (via springs, and as bed-gain entering streams etc.) which can become depleted, or even caused to dry up altogether, if drawdowns reach them. For this reason, a local evaluation of the groundwater system should always be made to identify the major points of groundwater outflow, so that (where these are important sources of water for supply or ecological purposes) they may be monitored also to ensure that they are not suffering from negative impacts.

Monitoring of groundwater levels and the hydrological behaviour of related surface water features provides powerful evidence for the scale of impacts once they begin to occur. However, prior to the development of a mine dewatering system, it will usually be necessary to make predictions of both the quantities of groundwater that will need to be pumped *and* the likely drawdown impacts of pumping these quantities of water. This sort of investigation is usually required by regulatory authorities. In the case of surface mines in areas prone to intense rainfall, dewatering requirements for storm conditions will also need to be analysed. Numerical techniques appropriate for modelling drawdown in response to pumping are now widely and routinely applied in water resources investigations (e.g. assessing the likely impacts of a proposed well-field). Such simulation methods are increasingly being applied to the prior analysis (and post-hoc interpretation) of the impacts of mine dewatering. Application of numerical models for such purposes is a skilled job, demanding inputs from suitably qualified and experienced specialists.

When considering predictive modelling of dewatering impacts, it is important to bear in mind that fracturing induced by mining often causes very high permeability in the rock mass immediately surrounding the mine void, to the extent that groundwater flow close to the pit walls can be turbulent (in contrast to the laminar flow conditions typical of most groundwater flow systems). This phenomenon has a number of important consequences (e.g. Dudgeon 1985). Because the ratio between flow rate and head-loss is much steeper under turbulent flow conditions than it is where flow is only laminar, the profile of the water table immediately adjacent to

Table 2. Recommended frequencies and locations of groundwater monitoring wells around proposed mine dewatering operations; [1]: from outer boundary of mined area, [2]: includes truly karstified limestones and evaporites, plus volcanic aquifers containing lava tubes and networks of old underground mine workings (in strata of all types) which function similarly to karst flow systems

Hydrogeological settings of mining operations	Sand and gravel deposits			Karstic and quasi-karstic terrains[2]			Nonkarstified, indurated sedimentary rock sequences			Plutonic and metamorphic rocks		
Anticipated rate of dewatering [10^3 m^3 d^{-1}]	0.5	1	> 2	0.5	1.5	> 3	0.15	0.5	> 1.5	0.1	0.25	> 0.5
Minimum no. of radial lines of monitoring wells	1	3	3	1	2	3	1	2	2	1	1	2
Distance[1] to innermost monitoring wells [m]	10	15	20	20	30	50	10	5	5	15	15	15
No. of intermediate monitoring wells per line	1	2	2	1	2	3	1	3	4	0	0	0
Distance[1] to intermediate wells [km]	0.2	0.2, 0.7	0.2, 1	0.2	0.3, 3	0.5, 2, 5	0.1	0.05, 0.5, 2	0.05, 0.5, 5, 8	–	–	–
Distance[1] to outermost monitoring wells [km]	0.7	1	2	0.8	6	10	1	5	10	0.15	0.15	0.15

a mined void may be very much steeper than a conventional groundwater modelling code would predict. Failure to account for this factor when developing a model simulation will typically result in substantial over-prediction of the areal extent and magnitude of drawdowns induced by the dewatering operations. This is the root cause of many overpessimistic predictions of mine dewatering impacts, which experience has regularly shown to be far less widespread than one would assume from the application of an off-the-shelf groundwater modelling package that represents only laminar flow conditions.

One of the more esoteric impacts of drawdown is stimulation of land subsidence. Figure 3 allows some appreciation of the possible magnitude of subsidence induced by artificial drawdown of the water table. In extreme cases, total subsidence can reach several metres. There are two principal modes of subsidence potentially associated with mine dewatering:

- Point collapses of ground forming circular "crown holes", a few metres deep and up to 30m or so in diameter; these are typical of karst terrains, and can arise where mine dewatering drains caves, removing the buoyant support of the groundwater from the cave roofs. These collapses are difficult to predict, but are least likely in areas where there are few known *natural* karst features.
- A more general lowering of the ground surface over a wide area, sometimes up to several tens of square kilometres. This type of subsidence occurs only where the local geological sequence contains substantial thickness of non-indurated silts and muds, which tend to compact irreversibly when subject to dewatering (Poland 1984, Holzer 1984). Fortunately, such strata are rare in most mining settings, with the possible exception of some Cenozoic coal bearing sequences which are currently under exploitation in eastern Asia. Where it is clear that subsidence *will* be induced by mine dewatering, it may be necessary to implement engineering measures to protect important surface infrastructure.

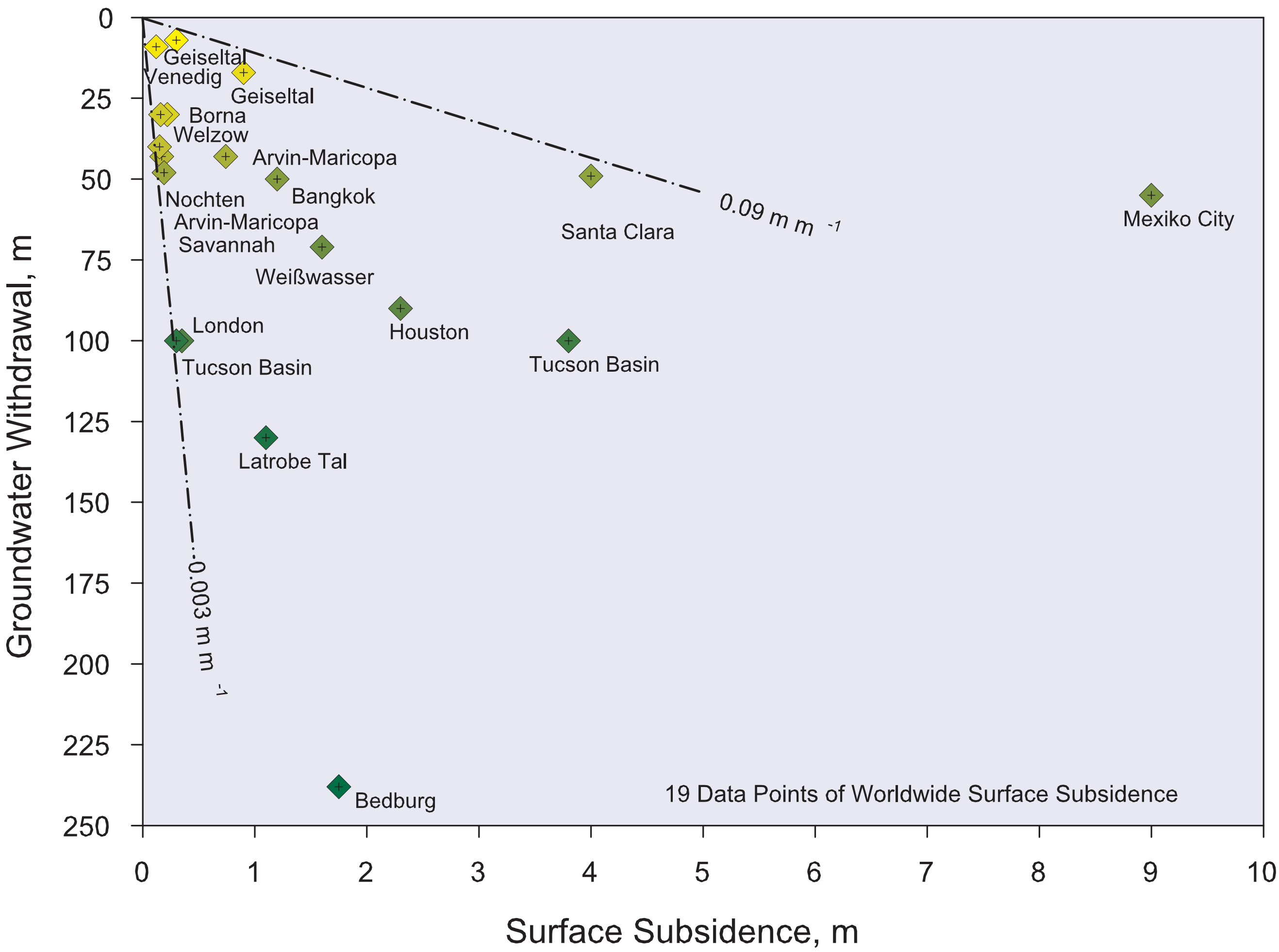

Figure 3. Surface subsidence of sediments due to the dewatering of mines and drinking water supply (modified after Wolkersdorfer and Thiem 1999)

In situations where drawdowns are predicted to spread widely and give rise to negative impacts on wells and springs etc., it may be advisable to reinject part of the pumped water around the perimeter of the dewatered zone to create a local "water table divide" to check the outward expansion of the trough of drawdown. This approach has been used with success on numerous occasions (e.g. Wardrop et al. 2000), with examples mainly relating to surface mines working sands and gravels in the UK and lignite mines in western Germany, close to the border with the Netherlands.

2.5 Mine Waste Management

2.5.1 Nature of Mine Wastes

The term "mine wastes" refers to all earth materials and associated residues of processing reagents, which the operator of a mine and/or mineral processing plant discards (or intends or is required to discard). As such, mine wastes are a very heterogeneous category of materials, albeit they principally comprise naturally occurring assemblages of minerals (Lottermoser 2003). A number of subdivisions of mine wastes are recognised in practice, amongst which the following are most important (Figure 4):

- **Topsoil** – Biologically active, humus-rich layer of material supporting the vegetation cover above a mineral resource. Top soil must be stripped prior to commencement of mineral extraction mining using surface mining techniques.
- **Overburden** – Unconsolidated sediments (other than top soil) or indurated rock overlying the mineral resource (i.e. coal seam or ore body). As with top soil, in the case of surface mining operations, overburden has to be removed prior to extraction of the mineral resource.
- **Waste rock** – Although often identical to overburden, waste rock is typically regarded as non-payable rock intimately associated with the mineral resource, which must be disturbed in order to extract the valuable minerals. Waste rock may be completely barren of economic minerals, or else may contain such minerals at a grade too low to for it to be mined and processed profitably.
- **Tailings** – Residues arising from *ex situ* processing of the run-of-mine ore, comprising crushed and sorted rock fragments from which as much as feasible of the desired minerals have been removed. Tailings consist mainly of gangue minerals, but may also include residual process water, process chemicals and portions of the unrecovered minerals.

Each of these different categories of mine waste merits a particular mode of handling. For instance, top soil is typically stored separately from other overburden, so that it can be reconditioned for biological productivity and replaced after mining

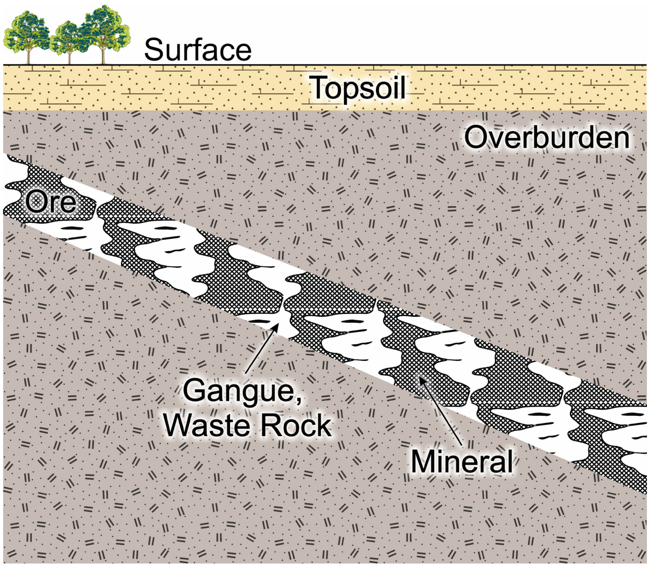

Figure 4. Schematic drawing of an ore body

finishes. Overburden and waste rock may either be continuously backfilled into the void (as in opencast mining operations), stored for later backfilling, or else discarded to form large piles of rock known as "spoil heaps". Because most tailings are separated from crushed ore in operations involving the use of water, they are frequently transported in aqueous suspension and settled out as sediments in constructed impoundments behind "tailings dams". The modes of environmental impact of these various mine waste disposal routes has already been outlined (Section 1.1), and extensive technical discussion is given by Younger et al. (2002) and Lottermoser (2003).

2.5.2 Recent and Emerging Guidance on Mine Waste Management

The best practices for the management of mine wastes has been definitively collated and systematised in the European Commission's "Reference Document on Best Available Techniques for Management of Tailings and Waste-Rock in Mining Activities" (European Commission 2003a). We have no intention of reproducing the detailed recommendations of this comprehensive reference document. Indeed, many of the most important recommendations of this reference document are already enshrined within the current proposal of the European Commission for a Directive on the management of waste from the extractive industries (European Commission 2003b). Before moving on to a brief discussion of some aspects of mine waste management that require particular attention from the particular catchment management perspective of the present discussion, a few comments are offered on the proposed Directive.

Many of the provisions of the proposed Directive echo established best practices in preexisting national-level management regimes. However, as it focuses primarily on improving the management of future mine waste disposal activities (a stance which it shares with the reference document), the proposed Directive largely ignores the problems associated with abandoned mine waste repositories. Given that European mining has been in net decline for almost a century, and few new wholly new mining ventures can be realistically anticipated in future, this is a significant oversight: the finest regulations in the world for future mining developments will not make a substantial impression on the legacy of existing environmental problems associated with old mine waste disposal facilities in many former mining districts of Europe. With few exceptions, the only realistic prospect of cleanup of these old mine waste deposits lies in their adoption and remediation by public authorities. To this end, it is recommended that each European Union Member State characterise their full inventory of problem sites, prioritise them for remediation (i.e. "worst first", wherever feasible) and then commit to a rolling programme of remediation over a decade or two. Only the UK, Portugal, the Czech Republic, and Bulgaria (for U-mines) have launched such initiatives to date. If suitably amended, the proposed Directive could help encourage other European Union Member States to do likewise.

As currently formulated, the proposed Directive rightly emphasises regulating most stringently those mine waste operations that involve reactive waste materials, such as sulphide minerals that tend to decay in the presence of the atmosphere to release ecotoxic metals and acidity to solution. The proposed Directive provides sweeping exemptions from regulations for operations dealing only with "inert wastes". In doing so, the MWD runs the risk of overlooking the hazards posed by fine-grained silts of any mineralogical composition (including geochemically "un-reactive" carbonate and/or silicate minerals). Where silt is released en masse to receiving watercourses, it tends to clog streambeds, cutting off light penetration to benthic algae and thus halting primary production. Vigilance must therefore be maintained at any mine waste operation that handles fine-grained materials, whatever their composition.

While the proposed Directive commends efforts to minimise waste generation in the mining sector, it is not prescriptive in this regard. Equally, the reference document on best practice concerns itself solely with mineral processing, and as such does not offer guidance on how mining activities can be adapted to promote a waste minimisation agenda. Some comments on this topic are therefore offered below.

2.5.3 Waste Minimisation

Minimisation of waste generation is an important tactic in all industrial sectors. In mining operations, waste minimisation may be implemented using the following approaches:

- Techniques that reduce ore dilution during the ore excavation process.
- Cut-and-fill mining techniques involving almost immediate re-tipping of waste rock/overburden as mining proceeds.
- Techniques allowing for the separation (with the aim of selling) excavated overburden/waste rock and/or tailings.

Techniques that reduce ore dilution during the excavation process differ between different types of mining operations. In underground mining of veins or lens-like ore bodies, the greater the precision with which the ore can be mapped and excavated, the less dilution of ore will there be. In the mining of coal seams/seam-like (stratiform) ore bodies, waste minimisation may require more careful planning of caving techniques to minimise the inclusion of barren roof strata in the run-of-mine product; particular care is needed in caving operations in the vicinity of faults, and in mining faces beset by significant inter-seam partings. Waste minimisation in surface mining of disseminated porphyry ore bodies is more challenging, but increases in the precision of planning and implementing magazine mining production methods do offer some scope for waste minimisation in many cases.

Cut-and-fill mining techniques, which allow almost immediate re-tipping of waste rock/overburden as mining proceeds, is only really possible in situations where the contrast between valuable mineral resources and waste rock is readily apparent to the naked eye of the miner. There are relatively few instances in which such situations arise. The most notable example is in coal mining, in which the coal is usually easy to distinguish visually from the non-coal strata (though sometimes even this can prove difficult where very black shales enclose the coal seams). Exploiting this fortunate circumstance, opencast coal mining may disturb very large overburden-to-coal ratios (values in the range 10 to 30 are common), but they produce very little permanent waste, as all non-coal rocks are backfilled into the void as the working bench advances. In other circumstances, the separation of valuable mineral from waste rock takes place *ex situ*, and expensive double-handling of waste rock may be necessary if backfilling is to be employed. The economics of mining operations may not always accommodate backfilling readily at all stages of the productive phase of mining. However, in a growing number of mature underground mining operations in Europe, cemented backfill is being economically introduced to underground workings to create artificial pillars, providing additional roof support that facilitates extraction of valuable mineral resources previously left in place as support pillars.

With a little lateral-thinking and creative marketing, it is occasionally possible to plan mining operations so that overburden and waste rock can be selectively handled so that rock with potential use as building stone or aggregate is stock-piled in a convenient manner, so that it can be sold without further processing. One example relates to the South Tyne Coalfield (Northumberland, UK), where *none* of the many extensive coal mines have generated significant spoil heaps despite over two hundred years of mining; this is because nearly all of the waste rock removed from the mines is limestone, which has been variously used as feedstock for lime kilns and/or as aggregate throughout the productive life of the coalfield. Similar approaches to potentially saleable non-ore fractions of tailings are also occasionally feasible, with tailings of various compositions having been successfully marketed as construction materials, bulking agents in the chemical industry, or as sources of lime (e.g. washout lime from gravel pits).

The above techniques need to be considered within the wider context of efficient mine planning (Section 2.3). Furthermore, a number of the techniques for long-term mine waste management discussed in Section 3.2 below are far more likely to be effective if they are managed interactively with waste minimisation strategies during the extractive phase of the mine life cycle.

2.5.4 Other Catchment Management Considerations in Tailings Disposal

Effective water control is central to the operation of many mine waste management facilities. In particular, water management has important implications for both the safety and environmental impacts of tailings dams. From the catchment management perspective, the following aspects of tailings disposal merit emphasis:

- Paste disposal is increasingly being promoted as an alternative to traditional waterborne tailings disposal in dams. While paste disposal certainly does offer significant advantages in terms of geotechnical stability of the disposal facility, a number of *caveats* need to be borne in mind, including:
  - The limited range of tailings streams physically amenable to disposal in the form of pastes.
  - The long-term behaviour of paste deposits after emplacement, when they may be subject to desiccation cracking, frost damage, root penetration or other processes that counteract the benefits of their inherently low intergranular permeabilities, potentially rendering them prone to acidity-release over time.
- Where conventional tailings disposal is used, it is important to realise that the use of water to transport the tailings to the dam is not merely a matter of operator convenience; subaqueous emplacement of tailings is a very effective means of controlling acidity-release from sulphidic fractions of the tailings (because submerged sulphides will not easily oxidise). It is therefore good practice to encourage subaqueous emplacement of tailings, all other things being equal.
- A number of construction strategies exist for tailings dams. Two main options are ring dykes (i.e. encircling bunds entirely enclosing the sedimentation lagoon) and valley dams (i.e. tailings dams constructed by impounding all or part of a natural valley). Three subtypes of valley dams are recognised:
  - cross-valley dams, in which the entire width of a natural valley is impounded,
  - valley side dams, in which a dam is constructed on one flank of the valley, using the natural ground to form one side of the lagoon,
  - valley bottom dams, which are constructed where a valley is too wide or too flat-lying for a cross-valley dam to be effective. In these circumstances, part of the width of the valley is impounded, and a further dyke parallel to the valley side is constructed to enclose the impoundment.

From the perspective of long-term catchment management, cross-valley dams and valley bottom dams are unlikely to be preferred technologies in the majority of cases, as they generally involve permanent culverting of the natural watercourse previously present in the valley, introducing darkened reaches of streams that are barriers to the development of aquatic flora and fauna. Even where streams can be diverted at the ground surface, very careful design, construction and maintenance will be required to ensure that floods do not give rise to damaging erosion of the deposited tailings. Hence, in general, catchment managers should argue in favour of ring dykes or valley side dams.

Once a tailings management facility is in existence, ongoing inspection and maintenance is the key to satisfactory long-term performance. Central to these activities ought to be the development and constant updating of water balances for tailings dams. Development of a water balance for a given tailings dam (Figure 5) involves quantification and comparison of all inflows and outflows of water. Variables that need to be quantified include precipitation, infiltration, evaporation, seepage, groundwater flow and possible flood events (in the case of valley dams).

Evaluation of seepage and other groundwater flow aspects of the dam water balance can be particularly challenging, as it requires mathematical manipulation of data on piezometry and *in situ* permeability, both of which can be expected to change significantly over time. There are few tailings dams in which the piezometric infrastructure is as extensive as would ideally be needed in order to quantify subsurface water fluxes to any great degree of accuracy.

3 Minimisation of Impacts: Closure Phase

3.1 Introduction

Even the longest-lasting mining venture is still a short-term land use compare to the millennial timescales over which natural changes tend to affect water catchments. However, recent studies of some of the ancient mine workings mentioned in Section 1.6 have demonstrated that post-mining water pollution can indeed persist over millennial timescales. Similarly, where mine dewatering was achieved using under-drainage by adits, the drawdowns achieved during mining can be expected to be maintained indefinitely, at least in the absence of substantial reengineering of the mined system. Given this fundamental mismatch between the decadal timescale of individual mining operations and the millennia over which their impacts on the water environment can persist, it is essential that the cessation of mining be undertaken to minimise negative impacts as much as possible.

Implementation of the various recommendations given in Chapter 2 of this report would go a long way to ensuring that few further interventions would be needed during closure to ensure minimal long-term impacts on catchment flows and water quality. However, many of the largest mining operations in Europe have been in operation for many decades or even centuries, so there is already a considerable number of mines for which adequate closure planning now needs to be retrofitted if compliance with the Water Framework Directive is to be achieved. In this section, two major elements of closure planning are discussed, namely engineering of mine drainage pathways to minimise post-closure environmental damage, and planning for the long-term management of tailings and waste rock. Other elements are also relevant to planning activities during closure, most notably the design of post-closure water treatment facilities; however, as these tend to be implemented only *after* final closure of the mine, discussion of these matters is reserved to Chapter 4 of these guidelines. Furthermore, wider issues of managing multiple potential mine discharges is held over to Chapter 5.

3.2 Engineering of Mine Drainage Pathways

3.2.1 Why Engineer Mine Drainage Pathways at Closure?

The final closure of a mine usually also involves a cessation of dewatering, and as such provokes substantial changes in local hydrological pathways. Where these changes are all likely to be for the better, there may be no need for any engineering works

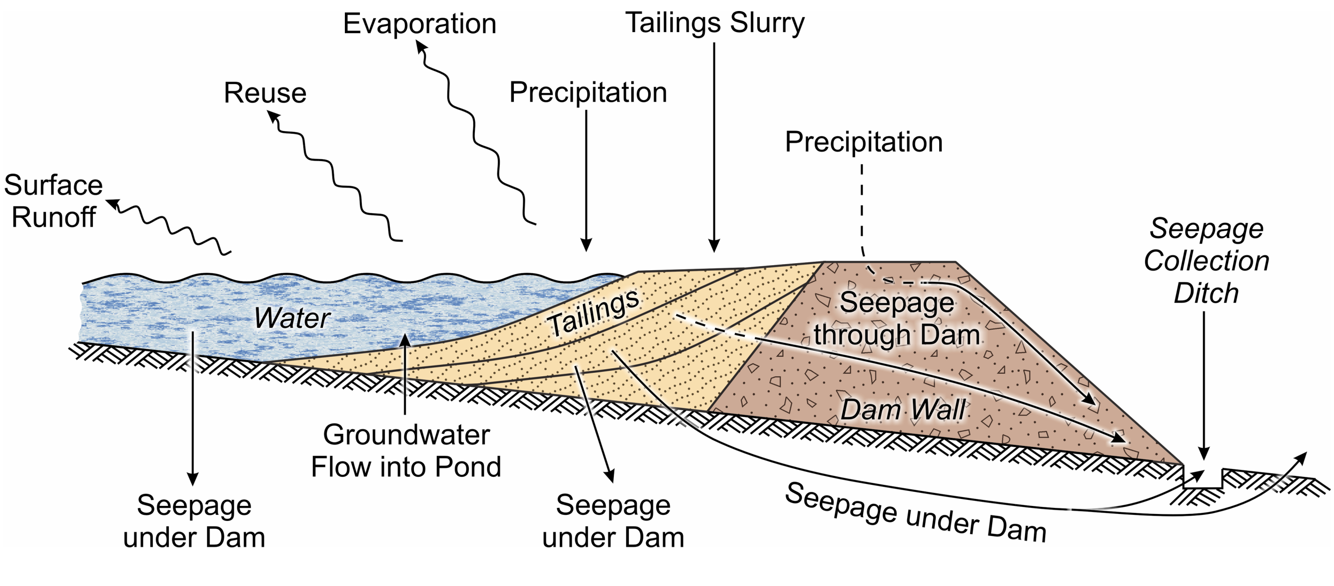

Figure 5. Elements of the water balance of a tailings dam (after European Commission 2003a)

during the closure phase of a mine. However, where the changes are likely to be negative, engineering interventions may well be advisable. Typical interventions for engineering mine drainage pathways before final closure of a mine include:

- preventing the contact of unpolluted waters with contaminant sources,
- ensuring that the pathways which waters will take after completion of flooding of the mine voids will drain freely, without causing dangerous build-ups of mine water above the local base level of drainage (with attendant flood risks).

Possible engineering interventions to achieve these ends were originally developed for use *during* mining, to assist the various measures for minimising water ingress to mine workings (as outlined in Section 2.3). The various technologies include underground dams, grout curtains and other forms of grouting, diversion ditches or channels, diversion wells, pipes, or adits, back filling of open voids and sealing of mine entrances. It needs to be clearly understood that, like any engineering measures, these technologies ought not to be viewed as “walk-away options” to facilitate mine closure. Where engineered structures will become inaccessible after closure, they should be designed so that their eventual failure will only occur after external conditions (e.g. the water level following complete flooding of the workings) will have changed in such a manner that failure will not have grave consequences. Requirements for maintenance or planned obsolescence of engineered structures should all be specified in a Quality Control Plan prepared during mine closure.

3.2.2 Diversion of Discrete Sources of Water

It may prove advantageous to divert surface waters and/or groundwaters. Means to divert surface water include channels and pipes, which are typically installed to:

- prevent direct leakage of watercourses into crown holes or other features which funnel recharge into the mined strata, or
- to prevent already contaminated mine waters from affecting sensitive streams or lakes.

Effective water diversion systems for these purposes can be very extensive, frequently extending over several kilometres, and sometimes even diverting waters from one surface water catchment (or subcatchment) to another. Sizing of channels or pipes must be based on conventional hydraulic analysis (see PIRAMID Consortium 2003), bearing in mind the local hydrological extremes that must be accommodated. Regular inspection and periodic maintenance for such diversion systems is highly recommended.

Groundwater diversions can comprise either physical barriers to flow or boreholes that intercept water and divert it elsewhere. Examples of physical barriers for groundwater diversion include dams in mine roadways, grout curtains, and various other concrete-based structures. In eastern Germany, near the Polish border, subsurface concrete diaphragm walls are used to prevent the River Neiße from flowing into several large open pit lignite mines. These walls are up to 100m deep and several kilometres long, and have been effective ever since they were first installed in the 1970s (Mielcarek 1988; Fahler and Arnold 1996). In the Idrija mercury mine (Slovenia) grouting of abandoned mine workings served both pollution prevention and geotechnical stabilisation purposes (Dizdarevič and Režun 1998). As has been shown at the M. Mayerova Mine (Czech Republic), grouting *after* mine flooding can also be used to halt the drainage of polluted waters (Kipko et al. 1993).

A number of national jurisdictions specify standard construction techniques for undergroundwater dams in mines. The most typical construction method comprises the following steps:

- pressure grouting of the surrounding rock mass using purpose-drilled grout injection holes
- construction of two brick or block walls across the mine roadway, and between 10m and 20m apart (depending on ground conditions), with injection pipes passing through the “downstream” wall
- injection of concrete to completely fill the space between the two walls, followed by final sealing of the injection pipes.

From tracer investigations in already flooded mines, it is known that underground dams constructed prior to flooding often fail, probably due to pressure differences across the dam during the flooding process. This is not surprising, given the very great heads which some underground dams have to withstand, which often greatly exceed the maximum heads retained by even the highest surface water dams (Figure 6). These pressure differences can weaken the structure of the dam and/or open groundwater flowpaths in the rock mass surrounding the dam (Wolkersdorfer 2001).

In a few cases, directional drilling of boreholes has been used to intercept “clean” groundwater and remove it (by gravity drainage) from a mined system before it can drain to deeper levels and become contaminated. This was accomplished, for instance, at the Dalquharran mine in Ayrshire, Scotland. However, circumstances in which this will prove easy to implement are few and far between.

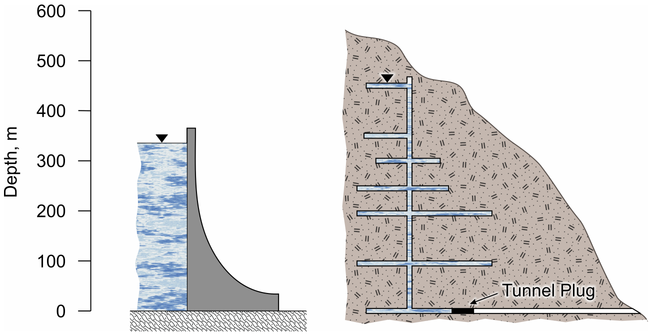

Figure 6. Comparison of hydraulic head above a mine plug with the world's highest dam (Rogun Dam, Tajikistan, 335 m)

3.2.3 Ensuring free Drainage Pathways for Future Drainage

Three recent experiences in the UK have underlined the importance of ensuring that waters that will emerge from closed mine workings after they have flooded up to base level have a free flowpath to the surface environment. Temporary impoundment of mine waters behind roof-fall debris in an adit led to a spectacular outburst of some $50 \cdot 10^6$L of highly polluted water from the Wheal Jane mine in Cornwall when the pile of debris suddenly collapsed. To avoid such problems during closure of the nearby South Crofty mine, the decant pathway to the surface was carefully mapped. When it was discovered that backfill had been stored in a crucial branch of the adit that would provide the most obvious drainage route for mine waters after flooding, this backfill was mined out to create a freely flowing pathway (Younger 2002a). Finally, an old underground dam in an adit at Mynydd Parys (which had originally been used to alternately flood and drain a portion of workings as part of an *in situ* leaching process) was found to be impounding more than $150 \cdot 10^6$L of acid water, perched above the town of Amlwch, which has expanded in the flat land downhill from the adit portal in the 50 years since mine closure. To eliminate the substantial flood risk, the mine water was pumped out from behind the dam, and then the dam was mined out, allowing the mine to drain naturally. Clearly, if greater thought had been given to this issue during mine closure, these expensive interventions would not have been required 50 years later.

3.2.4 Mine Entrance Sealing

Permanent sealing of mine entrances has long been a standard practice in many European countries, mainly to prevent unauthorised access to the mine voids (which are often dangerous when left unmaintained) and to minimise public health threats from dangerous mine gases etc. Some mine entrances have also been sealed in order to "prevent" polluted drainage, though in most cases this results not in prevention but in transfer of the discharge to another point, as shown in Figure 7.

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Indeed, there are a number of compelling hydrological reasons for considering alternatives to complete sealing of redundant mine shafts, including the following:

- the very act of backfilling shafts (which is typically achieved by tipping crushed rock from the shaft collar) often damages shaft liners, creating even more hydraulic connections between the shafts and the surrounding aquifers,
- tipped backfill settled in a shaft is almost never impermeable, and will thus still tend to provide a pathway for mine water movement post-closure (while backfill could in principle be pressure grouted to reduce its permeability, the costs of this are likely to be prohibitive for all but the shallowest of shafts),
- once a shaft is backfilled, options for using it for purposes of hydrogeological monitoring and/or as a pumping shaft for a mine water management scheme have been lost.

For these reasons, we recommend that shafts *are not* routinely backfilled, but are instead made safe (for reasons of public safety) by the installation of substantial reinforced concrete caps, through which large-diameter access pipes are maintained to facilitate hydrogeological monitoring and/or pumping post-closure.

The long-term stability of shafts is a concern, especially when the lack of ventilation, secondary egress routes and appropriate winding equipment in the post-closure phase make routine shaft maintenance impossible. However, there are plenty of examples in which abandoned shafts have been retained for hydrogeological uses many decades after mine closure, without any major shaft stability problems, and when an old shaft finally *does* collapse, it will at least leave behind an annulus of permeable ground through which a replacement borehole may be readily drilled.

In contrast to shafts, which being vertical are useful for level measurement and/or pumping, the less-permanent, sub-horizontal mine access structures

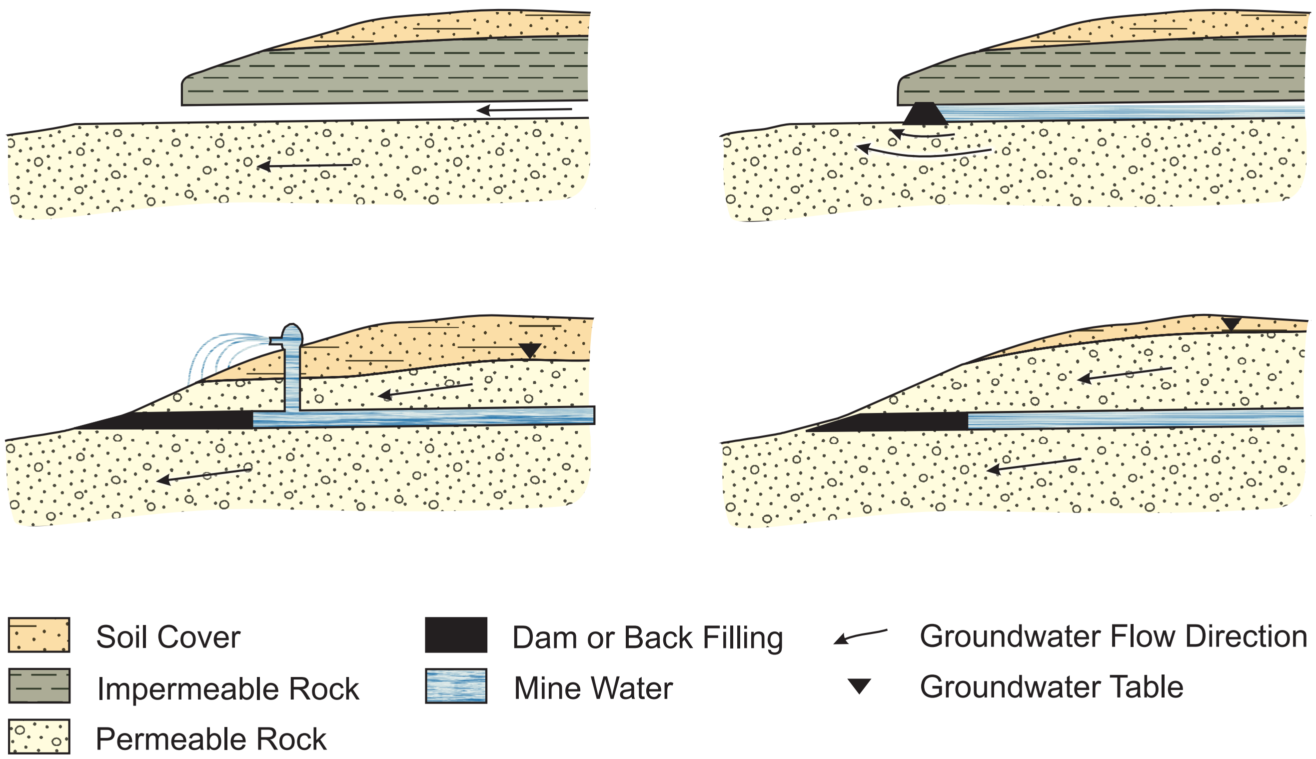

Figure 7. Sealing of mine entrances leading to the diversion of mine water discharges (after Fernández-Rubio et al. 1987)

(inclines and adits) which are not very useful hydrogeologically, should be sealed soon after closure for reasons of public safety. However, in many cases, it will be prudent to include a length of pipework through the stopping, with a flange plate on the daylight end that can be opened so that free drainage of excess mine water can be permitted should this prove necessary/advisable (Figure 8 and Figure 9). Complete sealing of an adit that would tend to serve as a natural drain can only lead to diversion of the water to another discharge point that might be less convenient (Figure 7). Furthermore, the build-up of excess head behind the seal has been known to compromise the stability of such adit seals, giving rise to eventual failure and sudden out rushes of large volumes of water that usually carry very high suspended loads, causing severe pollution in receiving watercourses.

It is also worth noting that grated barriers to shafts/adits might be most appropriate in areas where protected species of bats are known to colonise old mine workings. Such grates can be readily designed to permit both access for the bats *and* free drainage of water. Local environmental protection groups/bat protectionists can help to maintain such facilities in the long-term.

3.3 Planning for Long-term Fate of Tailings and Waste Rock

Water pollution related to the long-term fate of tailings and waste rock from ongoing and historic mining of sulphidic ores is primarily related to the release of acidic, metal- and metalloid-laden leachates. Water pollution may occur in the form of perennial release of contaminated leachates, reflecting insufficient remediation of the mine waste repository, or as specific events, either due to seasonal runoff of polluted surface waters from unreclaimed waste surfaces, or in extreme cases, due to impoundment failure (e.g. Macklin et al. 2003). In the case of either perennial or sporadic releases of pollutants, the timing, duration, and degree of water pollution at any point downstream are determined both by the source emission and by the subsequent hydrological transport and retention/attenuation of emitted pollutants (see Figure 1).

It is possible to incorporate representations of source emissions, transport and retention/attenuation processes downstream of sources within economic decision-making rules for determining and choosing the most efficient abatement measure allocation amongst the many source/downstream possibilities within a given catchment (Appendix III). In many cases, the possibilities will fall into two primary types of measures: direct intervention to minimise pollutant release from bodies of tailings/waste rock (e.g. the use

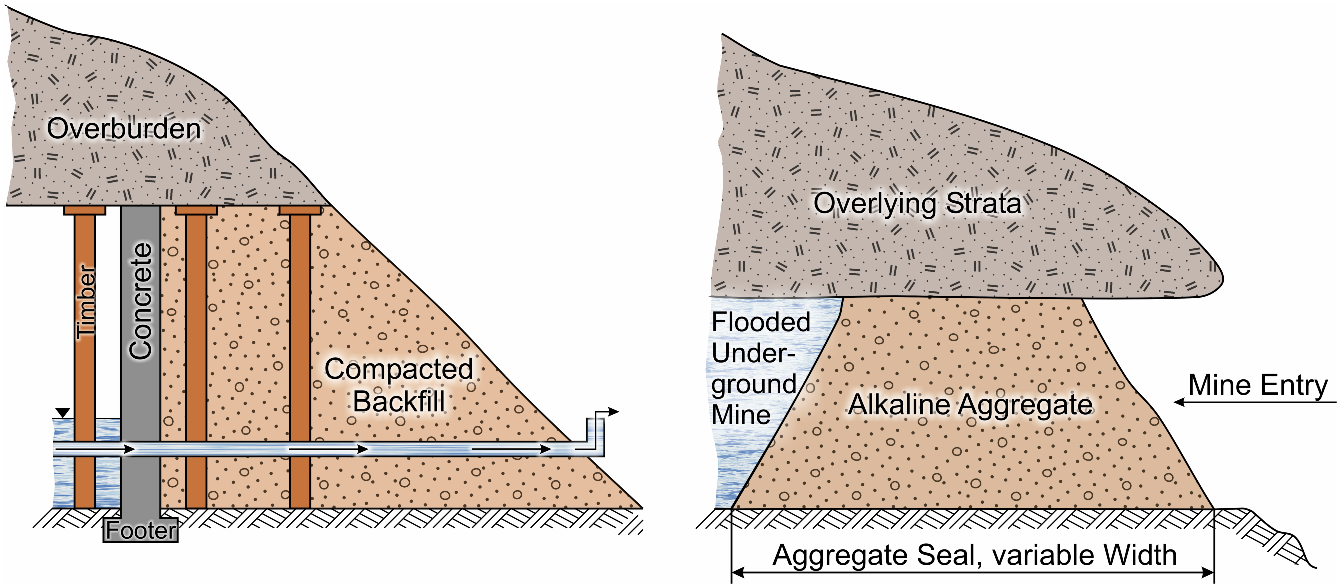

Figure 8. Possible mine sealing (after Scott and Hays 1975)

of soil and water covers) and passive treatment technologies installed downstream of the waste facility (e.g. wetland construction, see Appendix II). It should be noted that the best solution may not be an "either/or" choice between these two, but staged application of both. For instance, a series of papers concerning the Quaking Houses mine site in northern England (Younger et al. 1997b; Jarvis and Younger 1999; Gandy and Younger 2003) relate a series of post-closure interventions in which a compost wetland was first installed to treat acidic leachates. Site investigations were then implemented to identify the pyrite-oxidation "hot spots" within the spoil heap giving rise to the acidic leachates, allowing localised clay-capping of the heap over the "hot spots". Pollution generation has declined, thus reducing the stresses on the wetland and increasing its longevity.

Table 3 summarises the various forms and functions of soil and water cover technologies, which are effectively passive remediation measures aimed at minimising pollutant release from redundant spoil heaps and tailings dams. The alternative, downstream passive mine water treatment (commonly involving wetland construction) enhances natural pollutant attenuation by using natural materials that promote targeted chemical and biological processes. Environmental conditions are manipulated in the passive mine water treatment system, so that particular pollutant removal processes are optimised, as explained and discussed in detail in Appendix II.

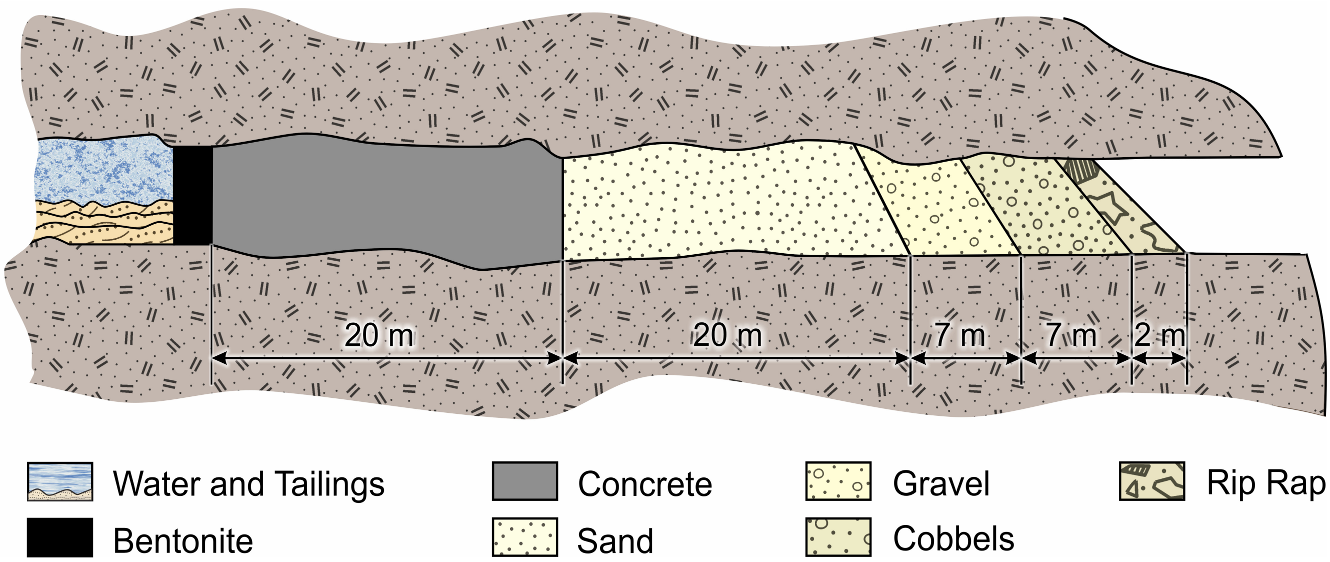

Figure 9. Example of an underground mine dam (after Lang 1999)

Soil and water covers are sometimes referred to as "walk away" solutions to the long-term problem of water pollution from mine tailings and waste rock. This implies that these covers will be stable and sustainable in the long-term, and not require any monitoring and maintenance after construction. However, the long-term performance and effects of such covers must still be studied (Salmon and Destouni 2001), with theoretical studies (e.g. Werner 2000) indicating and actual accidental events demonstrating (e.g. Macklin et al. 2003) important long-term risks and dangers. Hence, it is rash to assume long-term stability; systematic inspections, monitoring and maintenance should therefore be planned for, regardless of the choice of mine waste remediation method.

As an integral part of the planning and decision-making for the long-term fate, remediation/abatement, monitoring, and maintenance of mine wastes and associated leachates, the catchment management perspective also demands the investigation and quantification of the downstream transport and attenuation processes. This will allow evaluation of water influence zones (Figure 1), natural attenuation rates, and cause-effect relations for different remediation/abatement action scenarios (Appendix III), for example. Mine water geochemistry is, however, relatively complex, involving multiple components, high concentrations of solutes and nonlinear geochemical processes. Such complex interactions between solutes and solid phases result in a system of moving reaction fronts that govern trace element as well as major component chemistry and geochemical master variables (Zhu et al. 2001; Malmström et al. 2003). In surface waters (Berger et al. 2000), as well as in groundwaters (Zhu et al. 2001), attenuation of acidity occurs through reaction with buffering minerals and by mixing with other waters that contain alkalinity. The tendency is for pH to rise to successively higher levels at larger distances from sources (mine wastes deposits). In groundwater systems, the rate of propagation of low pH zones is determined by a complex interplay between physical transport processes and geochemical reactions (Zhu et al. 2001), an interplay that is sensitive to mineral abundances, and in particular to the precipitation/dissolution behaviour of secondary mineral species (Zhu and Burden 2001). As pH is a master variable for pollutant retention, a similar sensitivity should be expected in heavy metal transport and concentrations. Modelling studies of groundwater (e.g. Zhu et al. 2001; Berglund et al.

Table 3. The function of soil and water covers that may used in different possible forms and varieties for limiting/preventing acidic/metalliferous drainage from mine waste sites (see PIRAMID Consortium 2003 for details on their engineering design)

Cover type	Primary function
Soil cover as oxygen diffusion barrier	To limit oxidation by acting as a barrier against the diffusion of atmospheric oxygen into the mine waste. The soil pore water effectively slows down the oxygen diffusion rate, compared to the diffusion rate in air. Establishment of vegetation on soil covers helps stabilisation and enables reclaiming of waste sites.
Oxygen consuming barriers within a soil cover	To limit the transport of atmospheric oxygen through the soil cover, by promoting reactions that consume the penetrating oxygen.
Low permeability barriers within a soil cover	To limit the formation of leakage, by acting as a barrier against water infiltration from precipitation into the soil cover, and limit the oxygen diffusion rate by maintaining a high water content.
Geochemical barriers within a soil cover	To provide a (bio)geochemical environment that limits metal release rates and metal mobility.
Underwater deposition, or creation of a surface water cover over the mine waste by hydrologic-hydraulic engineering methods	The water cover above the deposited mine waste reduces the amount of atmospheric oxygen that is available for oxidation reactions, by slowing down the oxygen diffusion rate considerably, compared to the diffusion rate in air.
Groundwater saturation of mine wastes	The groundwater table is raised until it covers the mine waste, thereby creating an oxygen diffusion barrier by the same principle as for a surface water cover, or underwater deposition.
Wetland construction upon mine wastes	To form an oxygen diffusion barrier, by the same principle as surface/groundwater covering and underwater deposition, with wetland plants providing further protection and/or possibly requiring less water covering, by stabilising sediments and thereby hindering re-suspension of water pollutants.

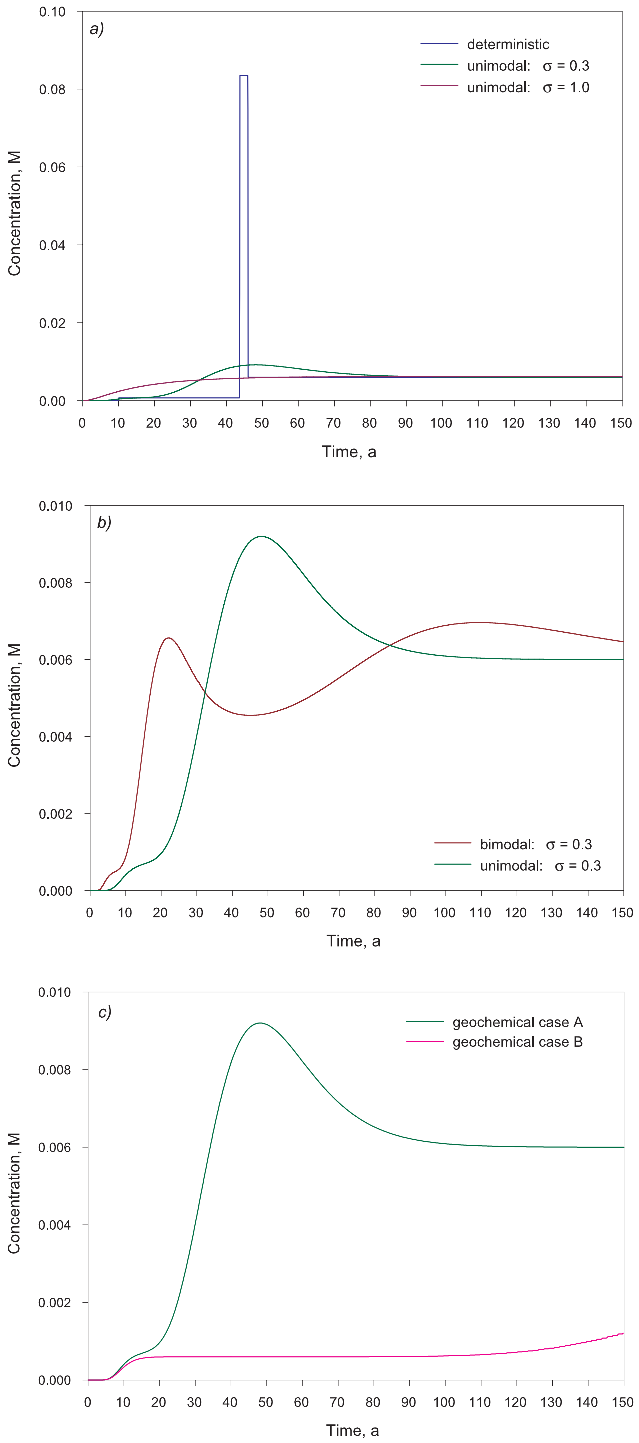

Figure 10. Modelled breakthrough of Zn^{2+}, transported and attenuated by precipitation of $ZnCO_{3(s)}$ in groundwater, at a compliance boundary 200m downstream of a mine waste deposit. *a)* effect of variable (unimodal) groundwater flow; *b)* effect of presence (bimodal) or absence (unimodal) of preferential flow paths; *c)* effect of major ion reaction model (from Berglund et al. 2003)

2003; Malmström et al. 2003) and field studies of surface water (e.g. Berger et al. 2000; Yu and Heo 2001), for instance, indicate that pollutants, such as heavy metals and metalloids, can be significantly retained by precipitation of secondary phases and/or sorption to solid phases. Important differences, however, are to be expected in the retention of redox sensitive elements, such as As, Cr, and Hg, between oxygenated surface waters and anaerobic groundwater.

Furthermore, all the above mentioned complex reactions occur in flow systems (groundwater and surface water) that are characterised by highly variable properties, implying uncertainty in model input and model results. In the context of planning and decision-making for the long-term fate of tailings and waste rock, relevant cause-effect relations must therefore be established and quantified regarding both the geochemical reactions and the physical transport processes in such highly variable flow systems (Berglund et al. 2003). Using transport and attenuation of Zn as a relatively simple example of heavy metal pollutant fate in groundwater, Figure 10 demonstrates prediction errors that can be made regarding the temporal evolution of Zn concentration at some control plane (for instance, a compliance boundary) downstream of a mine waste site (200m downstream in the example case of Figure 10) by use of erroneous transport and reactions process models. Figure 10a illustrates that Zn^{2+} concentration predicted by a model that neglects natural groundwater flow heterogeneity (the so-called deterministic curve in Figure 10a) may be quite irrelevant even in flow fields with a relatively low degree of heterogeneity (curve indicated by $\sigma = 0.3$ in Figure 10a, with a curve indicated by $\sigma = 1.0$ implying a relatively high degree of flow heterogeneity).

Figure 10b further exemplifies the effect of often observed (Simic and Destouni 1999; Kimball et al. 2002) preferential flow paths in groundwater (bimodal curve in Figure 10b), resulting in considerably earlier pollutant arrival, as well as increased long-term tailing in pollutant transport, relative to predictions that neglect such preferential flow paths (unimodal curve in Figure 10b); the local minimum in modelled Zn^{2+} concentration for the former (bimodal) prediction scenario, which coincides with the maximum concentration in the latter (unimodal) prediction, could be quite misleading for management decisions. Finally, Figure 10c compares the predictions of two different geochemical reaction models for the major components. In case A, the thermodynamically favoured transformation of calcite ($CaCO_{3(s)}$) to siderite ($FeCO_{3(s)}$) has been assumed to be fast, such that calcite is depleted, with remobilisation of Zn^{2+} occurring as consequence of a connected decrease in pH. In case B, the transformation of calcite to siderite is assumed slow, such that calcite remains in the groundwater system over the simulation period, and, thus, Zn^{2+} is continuously attenuated. The large discrepancy between predicted Zn^{2+} concentrations for the two reaction models indicates the importance of accurate establishment and quantification of geochemical processes in reactive transport models for reliable predictions of mine water pollution effects.

In general, model results such as those illustrated in Figure 10 show that emitted heavy metals may be substantially attenuated downstream of the emission sources, by processes including precipitation and sorption onto secondary ferric (oxy)hydroxides (Malmström et al. 2003). Moreover, natural groundwater flow variability acts in consort with geochemical processes to determine the extent of water influence zones (Figure 1) downstream of mine waste deposits (Berglund et al. 2003). In the planning and decision-making for the long-term fate and management of mine waters from tailings and waste rock, it is critical to account for such downstream geochemical and transport processes and their dynamics, to correctly: evaluate the current situation of different water environments in a catchment, as the effect of active and old mine wastes within and possibly outside of the catchment (Figure 1); position the present situation as one point in time in the long-term evolution of water pollution (Figure 10); and evaluate the long-term effects at different downstream water environments from present and possible new, future mine wastes under different action scenarios (Appendix III). In Chapter 5, we propose a general planning and decision framework for long-term mine water management, that provides a *modus operandi* for achiving these aims, coupling quantitative model results with stakeholder interactions and negotiations at different stages in the planning and decision process.

4 Minimisation of Impacts: Post-Closure

4.1 Introduction

In some halcyon age in the future, it is to be hoped that all mines opened and closed *after* the start of the third millennium will have been operated so closely in harmony with the recommendations made in Chapter 2 and 3 of this report that unplanned post-closure interventions will never be necessary. However, present reality is that the bulk of mine water problems in Europe are already associated with abandoned mine sites, and if these problems are not systematically addressed (for the most part by public authorities, where responsible “problem owners” cannot be identified) then it will be impossible to achieve the overall goal of the Water Framework Directive (European Commission 2000, i.e. “good” status for all of Europe’s rivers by 2015 AD), so numerous are the catchments in which pollution from abandoned mines is the single greatest cause of freshwater pollution.

In this section, we therefore describe a systematic approach to characterising and prioritising discrete sources of mine water pollution, and the development of “rolling programmes” of remediation to gradually lessen the burdens of the past in Europe’s rivers.

4.2 Management of Large Coalfields

Some of the most pressing issues which militate against the sustainable management of mining impacts on water catchments in Europe are associated with the abandonment of the many, very large Carboniferous coalfields that have long formed the principal energy resources in many European countries. We speak here of the major coal basins, including those of western Germany (Ruhr, Saar, Aachen), France (Nord-Pas de Calais, Lorainne), Spain (Central Asturian Coalfield), Belgium (Liège), the Netherlands (Limburg), Poland (the Silesian Coal Basins), and the UK (Central Scotland, North-East England, Yorkshire, East Midlands, and South Wales).

The catchment management problems arising from the abandonment of these major coalfields have been recently reviewed by Younger (2002b). The root cause of the catchment management problems associated with coalfield abandonment lies in the fact that centuries of mining have effectively led to the creation of “man-made aquifers”, which are extensively interconnected underground over contiguous areas of many tens, hundreds, or in some cases even thousands of square kilometres. During the working of such coalfields, elaborate regional dewatering systems have typically developed over time, with old shafts often being refitted for use as pumping stations as the areas of active mining migrate to ever deeper districts of the coalfields. Pumping at old mine shafts was typically carried out for safety reasons, to prevent migration of hazardous quantities of water to mines that were still in production. As more and more mines have closed over the years, the economic burden of dewatering has gradually been passed on to ever fewer remaining collieries, until the last working mines in a coalfield can carry economically insupportable pumping rates, approaching 15t of water being pumped from the workings for every tonne of coal raised.

Given this typical historical trajectory in the development of regional scale dewatering operations, final closure of an entire coalfield is usually accompanied by the termination of pumping, which can have diverse consequences over very large areas, in many cases extending through several adjoining surface water catchments and sometimes also affecting one or more regional aquifer systems. The consequences of regional-scale cessation of dewatering in major European coalfields include:

- **Relief from some of the negative side effects of dewatering, including:**
  - Excessive drawdowns in surrounding aquifers. For instance, the abandonment of coalfield dewatering south of the Butterknowle Fault in County Durham (UK) in the mid-1970s resulted in the water table rising about 10m in the water table in the overlying Magnesian Limestone Aquifer, an important public water supply source. This arguably means that the cessation of dewatering restores resources to an important aquifer, although (as will be seen below) issues of water quality cloud the issue substantially.
  - Contamination of surface waters by dewatering effluents. While dewatering effluents are commonly treated to prevent them from contaminating receiving watercourses (see Section 2.4.2 and Appendix II), highly saline mine waters may not be amenable to sufficient treatment to prevent degradation of receiving watercourses.
- **Loss of some former *benefits* of dewatering.** Dewatering effluents have frequently played valuable roles in sustaining flows in surface watercourses and in diluting other more noxious pollutants associated with sewage effluents. A cessation of dewatering usually results in the abrupt loss of these benefits.
- **Geotechnical problems relating to land subsidence and mine gas hazards.** The reactivation of void collapse and seismicity, sometimes leading to land subsidence, has been causally linked to the flooding of mine voids in the UK, Germany, France, and Ukraine. The physical mechanisms responsible for these processes are varied, and include the erosion of mine voids by rapidly flowing waters, the slaking of seat-earths and other incompetent strata, and the lowering of effective stress in fault planes due to the increase in pore pressure. On the other hand, in the Limburg coalfield (Netherlands), flooding of workings has reportedly caused "upsidence" (i.e. the raising of the ground surface), due to buoyancy effects and physicochemical expansion processes as strata containing swelling clays are flooded. The process of flooding can also temporarily accelerate mine gas emissions as gases are pushed ahead of the rising water table. This process has been documented recently from mines in Italy, Poland, and the UK, with fatal incidents being recorded in the UK.
- **The discharge of water from flooded workings to adjoining surface and subsurface water bodies.** Overall, this is the most common, most sustained and most environmentally- and economically-damaging consequence of the cessation of coalfield dewatering. Problems associated with mine water discharge from abandoned workings are of two basic types (both of which may be manifested by the same discharge), surface flooding, and aquatic contamination. Of all of the above potential consequences of a cessation of regional dewatering, the latter (aquatic contamination) is pre-eminent and will thus be considered in further detail below.

Although much of the literature focuses on "acid mine drainage", it is important to realise that many alkaline coal mine water discharges are still sufficiently rich in iron to be highly contaminating. As noted in Section 1.5, biological studies have revealed that the damage caused to benthic invertebrate faunas by alkaline, ferruginous discharges is often just as severe as that caused by acidic mine water discharges, due to the smothering of the benthos with ochre precipitates, which prevent photosynthesis, and therefore locally removes the foundations of the food chain. Many abandoned coalfield waters also contain sufficient sulphate and/or manganese that they compromise the usability of receiving watercourses for drinking water supplies. While most documented instances of aquatic pollution from abandoned coalfields relate to surface waters, it is important to realise that polluted mine waters can also migrate into adjoining fresh water aquifers, jeopardising their utility as water resources. Examples of aquifer pollution by migration of mine water from abandoned coal mines are surprisingly sparse, probably due to a lack of investigation. One of the few documented examples relates to the southwestern area of the Durham Coalfield (UK), where water levels recovered over the eight years following the final cessation of pumping in 1975. Contaminated mine water subsequently migrated into the overlying Magnesian Limestone aquifer, giving rise to an extensive sulphate-rich plume of groundwater and rendering tens of square kilometres of the aquifer unusable for water supply abstractions. Similar instances of aquifer contamination by migrating mine waters are anticipated to be one of the most important environmental consequences of the recent and future closure of large coalfields in a number of European coalfields, especially where the sub-aquifer coalfield is laterally continuous with exposed coal measures at higher topographical positions.

One aspect of the abandonment of major coalfields that makes their management at catchment scale particularly challenging relates to the timescales required for complete flooding of very large systems of interconnected underground workings, and the long-term persistence of water pollution after completion of flooding.

The vast scale of many coalfields means that, despite very high water makes, it normally takes a number of years (and in some cases even decades) for completion of flooding. For instance, in the central Fife Coalfield, Scotland, complete flooding of the collieries of the Ore Valley area took nine years. Similarly, the Northumberland Coalfield of England required 17 years to flood-up to sea level. Current predictions for large coalfields with modest water makes suggest timescales on the order of 30 to 40 years. With such long timescales, it is very easy for public authorities to become distracted from vigilance, so that the onset of polluted discharges comes as a nasty surprise. The demanding logistical measures needed to avoid such calamitous developments should not be underestimated.

In terms of post-flooding control of mine waters in large, interconnected coalfields, it will often be possible to prevent surface water pollution by allowing gravity outflows to the surface. These discharges can then be intercepted and treated prior to final discharge to receiving streams. In cases where the sensitive target is an overlying aquifer, there may be no alternative to long-term pump-and-treat operations that will maintain water levels within the abandoned coalfield below the minimum head level in the overlying aquifer. Such a system is currently undergoing commissioning in the sub-aquifer coalfield of eastern Durham (UK), and is predicted to require perpetual pumping of more than $200 L \cdot s^{-1}$ of contaminated mine water in order to prevent wholesale contamination of the overlying Permian Yellow Sands and Magnesian Limestone aquifers. The alternative would lead to the sudden loss of well-fields that represent the sole source of public water supply for several hundred thousand people.

With regard to the long-term persistence of water pollution after onset of overflow (or pumping) from fully flooded coalfields, contaminant concentrations are usually highest shortly after completion of flooding, (the "first flush"), with gradual improvement in water quality then being observed over a time period (Figure 11) that is typically about four times as long as the time that it took the mine to flood (Younger 2000a). However, water quality may or may not improve that the point that the water can be discharged untreated without adverse consequences. Hydrogeochemical modelling typically suggests that long-term levels of pollution may well endure for hundreds of years, until such time as pollutant source minerals are finally exhausted (Wolkersdorfer 1996; Younger and Banwart 2002). The development of appropriate, sustainable engineering responses to mine water pollution requires that this temporal persistence be taken fully into account when planning remedial works. In particular, it will frequently be necessary to use intensive "active treatment" techniques (Appendix II) during the first flush period, with passive treatment being used to provide long-term treatment for the mine waters post-flush.

4.3 Characterisation and Prioritisation of Problematic Discharges

4.3.1 Catchment Surveys to Identify Point and Diffuse Sources of Pollution

Before attempting to remediate problematic discharges, it is necessary to adequately characterise them over a sufficiently long time period, and prioritise. By doing this, the potential success of any remedial action is optimised. Prioritisation can be assigned to each discharge based upon various impact-derived criteria (Section 4.3.4).

In several cases, former mining areas are now remote and underpopulated, and though it may be known that a given river is heavily contaminated with metals, the locations and nature of the various pollutant sources may not be accurately known. Where this is so, it is necessary to identify and quantify the principal sources of pollution entering these rivers, so that catchment-based decision-making tools (see Chapter 5 and Appendix III) can be applied, and the most appropriate means of remediation (Section 4.4 and Appendix II) can be deployed. Identification and quantification of pollutant sources can be achieved by applying traditional geochemical exploration methods, identical to those used in mineral prospecting in the first place! The most widely used geochemical reconnaissance technique involves surveying whole catchments by sampling stream sediments and/or waters of each tributary, and following anomalies upstream (Leutwein and Weise 1962; Rose et al. 1979; Levinson 1974). Younger and Bradley (1994) successfully adapted these geochemical exploration techniques to locate point discharges from abandoned mines by considering the following factors:

- Mine water discharges produce characteristic sediments that may have a visual impact (e.g. ochre deposits), albeit small quantities of finely grained ochre mixed with clastic sediments in areas of a large drainage basin distant from sources can be hard to detect because iron is so ubiquitous an element that subtle anomalies are hard to distinguish.

- Uncontrolled mine water discharges often exhibit relatively low pH values, high sulphate concentrations and high salinities relative to "normal" shallow groundwater. These chemical attributes can assist with the search for such discharges.
- The influence of these chemical attributes of mine waters will be most marked during periods of little or no rainfall, when most of the water in the stream channels will have originated from groundwater sources (including mine water; sampling whilst the receiving waters are in spate would tend to subdue the geochemical anomalies).

It is necessary to delineate a "target area" for the study which must take into account certain mining related features, such as outcrop areas of formerly mined strata, beyond which uncontrolled surface discharges of mine water are unlikely. This type information can often be gleaned from geological maps or mining records. It is also important to understand what was being mined in a particular catchment. For example if Pb and Zn were mined, the laboratory analysis should be directed towards ecotoxic (semi-)metals such as Zn, Cd, Pb, Cu, and As (see also Section 4.3.2 and Appendix I).

Once a target area has been located, and a period of dry weather has led to baseflow conditions in the catchment drainage system, the survey can be undertaken. The basic strategy is to commence with the main river in the area and move upstream systematically sampling water in the main channel and all tributaries. At each tributary, one should sample the main river immediately upstream and a little downstream of the confluence. Note that the downstream measurement point is not right at the confluence, but at a distance equal to three to five times the width of the main river at this point, to allow for mixing of the tributary water and the main river water. Having sampled all tributaries in this way, those that display anomalously high conductivities compared with the main river water or other tributaries should be targeted for the next sweep of sampling. Surveys up each of the "suspect" tributaries carry on as above, until eventually one will be dealing with first order streams. By that stage in the survey, several major mine water inputs will likely have been discovered, and the remainder will be identifiable as inputs to the first order tributaries.

While making these field surveys, it is possible to measure some determinands (e.g. pH and conductivity) in the field, greatly expediting the identification of blatantly anomalous tributaries. Streams affected by mine waters will often have baseflow chemistries identifiable by elevated conductivity (i.e. in excess of around 700$\mu S \cdot cm^{-1}$) and may or may not be associated with a low pH (the existence of net alkaline mine water discharges means that reconnaissance surveys should not be carried out using pH alone, as not all mine water discharges are acidic). The results from the field catchment survey should always be supported by laboratory analysis of samples collected wherever pH and conductivity anomalies were detected (recalling that mine water discharges tend to show elevated sulphate and/or metals concentrations).

Stream sediments can also be collected at sites where field determinands and water samples are taken. Sediments are then sieved through a nylon mesh (62µm is typically used) with river water to attain a standardised sample size. Higher quantities of metals are associated with the < 63µm fraction because smaller grain size means a higher surface area grain size ratio. This particle size has also been used by others and is therefore useful for comparison between different sites (Gray 1997). The samples are then centrifuged and dried before being ground with a mortar and pestle and weighed. There are two methods of sediment digestion that can be carried out: partial and total. Total sediment digestion involves total breakdown of the samples with strong acids so that all of the metals present are liberated into solution. The total dissolution method is useful to gain an idea of the total amounts of metals associated with the sediment but it is unlikely that natural environmental conditions will ever become harsh enough for total sediment dissolution and subsequent full metal release to take place (Nuttall 1999). However partial dissolution (Harper et al. 1989) liberates most of the metals in the sediments (except for some refractory and silicate minerals) and gives a more realistic picture of what metals may be released under natural environmental conditions. Following digestion and analysis for metals, sediment concentrations can then be plotted on a catchment map and used to supplement the water quality survey data.

Using the above approach, Younger and Bradley (1994) were able to rapidly identify the most important sources of mine water pollution in the western part of the Durham Coalfield (England), paving the way for subsequent detailed characterisation, prioritisation and remediation of the predominant sources.

4.3.2 Hydrogeological and Hydrochemical Methods to Characterise Discharges

Hydrometric techniques for measuring the flow rates of springs, streams and rivers are well established and widely described in the literature (e.g. Brassington 1999; Younger et al. 2002). The PIRAMID Consortium (2003) offers substantial practical guidance on the application of these methods to mine waters, in which clogging of structures with ochre is a particular problem. To cope with this problem, it is recommended that, wherever possible, structures such as weirs and flumes are designed to allow rapid, easy removal of ochre build-up, and that water level sensors that do not require immersion in the water be considered. Further discussion on these matters, and on advanced hydrometric techniques such as the use of tracers in mine waters, is reserved for Appendix I.

Groundwater hydrometry involves the determination of elevations of groundwater levels (as measured in boreholes, shafts etc.) relative to a nationally agreed surveying datum (usually some particular measure of mean sea level). Spot measurements of groundwater level are made using electronic tape measures known as “dippers”, while measurements of temporal changes of head are best obtained using solid state digital loggers suspended in the boreholes.

Hydrometeorological measurements are required for all hydrological investigations, and mine water studies are no exception. The aim is to have sufficient data to allow calculation of hydrologically effective precipitation (HEP) in the catchment of interest (bearing in mind that this will vary considerably in hilly catchments). HEP is defined as the difference between total rainfall in a given period and the losses to satisfy any soil moisture deficit (which is essentially a measure of crop water use prior to the onset of rainfall) and evapotranspiration. To evaluate HEP, it is therefore necessary to have a record of changes in soil moisture content (usually obtained by continual application and upgrading of moisture budgeting models) and those meteorological variables that govern the rate of evapotranspiration (i.e. solar and net radiation, wind speeds and directions, air temperature, etc.). In European countries, the data required to calculate HEP are generally gathered by public authorities, and the mine water investigator will be able to purchase time series of quality controlled HEP values. However, publicly available HEP time series are generally given as spatially averaged values for a region, and are rarely readily available at any temporal resolution greater than weekly. While weekly or even monthly averages may suffice for analyses of subsurface flow, studies of storm runoff for surface mine drainage design will generally require much finer resolution. For detailed mine water studies, consideration should be given to the installation of rain gauges and automatic weather stations.

Appendix I provides an overview of the hydrochemical methods appropriate to mine water investigations, and nothing further need be added here.

4.3.3 Biological Methods to Characterise Discharges

Biological methods include the use of aquatic flora and fauna to assess or evaluate the impact of mine water discharges. The most widely used method involves sampling for benthic invertebrates. This process is known as “kick sampling” (APHA 1998). A standard pond net is used and the operator faces downstream, with the pond net held in front vertically against the substratum. The stream bed is then vigorously disturbed by kicking for a standard three minute period (Mason 1996). Roughly the same physical stream characteristics should be sought at each site (e.g. riffle zones). Samples are collected from the net and transferred to screw top jars and preserved in 70% ethanol with a few drops of glycerol (if they are to be analysed in the laboratory).

In the laboratory, samples should be sieved to remove fine debris. After sorting, samples can be observed using a binocular microscope with incident light, with identification of invertebrates typically being pursued as far as the “family” taxonomic level (Firth and Davies 1997). The data can then be used to calculate a standardised index of invertebrate diversity and numbers for each site, such as the average score per taxon (ASPT), the Biological Monitoring Working Party (BMWP) score (Chesters 1980), or any other national water quality index based on organisms. The BMWP score was originally designed to give a broad indication of the condition of rivers in the UK. Following identification of invertebrates to family level, each family is assigned a score (between 1 and 10) according to their pollution tolerance, with pollution intolerant families, such as the mayflies and stoneflies, being given a score of 10. The individual scores are then summed to give the overall BMWP score (Mason 1996). This biotic index is the one currently used by the UK Environment Agency for benthic macroinvertebrate analyses. Because it was originally devised for biological monitoring of “organic” pollution (such as that associated with sewage), there has been some scepticism over the applicability of the BMWP index to systems affected by mine waters. However, in practice, the BMWP score has been found to correlate directly with severity of chemical pollution in streams affected by both acidic and neutral-pH mine water discharges (e.g. Jarvis and Younger 1997). Most recently García-Criado et al. (2002) have supported the use of the BMWP index for the assessment of rivers affected by coal mine discharges in Spain, while proposing that the scores assigned to some families should be modified for mine water applications. This is because invertebrates such as *Leuctridae*, *Nemouridae*, and *Rhyacophilidae*, which receive high scores in the original BMWP scheme on account of their high sensitivity to eutrophication, are actually fairly tolerant to pollutants found in coal mine drainage (García-Criado et al. 2002).

Other biological methods appropriate to mine water impact assessments include:

- Sampling vegetation surrounding water courses or sampling algae or other aquatic plants within the watercourse. The material is dried, weighed, and then digested with acid (using a similar method as has been described for stream sediments). The residual liquid is then analysed for metals.
- Examining the effects of mine water discharges on higher organisms such as fish, by means of toxicity tests. This involves either assessing the fish stock density *in situ* using methods such as electrofishing, or by collecting fish and taking them back to the laboratory and exposing them to different levels of toxicity in controlled experiments. It is also possible to plant a certain number of fish eggs (usually three batches of 200 eggs) in mesh baskets (usually salmonid eggs) in the stream and count the number of individuals that have hatched/perished over an allotted time period.

4.3.4 Impact Assessment Methodology

Having identified the principal pollution sources (Section 4.3.1) and characterised the flow rates, chemistry and biological effects of individual discharges and receiving watercourses (Section 4.3.2) the next step it to use all of the data to assess the impacts of the various pollution sources. Methods commonly used for assessing the environmental impacts of mine waters tend to use either chemical or biological measures of impact, whereas in most cases the overall impact is better gauged on the basis of chemistry, biology and visual impacts (Jarvis and Younger 2000). One of the best developed, officially recognised methods for assessing mine water impacts is the procedure used by the Environment Agency in England and Wales. This method comprises two phases of investigation and has the virtue of considering both the chemical and biological impacts, as well as quantitative measures of unsightly staining of affected streams by ochre deposits. In the first phase of this methodology, a preliminary assessment of the impact of the mine water discharge on the receiving watercourse is made using the following six criteria:

- a visual assessment of the area of streambed affected by deposition of metal precipitates (especially ochre),
- the length (or reach) of watercourse that is demonstrably affected by the discharge,
- the quality of the existing substrate in terms of its suitability for salmonid reproduction,
- the intensity of any discoloration caused by iron deposition,
- the total iron concentration (in $mg \cdot L^{-1}$),
- pH, dissolved oxygen (in $mg \cdot L^{-1}$) and aluminium concentrations (in $mg \cdot L^{-1}$)

For each of these criteria, the impact is categorised by magnitude (i.e. high, medium, low or none). The second phase of the investigation involves assessing the impacts of the mine water discharges on the benthic macroinvertebrate community, in this case using the BMWP score, which is calculated for a number of sample positions downstream from each mine water discharge, until the point is found where the impacts of the mine water input have diminished to such an extent that the BMWP score matches that obtained for a reference stream reach (either upstream of all mine water inputs on the same stream, or else in a similar nearby catchment that does not receive mine drainage). Depending on the size of the watercourse, it may also be appropriate (and if so, it is important) to assess the fisheries potential of a receiving watercourse, as this can be useful information when prioritising discharges for treatment (see Section 4.3.5).

4.3.5 Ranking and Prioritisation

Using an impact assessment methodology such as that described in Section 4.3.4, it is possible to directly rank mine waters according to the severity of their environmental impacts. This ranked priority list can be used to guide the timing of investments to clean up polluted discharges so as to maximise the environmental benefits in the shortest period of time, given resource constraints that apply to all governmental agencies. Nevertheless, it is appropriate that prioritisation for clean up consider factors other than just those outlined in the impact assessment methodology outlines above (Section 4.3.5). For example:

- In the River Nent in the North Pennines, UK, several Zn rich mine water discharges impact the entire river, restricting fish stocks. However, there is almost no ochre precipitation in this river and its tributaries, so that discharges in this catchment would tend to score unreasonably lowly in an evaluation scheme that is dominated by quantification of the extent of ochre staining. To complicate matters further, however, a large waterfall low down in the catchment represents a barrier to migratory fish, preventing the establishment of a high quality salmonid fishery on this river even if water quality were restored to the highest levels.
- In some cases, a mine water that has relatively little chemical or biological impact on a watercourse happens to emerge in a highly visible location, near

residential properties or in a public recreation area. In such cases, a high priority tends to be accorded to remediating such discharges on sociopolitical grounds, to respond to the sensibilities of the local residents.

On occasions, it may also be necessary to prioritise discharges that have not yet emerged. This is in line with the requirement of the Water Framework Directive that there be no further deterioration in the status of rivers. Where abandoned mine workings are currently flooding up beneath catchments with very high quality rivers or vulnerable aquifer resources, the high prioritisation of future discharges can be extremely important. For instance, this occurred in the River Aln catchment in northern England in the late 1990s when abandonment of a medium sized colliery threatened to lead to uncontrolled pollution of a very beautiful reach of river that not only hosts one of the finest salmonid fisheries in England, but is also used as the sole source of drinking water supply for more than 150,000 people. In such cases, the timing of potential impacts can be assessed by studying the rate of mine water recovery and calculating its depth below predicted emergence (decant) points. If several mines are in the recovery phase, then it is possible to predict which discharges should be dealt with first. Some indications of emergent water quality can be made by taking samples at depth within a recovering system. Samples should be taken near shaft insets (as water quality often changes in these regions) and at the bottom and top of the shaft water column. The worst quality water can then be considered when designing a remedial strategy.

4.4 Developing a Rolling Programme of Remediation

4.4.1 Why a "Rolling Programme"?

Once discharges have been located, characterised, and prioritised, the next step is to implement remedial measures. However, it is not realistic to expect any public administration to sink all of its resources into mine water remediation in one fell swoop. Rather, public funding is more likely to be made available year-by-year, so that a "rolling programme" of remediation needs to be developed in order to decide in what order remediation should be applied to the various polluting sites in a catchment. To make such decisions, it is important to consider the ranked severity of mine water discharges at the catchment scale, and to thoroughly analyse how any environmental improvements are likely to benefit the catchment as a whole. How this can be achieved in practice, within the provisions of the Water Framework Directive, is explored in detail in Chapter 5.

Having embarked on the development of a rolling programme of remediation, it is important to appreciate that the character of a mine water discharge may well change over time, so that periodic reassessment of a chosen remedial strategy may be required. The following chapter provides more detailed insights into this and related issues.

4.4.2 Temporal Changes in Mine Water Quality

Combined laboratory, field and modelling studies of a number of mined systems are revealing the relative importance of geochemical attenuation and hydraulic flushing processes in determining the temporal evolution of groundwater quality at abandoned mine sites. Of particular importance in the short term is the flushing from the system of the products of the dissolution of hydroxy-sulphate salts, which are taken into solution during flooding of mines or waste rock repositories. During a period of time known as the "first flush" (Younger 1997), the quality of the mine water tends to improve, until a more steady long-term level of acidity remains (Figure 11).

The duration of the first flush can be estimated as follows (Younger et al. 2002):

$$t_{\mathrm{f}} = (3.95 \pm 1.2) \cdot t_{\mathrm{r}} \qquad (2)$$

where, t_{f} = duration of the first flush and t_{r} = rebound time, i.e. the time it took for the mine workings to flood to the drainage adit after the cessation of dewatering.

The ratio of peak contaminant concentrations to long-term (post-first flush) concentrations is typically on the order of about 8—10. In the even longer term, the interplay between sources of acidity and alkalinity is the dominant factor in determining bulk mine water quality. While any mine waters tend to remain circum-neutral after the first flush, cases are known in which discharges that were alkaline for several decades suddenly became acidic once more. The conclusion is that early changes in water quality are likely to be for the better, and can be predicted to some extent. However, vigilance must always be maintained and remediation systems designed to cope with reasonable contingencies in relation to changes in mine water quality.

4.4.3 Selecting the Appropriate Remedial Approach(es)

Based on the assessment of the key hydrological and geochemical attributes of mine water discharges, a rational decision-making framework has been developed for deciding which (or which combinations) of these options to implement in a specific case. There are three principal options (of which the latter two are not mutually exclusive; Figure 20):

- Monitored natural attenuation, in which natural processes are deemed sufficient to deal with the

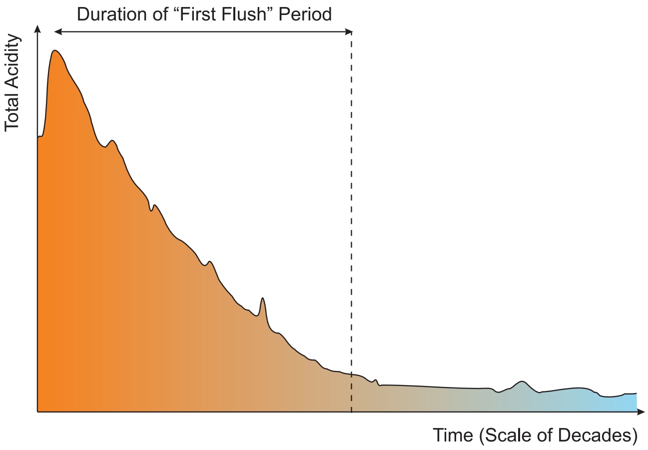

Figure 11. Typical temporal changes in mine water acidity in the years following first emergence of water from newly flooded mine workings (adapted after Younger 1997)

contamination, and monitoring is put in place to ensure that this conclusion is (and remains) valid.

- Prevention/minimisation of pollutant release processes.
- Mine water treatment, by either active (e.g. chemical dosing) or passive (e.g. wetlands) means.

These options are discussed in further detail in Appendix II, as well as in recent publications of Younger et al. (2002) and the PIRAMID Consortium (2003). Suffice it to note here that the choice between these three options is usually not a matter of simple personal preference. For instance, we simply do not yet have passive treatment technologies capable of coping with the most voluminous and most contaminated of mine water discharges, so nothing less than intense active treatment is likely to suffice for such cases.

For both active and passive treatment options, the long-term sustainability of the operation is a key consideration. Factors controlling overall sustainability include the balance between capital expenditure (capex) and operating expenditure (opex), the availability of finance for each of these, and the scale and frequency of essential maintenance operations. Sludges arising from mine water treatment are often voluminous, and at many pump-and-treat schemes, sludge handling and disposal commonly accounts for 25—50% of the total treatment cost. At present, landfilling of such sludges is the usual recourse. However, for the long-term, reuse of solid residues from treatment systems must become a reality if mine water treatment is to match up more closely than at present to the ideal of sustainability (possible ochre reuse strategies are outlined in Section 2.4.2 above).

5 Decision Framework for Long-term Mine Water Management at the Catchment Scale

5.1 Description of a Decision Framework

We describe here a decision framework for long-term mine water management, which can be conceptualised in a single flowchart, and which can be repeatedly used for addressing and solving both of the two primary water management tasks defined in Section 1.7, i.e. task 1: to manage mine water pollution within general water management plans and action programmes, and task 2: to evaluate environmental permit applications for individual mining/mine waste disposal projects. These tasks will be repeated at predefined time intervals for task 1, and whenever permits are required for task 2. Such a flow chart is proposed for general integrated water management by Destouni et al. (2003) and its key components are illustrated schematically in Figure 12. The flow chart includes three main water management questions that need to be answered for every relevant (mine water affected) water environment within the considered catchment, or water district (possibly consisting of several catchments) for task 1, or within the water influence zone of the considered mining/mine waste disposal project for task 2. These three main questions are:

I. Does/will this water environment comply with relevant environmental standards, e.g. in terms of ecological, chemical and/or hydrological status, related to mine water pollutants now and in the future?

If the answer is yes, task 1 does not require any actions to be taken, and task 2 yields a positive evaluation of the project application for this particular water environment. If the answer is no, the next step is to answer question II:

II. Are there any (for task 2: additional to the ones proposed in the permit application) feasible technological and/or socioeconomic measures that can be taken for achieving environmental compliance in the considered water environment?

If the answer is no, task 1 is solved by classification of the considered water environment as strongly modified (cannot be expected to be improved by any feasible remedial actions), and task 2 by negative evaluation of the permit application with regard to this water environment. If the answer is yes, the next step is to answer question III.

III. Which particular, measure allocation should be chosen (in the action programme for task 1, and as requirement for permit issuing in task 2) in order to achieve compliance with environmental standards?

The symbolic negotiating table in Figure 12, which is further detailed in Figure 13, illustrates that a relevant and sustainable answer to each of the three main questions I—III requires discussions, negotiations, and agreements among main stakeholders. The reason is that the answers to these main questions can seldom be expected to be self-evident, or easy to arrive at, because the problems are complex and different stakeholders may have their own, different knowledge base and interpretations of legislation, political decisions and interests, and scientific results for answering these questions. In general, use of relevant local knowledge and public participation have been identified as important for successful water management (e.g. GWP 2000; Olsson and Folke 2001; Belfiore 2002). Different stakeholders in Figure 12 and Figure 13, who can address the long-term fate of mine wastes as a catchment scale water management problem, are then competent authorities for the two different types of water tasks 1—2, along with mining and mine waste site operators and/or owners, and representatives of the general public.

Additional cornerstones for successful catchment scale water management, are:

- A shift needs to be made, from providing static information on just the present state of environment parameters (e.g. currently measured chemical/ecological data), toward pressure, impact and response information (e.g. Pentreath 1998;

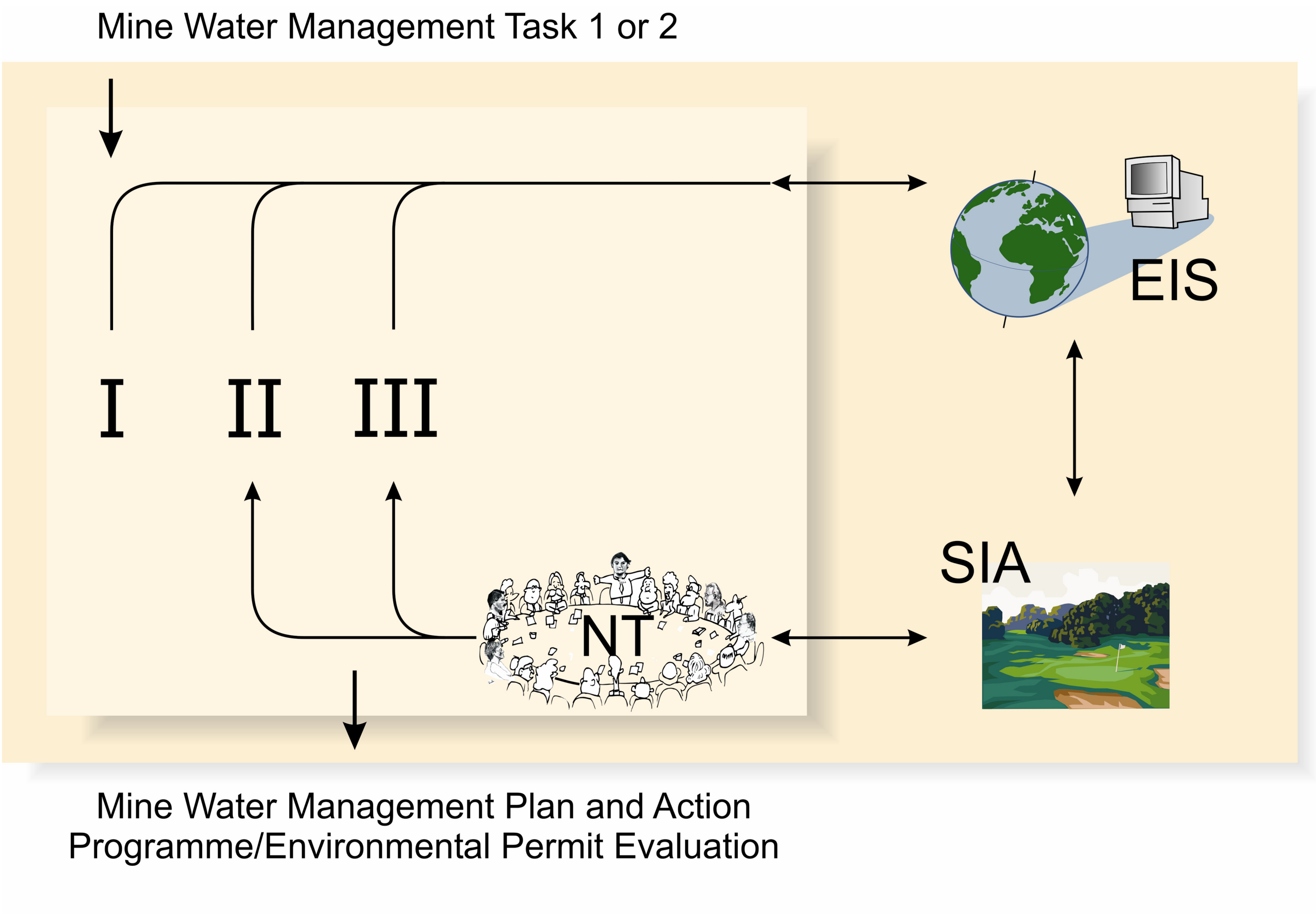

Figure 12. Schematic illustration of the main flow chart components (Destouni et al. 2003): Three main water management questions I—III; Environmental Information System (EIS); Stakeholder Interplay Arena (SIA); and Negotiating Table (further detailed in Figure 13), which are required for addressing and solving the two primary water management tasks: 1. formulation of water management plans and action programmes according to the WFD requirements; and 2. environmental permit issuing for new mining and/or mine waste disposal projects that also follows the WFD

Nilsson and Langaas 2004; IMPRESS WG 2003).

- There is a need for economic efficiency thinking and analysis on the catchment scale (Gren et al. 2000a, 2002; Koussis et al. 2002; Baresel et al. 2003).
- There is a need for better communication of expert knowledge to feed the decision making process (e.g. Denisov and Christoffersen 2001; Timmerman et al. 2002; Langaas et al. 2004).

Appendix III specifically proposes, and exemplifies for the concrete case study of the Dal River catchment, a scientific analysis approach for obtaining relevant dynamic characterisation and economic optimisation results, which can provide a rational basis for discussions, negotiations, and agreements in the process of answering the main water management questions I—III. This approach includes both dynamic characterisation of the catchment scale mine water pollution problem and economic optimisation among different possible pollution abatement measures. By dynamic characterisation, we mean here the interpretation and extrapolation of available (historic and new) data and information on mine water sources and their possible water quality effects, presently and in the future under different possible cause-effect and abatement measure allocation scenarios (see Figure 14 for general illustration). The entire set of such data and information of relevance for a given mine water pollution problem and/or catchment site may then be viewed as an Environmental Information System (EIS), symbolised in Figures 12 and 14 indicating that the EIS consists of a database, from which increasing information concentration is achieved by data interpretation and processing into expert (such as the prediction models underlying Figure 1) and non-expert (such as the resulting graphs shown in Figure 1) systems. This set of data and information may not presently be formally organised and handled as a coherent information system and also may not be generally available to all stakeholders and the general public. Nevertheless, such data and information sets do exist, to greater or lesser degree for most mine water pollution-sites (see Dal River catchment example in Appendix III), and both the WFD (2000/60/EC) and the new European Union Directive on Public Access to Environmental Information (2003/4/EC, European Commission 2003c) call for organisation and handling of such data and information, in the form of effective environmental information systems that are generally open, transparent, and accessible.

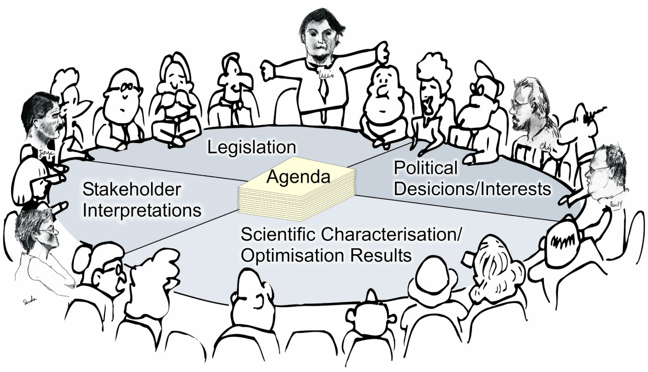

Figure 13. Detailed illustration of the negotiating table included in Figure 12 symbolising the stakeholder discussions, negotiations, and agreements that are required for arriving at generally accepted answers to the three main water management questions in the planning for the long-term fate, remediation and mine water pollution abatement of mine tailings and waste rock. The negotiations are required because there may exist different interpretations and relevance evaluations of the scientific analysis (dynamic problem characterisation and abatement measure optimisation), the legislation, and the political decisions and interests that determine the answers to the key water management questions

The general mine water management flow chart illustrated in Figure 12 implies that the EIS and dynamic characterisation results (Figure 14) must first provide a basis for answering the main water management question I. For a negative answer to this question, agreed upon among the different stakeholders (role of negotiating table, Figure 13), the EIS and dynamic characterisation results may further be used in economic optimisation analysis (Appendix III), thus providing a scientific basis for answering the main water management question II, in terms of a series of optimal abatement allocation solutions for different interpretations of cause-effect relationships (e.g. different relative water quality impacts of known mine waste sites and of other possible, diffuse mine water sources, such as abandoned mine voids) and environmental targets (e.g. different targeted pollutant load reductions, compliance boundary locations, and scale/timing of water quality measures). These optimal abatement allocation solutions provide lower limits of costs associated with achieving different environmental targets under different possible scenarios of cause-effect relationships, technological efficiency and cost-benefit functions, and possible technological limits to feasible environmental target achievement (see Appendix III). Among these optimisation results, it may then be possible to identify a (sub)set of measure allocations that the different stakeholders can agree upon as being both technologically and socioeconomically feasible (negotiating table, Figure 13), thus yielding a positive answer to main water management question II and a basis for choosing a final preferred measure allocation, as an answer to the final main water management question III.

The water management flow chart in Figure 12 further illustrates that different individual or groups of stakeholders may use the EIS (or different parts of the EIS, with possible different accessibility to different stakeholders) independently from each other and from the competent authority and arrive at different dynamic characterisation and economic optimisation results, which they will bring to the negotiating table. Such actions and possible interactions of different stakeholders outside the negotiating table are here referred to as the Stakeholder Interplay Arena (SIA, Figure 12). In addition to possible differing scientific analysis results, or different practical interpretations of such results, different stakeholders may also value certain unquantifiable aspects differently, such as social, esthetical and cultural values associated with different water environments, or with the mine waste sites. Moreover, there may exist stakeholder interests that are not at all related to the water environment, such as political interest pressures, or political popularity aspects, and associated stakeholder interactions in the independent SIA, which may highly affect a final action or non-action decision for the long-term mine water management of tailings and waste rock. Nevertheless, even if the decisions have been made due to other considerations than environmental water management, the scientific components, EIS, dynamic characterisation and optimisation proposed here as key parts of the general decision framework (Figure 12) will always have an important role to play, at least as important mechanisms for making decisions, their future effects, and the future monitoring of these effects. It may be useful and necessary to evaluate all stakeholder preferences, costs and uncertainties with multicriteria methods (e.g. Lahdelma et al. 2000) and both the WFD and the European Union Directive on Public Access to Environmental Information appear to

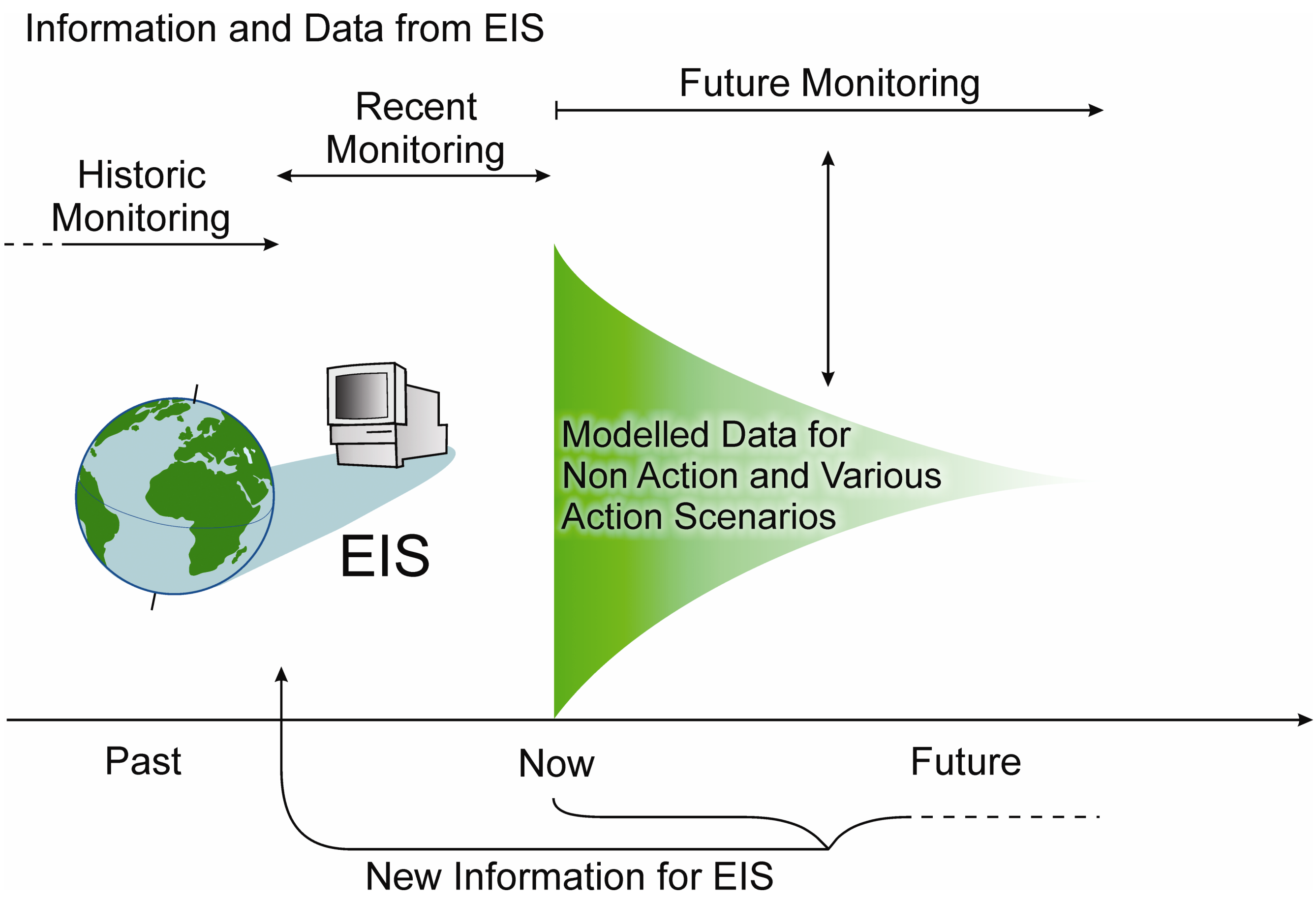

Figure 14. Schematic illustration of dynamic characterisation of the water pollution problem, using the Environmental Information System, consisting of the entire set of available (to some or all stakeholders) relevant information about this problem, including recent and historic monitoring data, as well other types of information and information systems, such as basic geomorphologic, hydrologic and hydrogeologic data, socioeconomic data on water, land and geologic uses and potential pollution sources, modelling tools for data and information interpretation and extrapolation, and technological feasibility and cost data

require transparency and quality control in the decision-making, handling and general accessibility of water and other environmental management, data and information, in ways consistent with the proposed decision framework.

5.2 Recommendations

We recommend that future planning and decision-making for the long-term fate of mine water pollution (from mine voids and mine waste deposits) be handled as an integral part of general catchment scale water management at different relevant levels (local, regional, national, international), considering that different relevant catchments can be defined on different scales for different water environments. We further propose a basic methodological decision framework for integrating the management of polluted mine water from tailings and waste rock with general water management. Main characteristics of this framework are that:

- it is generally applicable to the handling of mine water pollution abatement problems within different types of water management tasks, including the formulation of water management plans and action programmes, as well as the evaluation of environmental permit applications for individual mining/mine waste disposal projects,
- it integrates different types of quantitative analysis tools (environmental information system, dynamic characterisation, economic optimisation) and clarifies their use at different decision stages by competent authorities and other stakeholders,
- it integrates quantitative analysis tools and steps with qualitative stakeholder interaction, clarifying at which decision steps and with which quantitative scientific basis, the different stakeholders come together for common discussions, negotiations, and agreements.

On the basis of the present analysis and proposed water management framework, combined with the general and specific conclusions obtained from the complementary economic analysis study of Appendix III, we recommend, as a scientifically sound basis for negotiating action choices, taking decisions, and evaluating their effects, that:

- allocation of mine water pollution abatement measures be quantified on a catchment and/or water influence zone perspective and optimised based on economic efficiency at the catchment scale,
- expected long-term temporal changes in water pollution be considered by quantitative dynamic analysis of different possible mine waste remediation and/or mine water pollution abatement measure scenarios,
- cost scenarios for mine water pollution abatement be estimated with respect to different possible compliance boundaries, different possible water quality measures for judging compliance with different possible water quality targets, and different possible acceptance levels for risk/probability of not achieving targeted water quality improvements.

Efficient and generally accessible environmental information systems (EIS) are developed that couple mine waste/mine water with general water management information and enable provision of timely and relevant information of urgent policy need. In general, transparent and quality controlled data, information and results on the long-term dynamics of water systems and on the particular effects of mine waste/water on these systems in mining areas are indispensable for rational and cost-effective mine waste/and general water management.

Appendix I Mine Water Monitoring

AI.1 Introduction

The aim of this appendix is to briefly explain how to obtain reliable measurements of mine water flows and hydrochemistry, which are a prerequisite both for impact assessment and for the design of mine water remediation facilities. First, the monitoring of flows is discussed, with due emphasis on the use of tracer tests in mine water environments. Second, an overview of those hydrochemical measurements that must be made on-site is given, together with recommendations on sampling of waters for subsequent laboratory analysis. Finally some procedures for data manipulation and presentation are suggested. All of the techniques described in this appendix have been developed from more generalised approaches used for monitoring surface or groundwaters. Nevertheless, there are a number of pitfalls associated with applying these techniques to typical mine waters, as this presentation makes clear.

The importance of collecting synchronous measurements of flow and water quality can never be overemphasised (e.g. PIRAMID Consortium 2003). This is because both ecological impact assessments *and* the design of water treatment facilities are most usefully based on contaminant *loadings* rather than simple concentrations. Loadings are calculated by multiplying the flow ($L \cdot s^{-1}$) by the concentration ($g \cdot L^{-1}$):

$$\begin{aligned} &\text{Loading } [g \cdot s^{-1}] = \\ &\text{Concentration } [g \cdot L^{-1}] \cdot \text{Flow } [L \cdot s^{-1}] \end{aligned} \tag{3}$$

Cursory consideration of the above expression leads to the important conclusion that low concentrations of pollutants at high flow conditions can be just as damaging ecologically as high concentrations under low flow conditions. This fact is often overlooked by inexperienced scientists. Where diligent management of catchments is to be achieved, there is no excuse for such an oversight.

AI.2 Monitoring of Flows

AI.2.1 Using Existing Data

Flow measurement is carried out by public authorities in many states. Before investing time and effort in the collection of more flow data, it is always prudent to assess all relevant pre-existing data, to ensure that any further data-gathering complements rather than duplicates data already being collected. There are two principal uses of existing flow data:

- Where the flow measurement ("gauging") stations used to gather these data happen to be located at a point of direct interest in the river system, the data can be used directly for various forms of analysis useful in assessing impacts and designing remedial measures. However, given that long-term gauging stations tend to have been designed with quantification of water resources and/or flood warning in mind, they are rarely located in ideal locations from the point of view of mine water evaluation.
- The existence of a long, continuous, accurately measured hydrograph (i.e. a record of how flows have changed over time) for one or more gauged points in a catchment provides a foundation for the application of the techniques of "synthetic hydrology", in which hydrographs for other points in the catchment (or even in adjoining catchments) are estimated using various mathematical "transfer functions". Application of these techniques is only valid where the local vagaries of topography, geology, geomorphology and/or meteorology have been taken into account when developing a bespoke transfer function. Rainfall-runoff modelling can also be used to generate data for ungauged sites, provided satisfactory calibration against flows at the long-established gauging station is obtained. Full details of such techniques are beyond the scope of the present document; the interested reader is referred to texts such as those of Linsley et al. (1988) and Shaw (1994).

In many cases, existing flow data will be useful in catchment scale studies, but insufficient for full characterisation of mine water impacts, which tend to be most marked in small first- to third-order tributaries. It is therefore important to evaluate the existing gauging network thoroughly and then to install new flow measurement infrastructure to characterise mine water discharges or localised flows in tributaries to provide data appropriate to the specific needs of the project in question. It is worth recalling, however, that once data collection at these new measurement points has been undertaken for a year or more, it may be possible to develop robust correlations between these data and the synchronous data collected at the long-established gauging stations, providing the basis for extrapolating the hydrographs for the newly-measured points to give equivalent multiyear synthetic hydrographs, which (notwithstanding the observation of various *caveats*) provide a better basis for identifying extremes of flow than the relatively short, new records themselves.

AI.2.2 Physical Flow Gauging Techniques

It is important to note that flow gauging must be undertaken for at least one full annual cycle at any given point before committing to an engineering design for mine water remediation. If the flow measurement has taken place in a wet year (or, better still, in a wet year following several earlier wet years) then the flow data may be used to design for flood conditions with confidence. However, where the initial year's worth of data has been collected in a dry or average period, data collection must continue longer, with provisional design estimates of flood flows being obtained using synthetic hydrological techniques, as mentioned above.

Two options are available for flow measurements: discontinuous (or "spot") measurements, and continuous (logging) techniques. In many cases, a series of regular (i.e. weekly, or at very least monthly) discontinuous flow measurements will suffice for purposes of impact assessment and design. However, where the flow regime is notably "flashy" (i.e. prone to large fluctuations in response to rainfall) discontinuous measurements will be insufficient, and continuous logging techniques ought to be used.

Whether flow measurement is continuous or discontinuous, a key principle is that, except in the case of very small flows that can be measured using a "bucket-and-stopwatch" approach, we can hardly ever measure *flow* directly, but must rather measure water levels, and convert these into equivalent flows using hydraulic formulae (e.g. Linsley et al. 1988). Continuous logging of flows actually entails continuous logging of water levels, whence flows are calculated back at the laboratory.

A number of common strategies for flow gauging are used in practice. For a recent detailed discussion of the various techniques and their merits, the reader is referred to the PIRAMID Consortium (2003), who point out that measurement of mine water flows is often hindered by problems of ochre accretion on hydraulic structures and electronic sensors. Hence, although the following list of possible measurement techniques is the same as for any other natural water, special attention must be given to cleaning and maintenance of mine water gauging facilities.

Applicable flow measurement strategies for mine waters include the following (for further details, see Younger et al. 2002 and the PIRAMID Consortium 2003):

- Bucket and stopwatch (for modest flows, up to about 400 L min^{-1}).
- The velocity-area method, in which a current meter (impeller, heat-pulse or ultra sonic type) is used to determine the average velocities in a number of notional subsections of the width of an open channel, and the total flow for the channel is obtained by summing the products of velocity and flow for each subsection.
- Hydraulic structures, which cause the flow to behave in such a manner that there is a predictable relationship between water level (stage) and flow rate (discharge). The scale of the suitable structure varies with the magnitude of the flow to be gauged, and can thus range from large weirs in main river channels to large flumes, V-notch weirs (Figure 15) or small H-flumes. Regular maintenance of such structures is needed to counteract ochre clogging. Water levels behind such hydraulic structures are typically measured using pressure transmitters, or else float and counterweight systems connected to optical shaft encoders. Electromagnetic and ultrasonic devices offer the potential for non-contact measurement of water levels (avoiding ochre clogging problems) though the resolution of such techniques is not always sufficient for accurate measurement of low and intermediate flows in open channels.
- Diversion of water through a pipe fitted with a conventional water meter (impeller type) or an alternative type of flow meter, such as doppler ultrasound loggers, venturi meters or electromagnetic "clamp" meters (noncontact devices clamped onto the outside of the pipe). Pipes are highly prone to diameter reduction due to ochre precipitation, and hence should be applied with caution to ferruginous mine waters.
- Where water is pumped, to accurately log pump run times, allowing these to be converted into equivalent flows using known pump ratings. While it is common practice to use the manufacturer's stated ratings for such purposes, actual pump ratings should be checked using independent methods of flow measurement, to take into account wear and tear and temperature effects (water viscosity decreases as temperature rises).
- Salt tracer method (dilution gauging) using the increase of conductivity in a surface stream after injecting a given amount of sodium chloride to the stream.

The best option to measure flow in any particular circumstance depends on the availability of personnel, equipment, and on water quality. The problems of ochre precipitation have already been mentioned: where mine waters are very acidic, corrosion of pipes and other metal fittings can occur very rapidly (with fittings becoming unusable within one to two weeks in extreme cases), favouring the use of nonstandard timber and plastic-based structures, which may need to be custom-made. At the other extreme, very hard mine waters (containing much Ca and/or Mg) can give rise to clogging problems due to precipitation of carbonate minerals.

Figure 15. Example of a V-notch weir measuring the overflow from the Blaydon Hazard Shaft, an abandoned coal mine near Newcastle upon Tyne, UK

AI.2.3 Mine Water Tracing

Supplementing physical flow gauging methods outlined above, tracer tests offer a means of deducing hydrological flow paths (especially hidden, subsurface flow paths) between one point and another. Indeed, where there are no flow gauging facilities, tracer tests can be used to estimate flows (so-called "dilution gauging"; see Shaw 1994), perhaps as the first step in the design of a suitable gauging structure. However, this application of tracer testing can only be used to directly quantify flows where a known flow path exists between two neighbouring monitoring points on the same water course. In complex mined systems, these flow paths are not actually known, and hence tracer testing is used not so much to determine the magnitude of flows, but rather to detect flow pathways. Despite their power, tracer tests have not been widely used in mine water hydrology hitherto, most likely because their application requires considerable expertise, and the uncertainties of tested systems can lead to apparently unsatisfactory results. Nevertheless, numerous recent applications (see Wolkersdorfer 2002) have demonstrated the utility of mine water tracer tests in the process of designing remediation measures. Although tracer tests are site-specific in nature, they can contribute significantly to the overall development of catchment scale management measures.

Tracer tests are most commonly applied in relation to mine water systems to identify surface-subsurface flow pathways, and as such they have been used for the following purposes:

- in the development of precautions against dangerous inrushes to working mines,
- in the optimisation of extractive strategies (minimisation of water makes),
- to evaluate the feasibility of backfill/other underground waste disposal proposals,
- as part of mine subsidence investigations, and
- in the design of treatment or other remediation strategies for polluted mine waters.

For all of the above applications of tracer testing, it is essential to have some quantitative information on the hydrology of the mined system (at least in terms of bulk outflow volumes) before designing the tracer test. This is because injection and sampling sites can only be rationally selected where the boundary conditions of the mine water flow system are known.

In all cases, the background level of the tracers in the waters to be traced must be evaluated before the start of the test. For this reason, artificial tracers are often preferable to naturally occurring chemicals. Of the range of artificial tracers used in hydrological research, the best tracers yet applied to *mine waters* are microspheres (Wolkersdorfer 2002). Most other artificial tracers are either not stable in acidic mine waters, are strongly sorbed by ochre (e.g. Na-fluorescein), are prohibitively expensive in large quantities, and/or their use is legally restricted (radioactive tracers). Besides artificial tracers, a number of natural tracers have been used successfully in mined systems, including algal/bacterial spores (e.g. *Lycopodium*), bromide, various stable isotopes, and some rare earth elements.

Before the tracer test starts, at least two months (and sometimes as much as four months) of work are necessary, to investigate the hydrogeological situation, to get the necessary permits, and to conduct laboratory experiments with the potential tracers. Successful tracer tests in mines typically need between two and eight weeks of field work to ensure recovery of as much of the tracer as possible. After the tracer test, a further month or two must be allowed for processing of samples and presentation of results. Thus, a well-designed tracer investigation in a mined system can typically require a total elapsed time period of four to eight months. Where such a timescale is not feasible, it is perhaps better not to consider tracer testing.

AI.3 Hydrochemical Monitoring

AI.3.1 On-site Measurements

Hydrochemical methods are key to characterising the quality of mine waters and impacted water bodies, providing the key data for use in conjunction with flow rates in the calculation of contaminant loadings. There are various parameters that can and should be measured in the field, most notably pH, temperature, and alkalinity. Where the information is needed, it is also advisable to measure redox potential (also known as Eh or ORP) in the field. Where the necessary equipment is available, it is also advantageous to measure dissolved oxygen on-site, as this saves the use of expensive preservatives and the haulage of a separate bottle for each measurement.

Finally, although electrical conductivity can normally be as reliably measured in the lab as in the field, it is an extremely useful parameter to know on-site (as it distinguishes between waters with different degrees of mineralization, which can help guide opportunistic sampling) and it is easily measured with robust, portable meters. We therefore recommend conductivity as a routine on-site measurement.

In terms of chemical determinands, it is now possible to buy field test kits for many parameters (e.g. Fe, Mn, and dissolved Si) and they may prove useful in the reconnaissance phase of a study in providing approximate estimates of contaminant concentrations. However, their accuracy is normally such that they can not replace full laboratory analysis.

AI.3.2 Sampling of Waters for later Laboratory Analysis

When sampling mine waters at surface outflow points, it is relatively easy to decide where and how to fill sample bottles. However, when sampling underground mine waters via shafts or boreholes, it is important to remember that mine water quality does not only change in time, but also with depth (see Nuttall and Younger 2004; Rüterkamp 2001; Wolkersdorfer 1996). Many cases are known in which mine water is/has been strongly stratified. Where this has been overlooked, treatment systems have been under-designed. It is also important to note that stratification can change significantly over very short time periods – even hours – when previously quiescent mine water systems are suddenly disturbed by pumping or natural overflow.

Having determined which points to sample, water should be collected in polyethylene bottles, with at least two bottles per sample, as follows:

- One bottle acidified with a few drops of concentrated acid (two to three drops of ultrapure acid for every 50mL of sample) to depress the pH to 2 or less, thus preserving cations in solution, i.e. preventing metals from precipitating during storage and transit. While most standard sampling procedures recommend that acidification of sample be achieved using nitric acid, the reactivity of nitrate with ferrous iron precludes this as an option where direct determination of dissolved ferrous and ferric iron is to be carried out in the laboratory. In such cases, reagent-grade hydrochloric acid should be used instead.
- A second unacidified sample for anion and main cation analysis.

A safe choice for sample bottle size is 500mL; however, given the weight of full bottles of water, it is well worth checking with the laboratory to ascertain whether they are happy to work with smaller samples (the best laboratories can cope with 60mL samples, which makes for far less strenuous fieldwork).

If total and dissolved metal concentrations are required, *two* acidified cation samples should be taken, with one of these being filtered in the field. Field filtration can be achieved using syringe-based systems (for small samples) or using vacuum pumps and large filter housings for larger samples. While most protocols for water sampling recommend filtration through a 0.45µm filter, it is normally preferable to use a 0.2µm filter. There are a number of considerations in the selection of appropriate filter apertures, most notably:

- Water contains particles, colloids, and solute constituents. Usually colloids are in the range from 0.001—1µm and thus solute constituents have a diameter of less than 0.001µm. To separate solute constituents from non-solutes, a filtration with 0.001µm pore size is required. Since this is very difficult in the field, 0.2µm pore size is an operational compromise, being at least preferable to 0.45µm. However, if Al is to be determined, filtration to 0.2µm is strongly recommended, since Al tends to form colloids that are smaller than 0.45µm in size.
- Water contains bacteria with an average size from 0.2µm to 6µm and by using 0.2µm pore size filters, the sample will be sterile (with the exception of viruses), so that microbiological reactions are less likely to occur in the filtered samples.

It should be noted that filtration is not a panacea: for instance, filtration to 0.2µm will change the concentrations of gases in the water (particularly that of carbon dioxide), thus affecting the distribution of inorganic carbon species. Because of these complications, filtration ought not to be applied unthinkingly, but should be applied judiciously, with due consideration of the aims of the sampling exercise.

For purposes of quality control, it is essential that, for every sampling point, the location, date and time of sampling be recorded, as well as the names of the personnel who took the samples. Similarly, the brands of all meters, filters, or other equipment used must be recorded, to ensure that potential mistakes can be traced back to the source.

AI.3.3 Analytical Suites and Laboratory Analysis Techniques

In an ideal world with unlimited funding, specification of determinant suites could be simplified to specifying analysis of the entire periodic table and all known man-made compounds. In reality, limited time and money, and a robust appreciation of which determinands are likely to be present above detection limits, encourage the specification of more limited suites of determinands, tailored to the end-use. Table 4 summarises a range of recommended determinant suites for mine water investigations of various types.

Table 4 lists a number of site-specific additional determinands, such as the more "exotic" metals (e.g. Cu, Cd, Ni, Co) and the metalloid arsenic (As), which are by no means ubiquitous in mine waters, but may be important contaminants in certain cases (usually abandoned metal mines in carbonate-poor country rock, or coalfields with peculiar secondary mineralisation, e.g. the occurrence of millerite, NiS as an accessory mineral in the South Wales coalfield). Others such as cyanide (CN^-), which is used extensively in gold beneficiation, are of anthropogenic origin, being associated with mineral processing operations. In old gold mining areas, and in areas where gold mining is currently being undertaken by the informal sector, mercury (Hg) could be added to the list for the same reasons. The radioactive elements (U, Ra, Rn) are most common in uranium mining settings, though can also be associated with gangue minerals in certain metalliferous and even coal mining situations. The "user defined" category can cover a range of other metals (e.g. Cr, Pb) which in some settings are occasionally present in mine waters at elevated concentrations, and organic pollutants. The

Table 4. Recommended analytical suites for various types of mine water study; [1]: items in bold are to be determined *on-site*; for items underlined, it is often worth analytically determining individual species, as well as simple total concentrations; [2]: depending on mining geology/known or suspected site history

Determinant[1]	Type of mine water study:			
	Reconnaissance survey	Geochemical investigation	Routine data for design purposes	Site-specific[2] additional determinands
H	✓	✓	✓	
conductivity	✓	✓	✓	
temperature	✓	✓	✓	
alkalinity (and thus HCO_3^-)		✓	✓	
dissolved oxygen		✓		✓
Eh		✓		
Ca		✓		
Mg		✓		
Na		✓		
K		✓		
Li				✓
<u>Fe</u>	✓	✓	✓	
Mn		✓	✓	
Al		✓	✓	
<u>Cu</u>				✓
Zn		✓		✓
Cd				✓
Ni				✓
Co				✓
<u>As</u>				✓
Cl		✓		
<u>CN^-</u>				✓
SO_4	✓	✓	✓	
NO_3^-				✓
NH_4^+		✓		✓
U				✓
Ra				✓
Rn				✓
organic carbon (TOC/DOC)				✓
user defined	✓			✓

nitrogen species are normally rare in mine waters, but are sometimes introduced through the use explosives and from NO_3^- leaching from overlying agricultural soils, and NH_4^+ does occur naturally in some of the more saline mine waters.

It should be noted that acidity, which is a key characteristic of many mine waters, is not listed as an analytical determinant in Table 4. In line with the PIRAMID Consortium (2003) we recommend calculation of acidity from the concentrations of key hydroxide-forming metals and pH, as follows (from Hedin et al. 1994):

$$\begin{aligned}&\text{Acidity}_{\text{calc}}\,[\text{mg}\cdot\text{L}^{-1}\text{ as CaCO}_3] = \\ &50\cdot[2\,\text{Fe}^{2+}/56 + 3\text{Fe}^{3+}/56 + 2\,\text{Mn}/55 + \\ &3\,\text{Al}/27 + (1000\cdot 10^{-\text{pH}})\end{aligned} \tag{4}$$

where the relevant metal concentrations are in $mg \cdot L^{-1}$. Other metals can be added into the equation if they are present in elevated concentrations, using their valency and atomic masses respectively as numerator and denominator factors in the terms inserted for that metal. In cases where alkalinity and acidity are determined in the field for calculating the carbon species distribution, it is highly recommended to use pH-meters instead of a colour indicators.

While mine waters generally contain negligible quantities of organic pollutants, there are a number of situations in that they *can* occur, such as where electrical transformers and switch-gear have been left in mine workings that later flooded [in which case, polychlorinated biphenyls (PCBs) and other contaminants may be detected in the mine waters], and where mine waters have flooded previously burning zones of coal waste [in which case, polycyclic aromatic hydrocarbons (PAH) may be present at significant concentrations]. Where these and other possible sources of organic contaminants exist, occasional analyses of the collective parameter TOC (total organic carbon) can be made to check whether

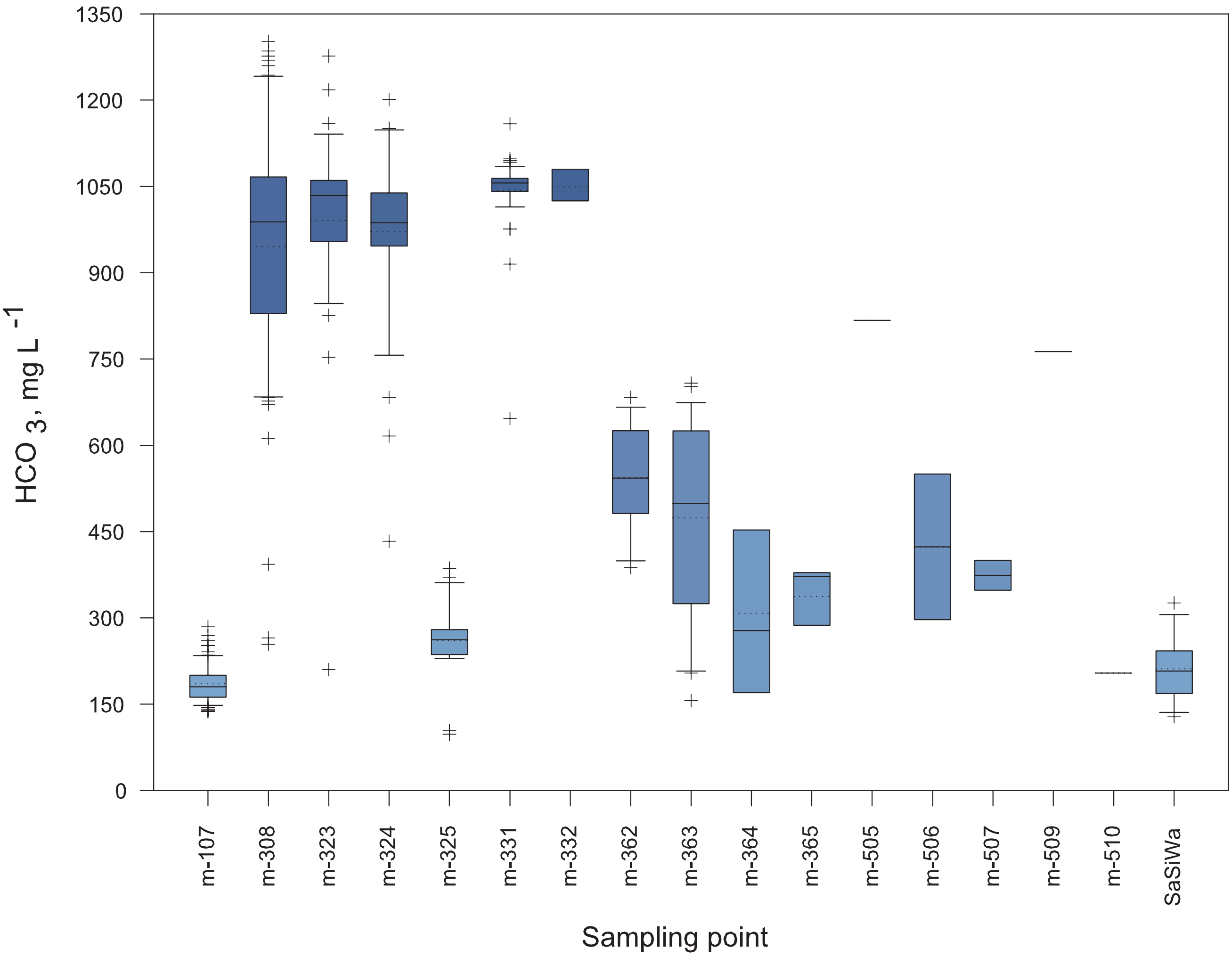

Figure 16. Example of a standard box-plot showing 312 data points at 17 mine water sampling points (Wolkersdorfer 1996)

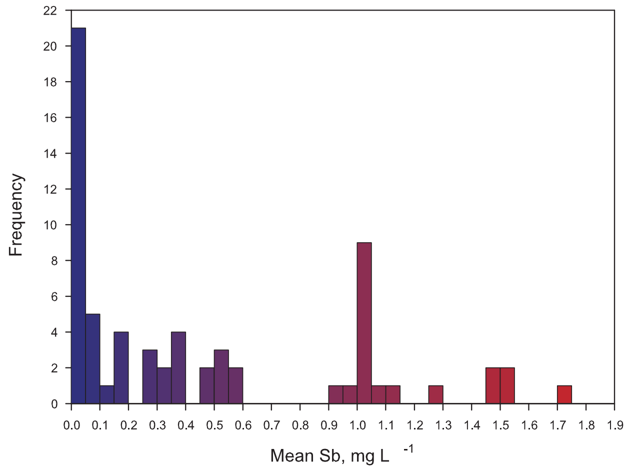

Figure 17. Example of a histogram, showing the frequency distribution of Sb and the clustering of the data

the organic fraction of a given water is significant, followed up by compound-specific analyses, as appropriate.

Laboratory analysis for metal cations is often carried out using atomic absorption spectrophotometry (AAS; with a graphite furnace attachment where low detection limits are needed for metals such as Al), or by inductively coupled plasma mass spectrometry (ICP-MS), the latter being particularly appropriate where sample throughputs are high. ICP-MS also offers the advantage of "screening" for a wide range of metals, perhaps helping to further refine the determinant suite for a particular mine water. Anions (e.g. SO_4^{2-}, Cl^- and NO_3^-) are usually analysed by ion chromatography, or by various colorimetric, turbidimetric, and/or gravimetric techniques. Whichever approach is used, the credibility of the data will be enhanced where it can be shown that it has been obtained in a manner consistent with internationally accepted analytical protocols, such as those of APHA (1998; see also Section AI.4).

AI.3.4 Presentation of Monitoring Results

Once all the data have been gathered, it is possible to present the data in various ways. For flow data, a simple graph of flow rate versus time, known as a hydrograph, is the most common form of presentation. For design purposes, it is often more useful to present flow data in other formats, such as flow duration curves (especially for defining low flow conditions), POT ("peaks over threshold") plots, and extreme-value plots for definition of flood flow magnitudes and probabilities of recurrence. The application of these hydrological presentation/interpretation techniques is beyond the scope of this document. The interested reader is referred to standards hydrology texts (e.g. Linsley et al. 1988; Shaw 1994).

With regard to hydrochemical data, extensive guidance on standard presentational techniques and their interpretative use are provided by Lloyd and Heathcote (1985) and Hounslow (1995), amongst others. Some of the less-specialised techniques are readily applicable to mine water data sets. For instance, the range of a single parameter is well represented using standard box-plots, which display the mean, median, 5, 10, 90, and 95% quantiles, and outliers in a transparent way (Figure 16).

Simple histograms also have great utility for revealing data clustering (e.g. Figure 17). However, more specialised hydrochemical plotting techniques, which are based on a presumed suite of major cations and anions (especially Piper and Durov diagrams) are less useful in mine water applications, in which Fe (or even Al) may be present at higher concentrations than Ca, Mg, Na, and K. In any case, in most mine waters, the principal applied interest is in the alkalinity-acidity balance. Bearing this in mind, Younger (1995) originated a presentational approach which is more appropriate than standard Piper or Durov plots in classifying mine waters (Figure 18). In this diagram the alkalinity-acidity balance defines one axis of a 2-D plotting field, while the other axis uses a general indicator of mine water provenance, manifested in the balance between SO_4^{2-} (generally representing the degree of pyrite oxidation) and Cl^- (elevated concentrations generally indicating ancient sedimentary waters). This diagrammatic format has been recently improved by Rees et al. (2002), by plotting the alkalinity/acidity balance in terms of the overall "net alkalinity", which equals the total alkalinity minus the total acidity (Figure 18).

AI.4 Quality Assurance/Quality Control

Design of remediation measures presupposes that the conceptual approach is founded upon high quality data. Accordingly, quality assurance (QA) and quality control (QC) are central to ensuring that monitoring data are actually useful. The distinction between QA and QC is that QA covers the "rule book" for the analytical programme as a whole, whereas QC relates to measures taken at the level of individual lab activities to be able to demonstrate compliance with QA. Most laboratories have their own QA/QC plans, though in many cases it will be necessary to extend these to cover field work (flow measurement, on-site analysis, and sampling). Some pointers on appropriate QA/QC measures for mine water monitoring are given below.

Documentary evidence is paramount in QA. Hence, all sampling and monitoring procedures must be precisely noted in field or laboratory books. These books must have properly bound sheets (adhesive binding should be avoided), preferably with numbered pages, and with no pages missing. Within the book,

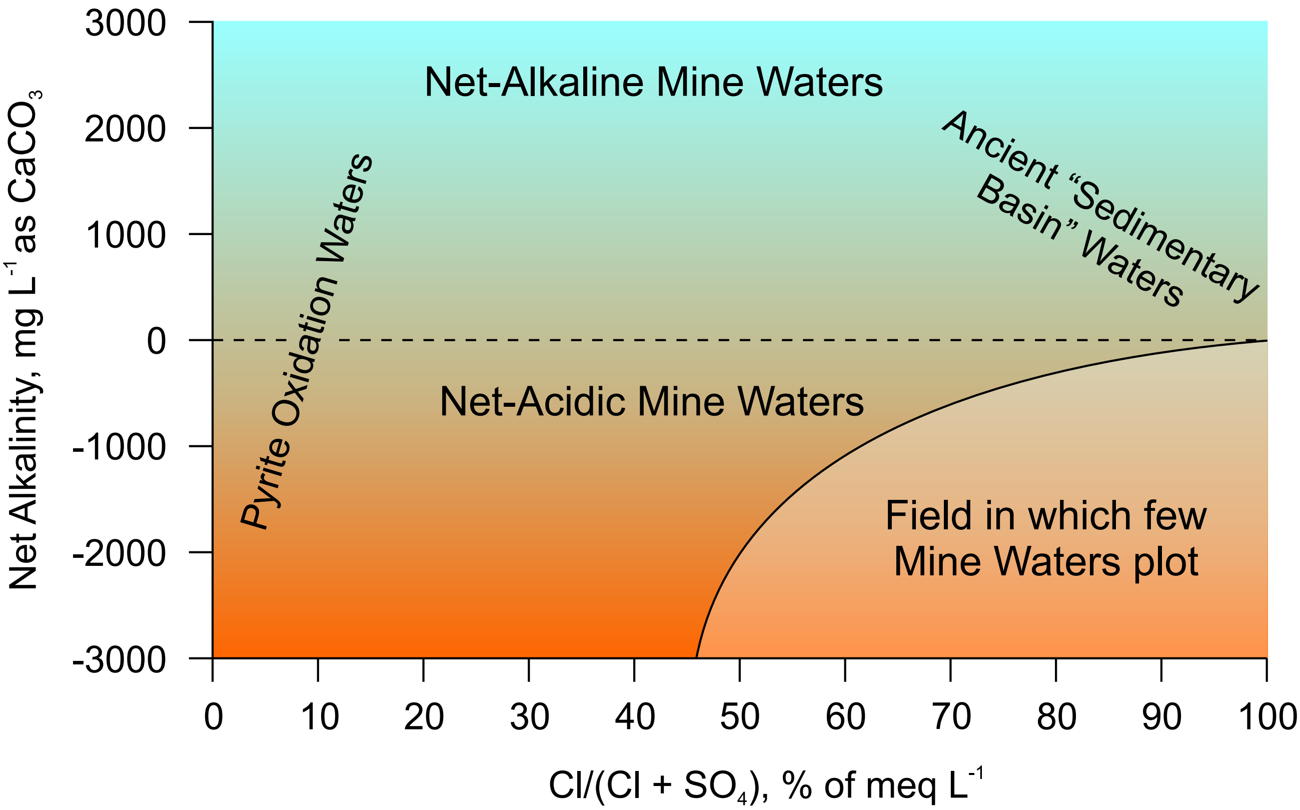

Figure 18. Mine water classification diagram in terms of net alkalinity and proportions of chloride to sulphate, following the scheme of Younger (1995) adapted in accordance with the proposals of Rees et al. (2002)

only field or laboratory observations and numbers should be written. Where a calculation is conducted, this part should clearly be noted as SC ("side calculation"). In no case is erasure allowed and no parts must be made unrecognisable. Wrong data should be "crossed" out with one single line, such that the wrong number can still be read. This guarantees that the potential reason for the faulty data can be reconstructed later (e.g. 7.45 instead of 7.54 or 7°C instead of 7°F). Besides the measured parameters, "soft" parameters have to be written down, too. These include an array of potentially relevant observations such as unusually high or low turbidity, ice covers, evidence of vandalism, presence of farm animals, or any other observation that might be relevant for co-workers.

Sampling points and samples must be numbered precisely throughout the whole project. It is advisable that all researchers and authorities involved agree to the same numbering system. Each sample must be labelled with the project's name, sampling site, sampling date, responsible research institution, and person who took the sample. Sample numbers containing a three letter project code, the date, and the sampling site provide a means to exclude mistakes during sample preparation (e.g. FDR-1202-MP1: Project code – short date – sampling point).

Chemicals or standard solutions used for equipment calibration or analytical procedures must be fresh, and the lifetime of the chemicals must be checked regularly. From time to time, blanks should be included in the analytical process to check for contamination or overlapping. Field equipment can be controlled by measuring standard solutions from time to time during the day.

Appendix II Mine Water Remediation Technologies

AII.1 Introduction

In general, there are three principal options for mine water remediation:

- 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.
- Prevention/minimisation of pollutant release processes.
- Mine water treatment, by either active (e.g. chemical dosing) or passive (e.g. wetlands) means.

Of these three options, the latter two are not mutually exclusive and may often best be applied in consort. We shall now consider each of these in outline.

AII.2 Monitored Natural Attenuation

Natural attenuation will often be the most "sustainable" remedial option overall, especially in the long-term, and particularly for the less heavily polluted mine water discharges. Selection of this option will be based on two key factors: the availability of dilution in the receiving watercourse, and 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 (Appendix I), 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, 1998a, 2000a). When workings are first left to flood, the water quality generally deteriorates markedly in comparison to that encountered during active mining and dewatering. However, in the long-term, significant ongoing pollutant release can only occur in strata that are permanently or temporarily (e.g. in summer and autumn only) above the water table in the flooded mine workings.

The applicability of monitored natural attenuation over extended periods of time requires site-specific evaluation of:

- the relative abundances of sulphides, carbonates, and silicate minerals within the mined systems and receiving catchment,
- pollutant release rates,
- rates of mixing processes, and
- the rates of geochemical attenuation reactions.

Guidance on the evaluation of these factors is beyond the scope of this document, but is extensively discussed by Younger et al. (2002).

AII.3 Prevention/Minimisation of Pollutant Release Processes

Strategies aimed at minimising the release of contaminants to mine waters must focus on limiting one or more of the following pre-conditions for the oxidation of sulphide minerals:

- the presence of oxygen,
- the availability of moisture,
- 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 0.5m or more of clay, compacted such that it retains a permeability no greater than $1.16 \cdot 10^{-9} m\ s^{-1}$), 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 2000b 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 of a single application rarely last longer than six months. Slow-release formulations are available, and have been successfully applied to highly reactive material, just before soil capping and revegetation. However, this is an expensive option, and has had limited application. This approach is not applicable to very large mine waste depositories nor underground mine voids.

AII.4 Mine Water Treatment Technologies

These have been extensively documented, together with detailed guidelines for their design and implementation (Younger et al. 2002, Brown 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:

- **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.
- **Dosing with alkali** (usually hydrated lime [$Ca(OH)_2$], 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.
- **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 recirculation 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 recirculation ("Rowdensity sludge").

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, applying conventional desalination technology (i.e. flash distillation and reverse osmosis) to mine waters may prove worthwhile. 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 turnkey operations, and are particularly suitable where very low metals 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. Zn/Cu). 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 $mg{\cdot}L^{-1}$ by equilibration of the water with respect to the Al sulphate mineral ettringite ($3\ CaO{\cdot}3\ CaSO_4{\cdot}Al_2O_3{\cdot}31\ H_2O$), 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 reuse of metals is an economic possibility. These approaches include:

- sorption and ion exchange processes
- solvent extraction
- electrochemical extraction
- biochemical extraction techniques
- the barium sulphide process (in which sulphate is removed from mine water by precipitation of $BaSO_4$)
- 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 refers to "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" (PIRAMID Consortium 2003). The working definition of "infrequent" in this context is currently around six months. Types of passive system currently in use include:

- aerobic, surface flow wetlands (reed-beds)
- compost wetlands with significant surface flow
- mixed compost/limestone systems, with predominantly subsurface flow [so-called Reducing and Alkalinity Producing Systems (RAPS)]
- subsurface reactive barriers treating acidic, metalliferous groundwaters
- closed system limestone dissolution systems for zinc removal from alkaline waters

- 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 2000a).

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 ecosystems 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:

- physical limitations on the areas available for treatment
- the frequent insistence of regulators that treatment wetlands be surrounded by flood defence bunds, which preclude two way exchanges of water, solutes, sediments and plankton with adjoining rivers
- engineering limitations, such as the need to allow freeboard at the perimeters of systems and the need to prevent erosion by extreme flows.

AII.5 Designing a Remedial Strategy

In the light of the foregoing summaries of mine water remediation technologies, it is now appropriate to summarise the logic that ought to underlie the rational selection and implementation of remedial options. AII.1 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 19, 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 flowchart 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 that 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 2000b) and predicted for others on the basis of the rates of weathering of sulphides and carbonates

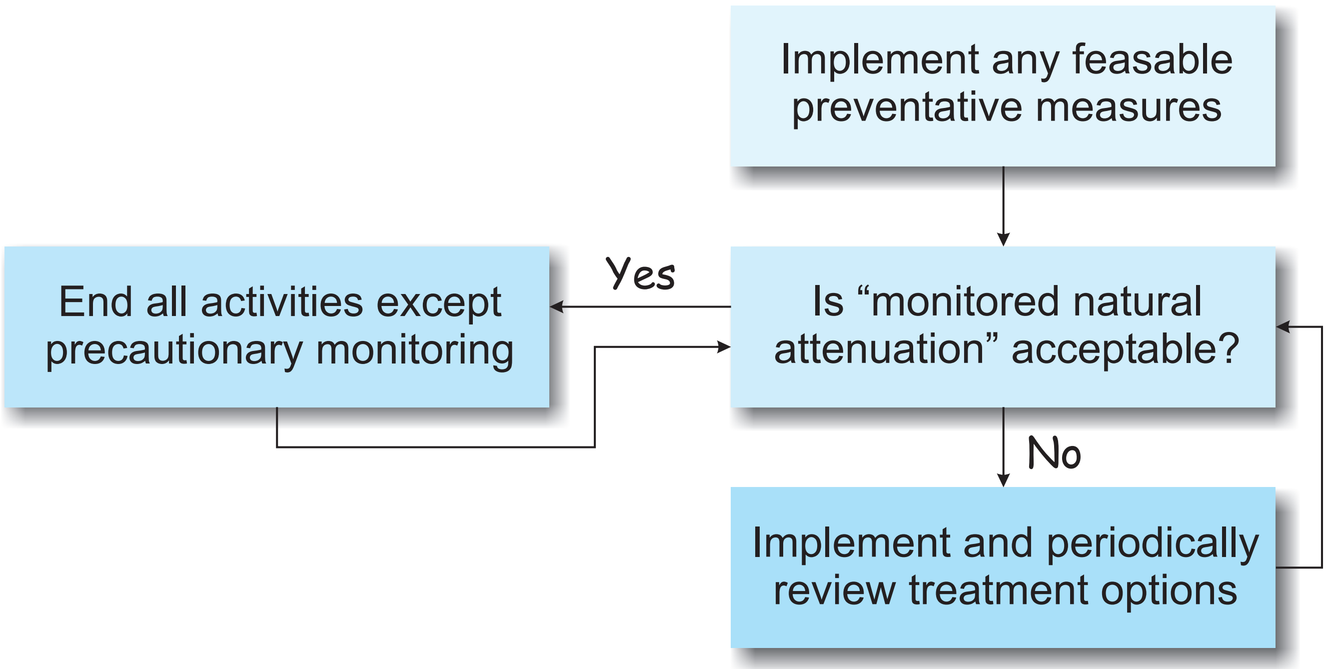

Figure 19. Flowchart giving the basic decision-making logic for the selection of the most appropriate remedial option for a polluting mine water discharge (after Younger 2002a). Note that this flowchart proposes no final endpoint for all activities, because precautionary monitoring (even if only by visual inspection) will nearly always be advisable

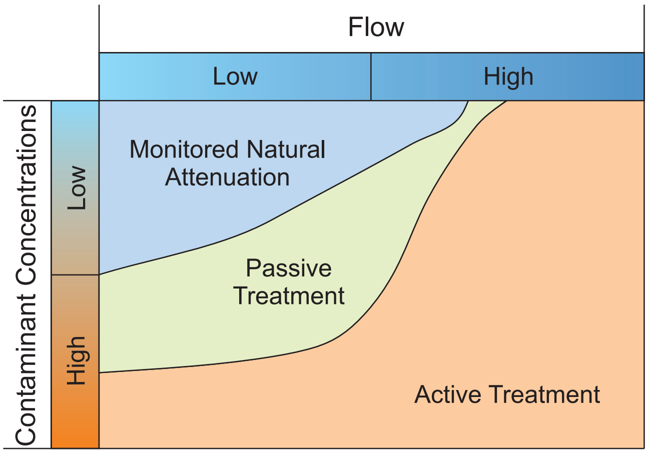

Figure 20. Suitability of monitored natural attenuation, passive treatment and active treatment options for remediation of polluted mine waters, expressed as functions of the magnitude of flow and intensity of contamination of mine water discharges (after Younger 2002a)

and their relative proportions in the leaching rock mass (e.g. Strömberg and Banwart 1994).

Figure 20 attempts to place the lower two thirds of Figure 19 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 20 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 (e.g. in terms of climate and runoff patterns), such that these categories will be reached at far lower absolute values of flow and concentration in a semiarid area than in a humid temperate area. For instance, under conditions typical of humid, temperate areas of western Europe, the split between low and high flows lies will lie at around $5L{\cdot}s^{-1}$, and very high flows are on the order of $200L{\cdot}s^{-1}$. 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 $50mg{\cdot}L^{-1}$ as $CaCO_3$ equivalent, for Fe at around 20 $mg{\cdot}L^{-1}$, and for Zn around 2 $mg{\cdot}L^{-1}$.

The key message of Figure 20 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 nothing less than intense active treatment is likely to suffice for such cases.

Appendix III Economic Analysis of Mine Water Pollution Abatement in a Catchment

AIII.1 Introduction

The water flowing through deposited mine wastes and abandoned mines are integral parts of the hydrological cycle in a catchment, and may become polluted, for instance, by sulphate and heavy metals. On its way downstream, through the variable water pathways of the catchment, such polluted mine water will then affect and may also pollute other water environments within the catchment (groundwater, streams, lakes), as well as the coastal and marine waters that are fed by the fresh water outflow from the catchment. Implemented and planned measures for water pollution abatement and remediation of contaminated land sites (e.g. mine waste sites) in a catchment are, in many countries, commonly based on environmental legislation that handles different pollution sources (such as different mine waste sites) uniformly. The new European Union Water Framework Directive (WFD), however, requires catchment scale tools for water quality management and decision-making that enable quantitative identification of economically efficient measure allocation for contaminated site remediation and water pollution abatement, which may be highly nonuniform among the different possible sources (such as different mine waste sites) of a certain water pollutant (e.g. a heavy metal) within a catchment.

In this appendix, which summarises the study and results reported in greater detail within the ERMITE report D5 (Baresel et al. 2003), we first discuss general economic decision rules for choosing mine waste site remediation and mine water pollution abatement measures within a catchment, such that given regulatory targets for water quality and/or pollutant loading are efficiently reached in chosen water recipients and water environments. In principle, there are two types of economic decision rules for water pollution abatement: choosing abatement that maximizes net benefits, or that minimizes costs for achieving prespecified (for example, by law or political decisions) water quality/pollutant load targets. These two economic rules for decision-making are consistent with an environmental regulation framework, such as the WFD, which focuses on water quality standards, for instance given in terms of maximum concentration levels (MCLs), or maximum pollutant loads (MPLs). A rational decision on where, when and which remediation and pollution abatement measures to apply within a catchment can then be based on identifying which measure or combination of measures can practically yield MCLs/MPLs at some compliance boundary (CB, associated with some water recipient) at minimum cost, or maximal net benefit.

In addition, this appendix exemplifies the application of such an economic decision rule to a particular case study. Specifically, we develop and apply a cost-minimization model for determining cost-effective abatement solutions within the Swedish Dal River catchment. This catchment is considered a 'hotspot' for water pollution by metals, caused by extensive historic and ongoing mining activities and mine waste deposition (HELCOM 1993). The water pollution of most general public interest is the metal leaching to the Dal River itself (Svensson 1988; Hartlén and Lundgren 1990; Lindeström 1999) and through the river to the Baltic Sea (HELCOM 1993). Also, the possible mine water pollution of local water bodies, in the near vicinity of mine waste sites, is of interest due to questions it raises in relation to local long-term sustainability and to the WFD (e.g. European Commission 2000 par. 13). In our case study, we consider Zn, Cu and Cd loads to different selected recipients (CBs), including the Dal River itself, and various abatement measure practices and designs that may be practically feasible for the (sub)catchments and mine sites associated with these recipients; investigated measures for water pollution abatement include soil and water covers at mine waste sources of metal emissions and wetland construction for metal load reduction close to the chosen CBs. The main aim of our investigation is to show how different metal load reduction levels at CBs can be achieved at minimum cost; the achievement of a certain load reduction then implies achievement of some related water quality standard, MCL or MPL, in the water recipient that is associated with a given CB.

AIII.2 Theoretical Basis

The application of the two different economic decision rules to the mine water pollution abatement problem on a catchment scale can be summarised as:

- maximisation of total net benefits of reducing the downstream water pollution by mine waters within the catchment, and
- minimisation of the total costs for achieving a prespecified water quality standard, such as MCL or MPL, at a given CB within the catchment.

Figure 21a illustrates the maximum benefits being achieved at the water quality improvement Q^* because any divergence from this improvement level would decrease net benefits (total benefits minus total costs), even if total benefits might then increase, or total costs decrease. The condition for identifying the optimal improvement point Q^* is that marginal benefits equal marginal costs. The cost efficiency rule is illustrated in Figure 21b, as a choice of abatement measure allocation that follows the third cost function, which yields a prespecified, targeted (for instance, given by law or political decision) water quality improvement (denoted Target) at minimum cost. The relation between water quality improvement levels Q^* and the Target may be such that Q^* is greater than the Target, if the benefits of increased water quality improvement are greater than the associated increase in abatement costs.

Economically efficient mine water pollution abatement is then defined from rule a), as the allocation of abatement measures within the catchment that maximises net benefits from pollutant reductions in one or several chosen water recipients, each associated with its own CB; reduced damages from metal loading into a recipient may, for instance, yield monetary benefits from improved recreational facilities and fishery (Gren et al. 2000b), which for a range of water quality improvements may exceed the associated abatement costs (Figure 21a). Needless to say, environmental improvements are, in practice, far from trivial to measure in monetary terms. Cost minimisation for reaching prespecified water quality targets, MCLs or MPLs, in chosen water recipients is then the alternative fruitful approach (Gren et al. 2000a, 2000b) to economic efficiency. In the cost minimisation approach, it is the water quality target (see Target in Figure 21b; given, for example, by political processes or by law) rather than the economically efficiency, but in many cases unquantifiable, water quality improvement (see Q^* in Figure 21a), which determines the required water quality improvement. The aim of a rational mine water pollution abatement decision is then to achieve this targeted water quality improvement at minimum cost. Following this section, we present and explain in a simple way the quantitative conditions for net benefit maximisation and cost minimisation of measures for abatement of mine water pollution from mine waste sites in a catchment. In our specific case study, presented in the subsequent section, we further develop and explain the cost minimisation approach and its specific model application to mine water pollution abatement at the Dal River catchment (see Section 3.3).

Monetary benefits of water quality improvement at some downstream recipient of mine water from different sources (e.g. from different mine waste sites) may be expressed as a function of average annual pollutant load decreases, $V(L'-L)$, with L' being the total level of unregulated, pre-abatement average annual pollutant load delivered by mine waters to a downstream CB, L being the achieved total post-abatement load at the same CB, and the function $V(L'-L)$ being decreasing and concave in load L, expressing that the more L is decreased, the smaller will the marginal net benefits be per additional unit load decrease (see benefit curve in Figure 21a). Also the cost of reducing pollutants to the water recipient must be accounted for, in both decision rules, including, for example, expenses for soil or water covering of mine waste deposits, or for construction of wetlands downstream of the sources, close or at the CB. A simple cost model may include $i = 1,\dots,n$ different mine waste sites, each associated with possible mine water abatement measures, such as covering the waste with soil or water, yielding a reduced average annual source emissions, L^i_0, or wetland construction that abates the resulting downstream average annual pollutant discharge, L^i, at the CB. Associated cost functions, $C^i_0(L^{i'}_0 - L^i_0)$ and $C^i_{CB}(L^{i'} - L^i)$, where $L^{i'}_0$ and

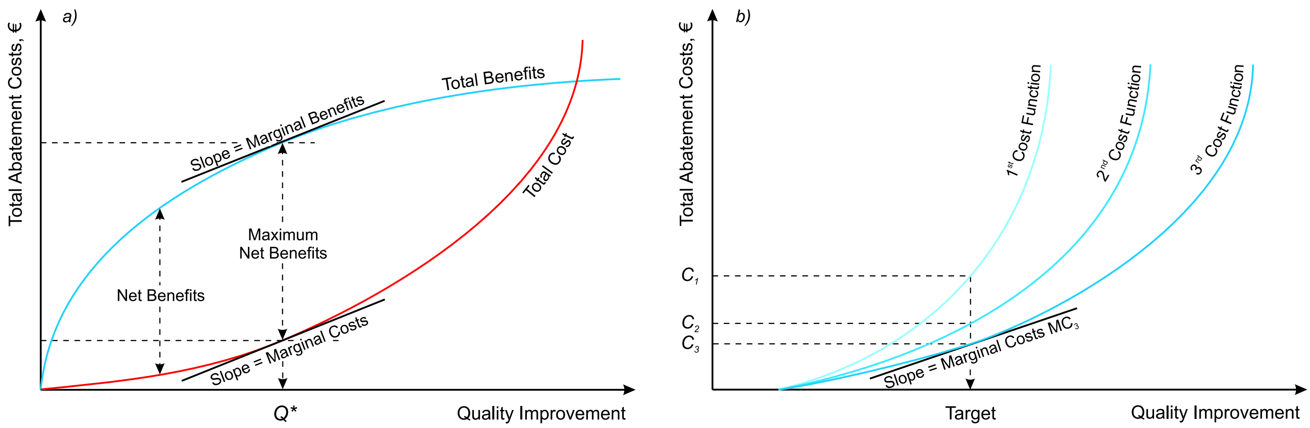

Figure 21. Schematic illustration of the two different economic decision rules: a) maximisation of total net benefits, i.e. total benefits minus total costs, which is achieved by choosing the abatement measure allocation that yields water quality improvement denoted as Q^* in the figure's abscissa; b) minimisation of the total costs of reaching a prespecified, targeted water quality improvement level (denoted as Target in the figure's abscissa), which is achieved by choosing the abatement measure allocation that yields this target following cost function 3

$L^{i'}$ are the unregulated pre-abatement emission level at the source and the resulting (also unregulated, pre-abatement) pollutant load at the CB, respectively. The cost functions may be assumed to be decreasing and convex in load, L, expressing that the more L is decreased, the greater the marginal costs will be for each additional unit load decrease (see cost functions in Figure 21).

The maximisation problem is then formulated as choosing the allocation of L^i_0 and L^i_{C0} that maximises net benefits, according to

$$Max\, V(L'-L) - \sum_{i=1}^{n}\left[C_0^i\left(L_0^{i'} - L_0^i\right) + C_{CB}^i\left(L_0^{i'} - L_0^i\right)\right] \quad (5)$$

where

$$L = \sum_{i=1}^{n} L^i = \sum_{i=1}^{n} \alpha^i L_0^i \quad (6)$$

and $0 \le \alpha^i \le 1$ are delivery coefficients, accounting for all pollutant retention, mass transfer and transformation processes on the pollutant's pathway from source to CB, which reduce the resulting average annual output load at the CB, L^i, relative to the corresponding source emission L^i_0. The corresponding cost minimisation problem is defined as

$$Min \sum_{i}^{n} C_0^i\left(L_o^{i'} - L_0^i\right) + C_{CB}^i\left(L^{i'} - L^i\right) \quad (7)$$

subject to

$$L = \sum_{i=1}^{n} \alpha^i L_0^i \le L^* \quad (8)$$

where L^* is a pre-specified maximum annual average load to the recipient through the CB; this maximum load may then correspond directly to a MPL, or to the load level required for achieving MCLs in the considered recipient.

The concavity and convexity assumptions of the benefit and cost functions, respectively, ensure that second-order conditions for an optimum are fulfilled and yield first-order conditions for maximum net benefits as

$$\frac{\partial C_0^i}{\partial L_0^i} = V_L \alpha^i, \frac{\partial C_{CB}^i}{\partial L^i} = V_L \quad (9)$$

with V_L being the monetary value of unit improvement in water quality. Equation (9) implies that mine water abatement for each source, with measures taken at the source itself, or at the CB, should be carried out as long as the increase in benefit per unit water quality improvement, V_L, exceeds the associated increase in abatement cost, expressed by terms $\partial C/\partial L$. The corresponding first-order condition for minimum cost is given by $\partial C_0^i / \partial L_0^i = \alpha^i \lambda$, $\partial C_{CB}^i / \partial L^i = \lambda$, with λ being the Lagrange multiplier for the target restriction (8), which may also be expressed as

$$\frac{\partial C_0^i / \partial L_o^i}{\alpha^i} = \partial C_{CB}^i / \partial L^i \quad (10)$$

The condition (10) states that, in the optimal abatement solution, the marginal costs of pollutant load reduction at the target CB should be equal for all abatement measures. Otherwise, if this condition were not fulfilled, it would be possible to reduce total cost for the same target, L^*, by reallocating abatement among measures, such that abatement at low cost is increased at some source or at the CB, and high cost abatement measures are decreased.

We emphasise here that for both decision rules (9) and (10), the delivery coefficient values, α^i, with $1\text{-}\alpha^i$, quantifying the natural pollutant attenuation along the different source to CB pathways within a catchment, are critical in determining the optimal solution for mine water abatement within the catchment. Source abatement costs are higher (equation 10) and benefits are lower (equation 9) for mine water sources emitting pollutants that are subject to low delivery coefficients α^i (α^i closer to zero and thus $1\text{-}\alpha^i$ closer to one, i.e. high natural attenuation), relative to sources with high α^i (α^i closer to one and thus $1\text{-}\alpha^i$ closer to zero, i.e. low natural attenuation), or to abatement measures at or close to the CB ($\alpha^i \approx 1$, $1\text{-}\alpha^i \approx 0$). For both decision rules, inefficiency would occur from erroneous assumptions about the delivery coefficients α^i, from the resulting in-optimal allocation of pollutant abatement among sources (such as water or soil covering of mine waste sites) and downstream measures close or at the CB (such as wetland construction). For instance, too much source abatement and too little downstream abatement may be chosen from erroneous α^i quantification, implying resulting inefficiency costs, the magnitude of which depends on the difference in marginal costs between abatement measures at the CB.

The net benefit maximisation rule requires, besides relevant estimates of delivery coefficient values, α^i, and abatement costs, information on the monetary value of the environmental benefits, associated with the water quality improvement (see, e.g. Gren 1995). The general difficulty of linking water pollutant abatement to, for instance, biological and health impacts measured in monetary terms (Gren et al. 1997) makes the cost-minimisation rule more fruitful for the yet relatively un-investigated problem of mine water pollution abatement on the catchment scale. Even the cost-minimisation rule is presently at a research stage with regard to this application problem and, in the following, we present and use a methodology for such application to the specific case study of the Dal River catchment in Sweden.

AIII.3 Cost-effective Mine Water Pollution Abatement in the Dal River Catchment

AIII.3.1 The Dal River Catchment

The Dal River catchment in the Swedish region Bergslagen has an area of 30,000km^2. While background levels of metal concentrations characterise the upper part of the catchment, where the river is divided into the West and East Dal Rivers, the central, industrialised area of the catchment is affected by mine water pollution, originating from Falun and Garpenberg, two major mining areas of historical and current importance (see Figure 22).

In Falun, mining started in the eighth century, producing many small and uncharacterized mine waste sites and possibly also mine voids that may all contribute metal leakage to the Dal River catchment. The city of Falun, with its preserved mining area, has been declared a Protected Area on the UN World Heritage List. Garpenberg also hosts numerous deposits of mining waste, many of which are protected for their historical value. Existing mine waste deposits around the mines consist mostly of heap and slag in lower and older deposits, and of waste rock and tailings in upper layers. All these mine water sources have influenced water quality, not only in the local recipients Runn and Gruvsjön, but also in the catchment's main watercourse, the Dal River (Lindeström 1999). The contribution of Zn and Cd from the Dal River to the Baltic Sea is estimated to be larger than for any other watercourse in Sweden (Svensson 1988). Therefore, the need of mine water pollution abatement is generally recognized for this catchment and various abatement measures are planned or have already been started.

AIII.3.2 Catchment Scale Cost Minimisation of Mine Water Pollution Abatement

We have carried out a cost minimisation analysis of the mine water pollution abatement in the Dal River catchment, along the conceptual lines described in the previous section. We considered different possible water recipients (and associated CBs), the Dal River on the regional level, and the lake Runn (subcatchment Falun) and the lake Gruvsjön (subcatchment Garpenberg; see Figure 22). Results for the cost-efficient measure allocation and associated costs for a variety of metal load reduction

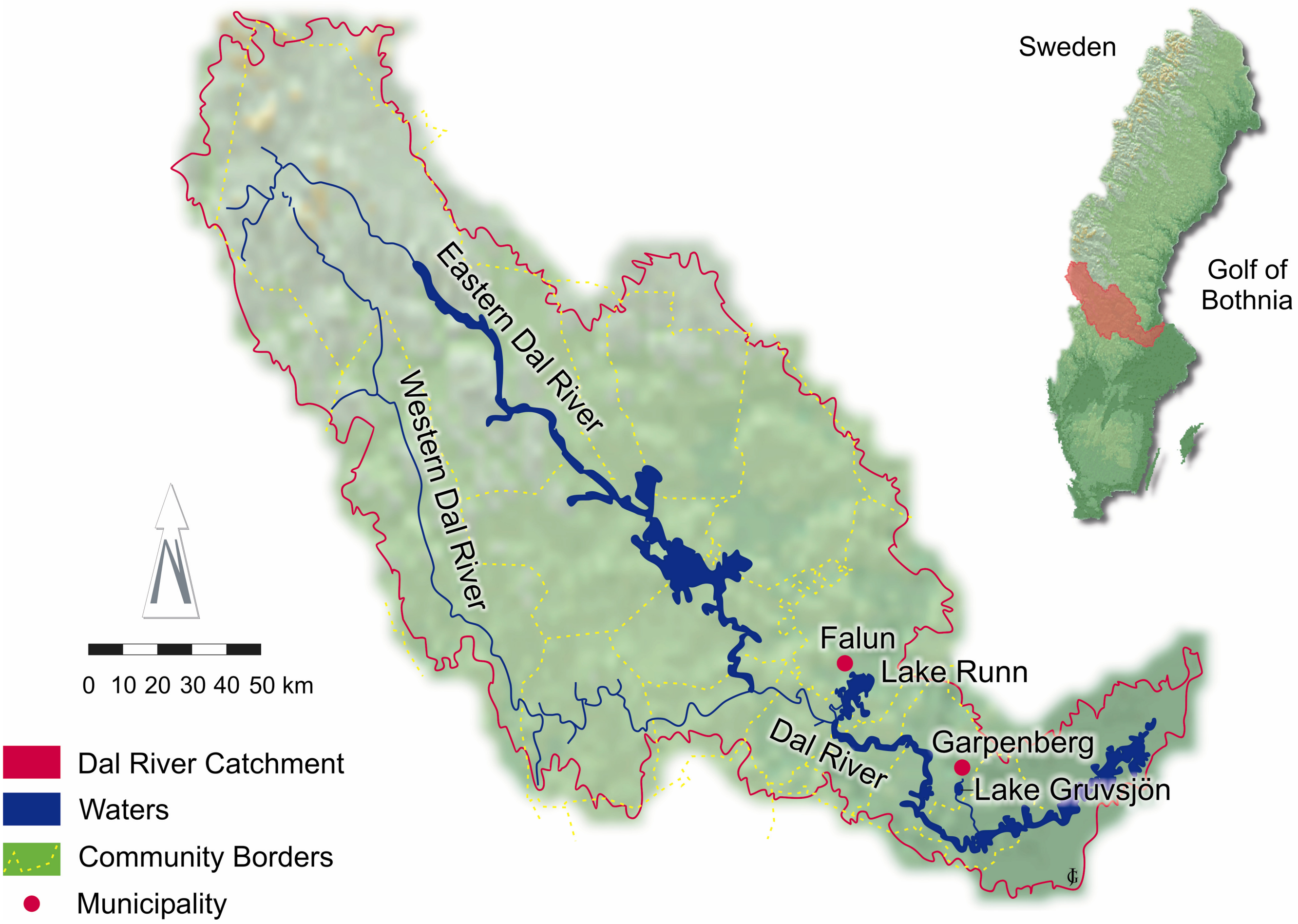

Figure 22. The Dal River catchment including the regions Falun with local recipient Lake Runn, and Garpenberg with local recipient Lake Gruvsjön

levels, ranging between 10 and 90% of the unabated (i.e. current) average annual metal loading to each recipient are presented. Currently observed metal loads are considered to be primarily derived from known major mine waste deposits though other possible and less known sources of metals leakage. We focus here specifically on the metals Zn, Cu and Cd, with Table 5 summarising different considered abatement scenarios for the Dal River and Table 6 different sources of metal leakage in Falun and Garpenberg, which are considered dominant in affecting both the Dal River and the local lakes Runn and Gruvsjön (a complete list of all known sources is reported by Baresel et al. 2003). Only one scenario was investigated for metal load reductions to Lake Runn and Lake Gruvsjön covering the contaminant sources with water and soil.

The cost-minimization problem, as discussed in connection with equations (7), (8), and (10) include soil and water covers are possible abatement measures at the mining waste sites and constructed wetlands as a downstream remediation method.

AIII.3.3 Results for the Dal River

Reductions of metal loading to the Dal River may be obtained by applying mine water abatement measures both at the Falun and the Garpenberg mine waste sites, as well as by constructing wetlands close to the Dal River CB. Figure 23 shows the resulting cost-efficient measure allocation solution and associated total annual costs for compliance with three different possible zinc load reduction targets, as a function of different possible wetland cost levels. All obtained cost-efficient solutions imply that measures for the Dal River load reduction compliance should only be taken in the Falun area, with total costs ranging, for example, between 123,000 and 1,200,000 SEK/yr for a Zn load reduction of 50% in different investigated scenarios. For the other investigated metals (Cu and Cd), all cost-efficient solutions involve only wetland construction (Baresel et al. 2003). Because only the zinc requires mine water abatement measures in addition to wetland construction for achieving cost-efficiency, the targeted Zn reduction level may dominate the overall measure allocation strategy and associated total costs. Wetland requirements to remove Zn will also reduce the loading of the other metals considerably.

The results in Figure 23 show that, if water covers are not hydrologically possible (base and worst case scenarios, Table 5, Figure 23a and Figure 23c, respectively), wetland construction is the only cost-effective abatement measure to be taken for all targeted Zn reduction levels, as long as the wetland

Table 5. Investigated abatement scenarios for the Dal River

Scenario	Description
Base case scenario	• abatement alternatives: soil cover and constructed wetlands • soil cover efficiency: 98.8% reduction of initial metal leakage • constructed wetlands uptake: 2g metal/day m^{-2}, lifetimes of 50 years, resulting in annual costs of 1 to 12SEK m^{-2} • soil cover lifetimes of 50 years, resulting in annual costs of 6.41SEK m^{-2} • retention of metals (natural attenuation) takes place in surface water sediments (with resulting delivery factors $\alpha^j<1$, with exact delivery factors as reported by Baresel et al. 2003)
ptimistic scenario	• water covering considered hydrologically possible (in addition to the soil cover and wetland construction measures considered in the base case scenario) at selected mine waste sites • water cover lifetime of 50 years, resulting in annual costs of 0.45SEK m^{-2}
Worst case scenario (combination of three different sub-scenarios, the individual results of which are reported by Baresel et al. 2003)	• estimated unabated (current) metal leakage from known mine waste sites reduced by 5%, (relative to base case scenario), with that 5% instead being considered to originate from unknown diffuse sources (for instance, reflecting possible metal leakage from abandoned mine voids, which may be highly significant metal sources (Younger et al. 1997a, 2002) that are not commonly accounted for) • no retention (natural attenuation) of metals takes place in river and lake sediments (i.e. all delivery factors are $\alpha^j=1$) • soil cover efficiency reduced to 85% of initial metal leakage
Variable wetland costs	• constructed wetland lifetime of 50 years, resulting in annual costs ranging from 1—12SEK m^{-2} (considering various reasons for wetland cost uncertainty, Baresel et al. 2003)

Table 6. Metal release from different sources in Falun and Garpenberg; [1]: Hartlén and Lundgren 1990, [2]: Lindeström 1999, [3]: compiled from Fällman and Quarfort 1990

	Release [kg yr^{-1}]		
	Zn	Cd	Cu
Falun			
Sum mine waste deposit leakage	289,600	363.0	15,720[3]
otal diffuse metal leakage in Falun[1, 2]	31,250	92.7	1,820
Garpenberg			
Sum mine waste deposit leakage[3]	6,031	8.04	138.4
Total diffuse metal leakage in Garpenberg[1, 2]	587	2.38	96.6

construction costs are relatively low. Soil covering, however, becomes an increasing part of the cost-effective measure allocation solution for increasing wetland cost levels, in particular in combination with increased required Zn load reduction. Furthermore, if water covering is hydrologically possible (optimistic scenario, Table 5, Figure 23b), this particular measure becomes part of the cost-efficient measure allocation solution for all considered wetland cost levels and targeted Zn load reduction levels, thereby also considerably reducing total abatement costs.

Figure 24 further illustrates the resulting marginal costs as functions of targeted metal load reduction level (additional results are reported by Baresel et al. 2003). The marginal costs for compliance to Cu and Cd reductions are constant because wetland construction is the only cost-efficient abatement measure over the entire range of investigated wetland cost levels and for all targeted load reduction levels. At an annual wetland cost of 1 SEK/m^2yr (Larsén 2002, and references therein), marginal costs are constant for all Zn load reduction targets because in this case, wetland construction is then the only cost-efficient mine water pollution abatement measure. Increasing wetland cost levels, however, imply that other possible abatement measures can come into play in the cost-efficient measure allocation solution. Marginal costs for reducing Zn loads to the Dal River then decrease, reaching a minimum somewhere around the 50% reduction level for the "Base case scenario" considered in Figure 24. The reason for this marginal cost behaviour is that soil and water covering measures, which increasingly come into play for higher wetland cost and Zn reduction level (Figure 23), are not continuous abatement measures. Specifically, it is not possible to obtain a continuously increasing load reduction level by increasingly covering a larger part of a mine waste deposit. Each mine waste deposit must be covered entirely to obtain a metal reduction effect, which then produces some discrete effect at each water recipient. If only part of a deposit is covered, oxygen diffusion into the mine waste will not be stopped, oxidation will continue, and hardly any metal reduction effect will be obtained. For the Dal River, and for mid-range to relatively high wetland cost, the cost-efficient Zn load reduction level is thus about 50% for the "base case scenario", regardless of any legal or political reduction targets possibly being set lower than that.

Constant marginal costs, as shown in Figure 24 for Cu and Cd (and in some cases, also for Zn), instead imply that total costs increase linearly and continuously with increasing targeted load reductions. Such a marginal and total cost behaviour is the result of the continuous abatement measure possibility offered by wetland construction, in contrast to the discontinuous mine waste (soil and water) covering methods.

Results for Runn (Falun Subcatchment)

Wetland construction is not considered as a possible mine water abatement alternative in the case of Lake Runn due to subcatchment area limitations, because there may not be sufficient space available for relevant wetland construction here. This area limitation implies that some combination of soil and water covering has to be applied for achieving metal load reduction to Lake Runn. Figure 25 then shows the resulting cost-efficient abatement measure allocation and associated total costs for compliance to different targeted metal load reduction levels. Zn is, at this local subcatchment scale, in contrast to the entire Dal River catchment scale, not longer the metal that dominates the overall mine water abatement solution, and requiring the greatest financial input for its abatement. Furthermore, load reduction can only be achieved here up to 80% of the pre-abatement (present) loading, even with all feasible abatement measures being used. In general, however, total costs of compliance with different targeted metal load reduction levels, up to the limit of about 70%, are in the same range as for the Zn load abatement in the Dal River

Figure 23. Cost-effective solution and associated total annual abatement costs for compliance to different targets of Zn load reduction to the Dal River in a) the "base case scenario", b) the "optimistic scenario", and c) the "worst case scenario", according to the scenario description in Table 5

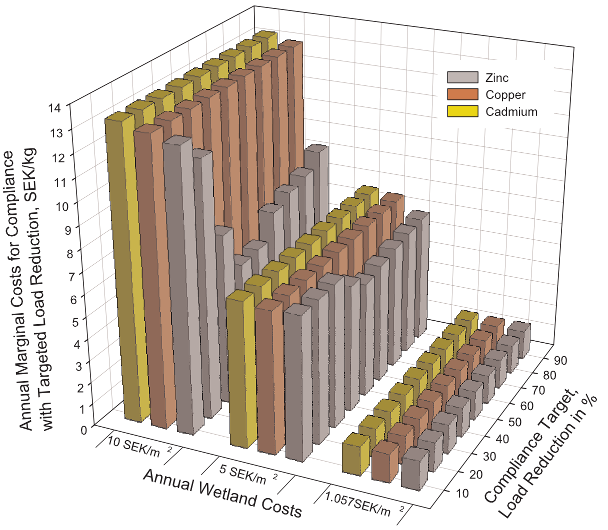

Figure 24. Annual marginal costs for compliance to different targets for reduction of Zn, Cd and Cu loading to the Dal River in the "base case scenario" (Table 5), for annual wetland cost of 1,057 SEK/m^2, 5 SEK/m^2, and 10 SEK/m^2

Figure 26 shows marginal costs of compliance with different metal load reduction levels. The discontinuity effects of soil and water covering, which are the only abatement measures considered feasible in the Runn case, are here evident for all investigated metals. Specifically, all marginal cost curves initially decrease with increasing targeted load reductions, until reaching the minimum point that represents a discrete, lowest possible cost-efficient load reduction level. Above this reduction level, marginal costs may increase rapidly, reflecting the required additional discrete mine waste site covering measure that must be implemented in order to get any additional metal load reduction effect at all. Thereafter, marginal costs decrease again until the point of this next, discrete cost-efficient load reduction level is reached (see particularly the Zn marginal cost curve in Figure 26).

From comparison of Figure 26 with Figure 24, we also note that marginal costs for compliance with different targeted Cd load reductions are, for the Lake Runn case, at least one, and in most cases more than two orders of magnitude higher than in the Dal River

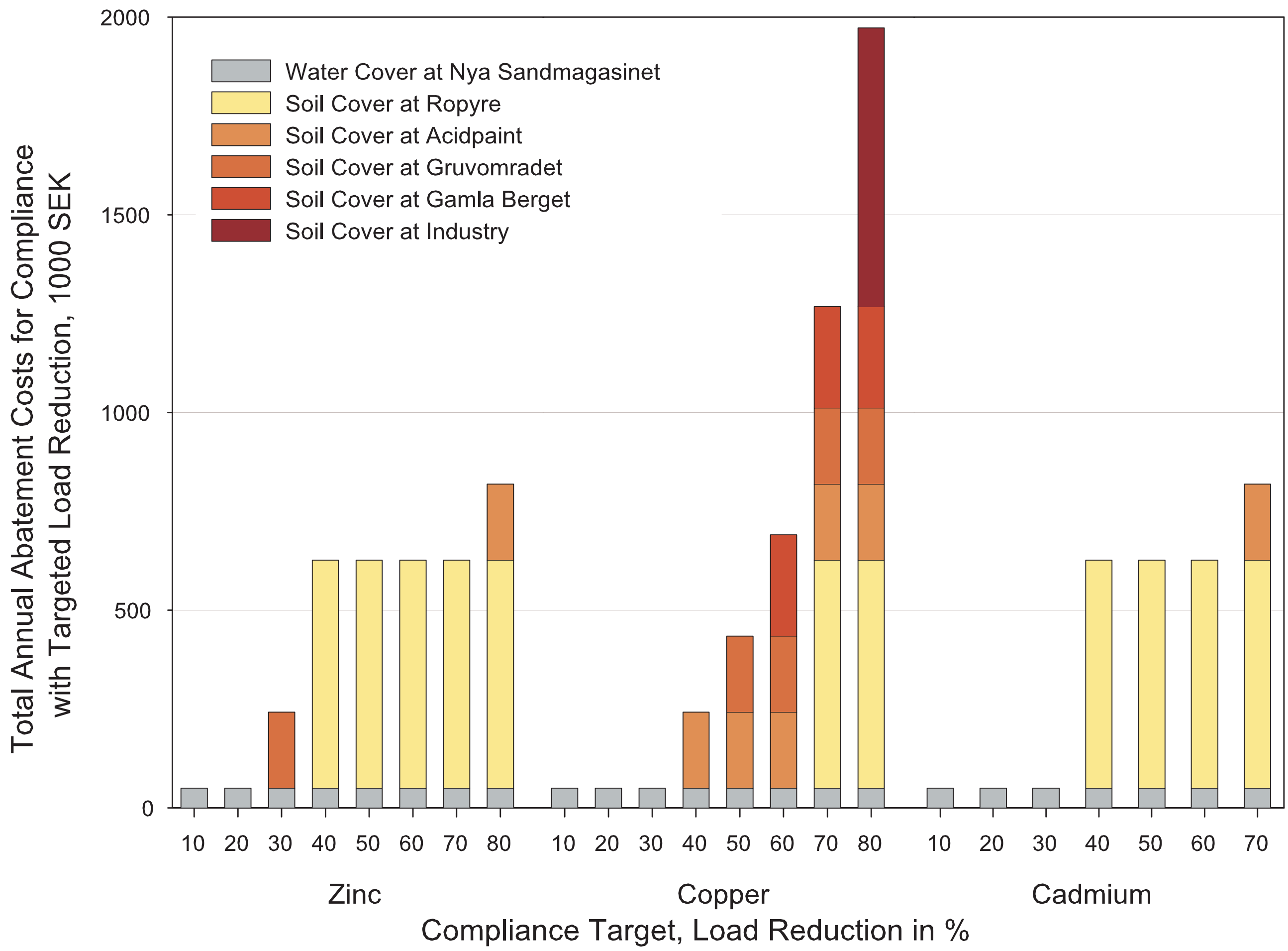

Figure 25. Cost-effective solutions and associated total annual abatement costs for compliance with different targets for reduction of Zn, Cd, and Cu loads to the Lake Runn

case. Also for Cu and for some conditions even for Zn, marginal costs of compliance to targeted load reduction levels are significantly higher than for the Dal River. These results are the direct effect of wetland construction, or any other abatement measure close to the Lake Runn CB, not being considered feasible.

Results for Gruvsjön (Garpenberg Subcatchment)

Pre-abatement (current) metal leakage from mine waste sites in the Garpenberg subcatchment are estimated to account for only 1—2% of the total metal discharges into the Dal River (Hartlén and Lundgren 1990). Nevertheless, metal discharges into the local Lake Gruvsjön may be considerable and require mine water pollution abatement, with Figure 27 illustrating results for cost-efficient allocation of abatement measures and associated total costs in this case. We can then see that already at a targeted 20% load reduction for Cu and Cd to Lake Gruvsjön, total abatement costs are comparable to those for the Dal River at a 70% reduction target, without any of the cost-efficient abatement measures for Garpenberg being included in the cost-efficient abatement allocation solution for the Dal River (Figure 26). At the 10% targeted reduction level, both total (Figure 27) and marginal (Figure 28) costs for Cu and Cd abatement are relatively low, due to the possibility of water covering at some mine waste sites. Similar to what was observed for Lake Runn, there is an upper limit in possible load reduction of 50% for Cu and 70% for Cd for Lake Gruvsjön, since no additional abatement measure alternatives are available for further load reduction, within the chosen set of measures investigated in this study.

Marginal abatement costs for Cu and Cd load abatement are much higher than in both the Lake Runn and the Dal River cases; abatement of 1kg of Cd may cost more than 240,000 SEK annually. Also in this case, marginal costs of compliance decrease as long as the cost-efficient abatement solution remains the same (Figure 27) due to the waste covering discontinuity effect; only at reduction level points where additional discrete measures must be applied do marginal costs rapidly increase and thereafter decrease

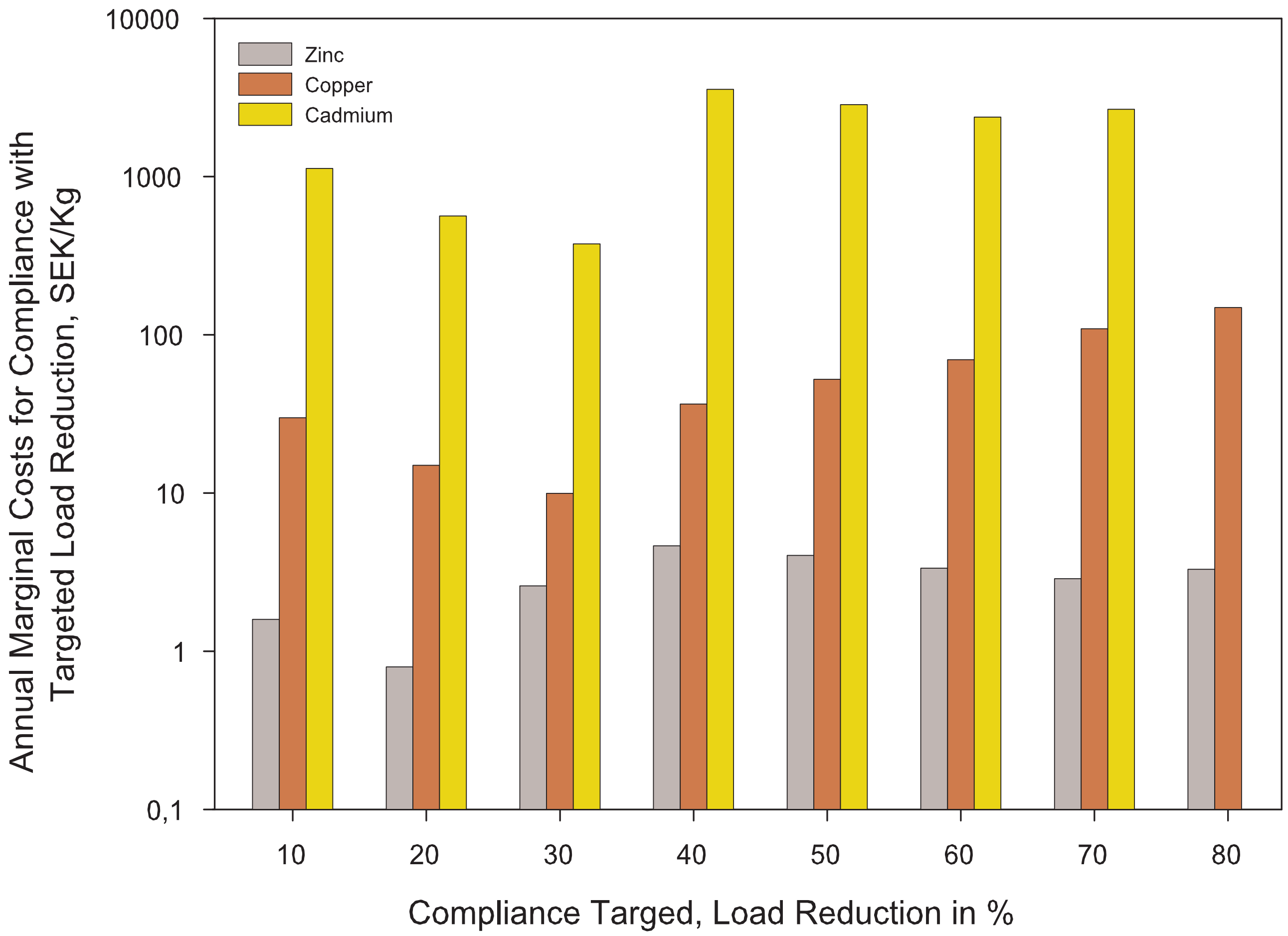

Figure 26. Annual marginal costs for compliance with different targets for reduction of Zn, Cd and Cu loads to the Lake Runn

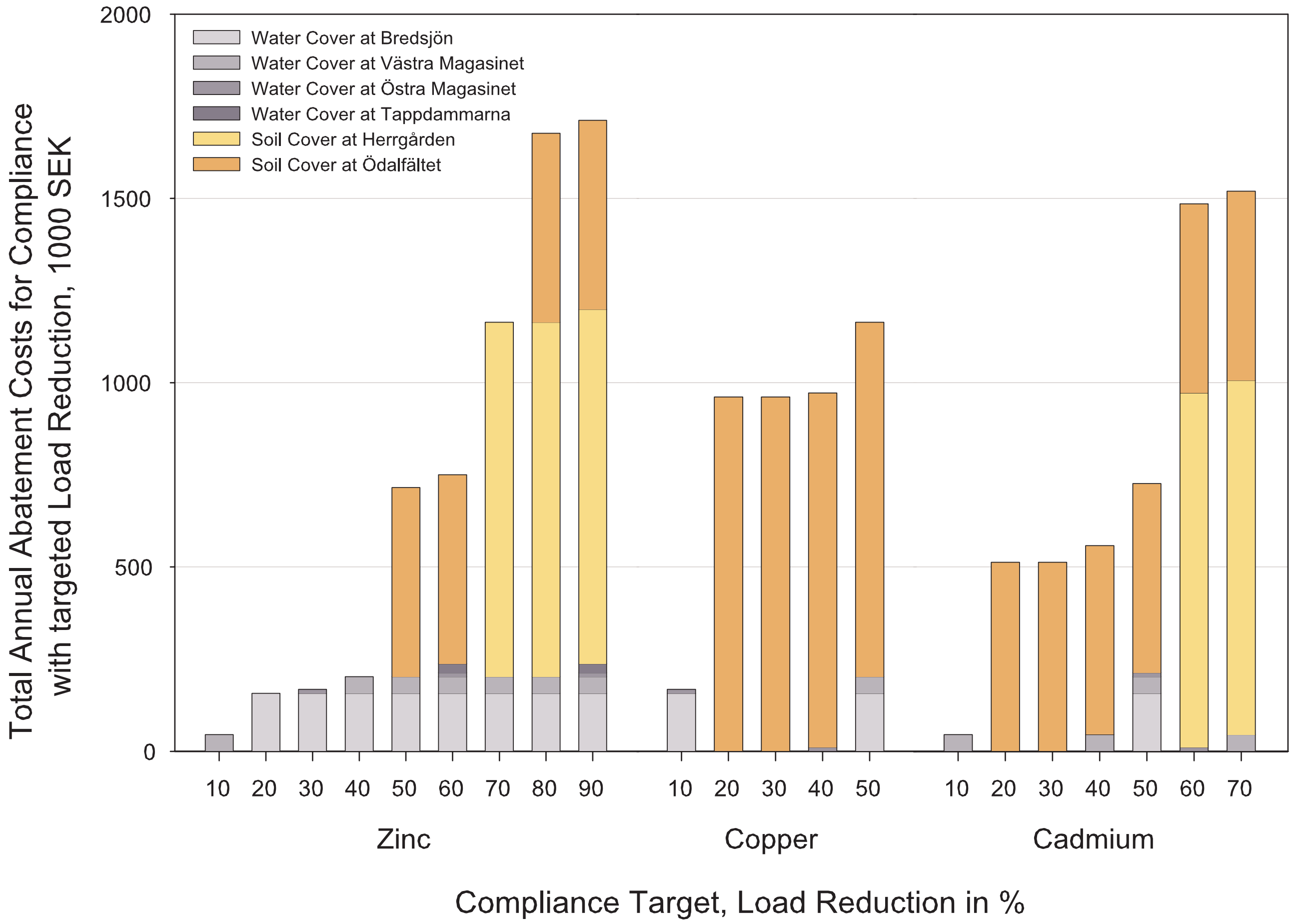

Figure 27. Cost-effective solution and associated total annual abatement costs for compliance to different targeted load reduction of Zn, Cd and Cu to the Lake Gruvsjön

again, until the next need of additional discrete abatement measures.

Case Study Conclusions

We have presented here results from a site-specific conceptual cost-minimization model applied to the determination of cost-effective allocation of mine waste remediation and/or mine water pollution abatement measures within the Dal River catchment, in order to achieve targeted Zn, Cu and Cd load reductions to the Dal River and to selected local mine water recipients. We have considered various, practically feasible remediation measures and designs, including soil and water covering of the mine waste deposits, and downstream wetland construction close to the compliance boundaries (CBs) associated with the different investigated mine water recipients. We calculate the cost-efficient measure allocation, and associated total and marginal costs for compliance to different environmental targets (ETs; in terms of metal load reduction) and CB locations (mine water recipients), and for different scenarios of technological efficiency and annual cost.

We show that total abatement cost for achieving a certain ET (load reduction) may be as high for a local water environment, as for the Dal River (entire catchment scale), thus implying much higher marginal costs for the former, local compliance. Furthermore, the cost-efficient abatement measure allocation solution for local compliance (here particularly for the Garpenberg – Lake Gruvsjön - subcatchment) may be completely different from that for the entire catchment scale (here the Dal River catchment). We note that the European Union Water Framework Directive allows for the possibility of using heavily modified waters, for instance close to sources, as pollutant sinks, and focusing remediation efforts to achieving good water quality downstream, in more practically restorable water bodies. The active choice of CB location is then of utmost importance for the cost-efficient allocation of mine waste remediation measures on a catchment scale.

Our site-specific results also show that discontinuity in the technical feasibility of certain remediation measures (here soil and water covers of mine waste sites) implies that relatively low chosen ET levels may

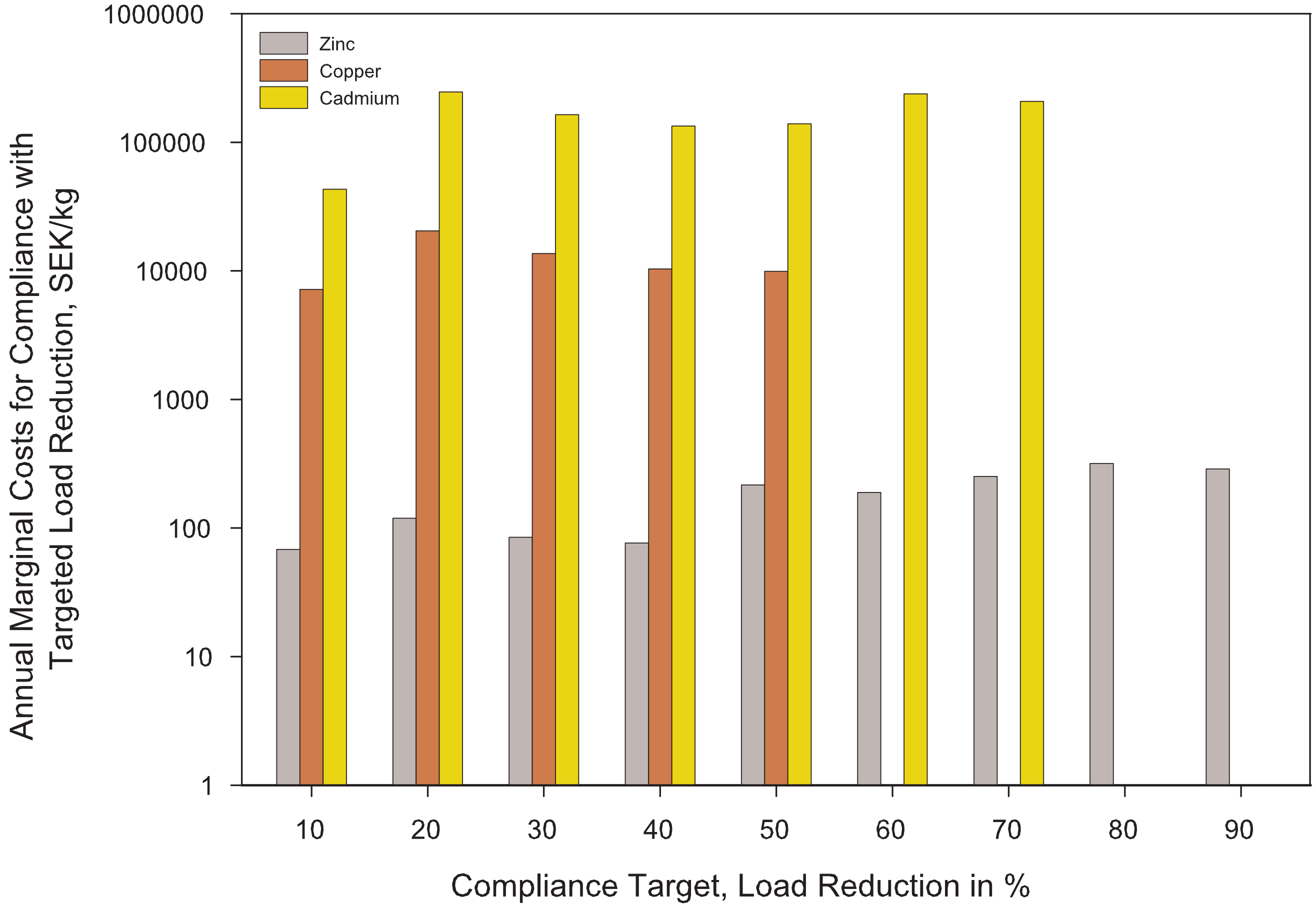

Figure 28. Annual marginal costs for compliance with different targeted reductions of Zn, Cd and Cu loads to the Lake Gruvsjön

not be achievable at relatively low cost. In general, local minima in costs may then occur only at certain, discrete ET levels, which must be identified and quantified for achieving economic efficiency. Wetland construction, or other possible abatement measures in the direct vicinity of CBs, may offer an alternative (to the discrete mine waste covering measures) continuous abatement measure possibility, which may be an important (or even the only, as shown here for Cu and Cd load abatement for the Dal River) part of a cost-efficient solution for abatement measure allocation within a catchment.

AIII.4 General Conclusions

As a general conclusion from the present economic analysis overview and specific case study, we identify the following main differences between today's practice in making decisions about mine water pollution abatement options and the rational economic approaches discussed above:

- Allocation within a catchment of mine water pollution abatement is not commonly quantified in a catchment perspective and thereby not optimised based on economic efficiency on the catchment scale. The present specific case study exemplifies the application of a quantitative hydrologic-economic modelling approach to catchment scale water quality management with a focus on the mine water pollution problem.
- Cost scenarios for mine water pollution abatement are not commonly estimated with respect to different possible CBs and different possible water quality measures with associated scales of analysis (e.g. locally measured vs. spatially averaged measures) for judging compliance with water quality targets, such as MCLs or MPLs. In the case study presented here, we explicitly quantify abatement cost related to different CBs, each associated with a different water recipient. We use as a relevant water quality measure, average annual pollutant mass discharging through the CB, and investigate a wide range of possible water quality improvements, quantified in terms of different reduction levels of the pre-abatement annually discharged pollutant mass. These measures of water quality and its improvement are common in and suitable for economic analysis of water quality abatement on a catchment scale (e.g. Gren et al.

2000a, 2002). For each water recipient, these measures can be related to other water quality measures, for instance average concentration or mass discharge values in(to) each considered water recipient, which may be more relevant for judgment of compliance to regulator standards, for instance, MCLs or MPLs. By providing cost-effective solutions for a wide range of discharge reduction levels in annual pollutant mass, the present case study thus also provides a basis for economic analysis of other possible water quality measures, each related to a different reduction level in annual pollutant mass discharge.

- Estimated costs for mine water pollution abatement do not commonly include cost components associated with measurement/prediction uncertainties, which imply finite risk/probability of abatement measures not achieving their targeted water quality improvements. This cost aspect has not yet been quantified for our specific case study either. However, the methodology and results already presented here for this case study, can readily be extended to include uncertainty effects, by use of methods presented by Gren et al. (2000a, 2002) and will be addressed in forthcoming work.

Expected long-term temporal changes in different mine water pollution scenarios are not commonly considered in any dynamic long-term analysis of efficient catchment scale mine water pollution abatement. There are also commonly no regulatory limits in time explicitly specified for compliance with water quality targets, MCLs or MPLs. The present specific case study deals to a large degree with water quality management related to old, abandoned mine waste sites, for which the current pollution load level, if unabated, may be considered to continue into the future, being more or less constant. The temporal problem of long-term variability in pollutant loads, however, may be highly relevant for the long-term planning and permit requirements for closure of active mines and should be addressed in future work.

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