Minewater Treatment

Technology, Application and Policy

Melanie Brown, Bob Barley and Harvey Wood

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Foreword

At the time of writing this preface it is almost exactly 7 years since the Coal Authority was formed in the UK on privatisation of the coal mining industry. The topic of minewater pollution, both existing and potential, was the subject of great debate in the formulation of both the Coal Industry Act 1994 and the Environment Act 1995, given the fact that an estimated 400 km of the nation's rivers are polluted by discharges from abandoned coal mines.

It was during the passage of the Coal Industry Act that the seeds of the Coal Authority's work on minewater pollution were sown. Since that time the Authority has worked closely with the Environment Agency and the Scottish Environment Protection Agency to formulate a prioritised programme of treatment for over 100 of the most significant discharges. At November 2001 thirteen schemes to treat or prevent minewater pollution have been completed, with another four coming on stream in the spring of 2002.

In the early days both the Authority, and the consultants providing advice, were heavily reliant on the work carried out by staff at the now disbanded US Bureau of Mines (Hedin *et al.* 1994). Although a fair amount of success can be claimed, many of our own lessons have been learned and we were delighted to hear of the proposal to carry out a comprehensive review of all completed minewater treatment systems in the UK.

Some of the lessons relate to the more scientific aspects, e.g. sizing of treatment systems and wetlands, but equally important are the practical lessons, such as the ability of ‘ochre’ to accrete in pipelines, or overly complex inlet arrangements in contradiction of traditional water engineering practice.

It is clear that, in the current climate of environmental awareness combined with European initiatives such as the Water Framework Directive, the need to address present and future minewater pollution will continue for some considerable time and will no doubt increase in scale. Perhaps the UK is ‘ahead of the game’ in Europe, but only so far as the problems arising from the closures during the late 1980s/early 1990s have required action to be initiated with the resulting learning curve.

I do hope that this publication will allow other countries, who have yet to go through the mine closure process, to gain a head start and that it will ensure their learning curve is less steep than that encountered by the Coal Authority over the last 7 years.

Keith Parker

Operations Manager
The Coal Authority

Introduction

Mining has been carried out in the UK since at least neolithic times, and quite possibly for longer than this. Associated with this historic mining activity there is a legacy of contamination of both land and water resources. In addition, recent widespread closures of coal mines has lead to water table rebound and outbreaks of contaminated minewaters. Thus there are watercourses throughout the UK that are affected by both historic and recent mining activity.

In March 1994, the National Rivers Authority (NRA) published a report in the Water Quality Series, No.14, entitled *Abandoned Mines and the Water Environment*. This report outlines what acid mine drainage (AMD) is, the scale of the problem (200 km of waters affected by abandoned coal mines and 400 km of waters affected by abandoned metal mines) and the legal framework in 1994.

Since this NRA report was published the reorganisation of the UK coal industry has lead to the formation of the Coal Authority and the NRA have become the Environment Agency (EA). These two bodies have worked closely together to identify minewater pollution problems, to prioritise incidents of pollution in terms of environmental impact and to implement mitigation strategies to reduce this impact. As a result of this co-operation there is an active programme of remediation and numerous schemes have been implemented throughout the coalfields of the UK. These schemes include physical, chemical and biological strategies to affect the amelioration of minewater.

There are also operational treatment schemes dealing with minewater from abandoned metal mines, and at operating and former coal mine sites that are not within the remit of the Coal Authority.

The aim of this book is to take an overview of these various treatment schemes, to assess their applicability, suitability, efficiency and cost-effectiveness, and to compile available monitoring data from the schemes. Together with an updated literature review and consideration of active treatment technologies and current research, this provides an assessment of the current situation in the UK, within a global context, of minewater treatment strategies. This will enable the selection of the most appropriate technique required to tackle a particular minewater discharge problem.

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1

The minewater problem

1.1 CHEMISTRY OF FORMATION

Contaminated minewater is generated when rock containing sulphidic minerals is exposed to water and oxygen, resulting in the production of acidity and elevated concentrations of metals and sulphate in the water. Sulphidic minerals are ubiquitous and many types have been implicated in the generation of contaminated minewater (table 1.1). The weathering of these minerals is a natural process, and thus generation of metal-contaminated waters can also be considered to be a natural process. However, the sulphide-containing rocks are generally beneath a mantle of soil and below the water table. Under natural conditions, therefore, contact between the rock and oxygen is minimal, and so acid generation occurs at a very slow rate. Mining activity causes exposure of the rock to oxygen and water, with a resulting acceleration of the weathering process.

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Table 1.1 Common sulphide minerals implicated in the production of contaminated mine water (adapted from NRA 1992a)

Mineral	Composition	Aqueous end-products of complete oxidation
Amorphous	FeS	Fe^{3+}, $SO_4{}^{2-}$, H^+
Arsenopyrite[a]	$FeAsS$	Fe^{3+}, $AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+
Bornite[a]	Cu_5FeS_4	Cu^{2+}, Fe^{3+}, $SO_4{}^{2-}$, H^+
Chalcocite[a]	Cu_2S	Cu^{2+}, $SO_4{}^{2-}$, H^+
Chalcopyrite[a]	$CuFeS_2$	Cu^{2+}, Fe^{3+}, $SO_4{}^{2-}$, H^+
Cinnabar	HgS	Hg^{2+}, $SO_4{}^{2-}$, H^+
Cobaltite[a]	$CoAsS$	Co^{2+}, $AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+
Galena[b]	PbS	Pb^{2+}, $SO_4{}^{2-}$, H^+
Mackinawite	FeS	Fe^{3+}, $SO_4{}^{2-}$, H^+
Marcasite[a]	FeS_2	Fe^{3+}, $SO_4{}^{2-}$, H^+
Molybdenite[a]	MoS_2	$MoO_4{}^{2-}$, $SO_4{}^{2-}$, H^+
Niccolite	$NiAs$	Ni^{2+}, $AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+
Orpiment[a]	As_2S_3	$AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+
Pentlandite	$(Fe,Ni)_9S_8$	Fe^{3+},Ni^{2+}, $SO_4{}^{2-}$, H^+
Pyrite[a]	FeS_2	Fe^{3+}, $SO_4{}^{2-}$, H^+
Pyrrhotite[a]	$Fe_{1-x}S$	Fe^{3+}, $SO_4{}^{2-}$, H^+
Realgar	AsS	$AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+
Smythite	Fe_3S_4	Fe^{3+}, $SO_4{}^{2-}$, H^+
Sphalerite	ZnS	Zn^{2+}, $SO_4{}^{2-}$, H^+
Tetrahedrite	$Cu_{12}(Sb,As)_4S_{13}$	Cu^{2+},SbO^{3-},$AsO_4{}^{3-}$, $SO_4{}^{2-}$, H^+

[a] Known to be oxidised by *Thiobacillus ferrooxidans*.
[b] Known to be oxidised by iron-oxidising bacteria.

The sources of contaminated minewater from mining operations include (Sengupta, 1993):

- drainage from underground workings;
- runoff from open pit workings;
- waste rock dumps from mining activities;
- mill tailings;
- ore stockpiles;
- spent ore piles from heap leach operations.

Contaminated minewater is often acidic, as can be seen by the presence of H^+ ions in the weathering products (table 1.1). However, the ability of a particular rock type to generate acid is determined by the relative content of acid-generating and acid-consuming minerals. If acid-consuming minerals, such as calcite and bauxite are present, the resultant water, while containing elevated concentrations of sulphate and metals, may be of circum-neutral pH. Thus, although there may be acidic conditions in micro-environments surrounding sulphide grains, the water draining from the rock or waste rock pile will not necessarily be acidic. The terms acid mine drainage (AMD) or acid rock drainage (ARD) are thus not universally applicable to all minewater discharges, although common usage means that these terms are often used even for cases of non-acidic minewater.

The generation of contaminated minewater is a combined chemical and microbiological process. The most widely studied mechanism in the formation of contaminated minewater is the oxidation of pyrite, which is the mineral most often implicated in this process. Oxidation of pyrite involves a series of reactions (Stumm and Morgan 1981), which can be considered to consist of an initiator reaction and a propagation cycle (Singer and Stumm 1970). The initiator reaction involves release of Fe^{2+} ions into solution, either by the simple dissociation of pyrite, or by oxidation of pyrite, which can be shown stoichiometrically as:

$$FeS_{2(s)} + 7/2\ O_2 + H_2O \rightarrow Fe^{2+} + 2\ SO_4{}^{2-} + 2H^+ \qquad (1.1)$$

The propagation cycle is then established, which involves the oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}):

$$Fe^{2+} + 1/4\ O_2 + H^+ \rightarrow Fe^{3+} + 1/2\ H_2O \qquad (1.2)$$

At pH values of greater than 2.3 to 3.5 ferric iron is not soluble in water and it will precipitate as ferric hydroxide ($Fe(OH)_3$), leaving little Fe^{3+} in solution and lowering the pH:

$$Fe^{3+} + 3\ H_2O \rightarrow Fe(OH)_{3(s)} + 3H^+ \qquad (1.3)$$

Any ferric iron that does remain in solution is subsequently reduced by pyrite thereby generating additional ferrous iron and acidity:

$$FeS_{2\ (s)} + 14\ Fe^{3+} + 8\ H_2O \rightarrow 15\ Fe^{2+} + 2\ SO_4{}^{2-} + 16\ H^+ \qquad (1.4)$$

The slow (or rate-determining) step in the above sequence is the oxidation of ferrous to ferric iron (equation 1.2) (Singer and Stumm 1970). Once ferric iron has been formed by this oxidation step, the subsequent oxidation of pyrite by ferric iron that remains in solution is rapid, and strongly acid-producing (equation 1.4).

The microbiological component of contaminated minewater generation is of major importance. Iron-oxidising bacteria, such as *Thiobacillus ferrooxidans*, can accelerate the rate of oxidation of ferrous to ferric iron by a factor larger than 10^6 (Sobek *et al.* 1990) and bacterial mediation throughout the weathering process can increase the overall rate of acid generation by a factor of up to 20. These bacteria tend to be most active between pH 2 and pH 4 (Filion *et al.* 1990). The weathering of pyrite to produce acidity is thus a self-perpetuating process. Initially, the pH may be such as to cause $Fe(OH)_3$ to precipitate (equation 1.3). However, this reaction releases H^+ ions, which lowers pH and results in more ferric iron staying in solution. This ferric iron is then involved in rapid oxidation of pyrite, which results in a further fall in pH (equation 1.4). Added to this chemically induced reduction of pH and acceleration of weathering is the increased activity of micro-organisms as the pH falls, which gives additional acceleration of these processes. The significance of ferric iron acting as an oxidising agent is that no molecular oxygen is required; thus even in flooded mine workings where air–pyrite contact is eliminated, pyrite weathering can still occur (Kalin 1989, Wildeman and Laudon 1989). However, this weathering will be at a much reduced rate, compared to that occurring in an aerobic atmosphere, since there will be no bacterial action (iron-oxidising bacteria respire aerobically). Nevertheless, natural waters can contain up to 10 to 15 mg/l dissolved oxygen, depending on the temperature, so flooding is no guarantee of anaerobic conditions.

The weathering of other sulphide minerals has been less well studied, and although the end-products are known (table 1.1) the pathways of oxidation are largely unknown. However, it is evident from the chemistry of discharges associated with mining activities (see table 1.2) that other sulphide minerals are indeed subject to mobilisation by weathering. Also, contact between acid water generated by pyrite oxidation and other minerals, not necessarily sulphidic, can cause the dissolution of these minerals and their subsequent mobilisation (Singh 1987).

Factors influencing the acidity and metal content of contaminated mine water include (Ritcey 1989):

- sulphide grain size and surface area;
- porosity and permeability of the deposit;
- nature of the gangue materials;
- nature of the sulphide ore;
- nature of acid-consuming minerals;
- various environmental factors influencing the activity of micro-organisms.

Table 1.2. Chemical composition of some contaminated mine water discharges (all concentrations are mg/l)

Constituent	A	B	C	D	E	F	G
Al	18	-	15	50	0.03	-	30
As	0.02	-	-	-	0.02	-	9.0
Cd	0.03	-	0.1	-	0.07	0.02	1.0
Cu	1.6	43	6	-	14	-	1.5
Fe	50	-	50	50–300	144	2.9	300
Pb	0.01	-	0.1	-	0.1	1.1	-
Mn	32	-	20	20–300	7	0.4	12.0
Ni	-	-	0.1	-	0.07	0.01	-
Zn	10	32	25	-	28	3.2	120
Sulphate	2100	-	-	20–2000	1000	33	
pH	2.6	7	3.6	3.0–5.5	2.7	7.5	3.5

A Big Five Tunnel, Colorado, USA (abandoned precious metal mine) (Wildeman and Laudon 1989).
B Britannia Mine, British Columbia, Canada (abandoned copper mine) (Ritcey 1989).
C Pennsylvania Mine, Colorado, USA (abandoned gold/silver mine) (Emerick *et al.* 1988).
D Typical coal mine drainage (Ritcey 1989).
E Parys Mountain, Anglesey, UK (abandoned copper mine) (NRA 1992a).
F Hafna Mine, Gwynedd, UK (abandoned lead/zinc mine) (NRA 1992a).
G Wheal Jane, Cornwall, UK (abandoned tin mine) (Dodds-Smith *et al.* 1995).

Generation of contaminated minewater is thus highly site specific, and can vary greatly even within a single mine site. Weathering of sulphides will continue as long as there are exposed sulphides present, and is thus likely to occur for many hundreds of years after mining operations have ceased. Much work has been done on methods of predicting the degree of acid generation that is likely (for example Erickson *et al.* 1985, Miller and Murray 1988, Miller *et al.* 1990, Botz and Mason 1990, Williams and Breward 1993, Lapakko 1994, Robb and Robinson 1995, ICARD 2000), although complex geology and hydrogeology make such work fraught with uncertainty. Continuous monitoring techniques have been shown to be effective in increasing the reliability of prediction of acid-producing potential, enabling early identification of potential trouble spots within a mining development and swift implementation of remedial action (Fytas *et al.* 1992). Attempts have also been made to predict the longevity and severity of mine water discharges, although once again there are too many unknown factors to make accurate prediction possible but general trends and probable outcomes can be suggested (Younger 1996).

1.2 OCCURRENCE AND SCALE OF THE PROBLEM

As mentioned in the previous section, sulphidic minerals are ubiquitous; the problem of contaminated minewater is therefore likely to occur in many situations where disturbance of rock causes exposure of the rock surface, and thus sulphidic minerals, to the air. Whereas some quarrying operations may be free of the threat of contaminated minewater generation (for example limestone or gravel quarries), the mode of formation of the majority of coal and metal deposits implies that sulphides are likely to be present. This latter statement encompasses the fact that the formation of these deposits almost invariably entails the formation of sulphide minerals. In the case of coal, the anoxic conditions in which the coal formed would have favoured the activity of sulphate-reducing bacteria, with the consequent formation of hydrogen sulphide gas and precipitation of pyrite, particularly in marine shales associated with coal deposits. In the vast majority of metal deposits, despite a great diversity of genetic origins, the ore comprises sulphidic minerals, which can be attributed to the greater thermodynamic stability of sulphides compared to other compounds of the metals.

In operating mines, dewatering of mine workings often obscures the effects of sulphide oxidation, since the water that is pumped is, initially, the original saline groundwater and, latterly, recently recharged water that has entered from the surface (Younger 1994). Such waters may be moderately saline and contain very few products of sulphide weathering. However, once dewatering ceases, as is the case for large areas of former coal and metal mining throughout the UK, rising water levels flush out the products of pyrite oxidation that have been forming in the ventilated mine workings since the earliest days of mining. Escape of this acidic water can have devastating environmental consequences. At Wheal Jane, in Cornwall, the unanticipated failure of an adit plug gave rise to the release of 54 million litres of highly acidic, metal-contaminated mine water into the Carnon River, with a consequent, ochrous plume affecting 6.5 million m^2 of the receiving tidal waters. Peak metal concentrations recorded included 450 mg/l zinc and 600 µg/l cadmium. The highly visible nature of the ochrous plume ensured that this event had national and international media coverage, with a consequent increase in public awareness. The political ramifications of this event are still continuing and the environmental consequences are yet to be fully realised. In another case, at Lower Ynysarwed Colliery, discharge of highly acidic, iron-contaminated mine water into the Neath Canal coated the canal bed with ochre and destroyed virtually all aquatic life over a 12 km stretch of the canal (Younger 1994).

The occurrence of the contaminated minewater problem in the UK is thus governed largely by the location of abandoned mine workings, although ongoing problems are to be anticipated in connection with the widespread closure of coal mines over the last 20 years (Robb 1994, Younger 1994).

1.2.1 UK metal mines

Metal mining in the UK has been carried out since at least Roman times, reaching its peak in the 18th and 19th centuries. The principal metals extracted were lead, zinc, tin and copper with lesser amounts of gold, silver and arsenic and minor amounts of other minerals. The main orefields were located throughout the southwest, central, southern and northern England, mid and north Wales and Scotland. The last remaining deep tin mine in Cornwall, South Crofty, closed in 1998. There is intermittent gold production in very small quantities from mines in Wales. Due to the very long history of metal mining in the UK, the underground workings can be extremely complex and their full extent is rarely known. Spoil heaps, which are widespread in former mining areas and are mostly unreclaimed, are also a major source of contaminated minewater. The scale of the problem is thus difficult to gauge although it is fair to say that in former mining districts, often whole water catchments, are affected from a multitude of diffuse sources.

Table 1.3. Discharges from metal mines which cause significant pollution problems (classified according to original NRA region, reproduced from NRA 1994a)

Region	Number of discharges	km affected	Working[1]	Abandoned
Northumbria	20	43	2	17
North West	5	36	1	4
Severn Trent	2	5		All
South West	Unknown	212	1	About 1700
Welsh	44	114	1	43
Total	> 77	410		

[1] Includes coal mines where these affect the same watercourse.

The NRA (now the Environment Agency) have attempted to quantify the impact of abandoned metal mines on the environment (NRA 1994a) by assessing the number of known discharges and the lengths of watercourses affected. These data are reproduced in table 1.3. It should, however, be noted that this assessment is a substantial underestimate of the actual scale of the problem, since many smaller watercourses and streams have not been surveyed in this context.

1.2.2 UK coal mines

Coal mining has been important in central and northern England, south Wales and south east England. The vast majority of coal mines are now abandoned although the exact number of such mines is unknown. The Coal Authority holds its own database of approximately 10,000 abandoned mines, but this is at best a rough estimate. In some recently closed mines, dewatering is still being carried out, either to protect operating mines within the same coalfield or to forestall the inevitable environmental impact that cessation of pumping will entail while research is carried out to assess the scale of this impact (Younger 1994, Connelly *et al.* 1995).

In 1994 the NRA published an assessment of the extent of the problem of contaminated minewater from coal mines with regard to its impact on the receiving watercourses (table 1.4). Comments made above in connection with the assessment of metal mines also apply to these data, meaning that there are many smaller and unclassified watercourses that are affected but not included in this survey.

Table 1.4. Discharges from coal mines which cause significant pollution problems (classified according to original NRA region, reproduced from NRA 1994a)

Region	Number of discharges[1]	km affected[2]	British Coal	Other Coal	Working	Abandoned
Northumbria	15 (all)	18 (12)	11	4	4	11
North West	24 (14)	57 (25)	19	5	Nil	24
Severn Trent	4 (all)	19 (4)	Nil	4	Nil	4
Welsh	21 (all)	54 (22)	6	15	Nil	21
Yorkshire	36 (most)	50 (11)	Most	Few	Nil	36
Total	100	198				

[1] Main occurrences from coal mines (originating from underground mines in brackets).

[2] Estimated total for main occurrences (numbers of waterways listed in brackets).

Also, since this 1994 survey, the progressive programme of minewater remediation that has been carried out by the Coal Authority, in consultation and collaboration with the EA and other bodies, has resulted in substantial improvements to the quality of many watercourses. The full extent of these improvements has not been quantified, although it is thought that currently, in 2001, there are approximately 40 km of rivers from which the visual impact of iron ochre staining has been eliminated. There has, however, been no work carried out to assess the improvement of ecological health in these watercourses, although there are often observational factors such as the return of fish to a previously fishless stretch of river.

1.2.3 Worldwide occurrence of contaminated minewater

As stated earlier, the nature of coal and metal deposits makes contaminated minewater a virtually ubiquitous problem associated with the extractive industries. Wherever mining occurs, there is likely to be an associated minewater problem, which is of most environmental significance if the mining operations have ceased. However, in developing countries, the environmental awareness, legislation and available technology may be such that even at active mine sites inadequate measures are taken to prevent influx of contaminated minewater into the environment. The legislative issues concerning minewater worldwide are summarised in section 1.5.

The best documented regions regarding the scale of the problem are Canada and the USA, while the most poorly documented regions are probably the eastern European countries, where environmental safeguards are likely to have been minimal (Reuther 1995).

In Canada, between 1984 and 1987, CANMET (Energy, Mines and Resources, Canada) and industry co-sponsored two projects to define the extent of the acid-generating waste problem at base-metal mining operations (Filion *et al.* 1990). This study identified a total area of over 15,000 hectares of acid-generating waste sites associated with both operating and abandoned mines. Most of these sites represented accumulation of waste from mining since World War II. However, this survey did not include gold mines, coal mines, uranium mines or abandoned mine sites for which responsibility had reverted to the Crown. The provinces of Ontario and Quebec, which have the longest history of mining, have recently completed an inventory of their abandoned mine sites. In Ontario, 100 abandoned mine sites were identified, 20 of which pose an acid-generating problem, with sulphidic wastes covering an area of 830 hectares. In Quebec, 107 abandoned mine sites were found, 21 of which had associated acidic drainage, with sulphidic wastes covering an area of approximately 4500 hectares. The total cost of remediating mine sites in Canada to minimise environmental impact is estimated to be in the region of $4 billion over the next 20 years.

In the USA, the mining industry spends over $1 million every day to treat acidic mine water (Kleinmann 1989). Based on information recently provided by the individual states, the US Bureau of Mines estimates that abandoned coal and metal mines adversely affect over 19,000 km of rivers and streams and over 73,000 hectares of lakes and reservoirs.

The scale of minewater pollution in other countries is less well documented, although it can confidently be stated that wherever extractive industries are operative, there are likely to be issues with minewater impacting the environment to a greater or lesser degree.

The foregoing discussion indicates the severity of the problem posed by minewater, although lack of documentation prevents a comprehensive quantification of the problem on a global scale.

There are numerous international minewater initiatives which, through the facility of the world wide web, form useful discussion and dissemination groups as well as being applicable in their country of origin. The names, and where applicable, website addresses of such initiatives are listed in appendix A.

1.3 LEGAL ASPECTS OF MINEWATER POLLUTION IN THE UK

1.3.1 Working mines

Discharges from working mines in England and Wales are controlled through consents issued by the Environment Agency (EA) under the Water Resources Act 1991 and, more recently, under the Environment Act 1995. Such consents generally include conditions relating to the quality, quantity, discharge regime and monitoring of the minewater. Separate consents are given for discharges from related above-ground activities. Failure to comply with the terms of these consents can lead to prosecution of the mine operators by the EA, with necessary reparations being made on the ‘polluter pays’ principle.

Planning legislation includes clauses relating to mineral extraction both above and below ground. The Town and Country Planning Act 1990, and amendments to this act in the Planning and Compensation Act 1991, require preparation of mineral plans by the Mineral Planning Authority. On the basis of these plans, and subsequent to necessary consultation, including with the EA, the Mineral Planning Authority can decide whether or not to grant permission to work minerals. An environmental impact assessment is often required, and permission is usually subject to a number of conditions concerning minimising environmental effects and carrying out adequate reclamation on cessation of extraction.

The legal situation with respect to working mines, or prospective mines, is thus well defined and has adequate provision to protect the environment. However, the legislation regarding abandoned mines is not so clear-cut, and statutes relating to environmental protection measures have some serious flaws.

1.3.2 Abandoned Mines

At the end of the 19th century, it became a legal requirement to notify a Mine’s Inspector of an abandonment of a mine within 2 months of ceasing operations

(Mines Inspection and Regulation Act 1870). In 1872, further coal mine regulations were introduced, which required all abandoned mines to be recorded for health and safety reasons. An accurate plan of the mine had to be submitted to the Secretary of State within 3 months for planning control and development purposes.

The legal requirement for notification of mine abandonment has remained similar since the 1870 Act, but is more concerned with the health and safety aspects of the closures than with environmental consequences. The Mines and Quarries Act 1954 requires notification, to the district Inspector of Mines, of any abandonments, either of whole mines or of veins or seams within the mines. Such notification must be made within 2 weeks of mining activity having ceased, or within a 2 week period of the mine, vein or seam having not been worked for 2 months (Section 139).

The Town and Country Planning Act 1971 provides much legislation which limits the extent to which an abandoned mine can become an eyesore (Robb 1994), but there was no obvious provision made in this Act to make discharges from abandoned mines accountable.

The Water Resources Act 1991 contains a clause which covers the event of a pollution incident from an operating mine (Section 85(1), previously Section 107(1) of the Water Act 1989). Thus liability is conferred on a person who:

> ... causes or knowingly permits any poisonous, noxious or polluting matter or any solid waste matter to enter any controlled waters.

However, within the same Act is a clause (Section 89(3)), that has been carried forward from the Control of Pollution Act 1974, that removes this responsibility from the mine owners in the case of an abandoned mine:

> A person shall not be guilty under Section 85 by reason only of his permitting water from an abandoned mine to enter controlled waters.

This ambiguity in the 1991 Act is a legal loophole which allowed mine owners to abandon mines without being held responsible for any subsequent pollution. There are essentially two ambiguities in the legislation of the 1991 Act: the term ‘abandoned mine’ is not defined, and the terms ‘causes’ and ‘permits’ are open to interpretation. The NRA (Welsh Affairs Committee 1992) was of the view that the cessation of pumping is a deliberate act which should therefore be considered to be ‘causing’ rather than ‘permitting’ a situation. However, this interpretation has yet to be put to the test in a court of law (Robb 1994). There has in fact only ever been one successful prosecution in the UK with respect to pollution from an abandoned mine (NRA 1994a). In this case, in 1981, the Crown argued under Section 22(1) of the River (Prevention of Pollution)

(Scotland) Act 1951 that the National Coal Board had caused pollution by closing Dulquharran colliery in Ayrshire, Scotland, that it had opened and operated from 1951 to 1977. There was thus irrefutable evidence that the pollution incident resulted from the fact that the Coal Board had opened, operated and then closed a mine, with the attendant dewatering and rebound of the water table that this entailed. However, discharges from abandoned mines can rarely be attributed to a sole operator, as was the situation in this incident. For example, at Wheal Jane in Cornwall, the long and complicated history of the large mining complex was considered to be unfavourable for a successful prosecution and no legal action was taken against the mine operators.

Under the Water Resources Act 1991 (Section 161(1)), the EA has powers to remedy and forestall pollution at its own expense or by recovering the cost of undertaking the work from the polluter. However, once again, there is the ambiguity that costs cannot be recovered if the pollution is ‘permitted’ rather than ‘caused’. This weakness in the 1991 Act clearly limits the powers of the EA to enforce pollution legislation in the case of abandoned mines.

Another possibility for assigning responsibility for a polluting discharge from an abandoned mine is the Salmon Fisheries and Freshwater Fisheries Act 1975, which states:

> ... that any person who causes or knowingly permits to flow, or puts or knowingly permits to put, into any waters containing fish or into any tributaries of waters containing fish, any liquid or solid matter to such an extent as to cause the water to be poisonous or injurious to fish or the spawning grounds, spawn or food of fish, shall be guilty of an offence.

Also, the Environment Protection Act 1990, Part II, Section 34 ‘Duty of Care etc. With Respects to Waste’ states:

> ... it shall be the duty of any person who imports, produces, carries, keeps, treats or disposes of controlled waste ... to take all such measures applicable to him in that capacity as are reasonable in the circumstances ... to prevent the escape of the wastes from his control or that of any other person.

Regarding the above clauses, it is not clear if a mine operator is responsible for waste discharging from a mine which he no longer operates, although it could be argued that the mining processes were responsible for ‘producing’ waste.

The announcement made by the Department of Trade and Industry, in October 1992, of plans to close 31 of the 50 British Coal-operated coal mines, served to focus attention on the potential threat posed by minewater (Allison 1994, Robb

1994, Walker 1994). The need for clarification of the legal situation regarding minewater became more significant as a result of plans to privatise British Coal.

In November 1993, a Memorandum of Understanding between the British Coal Corporation and the National Rivers Authority was drawn up (NRA 1994a). This was a five-point agreement regarding mine abandonment procedure, setting out liaison requirements and a voluntary code of practice for mine operators. A requirement of at least 14 days' notification to cease pumping at any mines was included in the Memorandum.

The coal industry privatisation bill (Coal Industry Act 1994), made no mention of responsibilities for minewater pollution or pumping. However, during the passage of this Bill through parliament, Lord Strathclyde made a statement in the House of Lords to the effect that the government would expect the Coal Authority to 'go beyond the minimum standards of environmental responsibility which are set by its legal duties and to seek the best environmental result that can be secured from the use of resources available to it for these purposes'.

A further Memorandum of Understanding was drawn up in July 1995 between the newly formed Coal Authority (the Coal Authority assumed its full range of functions on 31 October 1994) and the NRA. This document states that the Coal Authority will, except in circumstances outside its control, give at least 6 months' notice of any intention to alter or cease mine pumping operations. In the absence of any pertinent legislation, this voluntary agreement formed a crucial component of attempts to prevent pollution. In the same document, it was stated that a priority list of existing discharges would be agreed, with possible solutions identified on a priority basis according to the likelihood of environmental damage.

The NRA entered into separate agreements with R.J.B. Mining plc, which bought many of the operational mines after privatisation.

On 1 April 1996 the Environment Agency formally took over the responsibilities of Her Majesty's Inspectorate of Pollution, the NRA and the Waste Regulatory Authorities. The Agency was established by the Environment Act 1995, Part I. The Environment Act 1995 includes some provisions to clarify the situation regarding abandoned mines and responsibility for such. 'Abandoned' is defined as stopping all or some of the mining or related activities at a mine, or stopping or substantially changing operations for the removal of water from a mine. The requirement for mine operators to provide 6 months' notice of intention to abandon a mine became legally binding and failure to give the required notice became an offence (except if the mine is abandoned in an emergency to safeguard life or health and notice of such an abandonment is given to the appropriate Agency as soon as practicable). This requirement to give 6 months' notice of cessation of working any seam, vein or vein system, cessation of use of any shaft, or discontinuance of any or all operations for the removal of water from a mine was further ratified in the Mines (Notice of Abandonment) Regulations 1998 which came into force in July 1998.

Also included in the Environment Act 1995 are clauses to enable the Environment Agency to serve a 'works notice' on the person responsible for causing or knowingly permitting poisonous, polluting or noxious matter to enter, or be likely to enter, controlled waters. The notice would include details of the action to be taken to remedy the situation, and the timescale on which this action should be carried out. Failure to comply with a works notice is an offence subject to fines or imprisonment. However, a works notice cannot be served on an operator in respect of water from mines or part of a mine abandoned before 31 December 1999. The significant wording in this clause is the inclusion of 'knowingly permitting' pollution to occur in addition to 'causing' pollution – a distinction that had previously formed a line of defence against prosecution.

On 1 December 1998, Mr Battle, the Energy Minister, officially endorsed the major role played by the Coal Authority in addressing environmental challenges arising from minewater discharges from former colliery workings. In addition to continuing its work of dealing with pollution incidents from long-abandoned mines, the Coal Authority would, according to a DTI press release of 1 December 1998:

> … also give high priority to carrying out appropriate research, monitoring and preventative work and early treatment into pollution from more recently abandoned mines. These activities will be carried out in close collaboration with the Environment Agency and the Scottish Environment Protection Agency. The Coal Authority will have a separate minewater budget within its Grant-in-Aid to ensure that the Authority's other activities do not adversely affect its minewater programme, and that any emergencies can be dealt with.

This 'ring-fencing' of funds for preventative and treatment schemes for minewater both endorsed the Coal Authority's activities up to that time and facilitated budgeting and planning of future schemes, according to the priority lists.

In December 1999 a new and enhanced Memorandum of Understanding was signed between the Coal Authority and the Environment Agency, to build further on their co-operative working relationship. The new document reflects changes over the preceding five years and sets clear and common objectives for the two organisations working in partnership (Parker 2000). These include:

(1) seeking to prevent new outbreaks of pollution;
(2) enhancement of the environment by remediation of existing discharges on a prioritised basis;
(3) provision of a coherent operational framework to fulfil the above two objectives;

(4) seeking to ensure that operators deal with potential pollution from the closure of licensed coal mines in a responsible manner;
(5) furthering understanding of the processes involved in minewater rebound and sustainable prevention and treatment of minewater pollution.

Regional working groups, based on the Environment Agency regions, meet regularly and local concerns are fed into a national co-ordinating group. A series of programmes has been established to deal with four key activity areas, which are (Parker 2000):

(1) a monitoring programme, by which existing discharges and underground water levels are monitored and evaluated to establish the need for preventative or remedial action;
(2) a preventative programme, whereby schemes such as pumping are implemented to prevent future pollution where this is identified as necessary by the monitoring programme;
(3) a remedial programme to direct works to reduce pollution at significant discharges identified as priorities under the monitoring programme;
(4) operational works, under which completed schemes are managed and maintained.

Many of the treatment schemes reviewed in this report are operated and maintained by the Coal Authority and future schemes are under constant appraisal and evaluation in view of the dynamic nature of post-closure water table rebound in former coal mining areas.

1.4 LEGAL ASPECTS WITHIN A EUROPEAN UNION CONTEXT

The reduction and control of water pollution and contamination has always been an important priority in the European Union's environmental programme. Directives emanating from the EU are having an increasing influence on UK legislation. European Union controls on water pollution can be classified into three categories (NSCA 1996):

- discharge of dangerous substances (including wastes);
- quality objectives;
- by sector or industry (so far only the titanium dioxide industry has been covered).

The European Commission's Fifth Action Programme, *Towards Sustainability*, builds on current EU policies aimed at securing sufficient water supplies and maintaining and improving water quality. In particular, policies to the year 2000 have been aimed at (NSCA 2001):

- prevention of pollution of fresh and marine surface waters and groundwater;
- restoration of natural ground and surface waters to an ecologically sound condition;
- ensuring that water demand and water supply are brought into equilibrium on the basis of more rational use and management of water resources.

In September 2000 agreement was finally reached on a 1997 proposal for a Framework Directive on Water; some of the earlier Directives will be repealed 10 years after its formal adoption (NSCA 2001).

Framework Directive on Water Policy

The overall purpose of the Directive is to establish a framework for the protection of surface fresh water, estuaries, coastal waters and groundwater in the Community: the objective is to prevent deterioration and protect and enhance the status of aquatic ecosystems; to promote sustainable water consumption; and to contribute to the provision of a supply of water in the qualities and quantities needed for sustainable use of resources.

In February 2000, the Commission published a proposal for a Decision establishing a priority list of 32 hazardous substances to be included as an annex to the Framework Directive. These are primarily organic substances but include the following metals: cadmium and its compounds, lead and its compounds, mercury and its compounds and nickel and its compounds. Once agreed, discharge limits and environmental quality standards will be set for the substances, which have been selected because of the risk posed to the aquatic environment and to human health (NSCA 2001).

The Environment Agency has the task of incorporating the requirements of the Framework Directive on Water Policy into UK domestic legislation within 3 years. The approach currently being taken is to take a holistic view of the management of the water environment with the aim of achieving good ecological status in all waters. Thus the overall ecological health of water ecosystems will be considered, not just the specific, numerical values assigned to various contaminants within the ecosystem. This in turn will lead to a change in monitoring regimes defined by the EA to assess whole ecosystem health. A primary vessel for achieving this change in approach will be the compilation of Integrated River Basin Management Plans, which should be in place for water catchments throughout the UK by 2009. These plans will characterise the particular catchment and the pressures to which it is subjected, determine an appropriate monitoring regime to assess the ecological health of the water environment and formulate measures to combat pollution. This latter exercise will include both identifying primary measures, such as regulating point sources of pollution by way of discharge consents (see section 1.6), and considering supplementary measures, such as provision of best practice guidelines for land use within the catchment. The publication of a guidance booklet for farmers (*Best Farming Practices*, Environment Agency 2001a) marks the start of this more proactive approach whereby land users within the catchment are advised and encouraged to use best practice to avoid environmental harm, rather than being penalised after a pollution event. The role of the Environment Agency and Scottish Environment Protection Agency is considered further in section 1.7.

Dangerous substances

Discharges to surface and other waters are covered by the Framework Directive on pollution caused by certain dangerous substances discharged into the aquatic environment, and subsequent 'daughter' Directives. The Department of Environment Circular (DOE 7/89) gives guidance on the implementation of these Directives and emphasises that the aim should be the 'minimisation of inputs of the most dangerous substances to the aquatic environment'.

There are 129 Black List substances, that are considered to be so toxic, persistent or bio-accumulative in the environment that priority should be given to eliminating pollution by them. These substances are predominantly organic compounds; the only metals included in the Black List are cadmium and mercury, and their compounds. Control of these substances is to be achieved largely through the setting of limit values or environmental quality standards in subsequent 'daughter' Directives, thus formally giving the substance List I – or Black List – categorisation. From the original list of 129 substances, daughter Directives have now been adopted for discharges of mercury, cadmium and various organic compounds. Environmental Quality Standards relating to inputs of these substances to inland, coastal and territorial waters are defined in The Surface Waters (Dangerous Substances) (Classification) Regulations 1989. The remaining substances from the priority list which have not yet been given formal List I status in a daughter Directive are in the meantime treated as List II substances for regulatory purposes (NSCA 2001).

List II substances – the 'Grey List' – on the 1976 Framework Directive are those considered to be less harmful when discharged to water. Included in this list are metals such as Zn, Ni, Cr, Pb, As and Cu, as well as various organic compounds. It also includes those substances awaiting formal List I categorisation. The EU Directive requires Member States to establish pollution reduction programmes and to provide environmental quality objectives and standards for Grey List substances. There is also a requirement that concentrations of Grey List substances in water should not be at a level that freshwater fish cannot be supported. The Surface Waters (Dangerous Substances) (Classification) Regulations 1997 and 1998 set water quality objectives for a total of 20 List II substances. Schedules to the Regulations specify the limits of dangerous substances which must not be exceeded to meet criteria for classification as inland freshwaters and for coastal waters and relevant territorial waters (NSCA 2001). The Environment Agency is responsible for ensuring compliance with the regulations, including sampling and monitoring the effect on waters of discharges containing dangerous substances and for determining if the requirements of each classification are being met.

Quality objectives

Quality objectives are defined by a number of Directives for water intended for a variety of purposes. There are thus Directives concerning the quality of surface water intended for abstraction of drinking water, for bathing water, freshwater fish, shellfish and for water intended for human consumption.

Following agreement on the Framework Directive on water policy, the Directives on both surface and groundwater will be repealed and the proposed Directive on Ecological Quality withdrawn (thus the Freshwater Fish and Shellfish Waters Directives will also be repealed) (NSCA 2001). The Framework Directive will then provide the legislative and regulatory framework for water quality, once the previous Directives are repealed 10 years after its adoption.

1.5 INTERNATIONAL MINING LAW OUTSIDE THE EU STATES

Although mining has caused many significantly damaging effects on environments around the globe there are few internationally binding rules. The main exceptions to this are the international accords that afford the Antarctic region protection from mineral exploitation, and the United Nations Convention on the Law of the Sea (1982) which limits deep seabed mining.

Mining has impact on the environment from the exploration stage onwards. The effects on water are usually noted from the active operational stage through to the closure and in many cases beyond. Many minewater problems do not become apparent until after mine closure, in some cases only appearing several years after abandonment, which can be a significant factor in the legislative process.

The fact that awareness of environmental protection has developed in the last 20 years at national level in countries around the globe is in part due to the EU and World Bank economic aid programmes. Previously, mining had been seen as a major wealth creator and employer, with the result that the environmental degradation that ensued from the workings was considered to be of little consequence. When the provision of aid started to require sustainable environmental development, nearly every country world-wide developed legislation concerning the effects on the environment caused by working and abandoning mines and quarries. However, the historic liabilities of abandoned mines is a challenge that is as yet only beginning to be addressed.

1.5.1 Africa

Many African countries, including Bukina Faso, Botswana, Côte d'Ivoire, Ghana, Mali, Namibia, South Africa, Tanzania, Zambia and Zimbabwe, have developed core legislation to protect local environments from the operational consequences of new mining ventures. Environmental consequences of abandonment are now to be addressed at the planning stage of mine development. On top of this, any new mine is required to develop a bond that is in relation to the amount of mineral removed and in certain instances its value. This is to protect the governments in case of company failure.

In areas of political instability, compliance with these requirements is generally not observed.

Remediation of historical minewater pollution is at present not being addressed as it is in Europe, though there are some treatment schemes being put in place as part of good-neighbour arrangements.

1.5.2 Asia

The majority of Asiatic countries have developed regulations on similar lines to those in place in Africa, though some major countries have specific problems that are forcing certain environmental concerns on to their country's statutory or social agendas.

Bhutan, Brunei, Cambodia, India, Indonesia, Japan, Kazakhstan, Kyrgystan, Laos, Malaysia, Mongolia, Philippines, Sri Lanka, Uzbekistan and Vietnam all have reasonably sophisticated regulation of modern mining and its relations to water and pollution pathways. Most require full provision for reclamation of sites, and abandoned minewater remediation.

However, a few countries – Korea (both the Republic of Korea and Democratic Peoples' Republic of Korea), Myanmar, and Tajikistan – have little environmental legislation with regard to minewater. Any regulation is carried out at local level where agreements may or may not be formulated.

The largest mineral and minewater originator is China, which is at present developing legislation. However, the political structure of the mining industry hampers the nature of regulation. Modern mineral developments are of a European nature with regulation based on European legislative guidance. Older mines, however, are recognised as being of national, regional or local significance, which outlines the amount of practical regulation they can expect to have levied on them and from what level of the administrative bureaucracy.

The problems, for the Chinese, of abandoned and working minewater are only a part of the far larger problem of general contamination from historic and current industrial processes. The Chinese Environment Protection Agency is developing new legislation at regional and national levels.

1.5.3 Central and South America

Argentina, Brazil, Bolivia, Costa Rica, Ecuador, Guyana, Mexico, Peru and Venezuela have all developed significant minewater legislation over recent years, much of it in consultation with the larger mining companies. All address the issues of water management both during the operational life of the mine and at abandonment.

The central government of Brazil is at present having to shoulder the burden of historic liability of minewater pollution although they are looking towards the United States model of ‘Superfund’ or ‘polluter pays’ principle.

Chile has at present no environmental legislation appertaining to any mining operation.

1.5.4 United States

The Federal laws appertaining to mining in the USA developed from the Mining Law of 1872. Today there are more than three dozen environmental laws and regulations that cover all aspects of current mining activity. These acts include The National Environmental Policy Act which requires an interdisciplinary approach to environmental decision making, and The Federal Water Pollution Control Act (Clean Water Act) which directs standards to be set for surface water quality and for the controlling of discharges to surface waters.

In addition, each state has legislation that needs to be observed. Up to about 20 years ago most US mining legislation was developed to protect mining interests rather than the environment. An example of this is the protection of mining rights, such as one mine being debarred from discharging mine drainage into another.

Today all mining operations subscribe to a bonding fee system known as the Abandoned Mine Reclamation Fund, which is similar to the ‘Superfund’ scheme that appertains to contaminated land. These funds are collected centrally by the

Secretary of the Interior and passed to individual State Funds. This is the only piece of legislation in the world that is using present mining derived funds to pay for historic mining pollutions (Mining Control and Reclamation Act 1977). In reality, this fund is unable to cope with the liability and, in some instances, local river groups and other organisations have taken on the responsibility for the treatment themselves.

All modern mining operations, since 1977, have had to develop strategies for their eventual abandonment that will mitigate any pollution to watercourses.

1.5.5 Canada

Canada has mirrored the legal example of the United States with its mining legislation, except that there is no Reclamation Act to aid the funding of historic mining-derived pollutions.

1.5.6 Australia

Australian mining law is not prescriptive regarding abandoned minewater and liability. Mining companies are required to undertake plans for closure at the development stage of a new mine. It is felt that the mining companies themselves should be encouraged to apply innovative techniques to address the problems.

The Australian Environment Protection Agency, the community and shareholder expectation, alongside the globalisation of Australian mining company activity, encourage the industry to commit to best practice and due diligence with regard to all environmental liabilities.

The historic liabilities of mining are being addressed by the individual states.

1.5.7 Eastern Europe

Environmental legislation has been sadly lacking and the resultant degradation is evident in the waterways of most Eastern European countries. Romania, Poland, and parts of Russia have been particularly unfortunate. The World Bank and the Bank of European Reconstruction have had a huge effect in developing the will in all the countries of Eastern Europe to regulate environmental impacts of the mining industry. These two organisations have also developed an interest in the clean-up of historic minewater pollutions in the region.

1.6 CURRENT REGULATORY CONTROLS IN THE UK

The regulatory process for water quality issues within the UK is currently undergoing a major review, under the auspices of the EU-originated Framework Directive on Water Policy (see above). In the meantime, existing regulations and specified limits are applicable. These current regulatory controls are compiled in the *Pollution Handbook* (NSCA 2001). Regulations and numerical values of relevance to minewater impact can be summarised as follows.

Water Industry Act 1991

This Act consolidates various enactments relating to the appointment of water and sewerage undertakers, conditions of appointment, supply of water and the provision of sewerage services. The relevant part of this is the Water Supply (Water Quality) (England) Regulations, draft April 2000, part of which were due to come into force on 1 January 2001, with other parts being enforced at future dates up to 2004. These Regulations concern the quality of water intended for human consumption. Schedule 1 lists prescribed concentrations and values of microbiological and chemical parameters; Schedule 2 lists indicator parameters; Schedule 3 details monitoring, including parameters and circumstances for checking monitoring and annual sampling frequencies; Schedule 4 outlines the analytical methodology.

The parameters of relevance to minewater are detailed in table 1.5.

Water Resources Act 1991

The Water Resources Act (WRA) deals with some of the responsibilities of the Environment Agency, and includes the Regulations concerning Black and Grey List substances, referred to above, together with the following Regulations.

The Surface Waters (River Ecosystem) (Classification) Regulations 1994 prescribe a system for classifying inland rivers and watercourses (in England and Wales) in terms of their quality for fish populations. The classifications are:

RE1 water of very good quality suitable for all fish species;
RE2 water of good quality suitable for all fish species;
RE3 water of fair quality suitable for high class coarse fish populations;
RE4 water of fair quality suitable for coarse fish populations;
RE5 water of poor quality which is likely to limit coarse fish populations.

Eight parameters are used for assessing quality: dissolved oxygen, BOD, total ammonia, un-ionised ammonia, pH, hardness, dissolved copper and total zinc.

Table 1.5. Prescribed concentrations and values of substances, and indicator values, as defined in the Water Supply (Water Quality) Regulations 2000

Parameter	Concentration or value (max.)	Units of measurement
Schedule 1: **Prescribed concentrations and values** – Directive requirements		
Arsenic	10	µg/l
Cadmium	5.0	µg/l
Copper	2.0	mg/l
Lead		
from 25.12.2003 to 24.12.2013	25	µg/l
from 25.12.2013	10	µg/l
Nickel	20	µg/l
Nitrate	50	mg/l
Selenium	10	µg/l
Schedule 1: Prescribed concentrations and values - National requirements		
Aluminium	200	µg/l
Hydrogen ion	10.0	pH value
	6.5 (min.)	pH value
Iron	200	µg/l
Manganese	50	µg/l
Sodium	200	mg/l
Schedule 2 – **Indicator parameters**		
Ammonium	0.5	mg/l
Chloride	250	mg/l
Conductivity	2500	µS/cm at 20°C
Sulphate	250	mg/l

The Surface Waters (Abstraction for Drinking Water) (Classification) Regulations 1996 prescribe the system of classifying the quality of inland freshwaters according to their suitability for abstraction by water undertakers for supply (after treatment) as drinking water. The substances of relevance to minewater impact on surface waters and the corresponding values are shown in table 1.6.

The limits specified are the maximum allowable for each classification, and the classifications depend on the treatment required to bring the surface water up to drinking water standard:

DW1: simple physical treatment and disinfection;
DW2: normal physical treatment, chemical treatment and disinfection;
DW3: intensive physical and chemical treatment and disinfection.

Table 1.6. Criteria for classification of waters as prescribed by the Surface Waters (Abstraction for Drinking Water) (Classification) Regulations 1996

Parameter	Units	Limits		
		DW1	DW2	DW3
Nitrates	mg/l	50	50	50
Dissolved Iron	mg/l	0.3	2	
Copper	mg/l	0.05		
Zinc	mg/l	3	5	5
Arsenic	mg/l	0.05	0.05	0.01
Cadmium	mg/l	0.005	0.005	0.005
Lead	mg/l	0.05	0.05	0.05
Selenium	mg/l	0.01	0.01	0.01
Sulphates	mg/l	250	250	250
Ammonium	mg/l		1.5	4

1.7 THE ENVIRONMENT AGENCY AND THE SCOTTISH ENVIRONMENT PROTECTION AGENCY

The various Regulations regarding water quality are applied by the Environment Agency, which is charged with preventing deterioration of water quality and seeking its improvement. The EA also has a duty to promote the conservation and enhancement of inland and coastal waters, as well as various other water management roles. In Scotland, the Scottish Environment Protection Agency (SEPA) performs the equivalent function to the EA in England and Wales.

A tool of particular importance to the EA and SEPA for carrying out their role as custodians of the water environment is the issuing of Consents to Discharge. Applications for Consent to Discharge are made on a form provided by the Agency and are subject to consultation with all relevant public and private bodies. If consent is granted, conditions are applied to ensure compliance with statutory water quality objectives (including EU legislation), absolute limits set for discharges and other conditions which may include details of the nature, volume, rate, place of discharge, monitoring regime, record keeping etc. There are currently a number of different types of discharge consent, due largely to changes in legislation over many years.

Numeric Consents apply to discharges which have the greatest potential to affect the quality of the receiving waters. Numeric concentration limits are applied to individual substances or a group of substances to ensure compliance with current Regulations.

Descriptive Consents apply to discharges with a low potential to adversely affect the receiving water and which would be difficult to control by means of specific numeric values. They often define the nature of the effluent treatment plant to be used, and may specify discharge only to be allowed under specific weather conditions (e.g. storm conditions).

This system of varying stringency of consents may be formalised by the introduction of a ‘two tier’ licensing system across all the main regulatory regimes – integrated pollution prevention and control, waste, water and radioactive substances (as reported in the ENDS Report, issue 309, October 2000).

The upper tier, ‘major licences’, would be reserved for sites which ‘had a potentially significant adverse environmental impact, and/or were particularly complex or large’. A single licence would cover ‘all relevant activities’ at such sites.

‘Minor licences’ would be for sites ‘with lesser potential effect on the environment’. Standard licences would be drawn up for particular classes of site, and would be ‘simpler to apply for and to determine’.

The details and timescale for implementation of these proposals are not currently defined, and are likely to be the subject of ongoing consultation and legislation. In terms of abandoned minewater issues, the EA is currently redefining its position, but is tending towards taking a more pragmatic, less prescriptive approach. Thus, minewater discharges representing low risk, which are treated by ‘passive’ systems and are returned to the originally impacted watercourse, are likely to be subject to descriptive consents. However, discharges that are considered to represent high risk, for example where an ‘active’ treatment scheme is in place for a severely contaminated minewater discharge, or where the treated water is discharged to a previously unimpacted water course, a fully quantified, numeric consent is likely to be imposed. The EA believes that this proportionate approach to setting consents produces the greatest degree of environmental improvement overall and it does not involve those who install such remediation schemes in excessive cost. There is concern that pressure may come from the Europe Union to issue quantitative consents, in line with the Dangerous Substances Directive, for all discharges from minewater remediation schemes irrespective of the risk they pose to quality of controlled waters (Environment Agency 2001b)

Another change in the regulatory framework that is being considered and has been the subject of investigation and discussion since 1996, is the use of toxicity-based regulation (as reported in the ENDS Report, issue 309, October 2000). This would involve the regulation of effluent discharges on the basis of toxicity to wildlife rather than individual chemical constituents. This has been suggested as a result of research work carried out by the EA and SEPA with input from their counterpart in Northern Ireland and in collaboration with water companies and other industry groups. An important finding of this work was that there was little correlation between the toxicity of samples and the concentrations of chemicals, and the investigation concluded that this demonstrated the ‘limitations of a chemical-by-chemical approach and the need for a whole effluent sample direct toxicity assessment (DTA) method' (ENDS Report issue 302, March 2000).

In August 2000 the Environment Agency launched a 3 month public consultation on a proposed consenting policy for discharges of dangerous substances to surface waters. This suggested that four Control Regimes are necessary, depending on the nature of the substance, the type of discharge and the risk to the environment. The Regime applied to a particular discharge and substance will depend on:

- the nature of the substance, for example, whether it is in List I or List II of the Dangerous Substances Directive;
- the type of discharge;
- the risk to the environment.

All regimes apply the EA policy on 'no deterioration' and the 'precautionary principle'. Regime A is the most demanding of the four regimes, with Regimes B, C and D being less onerous. The specific instances where each regime would be implemented are set out on the EA website and in the consultation draft of the 'Consenting of Dangerous Substances in Discharges to Surface Waters' document.

According to the website press release of the launch of this consultation, the policy was due to start being implemented in April 2001. The process is to be rolled out gradually and fitted in with current consent review plans as far as possible, to minimise additional costs associated with this implementation.

1.8 ENVIRONMENTAL CONSEQUENCES OF MINEWATER

There are essentially four main characteristics (Kelly 1988) of contaminated minewater, not all of which will be applicable to all minewater discharges, that have the potential to affect the environment. These characteristics are listed below and will be considered in turn:

- acidity;
- ferric (Fe^{3+}) precipitates;
- trace metals;
- turbidity.

However, it should be noted that minewater is a multi-factor pollutant. The importance of each factor varies within and between affected ecosystems, and consequently it is not always possible to predict what effect a particular minewater discharge will have on the environment (Gray 1997). A review of water pollution in Scotland carried out by SEPA identifies the relative importance of classification parameters in determining the length of river polluted by minewater, with the following results: iron, 49%; invertebrates (i.e. paucity of species) 37% and ammonia 14%.

1.8.1 Acidity

Of all the environmental problems related to mine drainage, low pH is the most troublesome (Wildeman *et al.* 1991). Low pH means that not only does the discharge fail to comply with statutory requirements for drinking water, but also, as a general rule, that the solubility of trace metals will be higher, which in turn will increase their availability (Kabata-Pendias and Pendias 1992). It is important to consider 'total acidity' rather than just pH. Total acidity is a measurement of the base neutralisation capacity of a volume of water, and includes three types of acidity (Hedin *et al.* 1994):

(1) proton acidity associated with pH (a measure of free hydrogen ion concentration);
(2) organic acidity associated with dissolved organic compounds;
(3) mineral acidity associated with dissolved metals.

Minewaters generally have very low dissolved organic carbon content, so organic acidity is very low. However, many mine drainages, particularly coal mine drainage, have high concentrations of Fe^{3+}, Mn^{2+} or Al^{3+} ions, which contribute to the acidity of the water because they can undergo hydrolysis reactions that produce H^{+} ions. A significant environmental impact of high acidity is that it can destroy the bicarbonate buffer system that is an essential feature of natural waters. This buffer system involves a feedback mechanism which controls the magnitude of shifts in pH. If this system is destroyed, large shifts in pH are likely to occur which will have an adverse impact on the ecosystem. Another adverse effect of the loss of bicarbonate ions from solution is that many photosynthetic organisms use bicarbonate as their inorganic carbon source. Such organisms are therefore unable to survive in waters below about pH 4.2; this is the pH at which the bicarbonate ion is converted to carbonic acid, which in turn readily dissociates to water and free carbon dioxide. Acidity has an adverse effect on species diversity. As a general rule, the number of species present in an aquatic ecosystem reduces as the acidity increases.

1.8.2 Ferric (Fe^{3+}) precipitates

In oxidising conditions, ferrous iron oxidises to ferric iron (equation 1.2 of the pyrite weathering sequence). Ferric iron is then subject to hydrolysis reactions which result in the formation of ferric hydroxide. The solubility of ferric hydroxide is such that, under equilibrium conditions, negligible dissolved ferric iron exists (< 1 mg/l) unless the pH of the minewater is < 2.5 (Hedin *et al.* 1994). Thus, when iron-rich minewater enters receiving waters, the subsequent oxidation and hydrolysis result in precipitation of ferric hydroxide, the familiar ochrous floc that is so often associated with minewater discharges. Although this precipitate is not particularly toxic, it can have an adverse impact on the ecosystem by smothering benthic organisms and clogging up the gills of fish. If the precipitated ferric hydroxide remains suspended in the water body, light penetration will be reduced which will restrict the activity of photosynthetic organisms. There is also a tendency for As, Cu and Zn to co-precipitate with the ferric hydroxide (Johnson and Thornton 1987, Fuge 1993, Bowell *et al.* 1993). These potentially toxic elements can be considered to be temporarily unavailable to the biota, but are liable to be remobilised by a change in conditions, such as a pulse of low pH water (Tipping *et al.* 1986).

1.8.3 Trace metals

Metals, being elements, can be neither degraded nor metabolised; they are an example of ultimate persistence (Connell and Miller 1984). Once metals have been liberated from their relatively stable and unavailable state in the geosphere as a consequence of mining activities, they enter into the hydrosphere. The aquatic environment then acts as a conduit by which metals can enter the biosphere, where they may persist, through many reactions and cycles, until such time as they again become incorporated in sediments and ultimately rock. The fate of metals in the environment thus needs to be viewed within the context of this persistence and recycling, as well as on a more immediate timescale.

The impact of metals upon the biosphere is largely dependent upon the speciation of the metals, which in turn affects their solubility and availability to the biota. The solubility of trace metals in natural waters is principally controlled by pH, type and concentration of ligands and chelating agents, and the oxidation state of the mineral components and redox of the system. The free ion of a metal is usually the most toxic (Kelly 1988), but in fact most trace elements, particularly metals, do not exist in soluble forms for a long time in water. Rather, they are present mainly as suspended colloids or are fixed by organic and mineral substances (Kabata-Pendias and Pendias 1992). Thus, the mobility of metals in the environment depends on a more or less complex network of interactions between aqueous and heterogeneous chemical reactions, as well as particle coagulation and flocculation phenomena.

Hydrolysis and dissolved complexation tend to increase solubility of metals while precipitation and adsorption will delay metal availability and transport (Bourg 1995). The likely impact of contaminated minewater on receiving waters is therefore not immediately predictable just on the basis of the concentrations of potentially toxic elements that are present in the water. Detailed analysis is required of the forms in which these elements are present. There are many texts dealing with the detailed chemistry of metal speciation and availability in the aquatic environment (Salomons *et al.* 1995, Connell and Miller 1984, Forstner and Wittman 1981) and it is not intended to consider this aspect in any depth here.

All metals are toxic if present in an organism in excess, but some metals are also essential trace elements, taken up actively by the organism. Most micronutrients owe their essentiality to being constituents of enzymes and other important proteins involved in key metabolic pathways. Of these, Ca, Fe and Zn are needed in the largest amounts. Selenium and Cu, for example, are also essential elements but are needed in much smaller quantities. Some metallic elements are taken up by organisms by virtue of their position on the periodic table rather than because they are necessary for the organism's metabolism. This is due to the fact that the adsorption tendency of metals occupying the same relative position on the periodic table will be similar. Thus Zn and Cd behave similarly, so Cd is taken up with Zn, although Cd has no known role in living systems (Connell and Miller 1984).

Toxicity mechanisms for metal ions can be broadly divided into three categories (Ochiai 1977):

(1) blocking of the essential biological functional groups of biomolecules (e.g. proteins and enzymes);
(2) displacing the essential metal ion in biomolecules;
(3) modifying the active conformation of biomolecules.

Table 1.7 gives toxicity sequences for metal ions likely to be found in contaminated minewater. Similar patterns can be seen even for dissimilar organisms, which fact has been attributed to the classification of metal ions according to their binding preferences (Connell and Miller 1984). This classification is:

- class A – oxygen-seeking;
- class B – nitrogen or sulphur-seeking;
- intermediate.

Table 1.7. Toxicity sequences for metal ions in a range of organisms (modified from Connell and Miller 1984)

Organisms	Sequence[1]
Algae	Cu > Cd > Fe > Zn > Ni > Mn
Fungi	Cu > Cd > Ni > Zn > Fe
Flowering plants	Pb > Cu > Cd > Ni > Zn
Protozoa	Pb > Cu, Cd > Ni > Mn >Zn
Annelida	Cu > Zn > Pb >Cd
Vertebrata	Cu > Pb >Cd >Al >Zn > Ni
Mammalia	Cd > Cu, Pb, Mn, Zn, Ni, Fe >Al

[1] Atomic symbols represent tripositive ion for Al and dipositive ions for all other metals.

Class B metals are the most toxic, and include Cu^+ and Pb^{4+}. These elements exhibit a broad range of toxicity mechanisms, involving all of the three categories listed above. Intermediate metals, which include Cu^{2+}, Cd^{2+}, Fe^{2+}, Fe^{3+}, Mn^{2+}, Ni^{2+} and Zn^{2+}, are intermediate in toxicity rating between Class B and Class A metals. Their toxicity is generally due to replacement of ions in biomolecules (category 2 above), for example, the replacement of Zn^{2+} by Ni^{2+} in enzymes with a resultant loss of enzymatic activity. Toxicity among class A metals is less than for the other two groups and is also due largely to the mechanism shown as category 2.

The above consideration of relative toxicity can only be considered as a generalisation, since the development of tolerance mechanisms for specific ions in various organisms can result in exclusion or innocuous accumulation of potentially toxic ions.

1.8.4 Turbidity

The smothering and light-reducing effect of iron hydroxide flocs has already been mentioned when considering the impact of ferric precipitates on the aquatic environment. Other sources of turbidity associated with mine workings are the following:

(1) overflow from tailings dams;
(2) possible occurrence of calcium sulphate floc in waters that have been treated with lime;
(3) erosion from untarred haulage roads around mines;
(4) wash-out of fines from spoil heaps, which may continue long after mining has ceased, especially if vegetation establishment is poor.

1.8.5 Other factors

The foregoing discussion provides a broad overview of the ways in which contaminated minewater can affect the environment. Another factor which might sometimes play a role in harming the biota is the increased oxygen demand that may result either chemically or biologically. Chemical oxygen demand can be due to the presence of ferrous iron. Biological oxygen demand may be transported to downstream environs in the form of undecomposed organic matter, decomposition having been prohibited by acidity.

In addition to the reasons given above, contaminated minewater can be considered to have an adverse effect on the environment from an aesthetic viewpoint. This covers aspects such as the unsightliness of ochre-covered river beds and tracts of damaged vegetation, as well as the reduced aesthetic appeal (not to mention reduced ecological robustness) of a species-poor ecosystem. It is thus imperative that all possible steps are taken to ensure that contaminated minewater is prevented from harming the environment, or at least that the impact is minimised as much as is practicable.

Another consequence of mine abandonment (particularly where whole coalfields are involved, e.g. the Durham coalfield) that could have potentially drastic effects on land and property is the regional rebound of the water table following cessation of pumping. This could result both in widespread flooding and in subsidence caused by pillar collapse. The uncertainty surrounding the possible environmental impact of regional water table rebound is such that it seems preferable to continue pumping indefinitely, rather than face the widespread, and possibly catastrophic, consequences of cessation of pumping (Younger 1994).

2

Treatment options

Implementation of control measures for contaminated minewater can be carried out at various stages in the contaminated minewater generation process (Broughton and Healey 1992, Sengupta 1993):

- prevent/control contaminated minewater generation reactions (at-source control);
- prevent/ control migration of contaminants;
- collect and treat contaminated minewater.

Parallels can be drawn between this staged approach to minewater treatment and the new UK contaminated land legislation (Part IIA of the Environmental Protection Act 1990). This legislation takes a risk-based approach with the concept of *pollutant linkage* as fundamental to understanding the variety of remedial measures that are available. Thus, an area of contaminated land is only considered to constitute a risk if there is a plausible relationship (*pollutant linkage*) between a source, a pathway and a receptor, and remediation can be viewed as risk management where the objective is to control or break the source-pathway-receptor chain in order to mitigate any potential or actual risks (Martin 2001). In the context of minewater treatment, prevention/control of minewater generation reactions can be considered to

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be tackling the pollutant linkage by remediating at source; migration control is a form of pathway modification. Collection and treatment can be considered to be the last remaining option to break the linkage and protect the receptor if neither the source nor the pathway can be sufficiently controlled. Thus, although the Contaminated Land Regime Part IIA legislation is not directly applicable in most instances of minewater, a similar rationale can be applied in the approach to dealing with both types of contamination (i.e. in land water and minewater).

2.1 AT-SOURCE CONTROL

The preferred course of action is to prevent the oxidation of sulphide minerals. If primary control of this nature is carried out there is no generation of contaminated minewater and thus no need for the inconvenience and expense of treatment. Oxidation is prevented by eliminating one or more of the essential components, or by controlling the environment around the sulphide grains. The essential components for weathering of sulphides are:

- rock or waste containing reactive sulphides;
- oxygen;
- water.

The environmental factors that affect the rate of weathering of sulphides include:

- bacterial activity;
- temperature;
- pH.

By consideration of the above-listed factors, the following strategies for elimination or reduction of contaminated minewater generation can be formulated.

2.1.1 Sulphide removal or isolation

If sulphide minerals in waste rock or tailings are removed, reduced or isolated (Evangelou 1994, Georgopoulou *et al.* 1995), then oxidation of these minerals will not occur. High sulphide waste can be separated from the bulk of the mine waste and handled differently, with the aim of reducing acid-generation (Knapp and Welch 1991). Techniques are being developed to coat sulphidic minerals in mine tailings with a protective, inert layer of silicate (Mitchell *et al.* 1999, Fytas *et al.* 2000) or to treat pyritic wastes with other chemicals to reduce their activity (for example the DuPont process (De Vries 1996) that uses potassium permanganate and the UNR process (Mehta *et al.* 2000) that uses oxides of calcium and magnesium). Other proprietary products are being developed to coat and stabilise sulphidic minerals and metal-rich sludges, e.g. Envirobond^{TM} (Gobla *et al.* 2000, Littlepage *et al.* 2000).

A novel approach to prevention of AMD is the electrochemical manipulation of large bodies of sulphidic tailings to improve the thermodynamic stability of the sulphide minerals and thus prevent their oxidation (Brousseau *et al.* 2000).

2.1.2 Exclusion of water

Total exclusion of water to prevent acid generation is a difficult option but it may be possible in some circumstances. All water – surface water, infiltration due to precipitation and groundwater – must be prevented from coming into contact with the sulphidic rock or waste for this to be a viable control measure. Such dry covers may comprise cementitious material (Jones and Wong 1994), composite soil material (Bell *et al.* 1994, Swanson *et al.* 1995), organic material (Tremblay 1994, Pierce *et al.* 1994, Stogran and Wiseman 1995, Peppas *et al.* 2000), milling wastes (Aachib *et al.* 1994) or synthetic membrane material (Sengupta 1993). However, the longevity of such covers is uncertain since in the long term degradation is likely to occur, resulting in the infiltration of water and subsequent onset of acid generation. Steps can also be taken, as part of an overall mine closure strategy, to reduce the amount of water entering disused workings and backfilled opencast workings (Geldenhuis and Bell 1998).

2.1.3 Exclusion of oxygen

The elimination of oxygen from waste rock and exposed rock surfaces would prevent the oxidation of sulphide minerals or reduce the rate of contaminant production. Although it is possible that acid can be generated under anaerobic conditions (Sengupta 1993), this has not found to be significant in mining wastes. Significant reduction in oxygen level can be achieved through the placement of a cover with extremely low oxygen diffusion characteristics. Such covers could include those mentioned in section 2.1.2 for exclusion of water, and also water itself (Real and Franco 1990, Knapp and Welch 1991, Fraser and Robertson 1994, Lapakko 1994, Allan 1995). Wetland establishment on tailings dams is another means of excluding water on a long-term basis (Treacy and Timpson 1999, Beckett *et al.* 1999, O'Sullivan *et al.* 1999, Eger *et al.* 2000).

A technology is offered by the Green World Science company (www.greenworldscience.com, based in Idaho) that partially replaces oxygen in potential AMD-producing material with carbon dioxide, thereby inhibiting formation of AMD.

2.1.4 pH control

If the pH of the water is maintained above pH 7, acid generation can be inhibited. The pH may be controlled by the addition of alkaline materials to potentially acid-producing wastes. Blending of acid-consuming waste with acid-producing waste can be a successful approach (Day 1994). Adding and mixing imported alkaline material such as ground limestone or apatite (Renton *et al.* 1988) can be an efficient procedure. Co-disposal of tailings with suitable materials, for example alkaline residues such as cement kiln dust and residues from bauxite processing can help reduce acid mine drainage generation within the tailings (Fortin *et al.* 2000). Use of electric utility wastes, such as fly ash, as a grout in potentially acid-producing zones of abandoned, reclaimed or active coal mines has been shown to be an economical and effective way of ameliorating acid mine drainage (Kim 1999, Kim 2000, Rafalko and Petzrick 2000). These latter two examples have the added advantage of addressing two waste disposal/environmental problems at once.

2.1.5 Control of bacterial action

The significance of bacterial action in catalysing the oxidation of pyrite has been noted previously (section 1.1), and thus the use of bactericides can substantially reduce acid generation. Suitable bactericides have been found to be anionic surfactants (Watzlaf 1988, Benedetti *et al.* 1990, Sobek *et al.* 1990, Parisi *et al.* 1994), organic acids (Backes *et al.* 1988), food preservatives (Singh *et al.* 1990, Sengupta 1993) and organic waste products such as alkaline paper mill waste (Bellaloui *et al.* 1999). Proprietary bactericide products are available on the market for use in active stockpiles, waste dumps etc. and during reclamation implementation (e.g. ProMac bactericide produced by MVTechnologies, Inc., Ohio).

Alternatively, bacterial action that has a remedial effect on acid mine drainage can be encouraged *in situ*, so that the problem is tackled before the minewater migrates from the mine workings. This approach has successfully been employed in Montana, where a biological reactor was created within flooded subsurface mine workings by addition of organic matter in the shaft (Canty 1998, Canty 2000). Action of sulphate reducing bacteria reduced aluminium, cadmium, copper and zinc concentrations in the effluent minewater. Iron concentrations, however, were increased due to the reduction of ferric iron within the anaerobic environment and its mobilisation as ferrous iron. Zinc concentrations have successfully been reduced in an open pit lake in central Finland by addition of organic matter to stimulate bacterial activity (Riekkola-Vanhanen and Mustikkamaki 1997).

Methods for at-source control of contaminated mine water are reviewed by Kleinmann (1990), Sengupta (1993) and Ziemkiewicz and Skousen (1996). A comprehensive review of options, with case studies, is presented by Skousen and Ziemkiewicz (1996).

2.2 MIGRATION CONTROL

If it is not possible to prevent acid generation, the next level of control is to prevent or reduce the migration of the contaminated minewater into the environment. This is achieved principally by preventing ingress of water to the sulphidic rocks which are the source of potential contaminated minewater. However, increasing attention is now being paid to the development of 'zero-discharge mining' (Cale 1994).

Water entry may be controlled by (Broughton and Healey 1992, Sengupta 1993):

- diversion of all surface water flowing toward the sulphidic rocks (Fernandez-Rubio and Carvalho 1993);
- prevention of groundwater flow into the area of sulphidic rocks (Snow 1990, Scheetz *et al.* 1995, Scheetz *et al.* 1998);
- prevention of infiltration of precipitation into the area of sulphidic rocks (by use of covers or cementitious grouts);
- controlled placement of acid-generating wastes.

The feasibility of implementation of any of the above measures is strictly site specific. Although such measures can provide an efficient and cost effective way of minimising the impact of contaminated minewater on the environment, they often cannot be employed, especially in the case of complex, abandoned, underground mine workings, such as predominate in the UK.

A technique that is becoming increasingly used is the construction of permeable reactive barriers (PRBs) to intersect migrating, minewater-contaminated groundwater. A PRB is a permeable zone containing or creating a reactive treatment area oriented to intercept and remediate a contaminated groundwater plume. Thus, rather than employing grouting techniques to create impermeable barriers to flow, a permeable material is used that will have an ameliorative effect on the migrating plume. Two basic designs are being used in full-scale implementation of PRBs:

(1) funnel and gate; and
(2) continuous trench.

In the former, the contaminated plume is directed, by means of a funnel comprising impermeable walls, to a gate or gates containing the permeable reactive media. In the latter, a continuous trench containing permeable reactive media intercepts the whole contaminated plume (US EPA 2000b).

One of the first investigations of the potential for this technique was carried out in an aquifer down-gradient of a mine tailings impoundment at the Nickel Rim mine site at Sudbury, Ontario. Tests, using various sources of organic carbon to induce bacterially mediated sulphate reduction and subsequent precipitation of metal sulphides, were carried out in a small-scale test-cell and gave very promising results (Blowes *et al.* 1995). This test-work was followed by a full-scale reactive barrier, in which the organic carbon source was municipal compost, mixed with pea gravel to ensure sufficient permeability (Benner *et al.* 2000). This has given rise to dramatic improvements in the groundwater chemistry, with up to 90% decrease in Fe concentration, and an increase in alkalinity, both as a result of sulphate reducing bacteria activity and precipitation of iron sulphides. The main controls on barrier performance were found to be organic carbon reactivity, groundwater temperature and variable residence time within the barrier. Successful remediation of groundwater contaminated with cadmium, copper, nickel, lead and zinc has also been achieved using a similar technique and organic carbon source (McGregor *et al.* 2000). The technology is patented by Envirometal Technologies, Inc. (based in Waterloo, Canada) and has been investigated as part of the NATO/CCMS Pilot Study on Evaluation of Demonstrated and Emerging Technologies for the treatment of Contaminated Land and Groundwater (Phase III) (Burmeier 1998) and described in US EPA publications (US EPA 1998, 1999, 2000a and b).

Alternative substrates for use in reactive barriers have been investigated, with the organic carbon source being provided by varying amounts and combinations of sewage sludge, leaf mulch, wood chips, sheep manure, sawdust and cellulose (Waybrant *et al.* 1998). Statistical techniques have been applied to determine the most appropriate reactive mixture to optimise sulphate reduction (Cocos *et al.* 2000). Use of non-organic substrates in permeable reactive barriers has been investigated for the *in situ* removal of arsenic (basic oxygen furnace oxide, zero valent iron and activated alumina, McRae *et al.* 1999) and uranium (zero valent iron, Morrison *et al.* 2001). These latter workers have also emphasised the importance of understanding the geochemistry (or biogeochemistry in the case of organic substrates) occurring within the reactive medium, so that geochemical modelling can be carried out. This would enable residence times, flow paths etc to be optimised and also provide an estimation of the expected longevity of the treatment system. Work on a two-column siderite/limestone reactor for removal of arsenic and cadmium from wastewaters led the authors to suggest that these two materials could be used in a reactive wall setting to passively achieve the same remediation purpose (Wang and Reardon 2001). Modelling studies of flow through quartz sand and calcite in laboratory column studies are reported by Fryar and Schwartz (1994), who conclude that ferric iron, in the form $Fe(ClO_4)_3$, is effectively and rapidly removed from a dilute solution without fouling of the reactive surfaces.

Reactive barriers are being increasingly used in the UK, further details of which will be found in the case studies in Chapter 5. In particular, a reactive barrier is currently in the planning stage at Shilbottle, Northumberland, and a two-stage subsurface reactive barrier (first trench filled with wood bark, farmyard manure and limestone, and the second trench filled with limestone) was constructed at Renishaw Park, South Yorkshire, in 1999, to treat acidic spoil leachate.

Passive, gravity-flow geochemical reactors are also being developed; they are considered in the following section.

2.3 COLLECTION AND TREATMENT

The two primary aims of the treatment of contaminated minewater are to:

1. neutralise acidity; and
2. remove metals.

In addition it is necessary to remove suspended solids, which cause environmental problems due to turbidity, and to prevent discolouration of receiving waters by ochrous precipitates. This latter feature of contaminated minewater is detrimental both physically and aesthetically.

The end-products of minewater treatment will be water, which may have a varying degree of remaining contamination, and some form of solid residue or sludge. The efficacy of a treatment process can be considered in terms of the chemistry of the resultant water (whether it requires a further polishing stage or is suitable for discharge to a receiving watercourse, or perhaps re-use), the nature of the sludge (volume, toxicity, long-term stability, disposal requirements) and whether any marketable products can be recovered from the treatment process to set against costs (for example recoverable metals or potable water).

The types of available treatment can be divided into two broad categories:

1. active systems, which require continuous operation and maintenance;
2. passive systems, which are intended to be self-sustaining after an initial start-up period.

Active systems are often used in conjunction with operating mines, when there is a workforce on site to implement the treatment and the financial ability to fund it. However, in the case of decommissioned and abandoned mines, passive treatment systems have an obvious attraction since they do not have the ongoing technical and financial liabilities of active treatment. The following two chapters summarise the main characteristics of these systems.

3

Active treatment

This chapter considers the options available for active treatment of minewater, by first considering the general principles of the treatment method and then considering details and specific proprietary products. The relative merits and limitations of the techniques are considered. Contact details for providers of the various technologies are included in appendix B. For classification purposes the technologies considered are grouped according to the type of treatment, although there is sometimes overlap if a particular technology applies more than one treatment principle. However, technologies can be broadly subdivided into:

- pH modification;
- ion exchange;
- biology-based treatments;
- other adsorption treatments;
- electrochemical treatment;
- physical process technology.

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3.1 PH MODIFICATION

The conventional active treatment of contaminated mine water is that comprising neutralisation and precipitation. Materials used to achieve this neutralisation are most commonly lime (CaO), limestone ($CaCO_3$), caustic soda (NaOH) or sodium carbonate (Na_2CO_3), although other chemicals can be used (see table 3.1).

Table 3.1. Chemicals used for neutralisation of acid mine drainage (Skousen *et al.* 1996)

Chemical name	Chemical formula/active ingredient	Comments
Calcium oxide (quicklime, caustic lime)	CaO	Very reactive
Calcium carbonate (limestone)	$CaCO_3$	Used most successfully in anaerobic situations
Calcium magnesium carbonate (dolomite)	$(Ca, Mg)\ CO_3$	Similar to limestone, less reactive
Calcium hydroxide (hydrated lime)	$Ca(OH)_2$	Requires extensive mixing
Sodium hydroxide (caustic soda)	NaOH	Very soluble, available in solid form or in solution
Sodium carbonate (soda ash)	Na_2CO_3	Usually in briquette form, used for remote locations
Potassium hydroxide	KOH	Similar to sodium hydroxide
Magnesium oxide (magna lime)	MgO	Similar to calcium oxide
Magnesium hydroxide	$Mg(OH)_2$	Similar to calcium hydroxide
Ammonia	NH_3 or NH_4OH	Reactive and soluble, can be supplied as aqueous ammonia
Calcium peroxide (trapzene)	CaO_2	Neutraliser and oxidant, supplied as a powder or as briquettes
Kiln dust	$CaO.Ca(OH)_2$	Waste product of cement industry, contains lime
Coal-use residues (fly ash, FBC residues)	$CaO.CaCO_3$	Many coal-use residues contain lime or carbonate, neutralisation potential varies between sources

The basis of pH modification treatment is to raise the pH of the AMD causing first iron, and then other metals, to precipitate out of solution. The minimum pH values for the complete precipitation of metal ions as hydroxides are shown in table 3.2.

Table 3.2. Minimum pH values for complete precipitation of metal ions as hydroxides

Metal ion	Minimum pH
Sn^{2+}	4.2
Fe^{3+}	4.3
Al^{3+}	5.2
Pb^{2+}	6.3
Cu^{2+}	7.2
Zn^{2+}	8.4
Ni^{2+}	9.3
Fe^{2+}	9.5
Cd^{2+}	9.7
Mn^{2+}	10.6

Use of waste products is an attractive proposition in that it solves two waste disposal/environmental problems at once, but there is a concern about the inevitable variation in the availability and content of such reagents; the process design will need to be sufficiently robust to take this into account.

Use of conventional lime dosing treatment has the following advantages:

- It is a tried and tested technology.
- It is effective for the treatment of highly acidic wastes.
- Treatment is largely unaffected by seasonal temperature fluctuations.
- It requires relatively straightforward plant and operation.
- Decant water from the settling dam is of sufficient quality to be discharged into surface watercourses.
- The process can accommodate changes in water quality or quantity by relatively easy adjustments of the operating parameters.

However, there are also numerous drawbacks to liming:

- Equipment maintenance is relatively high due to scaling.
- The high pH that is needed to remove metals such as manganese may cause remobilisation of other metal hydroxides (e.g. aluminium).
- The sludge is chemically complex and unstable, making long-term disposal problematic.
- Sludges are low density and gelatinous; the resulting large volumes are difficult and expensive to handle and dispose of.
- The sludges generally have no commercial value – reclamation of metals is uneconomic due to the complexity of the mixture and the large excess of lime that is generally used to ensure complete precipitation.

The relative merits of the four commonest chemical neutralisation agents (lime, limestone, sodium carbonate and caustic soda) can be examined in terms of their properties based on calcium versus sodium and carbonate versus hydroxide, and are considered below (Skousen 1996).

Calcium versus sodium

Factors that may influence the selection of a calcium or a sodium compound for AMD treatment are listed in table 3.3.

Table 3.3. Factors influencing selection of calcium or sodium compound for AMD treatment (Skousen 1996)

Factor	Calcium	Sodium
Solubility	Slow	Fast
Application	Requires mixing	Diffuses well
Hardness	High	Low
Gypsum formation	Yes	No
High total suspended solids or clay particles	Helps settle clay	Disperses clay particles and keeps clay in suspension
Chemical cost	Lower	Higher
Installation and maintenance costs	High	Low

The calcium compounds are less expensive than sodium compounds, but they have low solubility in water (reaction rate is slow). Therefore, calcium compounds are generally used in large treatment systems where electricity is available to improve the reactivity of the calcium material (e.g. where aeration and mixing units are operated). If sulphate concentrations are above 2000 mg/l, then the calcium compounds will react with the sulphate to form anhydrite or insoluble gypsum. This calcium sulphate precipitate may clog pipes or other structures used to convey the water to the receiving stream after treatment. Some proprietary technologies make use of this precipitate as a marketable byproduct (e.g. Gyp-Cix, see below). The high calcium concentration in the treated water also creates greater hardness than sodium compounds.

The use of magnesium oxide as a neutralising agent is reported by Bekin and Matsuoka (1998) who conclude that it is an effective reagent for lowering iron concentration in minewater producing a relatively dense sludge.

Carbonate versus hydroxide

The pH of water during treatment affects the types and amounts of metals that can be removed or precipitated from the water. Carbonate compounds do not raise the pH of the water above 8.5, whereas hydroxide compounds can raise pH to above 10. Due to the different solubilities of metal hydroxides (see above), the metals present in the AMD will dictate which chemical reagent is most appropriate to raise the pH sufficiently. Thus, if ferric iron is the main constituent of the AMD, it is possible to remove it with a carbonate material, whereas manganese generally requires the higher pH obtainable with a hydroxide material.

Ammonia

Ammonia can be used for pH modification and is effective for manganese removal. The most promising aspect of ammonia for AMD treatment is its cost when compared with caustic soda (Skousen *et al.* 1996). However, it is a potentially hazardous material, both in terms of its handling requirements and possible harmful effects on the environment if strict controls are not enforced.

The selection of an appropriate chemical for pH modification is thus a matter of considering the chemistry of the water to be treated and the operational requirements and costs of the various chemicals available. In some instances, more than one chemical may be used to achieve the optimum treatment in terms of cost effectiveness and reliability.

Aeration

The valence state of the metal greatly affects the solubility of the hydroxide, as shown in the following order:

$$Fe(OH)_3 = MnO_2 > Al(OH)_3 >> Fe(OH)_2 > Mn(OH)_2$$
Insoluble – > Soluble

Since $Fe(OH)_3$ and MnO_2 are insoluble, whereas $Fe(OH)_2$ and $Mn(OH)_2$ are soluble, the best route to achieve a stable precipitate is to oxidise the iron and manganese prior to precipitation. Oxidation may be achieved by passive, physical means, i.e. cascading the minewater. However, if there is insufficient head differential or land area to facilitate cascades, or if the metal content of the minewater is so high that cascading alone would not introduce sufficient oxygen (a well-designed cascade system should introduce enough oxygen to oxidise about 50 mg/l of Fe^{2+}) oxidation may be achieved by addition of oxidants (peroxide, ozone (Sato and Robbins 2000), chlorine compounds or potassium permanganate), catalysts (trapzene), by in-line pre-aeration (Ackman and Kleinmann 1984, Jageman *et al.* 1988, Kleinmann 1989, Ackman 2000) or by using micro-organisms in a bioreactor (Omura *et al.* 1991, Diz and Novak 1998, Diz and Via 1998).

There is growing interest, particularly in the USA, in using alternative energy sources to fuel the aeration process, for example, use of windmills to inject air into minewater holding ponds. It is also noted (Watzlaf pers. comm.) that equally important to introducing air into minewater for oxidation purposes, an aeration step facilitates degassing of carbon dioxide from the minewater. This effects a rise in pH and a consequent increase in the iron oxidation rate.

Coagulation and flocculation

The sludge that is formed by alkaline addition is generally of a low density, with a solids content of as little as 2%; it is therefore very bulky for disposal purposes. Also, the precipitated hydroxides tend to have a small particle size and thus need substantial amounts of time to settle out in the settlement tanks or tailings dam.

The settling characteristics of the precipitated material can be improved by the use of chemicals to aid in the coagulation and flocculation processes, which in turn improve the settling efficiency. Coagulation reduces the net electrical repulsive forces on the surface of particles, thereby promoting consolidation of small particles into larger particles. Flocculation aggregates or combines small particles by bridging the space between particles with chemicals. Thus, both these processes result in larger particles that will settle more readily than the original small particles. Table 3.4 lists the various chemicals that are available to perform these functions in water treatment. The problem of low sludge density can also be overcome by using a proprietary process to increase particle size or alter the consistency of the sludge (e.g. high density sludge process, KEECO KB-1, see below) or by physical or mechanical dewatering of the sludge by means of filter presses or centrifuges, for example.

High density sludge process

The high density sludge process (HDS) is a process that has been developed to achieve greatly increased sludge densities compared to those resulting from conventional pH modification systems. The HDS process uses conventional reagents, but also includes a sludge recycling step which increases the particle size of the precipitated material and thus results in a denser sludge. The earliest documented uses of the HDS process were in the 1970s (Kuit 1980, Murdock *et al.* 1994, Zick *et al.* 1998). Ongoing refinement and development of the process has

Table 3.4. Chemicals for coagulation and flocculation in water treatment (Skousen *et al.* 1996)

Chemical	Chemical formula	Comments
Coagulants		
Alum (aluminium sulphate)	$Al_2(SO_4)_3$	Acidic, forms $Al(OH)_3$, which attracts other metals for precipitation
Copperas (ferrous sulphate)	$FeSO_4$	Acidic, usually slower reacting than alum
Ferric sulphate	$Fe_2(SO_4)_3$	Ferric products react faster than ferrous. Ferric sulphate coagulates particles over a larger pH range than ferrous sulphate and the precipitate is a heavy quick-settling floc.
Ferric chloride	$FeCl_3$	Also used as an oxidant, reacts similarly to ferrous sulphate
Sodium aluminate	$NaAlO_2$	Alkaline coagulant, high cost
Magnesium chloride	$MgCl_2$	Also used as an oxidant
Aluminium chloride	$AlCl_3$	Also used as an oxidant
Calcium chloride	$CaCl_2$	Also used as an oxidant
Flocculants		
Mineral flocculants:		
Activated silica	Na_4SiO_4	A negative charge on the surface. Attracts cations and metal hydroxides to form a dense, stable floc.
Clays	Bentonite	Negatively charged surface
Metal hydroxides	$Al(OH)_x$, $Fe(OH)_x$	Usually a positively charged surface
Natural flocculants:		Natural flocculants are only used in specialist situations, and are expensive. None are known to have been used in AMD treatment
Starch derivatives	Corn starches	Gelatinous materials
Polysaccharides	Guar gums	Sticky substances
Sodium alginates	$C_6H_7O_6Na$	Food thickeners
Synthetic flocculants:		Very large polymer molecules that produce highly charged ions when dissolved in water. Needed in much smaller quantities than mineral and natural flocculants
Anionic		Negatively charged surface
Cationic		Positively charged surface
Polyampholytes		Have both positive and negative charges on surface, based on pH

led to the existence of several proprietary processes, some patented, that use the same principles; these can be summarised as follows:

- pH modification is achieved with conventional chemicals (e.g. lime, caustic).
- Recirculation of sludge results in the formation of large, crystalline particles that have a low affinity for water and have much improved settling and dewatering characteristics as compared to conventional lime dosing.
- An aeration step is included to oxidise ferrous iron to ferric iron.
- Flocculant is added to improve the settling characteristics of the sludge.
- Water/sludge separation is carried out in a clarifier/thickener, with a proportion of the sludge being recirculated to the start of the process.
- Sludge densities of 20%, or more, solids are achieved without any further thickening or dewatering steps.

The different proprietary processes vary in the order in which various steps are carried out, the number of treatment steps or other operational differences, such as operational pH; however, there is also flexibility and variation within each proprietary process since the treatment process will need to be tailored to a particular minewater problem. A comparison of four HDS processes is made by Zinck and Griffith (2000), and a summary is given below.

Cominco process

The minewater is generally fed into a rapid mix tank (RMT; see figure 3.1), where it is brought into contact with a lime/sludge slurry to bring the pH of the combined slurry to 9.5–9.8. The RMT is often used to offer better pH control in the process, but it is not essential. The lime reactor tank (R3) has a retention time ranging from 30 to 90 min. Air is normally sparged in the lime reactor for ferrous oxidation. The polymer tank is used to contact the polymer to the precipitates for floc formation.

The sequence of reactor vessels in this process is considered in the literature as the 'conventional' HDS process (McLaughin *et al.* 1996, McCandless and Kuit 1998, Aube and Zinck 1999, Zick *et al.* 1998).

A portion of the sludge from the clarifier underflow is recycled to the lime/sludge mix tank. The sludge recycle ratio is controlled by the feed rate and a predetermined ratio of solids recycled to solids formed. This ratio is typically between 10:1 to 30:1, or 10 to 30 kg of solids recycled for each kg of solids formed in the process. This means that at any given time, at least 90% of the solids in the lime reactor are from recycled sludge. The lime addition is controlled to keep pH at the desired value (9.5), measured either in the RMT or the polymer tank.

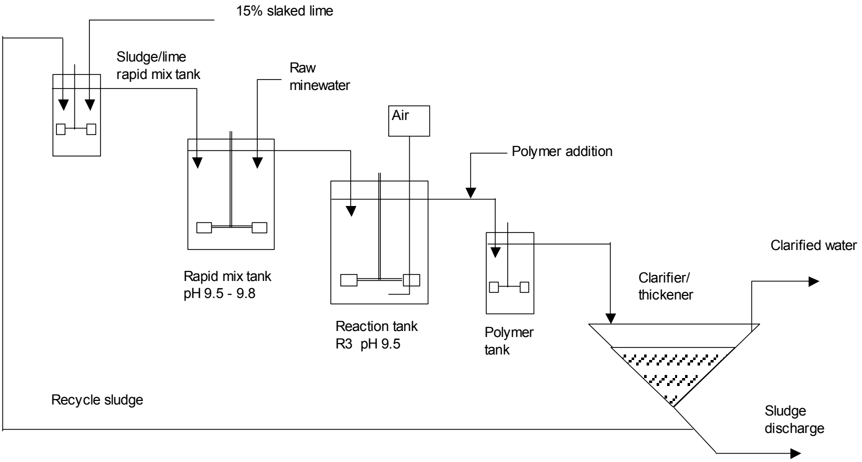

Figure 3.1. Cominco or 'conventional' HDS process.

Tetra HDS Process

Minewater is pumped into the first reactor (R1; see figure 3.2), and the pH is adjusted to 3.5 with recycled sludge from the thickener underflow. The amount of sludge recycled to R1 is controlled by the pH in R1. Reactor tank 1 overflows to the second reactor (R2), where air is sparged to oxidise ferrous iron to ferric. The pH in R2 is adjusted to 9.5 with the overflow from tank 1 and slaked lime from the sludge/lime rapid mix tank. Sludge is added to the sludge/lime tank at a pre-set rate, while the amount of lime added to the sludge/lime tank is controlled by the pH in the R2 tank. Reactor tank 2 overflows to the retention tank (R3) to complete metal precipitation and iron oxidation, with aeration. Polymer is added to the R3 overflow line, and the slurry enters the short retention polymer tank. The slurry from the polymer tank then flows by gravity into the clarifier.

Geco Process

In the Geco process (Aube and Payant 1997), the minewater is fed to the first reactor (R1; see figure 3.3) where it is contacted with recycled sludge. The pH in R1 is not controlled, as the recycle ratio is set to produce a specific recycle solids to feed solids ratio, and it is this that determines the pH in R1. The rapid mix tank is used for improved pH control and is not an essential part of the system. The third reactor tank (R3) completes the final neutralisation step.

Aeration is carried out in R3 to oxidise ferrous iron. The polymer tank is for mixing the flocculant with the slurry.

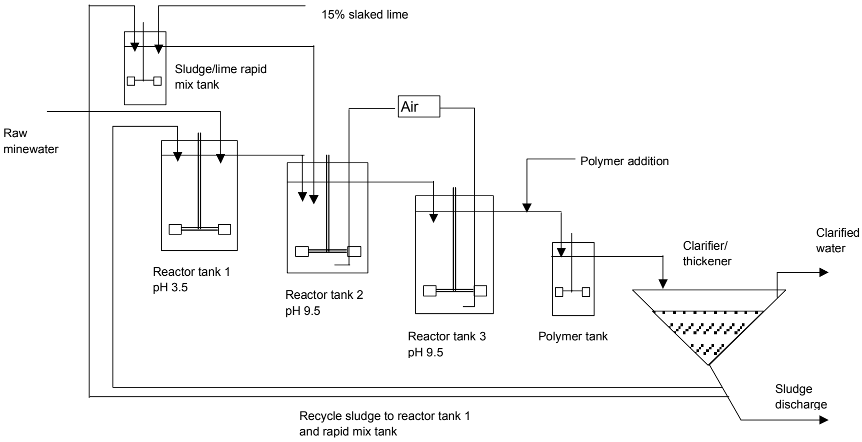

Figure 3.2. Tetra HDS Process.

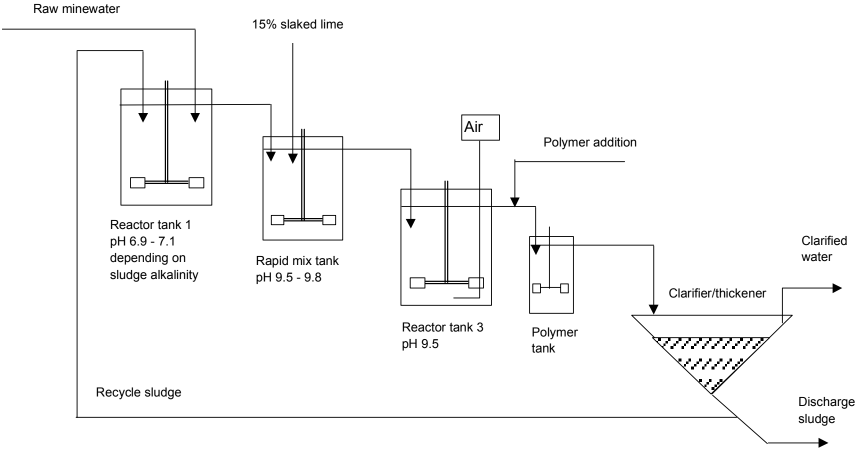

Figure 3.3. Geco Process.

Staged-neutralisation process

The staged-neutralisation process (Demopoulos *et al.* 1995, see figure 3.4) applies crystallisation principles to enhance precipitate crystallinity and reduce sludge volume. The process involves neutralising in a series of steps to control the level of supersaturation during metal precipitation. Using a supersaturation ratio of 10, the pH level in each reactor is calculated based on the composition of the minewater. Sludge is used for neutralisation in the first two reactor tanks and lime slurry is used to adjust pH in reactor tanks 3 and 4. The substantial reduction in lime consumption in this process compared to the other processes will reduce operating costs, but the larger number of reactor vessels will increase capital cost.

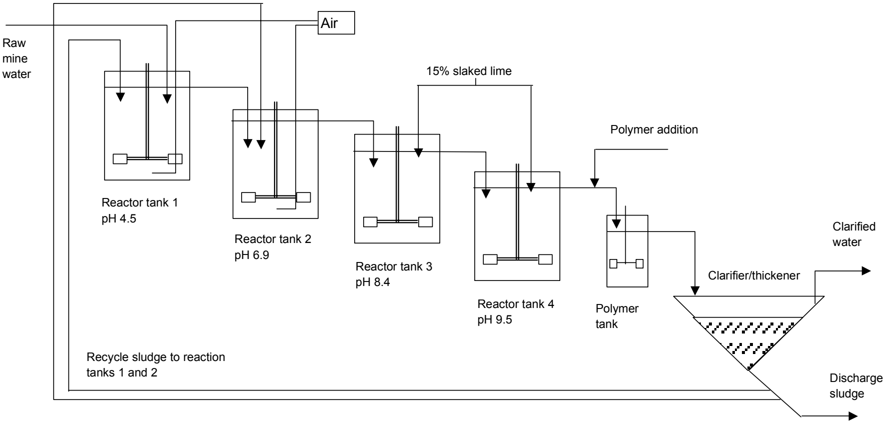

Figure 3.4. Staged Neutralisation Process.

All the processes considered above successfully treated a low-strength Zn–Fe minewater at pilot scale (Zinck and Griffith 2000) with no one process excelling in all aspects of treatment efficiency (e.g. sludge density, settling rate, lime consumption, sludge dewatering characteristics and stability). It thus seems that the high density sludge process, if deemed applicable for a particular minewater treatment scheme, needs to be tailored to the specific minewater problem and adapted and refined to provide optimum treatment and operation.

An example of an adapted HDS process is described by Aube (1999); in this instance it was found to be possible to dispense with the rapid mix tank, the polymer tank and aeration without any detriment to the treatment process (see diagram of simplified process, figure 3.5).

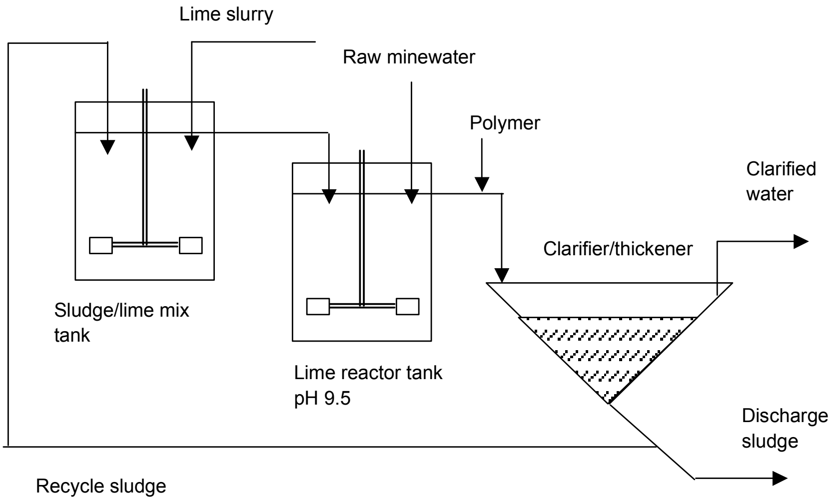

Figure 3.5. Simplified HDS process.

The active treatment process at Wheal Jane in Cornwall is an adapted HDS process, which has two reactor tanks and a clarifier; this differs from the scheme shown above in that the minewater and recycled sludge are mixed in the first reactor tank, with lime addition and aeration in the second (see details of Wheal Jane treatment plant in case study 5.35).

Unipure

The Unipure proprietary process uses similar principles to the HDS process, and is the process that the Wheal Jane treatment system is largely based upon. In wastewater streams containing metals but no ferrous iron, the Unipure system adds ferrous iron to the waste stream to initiate the reaction process of co-precipitation of the target metals with iron. In an acid minewater that contains ferrous iron, the addition of more iron is not necessary. In both cases, the ferrous iron will naturally coagulate, with ions of the other metals being included in the soluble chain-like structure that forms.

The next stage of the Unipure process is the reaction zone, in which alkali is added to raise the pH to 7.8, air is sparged through the minewater to oxidise ferrous iron to ferric and there is vigorous mixing to create a high shear environment. The insoluble ferric iron rapidly precipitates out of solution, incorporating the other metals in the dense matrix that is formed. In addition to this occlusion of metals within the matrix of the precipitated iron, the precipitated particles provide adsorption sites for other metals and for further ferric iron precipitation. The high shear environment ensures there is a high surface area to facilitate this process. The coprecipitation effect can be enhanced by the introduction of recycled sludge to the process (hence the similarity to the HDS process).

The resultant sludge from the Unipure process is dense, with good dewatering characteristics and low leachability (similar, in fact, to the sludge resulting from the HDS process).

Keeco – Silica Micro Encapsulation Technology

Klean Earth Environmental Company (KEECO) have developed a proprietary process for treating acid mine drainage which uses a reagent specifically developed for this purpose. The reagent, KB-1, comprises three major components:

- pH adjuster brings the material being treated to the iso-electric point, precipitating the heavy metals.
- A second component undergoes condensation polymerisation, chemisorbing the metals into a three-dimensional structure composed primarily of silica. This matrix completely encapsulates the metal and continues to strengthen with time, forming colloidal structures.
- Support medium for the first two components, offering a high surface area for maximum reactivity.

Thus the KB-1 reagent facilitates, in a single step, rapid precipitation of metals and forms a stable, easily dewatered sludge. There is no requirement for flocculants to be added as the precipitation and flocculation occur rapidly without such reagents. The sludge is of a sandy nature and has been demonstrated in leaching tests to be non-toxic. The encapsulation of the metals into a silicate framework means the sludge is more resistant to weathering processes or changes in the acidity level of the storage facility than sludge from conventional lime treatment.

The KEECO silica micro encapsulation technology has been tested in both pilot and full-scale applications, and has been proven to be highly effective (Rybock and Anderson 2000, Mitchell *et al.* 2000, Mitchell and Wheaton 1999). Minewater treatment schemes in the USA that use the KEECO process report favourable cost comparisons between this process and conventional lime-dosing, in terms of capital costs, operational costs and sludge management (Anderson and Rybock 2000). Work is ongoing to adapt the process to enable recovery of metals from the minewater, thus providing a marketable product to offset against costs (Mitchell and Wheaton 1999).

Pilot-scale trials have been carried out in the UK at Wheal Jane in Cornwall (Mitchell *et al.* 2000) and at two active quarries in south west England – Meldon Quarry and Venn Quarry (Mitchell *et al.* 2000, Mitchell and Whitbread-Jordan 2001). At Meldon Quarry the use of KB-1 compared favourably to a caustic-soda-based system. At Venn Quarry a full-scale treatment plant using the KEECO process was commissioned in November/December 2001.

Aquafix

The Aquafix system has been designed to be operated by a waterwheel and thus is suitable for use at remote sites where there is no access to electricity. Flowing water drives an auger that delivers lime to the stream of minewater to be treated. The system uses pebble quicklime (calcium oxide) as the pH modifier and is available in three sizes, in modular form. The smallest operational unit weighs less than 230 kg. Settling ponds are needed for precipitation of the metals.

There are over 100 Aquafix systems operating in the USA, and the system has successfully treated highly acidic, high-Fe and Mn-containing minewaters to standards set by state agencies in Pennsylvania, West Virginia and Kentucky. Cost comparisons made at one site (Pen Coal, Pennsylvania) showed substantial savings compared to the previous caustic-dosing system.

Hazleton Iron Removal System

This technology, developed by Hazleton Environmental, combines acid neutralisation, oxidation and precipitation of metals as hydroxides into one step in a MAXI-STRIP unit. The phenomenon that occurs is referred to as globular catalytic reaction and is described in the company literature as follows.

Lime slurry, raw minewater and recycled sludge are pumped to a Hazleton MAXI-STRIP where air is being aspirated and the mixture is sheared into microscopic globules of liquid. Some of the globules are at a very high pH, some highly aerated and some comprise raw minewater. A surface reaction occurs between the globules causing the surface film to oxidise and precipitate as a hydroxide. The surface film of the globule then catalyses the remaining liquid in the globule. As these reactions occur, the pH of the solution gradually drops to its equilibrium endpoint; for example, manganese, which normally precipitates only above pH 9, will precipitate simultaneously with iron as the pH of the solution drops to a pH 8.5 endpoint.

The Hazleton process includes a deep filter bed, the Floc Tunnel, which traps precipitated metal hydroxide sludge and uses it to filter the treated water flow. Sludge densities of 15% to 20% solids can be produced in this system.

Modular Environmental Technologies – the HARDTAC Process

The HARDTAC Process (High Aspect Ratio Draft Tube Agitated Crystalliser) was developed by Dupont and licensed by Modular Environmental Technologies Inc (MET) in 1995. The process uses conventional reagents but specially designed equipment to control the neutralisation and precipitation reactions. The high degree of control exerted over the rate of reagent addition and the degree and nature of reagent/minewater mixing means that crystallisation of the resultant precipitate is optimised. Thus primary nucleation is minimised so that fewer, larger crystalline particles are formed. This produces a sludge that has low final volume, and settles and dewaters rapidly. MET claim that mean particle sizes of up to 40–60 microns can be achieved, as compared to less than 1 micron for conventional lime-dosing and less than 5 microns for sludge recycle, i.e. HDS, processes. The larger particle size, and its implications for settling and dewatering characteristics, will have cost benefits in the sizing and operation of settling and dewatering equipment. Reagent use is also optimised, with usage typically less than 5% in excess of the theoretical stoichiometric amount, as compared to a 50–100+% excess requirement in conventional precipitation systems.

The HARDTAC process is a proven technology for treatment of metal-laden industrial wastewaters, having been installed in over 200 process applications and 20 wastewater treatment facilities over the past 20 years.

Virotec Bauxsol Process

The Bauxsol process is a pH modification process that uses seawater-neutralised bauxite refinery residues as the reagent. It thus fulfils the desirable criteria of using one waste to treat another, which is both environmentally friendly and cost effective. Bauxsol effectively neutralises acidity due to the presence of large quantities of carbonate minerals that form weak bases. Trace elements are stripped from the minewater through a binding mechanism, which commences as surface adsorption and is followed by the formation of new minerals. The contaminant elements are structurally bound within these new minerals. Removal efficiencies of 99.95% are typically achieved for a wide suite of elements.

This process is effective for the treatment of large bodies of acidic metal-rich water, as was demonstrated by trials of the process on water from Berkeley Pit, Montana, USA. Bauxsol has recently been successfully used to carry out *in situ* treatment of a 1500 million litre body of acid mine drainage at the Mount Carrington Mine in northern New South Wales, Australia. The effluent water from the treatment process meets stringent Australian standards for protection of aquatic ecosystems. The company marketing Bauxsol claims that the resultant sludge is stable and performs well in leaching tests.

Tests on using Bauxsol in a passive treatment system, similar to an open limestone channel, have demonstrated a potential for use in this context also. The hydraulic conductivity of the Bauxite fill was not significantly reduced by passage of Fe and Al-rich minewater, which suggests that the material was not becoming coated with iron and aluminium hydroxides which can block, or render ineffective, open limestone channels. The company literature claims that such Bauxsol drains would remain effective for over 20 years, although, due the recent emergence of this technology, this has yet to be verified.

Mintek Savmin process

The Savmin process has been developed by the South African research organisation Mintek to provide potable water (Smit 2000). The process has been developed over the past few years to tackle South Africa's twin problems of sulphate contaminated minewater and constrained water supplies.

Savmin uses precipitation reactions to purify minewater containing sulphate. The first stage is the addition of lime to raise the pH and precipitate out metals and magnesium as hydroxides. The hydroxides are separated and the resulting supersaturated calcium sulphate solution is contacted with gypsum crystals, which catalyse the precipitation of calcium sulphate (gypsum) from solution. The precipitate is thickened and filtered and leaves the process as waste or byproduct. The developers of the process found that, due to the slow settling rate of the precipitates, the most cost effective equipment for the solid–liquid separation stages of the process are parallel-plate cone clarifiers.

In the third stage of the process, aluminium hydroxide is added to the solution which causes formation of the insoluble salt ettringite, which removes calcium and sulphate from solution. The solution is then treated with carbon dioxide to lower the pH (process pH values in the first and third stages of the process need to be maintained at 11.6 to 12). The lower pH causes the precipitation of pure calcium carbonate which is separated from the water by filtration.

The final stage of the process is the recycling of the ettringite, which is treated with sulphuric acid to regenerate aluminium hydroxide.

Mintek claim that the process can effectively remove heavy metals from minewater to concentrations well below those required for potable water, and also remove 99% of calcium, 100% of magnesium and 98% of sulphate in the raw minewater. Univalent cations such as sodium, chloride, potassium and fluoride are not removed.

The Savmin process has been subjected to extensive laboratory tests, and trialed at pilot scale at Anglogold, Vaal River operations in South Africa.

Hydrometrics CESR process

The CESR (Cost Effective Sulphate Removal) process is a modified lime dosing system that has been developed to treat high sulphate effluents and successfully reduces sulphate concentration to less than 100 mg/l through the use of a proprietary powdered reagent (Reinsel 1999). The addition of this reagent to lime-pH-controlled water precipitates sulphate as a insoluble calcium-alumina-sulphate compound known as ettringite. The formation of ettringite can also provide a polishing effect, allowing precipitation of difficult-to-remove metals such as chromium, arsenic, selenium and cadmium, typically in a non-leachable form.

There are essentially four steps in the CESR process:

(1) Initial sulphate precipitation – for wastewater with high metals content and more than 8000 mg/l sulphate. Hydrated lime is added to precipitate most of the sulphate as gypsum. This precipitation is carried out at a pH value lower than that for metals precipitation so the gypsum sludge is uncontaminated with metals.
(2) Metal hydroxide precipitation – Hydrated lime is added to raise the pH to 10.5 to precipitate out metal hydroxides and further precipitates of gypsum, so that approximately 2000 mg/l sulphate remains in solution (determined by the solubility of gypsum).
(3) Final sulphate precipitation – The pH is raised to approximately 11.5 with hydrated lime, and the addition of the proprietary reagent causes precipitation of the remaining sulphate as ettringite.
(4) pH reduction – A final step to reduce pH, by addition of carbon dioxide, is necessary if the water is to be discharged to surface waters.

The CESR process is in use at up to 20 sites in Europe and the USA, at flow rates of up to 25 l/s. The process is used by various industries (electroplating, battery recycling, agriculture) and has been tested at bench-scale on mining-related effluents (at Berkeley Pit in Butte, Montana and on minewater from a closed gold mine).

The company marketing this process claims that it is less expensive than alternative technologies for comparative reduction in sulphate concentration, such as reverse osmosis and ion exchange, both in terms of capital and operating costs, especially if a lime dosing plant is already in place.

3.2 ION EXCHANGE

The presence of base metals such as zinc and copper in acid mine drainage gives rise to the possibility that costs may be offset against the value of the metal recovered.

The concentrations of the metals in sludge produced by conventional liming is not sufficient for smelter operators to pay for it, nor is the sludge of sufficient purity to make recovery a viable option and the excess alkali reagent in the sludge would also prove problematic. A possible answer to the problem of metals recovery is to use ion exchange technology to strip valuable metals from the minewater before conventional treatment. The metals can then be eluted from the ion exchange medium in a much more concentrated form. Laboratory studies prove that this process is technically feasible and that metal concentrates of suitable concentration for smelting can be produced. The problem is that the ion exchange media are often too expensive and the flows too great for the technology to handle economically. The cost of producing the metal is thus greater than the potential value, or the profit so small that initial costs in installing the technology would not be repaid for many years. Furthermore, in the case of post-closure minewater treatment, the amount of metal reclaimed, and therefore the economic gain, would tend to decrease over the life of the plant due to the gradual improvement of minewater quality that is frequently observed.

However, as novel cheaper ion exchange media are developed, the economics and practicality of such an approach substantially improve.

CHEMEFFCO GYP-CIX process

The GYP-CIX process was developed by Chemical Effluent Treatment Process Limited (CHEMEFFCO) in South Africa, as a desalination process for the bulk removal of scaling calcium sulphate, as well as other salts and heavy metals, to produce a re-usable product water and a manageable waste discharge. The GYP-CIX process is based on ion-exchange resins and uses low-cost chemicals including lime and sulphuric acid.

The process removes major ions (calcium, magnesium, sodium, sulphate, chloride, nitrate) and a large suite of metal cations (including iron, cadmium, copper, zinc, nickel, lead, arsenic, uranium) from solution. The process feeds limed AMD effluent into a counter current fluidised bed cation exchange reactor, followed by an anion exchange reactor to produce potable water. The ion exchange resins are regenerated by sulphuric acid and lime washes respectively, allowing the resins to be recycled without affecting their service life. The wastes of the regeneration are processed to produce gypsum, which may be a marketable byproduct.

The process was developed to desalinate mine waters with a high calcium content and a total dissolved solids concentration of up to 3,750 mg/l, which is typical of minewaters from South African mines. CHEMEFFCO claim that the process can be used in combination with other systems to treat waters with differing chemical qualities.

After 3 years of pilot-scale, proof-of-principle tests, CHEMEFFCO have built a demonstration plant at Amcoal’s Landau colliery, capable of treating 250 m^3/day of water. The company claim that the running costs of this demonstration plant, about 26 cents/m^3, are about the same price as purchased water.

KAD Technology

KAD Technology uses Kaolin Amorphous Derivatives as ion exchange media for removing metals from solution by ion exchange. The process is marketed by Mine Remediation Services Pty Ltd, which is a joint venture between Cardia Mining NL (an Australian mineral exploration company) and UniQuest Ltd (the technology transfer company of the University of Queensland).

The newly developed materials, KAD, can be readily synthesised from any form of kaolin mineral and most montmorillonites using simple chemical treatment methods and mineral processing methods. The reaction requires a caustic reagent and a process temperature of less than 150°C. The resultant KAD materials have a high surface area relative to the starting clay material (10–100 times higher) and have significantly enhanced cation exchange capacities in acidic solution (around pH 3) when compared to conventional ion exchange reagents (MacKinnon 1998). The KAD materials can be designed to specific adsorptive capacities for a wide range of metal species and concentrations. The natural basicity of the KAD effectively neutralises acid mine drainage and the KAD material can be regenerated and re-used. The water that is produced passes strict criteria for discharge to the environment, or may provide a potable water supply. The application of conventional metal recovery techniques can be used to gain a marketable resource in the form of metals reclaimed from the minewater, with the specificity of the KAD materials facilitating this objective.

Pilot-scale trials have been carried out on acid mine drainage extracted from the tailings toe dam at the Mount Carrington site, northern New South Wales, Australia (MacKinnon *et al.* 1997). A three-column counter-current mixing/decantation presentation rig was used. One column was used for preconditioning the minewater (pH control by caustic addition and sedimentation of suspended solids by using flocculants were investigated), and two columns were used to contact the minewater with the KAD. Used KAD slurry is removed from the bottom of the third column, regenerated then added to the top of the second column. The trials showed that complex, multi-element minewater can be treated to stringent Australian water quality standards, using one type of KAD in a ‘one-pass’ operation. Contact times of 30 minutes or less were found to be adequate for many minewater compositions. Most of the KAD materials showed strong proclivity for removal of common metal contaminants such as copper, zinc, iron and aluminium, dependent on the pH of the solution. However, removal of other elements in a multi-element solution, such as magnesium and calcium, varied considerably depending on the type of KAD used. Also, at pH values of less than 3, iron and aluminium are taken up preferentially by many KAD materials; this leads to the recommendation that for acid minewater with high concentrations of these two metals (e.g. in excess of 500 mg/l), use of a pre-treatment strategy to reduce these elements may be advantageous for conservation of KAD material during treatment. By controlling pH in successive stages of treatment with KAD, specific elements could be extracted from a multi-element solution, thus simplifying reclamation of metals, if such were deemed to be economically viable.

Successful pilot-scale trials have also been carried out at Berkeley Pit, in Butte, Montana (as reported in the Mining Journal in November, 1998).

Dynaphore, Inc. – FORAGER Sponge

The FORAGER Sponge is an open-celled cellulose sponge containing a polymer with selective affinity for dissolved heavy metals in both cationic and anionic states. The polymer contains iminodiacetic acid groups which enter into chelation bonding with transition-group heavy metal cations. The polymer's affinity for particular cations is influenced by solution parameters such as pH, temperature and total ionic content. In general, the following affinity sequence for representative ions prevails:

$$Cd^{2+}>Cu^{2+}>Hg^{2+}>Pb^{2+}>Au^{3+}>Zn^{2+}>Fe^{3+}>Ni^{2+}>Co^{2+}>>Al^{3+}>Ca^{2+}>Mg^{2+}>>Na^{+}$$

During absorption, a cation is displaced from the polymer, which may be H^+ or a cation below the absorbed cation in the affinity sequence.

The polymer also contains tertiary amine salt groups which exhibit selective bonding for anionic species. The absorption of certain anions can be enhanced by pre-absorption of a cation that will form a bond with the target anion.

Absorbed ions can be eluted from the sponge using standard exchange medium regeneration techniques. The number of times the sponge can be re-used depends on the nature of the absorbed ions and the elution technique used. Alternatively the sponge can be dried, incinerated or land-filled.

The sponge can be used in columns, fishnet-type enclosures or rotating drums. Placement of fishnet enclosures into wells or trenches facilitates a passive treatment of groundwater, similar in concept to the use of reactive barriers.

This technology has been used for acid mine drainage treatment at three sites in Colorado and in bench-scale tests to treat contaminated groundwater at various Superfund sites in the USA.

Octolig MRP

The Octolig system removes heavy metals from solution by chelation on immobilised ligands, called Octolig (a proprietary product). When high concentrations of the target metals are present, precipitation (by alkali addition) and filtration may be used prior to Octolig MRP treatment, which then removes the metals down to very low concentrations (parts per billion). Octolig is not strictly an ion exchange process but rather uses pH-responsive immobilised ligands, which reportedly have a higher affinity for heavy metals than ion exchange resins resulting in a highly selective process. Chelation reactions occur on the surface of the host silica gel and regeneration of the Octolig material can be achieved with a small volume of acidic solution. Regeneration can be carried out hundreds of times without compromising the performance of the treatment system.

The Octolig system has been tested at pilot-scale at various sites, for example, a drainage tunnel at Leadville, Colorado, where the primary target metals were copper and zinc; at the Berkeley Pit in Montana where a suite of metals were removed by Octolig (pH was first raised to 8.3); and at Summitville near Del Norte, Colorado, where the system was used to treat drainage water from a cyanide destruction process following cyanide leaching. An Octolig MRP system is currently installed at a gold mine in Colorado where the primary contaminant is zinc.

3.3 BIOLOGY-BASED TREATMENT

The importance of micro-organisms in both the generation and amelioration of contaminated minewater has long been known (as reviewed by Ledin and Pedersen 1996, Kuyucak 2000) and is the subject of much ongoing research, for example at the Wheal Jane LINK project in Cornwall (Halberg and Johnson 2001, Dennison *et al.* 2001).

Biology-based treatments include techniques that harness the remediation function of microbes in bioreactors, thereby facilitating a greater degree of control than is possible when the microbes are utilised in a passive wetland treatment context. Another example of biology-based treatments is the use of biologically produced materials as adsorbents for the removal of metals from a waste stream.

Bioreactors using sulphate-reducing bacteria

A novel approach to minewater treatment has been developed in the last few years that utilises biological amelioration in a highly controlled manner. This system uses sulphate-reducing bacteria (SRB) in a bioreactor to ameliorate contaminated minewater. In order to place this biological treatment process in context, there follows a very brief review of the nature of SRB and the development of their use for water treatment.

Sulphate-reducing bacteria are a diverse group of anaerobic, heterotrophic microorganisms that conduct dissimilatory sulphate reduction. In this process the sulphate ion acts as an oxidising agent for the dissimilation of organic matter, as does oxygen in conventional respiration. A small amount of reduced sulphur is assimilated by the organism, but virtually all is released into the external environment as the sulphide ion, usually substantially hydrolysed to free hydrogen sulphide gas (Postgate 1984, Odum and Singleton 1993). If this metabolic end-product comes into contact with metal ions, precipitation of metal sulphides occurs. It has been proposed that the precipitation of metal sulphides due to reaction with H_2S gas could be an adaptation by the SRB to an environment containing potentially toxic metals, since the precipitation process removes the free metal ions from solution and thus effectively detoxifies the environment (McClean and Beveridge 1990). Precipitation also removes H_2S gas which can be toxic to the SRB if levels are allowed to rise (Smith 1993).

The use of sulphate-reducing bacteria for human benefit has long been recognised and investigated. In the 1950s, bioreactors based on SRB metabolism were developed to pilot plant-scale to produce sulphur as an end-product (Butlin *et al.* 1956, 1960, Knivett 1960). These reactors used sewage sludge as the carbon source and calcium sulphate as the source of sulphur. The need was recognised to sweep out the H_2S gas from the reaction chamber to prevent toxic effects to the SRB population.

In the 1960s, the role of SRB in ameliorating contaminated minewater was investigated by damming a contaminated stream with wood shavings and carrying out subsequent laboratory microbiological investigations (Tuttle *et al.* 1968, 1969a and 1969b).

More recently the importance of SRB for amelioration of contaminated minewater in constructed wetland systems has been recognised (Hedin *et al.* 1988, 1989, McIntyre and Edenborn 1989, McIntyre *et al.* 1990, Chang *et al.* 1991). Numerous laboratory investigations using contaminated minewater have been carried out to quantify the rate of sulphate reduction and precipitation of heavy metals (Brierley and Brierley 1980, Bechard *et al.* 1989, Ueki *et al.* 1991, Belin *et al.* 1991, Machemer and Wildeman 1992, Kar 1992). Recent laboratory work has investigated the selection of suitable substrates for SRB activity (Duc *et al.* 1998, Prasad *et al.* 1999, Chang *et al.* 2000, Drury 1999), revitalisation of a depleted organic substrate in a bioreactor (Tsukamoto and Miller 1999), the ability of anaerobic bioreactors to function in acidic conditions (Elliot *et al.* 1998) and have attempted to model sulphate reduction in bioreactors (Drury 2000).

The work mentioned above is based on the principle of metal removal by indirect microbial action. Amelioration of contaminated minewater is achieved by metal ions combining with products of microbial metabolism, in this case, with H_2S gas, to form metal sulphide precipitates. This means that the process requires live biomass, unlike biosorption processes discussed elsewhere which could be achieved with non-living biomass. Thus the microorganisms in question could be considered to be low cost producers of the chemical compounds that immobilise metal ions (Rossi 1990). Anaerobic treatment of low-strength wastes is only economical if large volumes of waste can be forced through the system in a relatively short time period. This means that biomass retention needs to be controlled independently of the wastewater flow rate, i.e. steps must be taken to prevent wash-out of the microbial biomass (Lettinga *et al.* 1980). It is with this necessary design criterion in mind that the development of bioreactors has occurred over the past two decades.

In 1985, Maree and Strydom developed a packed bed reactor, in which biofilm formation on crushed stone facilitated faster processing of wastewater than a completely mixed reactor and gave better results than a sludge blanket reactor (Maree and Strydom 1985). Although this pilot plant-scale reactor was designed to treat contaminated minewater, the effluent H_2S was oxidised to S in a second, photosynthetic reactor, rather than being used to precipitate metals. A proprietary culture of microorganisms, immobilised on a porous ceramic medium, has been used in a pilot scale bioreactor to remove cyanide, nitrate and metals from mine process water (Canty *et al.* 2000).

In 1988, Nakamura developed a continuous flow, fluidised bed reactor in which H_2S gas was piped away from the reactor vessel and contacted with contaminated minewater to precipitate out metals (Nakamura 1988). This separation of the H_2S gas from the biological component of the reactor is a technique that was to be further exploited in later developments of bioreactors (see below).

Various workers developed pilot plant-scale reactors in which H_2S production and metal precipitation were carried out in the same reactor vessel. For example, in 1989, Bechard *et al.* made a continuous flow reactor containing straw and wood shavings as the organic substrate, Dvorak *et al.* (1991) used spent mushroom compost and Lyew *et al.* (1994) used a gravel substrate, seeded with SRB, and introduced the necessary organic nutrient source with the contaminated minewater that was to be treated. Herrera *et al.* (1993) removed sulphate from contaminated minewater in a sequencing batch bioreactor, the H_2S being flushed out and rendered non-toxic rather than being used for metal precipitation.

The above account gives a brief overview of the development of bioreactors over the past few decades. The most significant recent developments have been the commissioning of an operating bioreactor in the Netherlands and refinement of the Nakamura approach to allow very efficient water treatment and recovery of metals (see NTBC Biosulphide process, below).

In May 1992, a commercial-scale, bacterial sulphate reduction treatment plant began treating zinc-contaminated groundwater at the Budelco Smelter site in the Netherlands, with ethanol as the carbon source for the SRB – using the Paques process, see below (Scheeren *et al.* 1991, 1992, Barnes *et al* 1992). This plant uses a sludge blanket bioreactor, in which the zinc precipitates as sulphide, which forms the ‘blanket’ to which the microorganisms can adhere. Zinc sulphide sludge that settles to the bottom of the reactor vessel is sent to a nearby smelter for processing and recovery of Zn metal.

In most of the above-mentioned reactors, with the exception of the one developed by Nakamura (1988) and those not used for metal sulphide precipitation (Maree and Strydom 1985, Herrera *et al.* 1993), a single reactor vessel is used for both generation of H_2S and precipitation of metal sulphides. These relatively simple systems may be suitable if all the components in the wastewater are non-toxic to the bacterial population, and if the production of a mixed-metal sludge that contains organic matter is acceptable. However, metal concentration in mining or mineral processing wastewater often exceeds the tolerance levels of anaerobic bacteria (Smith 1993), and mixed-metal precipitates are less easy to process for metal recovery by smelting. An approach that avoids these problems is the use of an inert carrier gas to flush H_2S from the reactor vessel and transport it to one or more separate precipitation vessels, as was proposed by Nakamura (1988). In the precipitation vessels, H_2S reacts with the wastewater to form insoluble metal sulphides. The advantages of this approach are as follows:

- Potentially inhibitory or toxic metals in wastewater do not contact the SRB.
- Metal sulphide precipitation reactions can be carried out under conditions that are not conducive to the growth of SRB.
- Individual metal sulphides can sometimes be selectively precipitated by controlling the pH of the wastewater within the precipitator vessels.
- Metal sulphides that are processed for metal recovery are not contaminated or diluted with biomass or organic substrate.
- The organic carbon source used for SRB metabolism can be a waste organic material, such as food processing waste or sewage sludge, thereby solving two waste disposal problems in one operation.

This technology was developed to bench-scale pilot reactors, in the early 1990s, by two separate research groups working independently in Canada and the USA (Rowley *et al.* 1994, Hammack *et al.* 1994). Both bench-scale tests showed encouraging results, improving the quality of highly contaminated minewater to within discharge consent limits (and to a much higher quality than achievable through conventional liming), and producing copper and zinc sulphide concentrates of suitable quality for smelting. The pilot systems were also shown to perform reliably and consistently over prolonged periods. The biogenic H_2S treatment system would thus appear to offer an attractive treatment option, particularly in situations where there are:

- high concentrations of saleable metals in the contaminated minewater;

- low-cost or waste carbon sources available, or existing anaerobic sewage treatment facilities that could be modified for H_2S production (Butlin *et al.* 1956, 1960);
- concerns about sludge volume and toxicity;
- strict discharge limits for metal contaminants and sulphate.

Modifications of this process have been tested at pilot scale, for example a consecutive hydroxide-sulphide precipitation process was tested using synthetic minewater (Diaz *et al.* 1997). In this process, iron and arsenic were removed in the first step, by alkali addition, prior to subjecting the minewater to a linked biological/chemical treatment as described below (the biosulphide process).

The above is an account of the development of the use of bioreactors to treat AMD. The proprietary processes currently available on the market are now considered.

PAQUES THIOPAQ process

The Paques THIOPAQ process comprises two biological treatment steps that take place in separate reactors. First, sulphate is converted to sulphide by sulphate-reducing bacteria, then sulphide is converted into elemental sulphur by sulphur-oxidising bacteria. Due to the alkalinity production in the conversion of sulphide to sulphur, recirculation of the process water can save on alkali chemicals. The heavy metals in the minewater precipitate as sulphides in the first reactor and can be recovered separately by manipulating the process pH (due to different metal sulphide solubilities at different pH values). Sulphur cake of up to 60% dry solids and 95% purity is produced in the second reactor and may be marketable for sulphuric acid production (van Bodegraven 1997, de Vegt and Buisman 1996). Suspended solids are removed, then the effluent polished, in two or more subsequent polishing steps.

Van Bodegraven's April 1997 paper describes two pilot plants and four THIOPAQ reference installations. Two of the reference plants use ethanol and two use hydrogen as the electron donor for the first stage of the biological process.

The pilot plants were in operation at the US Bureau of Mines Pittsburgh Research Centre, where various simulated North American minewaters have been treated, and at a metal treating industry where wastewater containing metals, fluor borate, sulphonic acid, sulphate, nitrate and acetate has been successfully treated.

The reference plants are at Budelco in the Netherlands, as referred to above and described in more detail below; at Kennecott Utah Copper, where groundwater is treated to remove sulphate and metals, and the sulphides are used to selectively recover copper; at Akzo Emmen where washing water of a fibre industrial plant is treated for sulphate; and at EPZ Power Station where the process is adapted to treat flue gas.

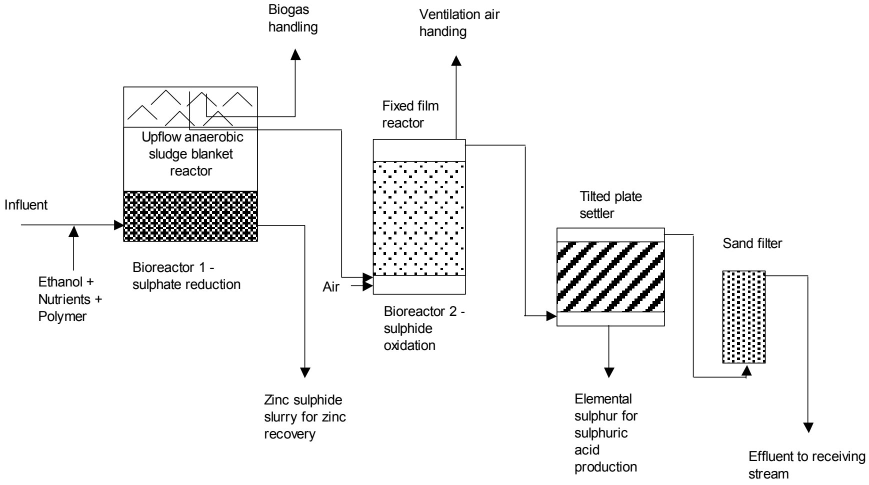

Figure 3.6. Thiopaq process at Budelco.

A schematic representation of the treatment system at the Budelco zinc refinery is given in figure 3.6. This process uses ethanol as the electron donor for the sulphate-reducing bacteria, and produces a metal sulphide slurry (comprising mainly zinc sulphide) and a sulphur slurry, which are re-used for production of zinc and sulphuric acid respectively. Further pilot plant tests have been carried out on three mining-contaminated waters, with the intention of testing the process on more severely contaminated waters than those treated at Budelco (Hammack *et al.* 1998). Generally, sulphate concentrations were successfully decreased to acceptable standards for discharge, as were concentrations of all metals except manganese.

NTBC Research Corp – the Biosulphide Process

The Biosulphide Process is an integrated chemical/biological process designed to treat metal-contaminated, sulphate-rich minewater (or other wastewater of suitable composition). The process can be divided into two stages: a chemical circuit in which treatment of the minewater occurs, and a biological circuit in which reagents (dissolved and gaseous sulphide and alkalinity) are generated for use in the chemical circuit. Thus each circuit can operate at maximum efficiency independent of the other and the full flow of minewater does not have to pass through the slowest stage of the process (the bioreactors). This in turn has implications for the size of reactors: they can be considerably smaller, and thus less expensive, than bioreactors which have to receive the entire waste stream. The process is generalised in figure 3.7.

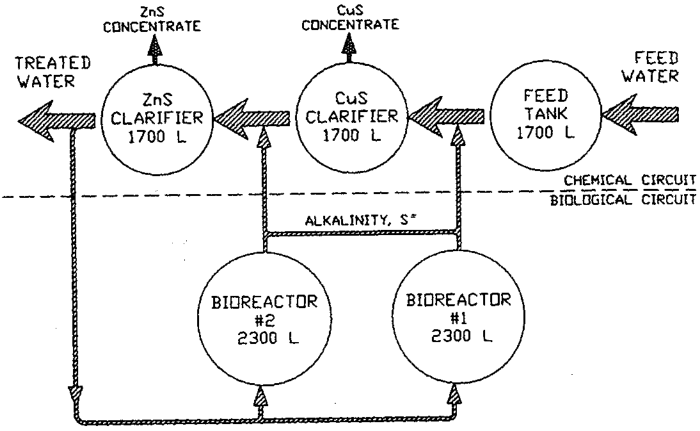

Figure 3.7. Generalised process flow diagram of the Biosulphide Process (after Rowley *et al.* 1997).

The biological stage utilises sulphate-reducing bacteria to convert sulphate to sulphide, with the simultaneous generation of alkalinity. These reagents are then added to the chemical stage to precipitate out metals and neutralise acidity. The bacteria in the bioreactor need a supply of nutrients (sulphate, carbon and energy with lesser amounts of phosphorous, nitrogen and potassium) to survive. Sulphate is generally provided by the water being treated; carbon and energy requirements are met with a system developed by NTBC to provide a reliable, inexpensive supply from natural gas, propane or other low cost fuel; nitrogen, phosphorous and potassium are supplied by small additions of inexpensive fertiliser.

In the chemical stage, the entire flow of minewater is treated by addition of the sulphide and alkalinity generated in the biological stage. By use of a series of precipitation stages, each maintained at a specific pH, high purity metal sulphide products can be recovered and used to generate income, either through on-site processing or sale to smelters, to set against operating costs. The final precipitation stage removes any remaining metals that are economically less viable or inseparable with current process technology. The waste sludge is thus reduced in volume (due to removal of reclaimable metals, and greater density of sulphide minerals as compared to hydroxides as would be achieved from alkali-addition treatment) and also reduced in toxicity (due to removal of reclaimable metals).

The Biosulphide Process has been tested at three large-scale pilot plants, including one at the former Britannia Mine in British Columbia (Rowley *et al.* 1997). Evaluation of the process was made at this site, with comparison with an HDS process, a modified HDS process (O'Hearn and Klein 2000) and conventional lime dosing (NTBC website). In terms of costs, the Biosulphide Process compared very favourably with lime dosing and a combined HDS/biosulphide process was considered to be the most cost effective scenario in the O'Hearn and Klein study.

Other SRB bioreactor processes

Anaerobic bioreactors have been used in various treatment scenarios, sometimes as part of a larger, integrated system (e.g. Gusek *et al.* 2000), and sometimes as a stand-alone process, based on the principles discussed above. Processes that have been developed using SRB bioreactors for minewater treatment are included below.

Lawrence Consulting Ltd

Technology is based on the Paques process, but is less capital intensive and uses a novel partial extraction burner to generate the hydrogen used as the electron donor. This is claimed to be a distinct advantage over the hydrogen generated by steam reforming as used in the Paques process. Projects are well advanced in three North American locations (Alaska, Montana and new Brunswick) and was to be commercially implemented during 2001.

Hydrometrics

Hydrometrics have carried out a pilot-scale test using a passive bioreactor with a substrate containing organic carbon at a mine with a small flow of acid rock drainage; metals and sulphate were substantially removed and pH was increased. Initial concentrations were approximately 80 mg/l iron, 25 mg/l zinc and 90 mg/l manganese and removal rates of 99% were achieved for iron and zinc, 78% for manganese. Sulphate removal was approximately 90% to achieve a discharge concentration of 250 mg/l. pH was increased by approximately 1 unit to pH 6.7. A full-scale passive bioreactor is currently being designed for the mine site where this pilot-scale test was carried out.

The ABC (Anoxic Biotreatment Cell) process has been developed by Hydrometrics principally for the removal of nitrate in a denitrifying anoxic biotreatment reactor, and is also effective for selenium removal (Reinsel and Botz 1999). However, bench-scale tests on heap leach water from a large gold mine, in which a longer residence time was used, achieved drinking water standards for aluminium, arsenic, cadmium, iron, selenium and zinc, with removal rates in excess of 90% for all these elements.

Waste stabilisation ponding process

The waste stabilisation ponding process has been developed over the last 40 years for the treatment of a wide range of wastewaters (Mara *et al.* 1996), but little work has been carried out on using this technique for minewater treatment. Essentially, the process entails retaining large volumes of wastewater within earthwork ponds. By adding an organic carbon source, algal growth on the surface and SRB at depth will become established. Thus the algae perform a function in metal immobilisation and also provide a renewable and sustainable biomass/carbon source for the SRB. Rose *et al.* (1998) describe investigations into use of this technique on mine and zinc refinery wastes, using co-disposal of tannery waste as the carbon source. They present a flow diagram of a staged treatment process (figure 3.8) and suggest that this process would be suitable for larger volumes of flow than could be passed through other SRB bioreactor processes.

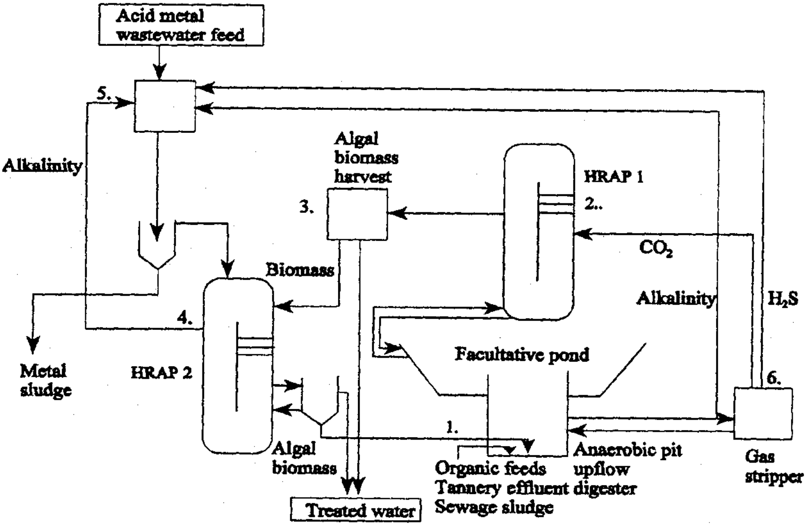

Figure 3.8. Waste pond stabilisation process flow diagram (after Rose *et al.* 1998).
2 and 4 HRAP = high rate algal pond, 5 = pre-treatment mixer.

Treatment of acid lignite minewater

A pilot-scale process to treat minewater from a lignite mine in Germany is described by Glombitza (2001). SRB were immobilised on porous media and methanol was used as a carbon source. Metals are precipitated as sulphides, and excess sulphide is oxidised to elemental sulphur by addition of hydrogen peroxide. Metal removal rates approaching 100% were achieved over several months of pilot plant operation, and the pH was increased from 3.0 to 6.9.

Use of acidophilic bacteria

The SRB bioreactors described above use neutrophilic bacteria (i.e. bacteria requiring circum-neutral pH in order to function effectively), meaning that care must be taken not to expose them to low pH minewaters. Johnson *et al.* (2000) describe work carried out using acid- and metal-tolerant bacteria, that occur naturally in acid minewater-impacted environments, in bioreactors. They investigated two types of fixed bed bioreactor, one containing acidophilic (acid-loving) iron-reducing bacteria and fungi and the other containing acidophilic and neutrophilic SRB. The bioreactors were investigated when operating independently and in tandem. Successful iron reduction was achieved using paper pulp as the sole carbon and energy source, and sulphate reduction was observed in minewater with pH 3 and above, although the rates were lower than reported for circum-neutral pH systems.

Other bioreactor processes

Bioreactors used for treatment of minewater are not solely anaerobic. Investigation has also been made into the use of aerobic bioreactors to oxidise ferrous iron, often as part of a multi-stage process to treat minewater (Diz and Novak 1998, Diz and Via 1998, Mazuelos *et al.* 1999). Both fluidised bed and packed bed reactors have been investigated. Most of these integrated systems are currently at an experimental stage and have yet to be tested at pilot scale or full scale. A system in operation at the Nickel Plate Mine in British Columbia has been successfully removing cyanide, thiocyanate, ammonia, nitrate and metals from tailings water since 1996. This system consists of aerobic bioreactors (where bacterial oxidation of cyanide, thiocyanate and ammonia occurs), followed by anaerobic bioreactors (where nitrate is converted to gaseous nitrogen) with a final high density sludge stage to remove the metals (www.homestake.com, www.microbialtech.com).

Other biological treatment processes

As mentioned in the introduction to this section on biology-based treatment processes, the biological component can either utilise the metabolic processes of microorganisms, as in the bioreactors discussed above, or use living or non-living biomass for the absorbent/adsorbent properties of the biomass. Research has been carried out into the use of biological materials for adsorption of metals from wastewaters (Brierley *et al.* 1989, Brierley 1990, Tsezos 1990, Gadd 1992, Schultze-Lam *et al.* 1993), using materials such as waste biomass (Mattuschka and

Straube 1993), peat (Lappako and Eger 1998, Viraraghavan and Rao 1992), algal biomass (Kratochvil and Volesky 1998), filamentous bacteria (Shuttleworth and Unz 1993) or bark (Vasquez *et al.* 1994). These materials are cheap, but cannot be re-used, as can ion exchange materials, meaning that there is a disposal problem to be overcome.

Work was carried out by the US Bureau of Mines to develop biosorption treatment that incorporates biomass into polysulfone beads that can be re-used many times for adsorption of metals (Jeffers *et al.* 1989, Ferguson *et al.* 1989, Bennett *et al.* 1991, Spinti *et al.* 1995). A review of the potential application of immobilised microbial biomass to treatment of metal-bearing wastewaters was compiled by Tsezos *et al.* (1998) which considers that the technique is particularly suitable for high flow/low concentration wastewater streams. The various biological materials, immobilisation methods and contacting configurations are considered, and modelling of the adsorption process is briefly discussed.

Biosorption techniques that are currently commercially available are considered below.

Resource Management and Recovery – AlgaSORB Biological Sorption

The AlgaSORB sorption process uses algae to remove heavy metal ions from aqueous solutions, making use of the natural affinity for such ions exhibited by algal cell structures. The AlgaSORB medium consists of dead algal cells immobilised in a silica gel polymer. This immobilisation serves two purposes: it protects the algal cells from decomposition by other microorganisms and it produces a hard material that can be packed into columns that, when pressurised, still exhibit good flow characteristics.

The AlgaSORB medium functions as a biological ion-exchange resin to bind both metallic cations (positively charged ions such as copper, zinc) and metallic oxoanions (negatively charged, large, complex, oxygen-containing ions, such as selanate). Anions such as chlorides or sulphates are weakly bound or not bound at all. In contrast to other ion exchange technology, divalent cations, such as magnesium and calcium, and monovalent cations, such as sodium and potassium, do not significantly interfere with the binding of the heavy metals to the algae-silica matrix.

Like ion exchange resins, AlgaSORB can be regenerated by use of acids, bases or other suitable reagents. This regeneration process generates a small volume of highly concentrated solution which will need to undergo treatment prior to disposal.

The process was tested on mercury-contaminated groundwater at a hazardous waste site in Oakland, California in 1989 and is being commercialised for groundwater treatment and industrial point source treatment.

Oak Ridge National Laboratory – Ligand-grafted polymer

Many organic molecules, particularly biological molecules, contain functional groups (ligands), that actively interact with metal ions in solution by adsorption, ion exchange or chelation/coordination/complexation. Oak Ridge National Laboratory in Tennessee, USA, has developed a biosorbent comprising ligand molecules grafted onto an insoluble matrix, which has been tested in stirred-tank and packed-column reactors. The choice of ligand functional group, for example amino, hydroxyl or carboxyl group, enables selective recovery of targeted metal ions. The material can be regenerated by sodium bicarbonate or sodium carbonate solution and re-used. A demonstration project in which uranium was selectively recovered from an aqueous solution that also contained iron, copper, zinc and manganese is described in Hu and Reeves (1999), and these authors conclude that this technique shows great promise for metal separation from aqueous waste streams, although they note that further research is required.

3.4 OTHER ADSORPTION TREATMENTS

The use of non-biological adsorbents is another technology option that has been investigated and proposed as a solution to the treatment of acid mine drainage. The aim of this process is to utilise particles of known size and density to adsorb the metals from solution and to exploit knowledge of physical processes in the separation of the solids later in the treatment.

Nalco – Nalmet Heavy Metal Removal System

NALMET is a patented polymer capable of precipitating soluble heavy metals down to extremely low levels. The metal precipitating groups are incorporated onto a solution polymer backbone. This allows for simultaneous precipitation/clarification, without the need for an additional step of flocculant addition and settling.

NALMET is effective on monovalent and divalent soluble heavy metal ions such as copper, nickel, zinc, lead, mercury, cadmium and silver. It exhibits a preferential binding, the following being a list in order of the strongest affinity to the weakest affinity:

Mercury>silver>cadmium>copper>lead>zinc>cobalt (II)>nickel>iron (II)>manganese (II)

NALMET is not effective on trivalent or higher valence metals such as chromium, aluminium and iron (III).

IBC Advanced Technologies Inc. – Molecular Recognition Technology (MRT)

Molecular Recognition Technology (MRT) makes use of specially designed ligands attached to solid support material such a silica gel, polyacrylate or polystyrene; the resulting products are marketed under the trade name SuperLig. The solid state ligand materials can be used in several formats: packed bed reactor; SuperLig-in-Pulp for slurries or unclarified solution treatment; and membrane cartridges for extremely high flow rates and as solvent extractants.

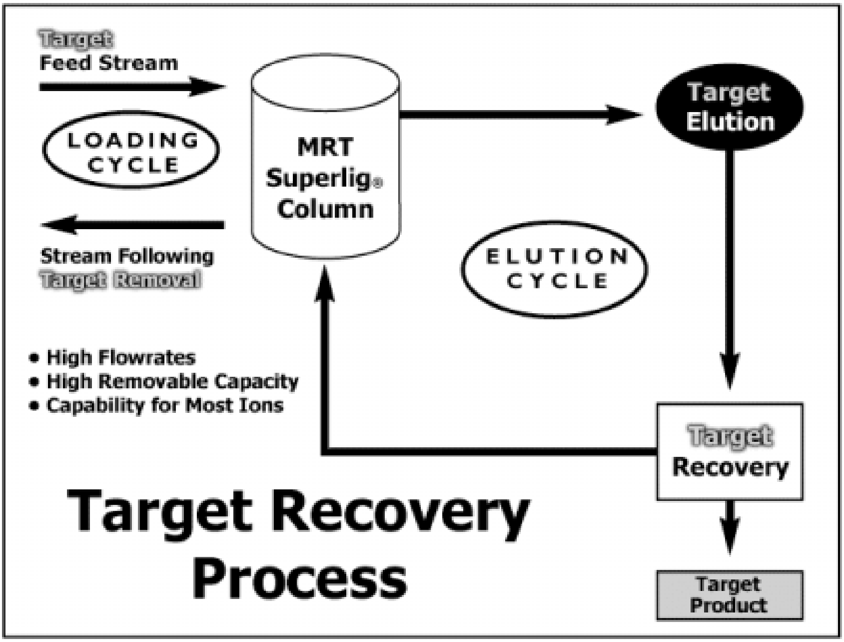

Figure 3.9. Schematic representation of the MRT process (from IBC Advanced Technologies Inc.; website: www.ibcmrt.com).

SuperLig materials exhibit high selectivity for the target ion, high loading capacity and rapid loading and release kinetics. IBC claim that these characteristics often result in substantially lower capital and operating costs than other separation technologies such as ion exchange, solvent extraction, and pH adjustment/precipitation, and list the benefits of the process as follows:

- There is high selectivity for the ion of interest, leading to relatively smaller quantities of SuperLig and capital footprint.
- Most ions removed by the process are in 99%+ purity and can be sold or recycled.
- The separations process is flexible with respect to acid or base content, organics level and higher levels of other ions in the solution matrix.
- The target ion can be removed to very low levels so that effluents can be readily discharged.
- Highly concentrated eluent solutions can be produced from which simple recovery of a high purity, high value product is possible.
- The SuperLig materials are robust and can be used for many cycles without replacement.

According to the company literature, the SuperLig process is suitable for many industrial applications, including, extraction and recovery of metals (e.g. copper, cobalt, nickel, uranium, zinc, manganese, iron and arsenic) from acid mine drainage, although it is not known if there are any such operations currently in existence.

Lewis Environmental Services, Inc. – Enviro-Clean process

The Enviro-Clean process adsorbs metals from the waste stream onto a bed of granular activated carbon in a series of packed bed filters. The chemically treated activated carbon selectively adsorbs the metal ions by a surface attraction/chemical bonding phenomenon. The metals are thus retained in the filter and the resultant water complies with US EPA regulations for discharge, or it can be re-used on site. Once the filters are loaded with metals, they are taken off line and sent back, as non-hazardous waste, to the suppliers (Lewis Environmental Services in Pittsburgh, Pennsylvania) for regeneration. The metals are recovered electrolytically and the carbon recycled back to the client for continued use, having an effective life of 25 to 30 cycles. The necessity to return the carbon material to the suppliers obviously limits the use of this process outside the USA.

The process is deemed suitable for use with single or multi-component metal streams and has been successfully used to treat waste streams containing chromium, copper, zinc, lead, silver, cadmium, mercury, iron, selenium, nickel and molybdenum. However, although the company literature claims it is suitable for treating acid mine drainage, there are no such operations currently in existence.

*Selective Environmental Technologies, Inc. (Selentec) – MAG*SEP*

The MAG*SEP process uses ion exchange principles and magnetised particles to separate heavy metals and radionuclides from water and aqueous waste streams. The process uses engineered particles for the selective recovery of specific contaminants. The very small particles (70 to 300 microns) have a core of magnetite with an acrylic coating. A functionalised resin that is similar to those used in ion exchange columns is applied to the acrylic. The resin then selectively adsorbs the target metals onto its surface. The particles and contaminants are removed from the waste stream by a magnetic filtration system. The process is suitable for treating high flow rates.

Selentec has a large array of functional resins and a combination of resins appropriate to the particular contaminants of concern is used. After treatment, the contaminated particles can be regenerated for re-use or disposed in a stable final waste form.

This technology has been tested at Berkeley Pit, Butte, Montana, where it successfully removed iron and copper from the acid minewater and on metal-contaminated groundwater at the Savannah River Site (Wagner 1997).

Electrochemical Design Associates, Inc. – Selective Ion Recovery System

The Selective Ion Recovery System (SIRS) has been developed to selectively recover arsenic, antimony, perchlorate, chromate, selenate and borate from solution. It uses highly selective ligands to extract the contaminants and a closed loop process uses a caustic flush to extract the recovered contaminants in a stable, non-soluble form. A proprietary electrochemical process is used to recover arsenic in an insoluble form. This process is currently being used on drinking water problems around the world.

3.5 ELECTROCHEMICAL TREATMENT TECHNOLOGIES

As the behaviour of metals in solution is often controlled by their electrochemistry, the use of electrical technologies in the treatment of acid mine drainage has received attention from researchers and process developers. Several electrochemical techniques are often used in combination to facilitate direct deposition of a metal from a solution of its ions.

Geokinetics – Rotating Cylinder Electrode cells

Rotating Cylinder Electrode (RCE) cells are effective for removing metals from both high and low concentration metal solutions. Very high mass transfer conditions are achieved by shearing the electrolyte between a rotating cathode and stationary anodes (and cell casing). Electrode potential governs the plating conditions for different metals, so that different metals can be plated out sequentially across several RCEs. The high degree of control in the process means that high purity metal dendrites (powdery deposits) are plated onto the rotating cathodes, and integral scrapers remove the deposits as powder during continuous operation. The metal powder is then separated from the electrolyte by mechanical filtering or cyclone processes and forms a marketable product of the process. RCEs have been tested at a former mining site where zinc and copper were separated from minewater by passing the water through two RCE units. In the first electrode potentials were set to plate copper and in the second to plate zinc. Both metals were recovered at better than 98% purity and recovery rate within the integrated system (www.geokinetics.com).

Geokinetics – Electrochemical Ion Exchange

The Electrochemical Ion Exchange (EIX) technique can be used on its own for metal recovery from dilute solution, or in conjunction with RCE cells, to produce high quality metal products and clean water. In the latter example, effluent from the RCE cells, which contains low concentrations of metals, is passed through an EIX process to remove the remaining metals. The resultant concentrated solution of metal ions can then be fed back from the EIX process to pass again through the RCE cells, thereby recovering more of the metals in a marketable form.

The Electrochemical Ion Exchange process uses ion exchange media (resins and special ceramic materials) attached to the anode or cathode in a free flowing electrolysis cell. The electrodes are used to create a field that aids adsorption or elution by simply switching polarity. Thus far superior adsorption rates are achieved as compared to conventional ion exchange methods, since the ions are effectively ‘driven’ into the exchange media. Chemical eluants are not required to regenerate the exchange medium so that the resultant concentrated metal solution can be recycled to the RCE cells for further treatment, and the exchange medium has a considerably longer effective life than media used in conventional ion exchange (up to 2000 cycles) since the electrochemical elution process is less stressful than acid base elution.

The drawbacks of this process are that there is virtually no treatment of acidity through the ion exchange, and that sulphate must be removed after the copper has been stripped but before zinc and iron can be plated successfully. Thus additional process mechanisms such as alkali addition and/or electrodialysis need to be incorporated into the overall treatment process.

Laboratory-scale tests using the combined RCE-EIX process have successfully treated minewater samples from Montana and California (Clarke *et al.* 1996) and the technique is being subjected to further tests in the USA and Australia.

Aprotek – Ion Conduction Agglomeration system

The high tension Ion Conduction Agglomeration (INCA) system is an electrolytic recovery process that removes or recovers soluble and particulate metals from aqueous solutions. Unlike other electrolytic technologies, the INCA system incorporates tubular cathodes that are coated with proprietary materials that are specific to the metals to be targeted. When power is applied, the targeted metallic ions in the solution destabilise and agglomerate to the tubes. The metals are collected on a special plate system, where they are consolidated into a sludge and fall into a collection vessel. The sludges, which contain approximately 66% water, are then drained and dried in a drum drier.

The standard flow-through system is a modular unit that can process up to 130 l/s. Larger volumes can be processed by increasing unit size or by adding additional modules. Each INCA module is individually configured to recover a desired element or series of elements in a separate stream, thus enabling selective recovery of metals from multi-metallic streams, such as minewater.

This process would theoretically be suitable for treating minewater and has been subject to demonstration testing, but there are no currently operating examples.

3.6 PHYSICAL PROCESS TECHNOLOGY

Physical process technology, for example by use of membranes, reverse osmosis and microfiltration, can be used for minewater treatment.

Ionics – RCC Brine Concentrators, Reverse Osmosis, Crystalliser, Spray Dryer

Ionics make a range of physical treatment process equipment, with applications for industrial wastewaters and minewater.

The RCC Brine Concentrator works on the principles of heat exchange, evaporation and distillation and enables up to 95% of the wastewater to be recovered as a high purity distillate. The remaining 5%, a concentrated slurry, can be reduced to dry solids in a crystalliser or spray dryer. The crystalliser is a forced circulation evaporator that uses a mechanical vapour compressor or plant steam as the energy source, and can recover specific salts from a mixed-salt waste stream. The spray dryer atomises the wastewater slurry inside a hot chamber, instantly vapourising the water droplets and leaving only dry salts behind.

Ionics also produce a Reverse Osmosis Membrane System which can be used to recover high purity water from highly saline minewaters.

A system using several of these components is in operation in Poland. Highly saline minewater from coal mines is sent to a reverse osmosis system in which about 75% of the water is recovered as drinking water. Reject from the reverse osmosis system is sent to two Brine Concentrators, which recover distillate for the coal mine power plant. Concentrated brine is then sent to a crystalliser; this recovers 99.5%

pure sodium chloride, which is marketable and helps offset the operating costs of the treatment plant (www.ionics.com).

USF Memcor – Continuous Microfiltration system

The Memcor Continuous Microfiltration (CMF) system is in essence a very fine filter with a patented self-cleaning air backwash mechanism. The CMF system is used in municipal, commercial and industrial applications to purify large volume fluid flows and is often used to ensure consistent feed quality ahead of reverse osmosis treatment. Although probably not directly applicable to minewater treatment, this technology can be used to obtain a potable water supply from minewater contaminated groundwater, for example. Such an application is currently in use at Homesford Sough in Derbyshire where a historic lead mine drainage discharge is used for public water supply. The CMF system removes bacterial contamination.

Other suppliers of filtration equipment are ***Enviro-Main Filter, Inc***. in Canada and ***Stewart Environmental Consultants, Inc***. in Colorado, USA. ***EPOC Water, Inc.*** in California provide a proprietary system that includes precipitation, microfiltration and sludge dewatering, and that has been tested on a highly acidic minewater seep at a Superfund site in California.

Ambient temperature ferrite process

The ambient temperature ferrite (ATF) process is a proposed treatment process for acid mine drainage that uses the 'seeding' principle of the high density sludge process, but which produces magnetic particles, thereby facilitating separation using magnetic filtration (Choung *et al.* 2000, McKinnon *et al.* 2000). The conventional ferrite process necessitates use of elevated temperatures and/or controlled redox potentials in the presence of catalysts, which makes it uneconomical and impractical to use for AMD treatment. However, in the ATF process, control of the solution chemistry, mainly the ferric-to-ferrous ratio, allows magnetic ferrite to form under ambient conditions. Interference by calcium and other non-ferrous metal ions is overcome by addition of carbonate and magnetic seed particles respectively – all metal ions present in the AMD co-precipitate on the magnetic seeds in the form of magnetic ferrite. Some of the resultant magnetic solid products can be recycled as seeds to the start of the process; the remaining solids can be separated as an easily dewatered crystalline spinel ferrite for disposal. The resultant water is generally of drinking water standard, with the exception of the manganese concentration; operating the process at a higher pH might make manganese removal possible.

The ATF process has currently only been tested at laboratory scale, although the process diagram shown in figure 3.10 has been proposed for a scheme to treat a large volume of AMD (McKinnon *et al.* 2000).

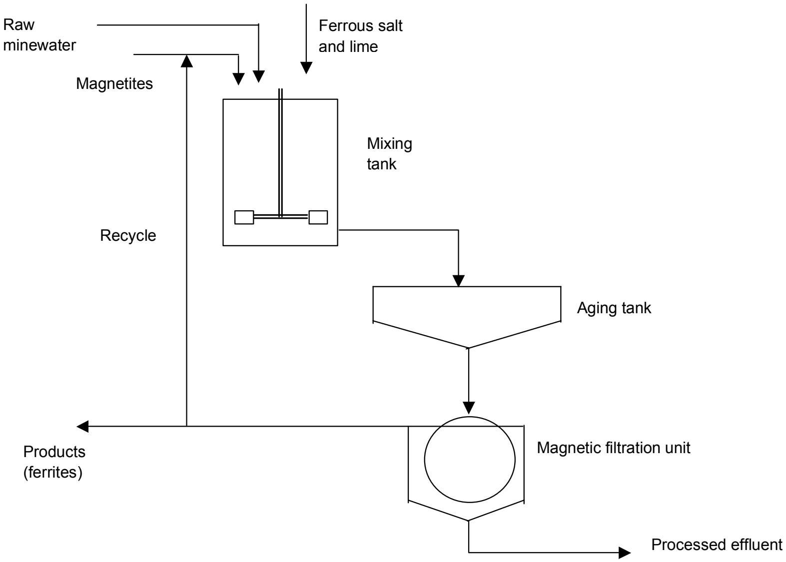

Figure 3.10. Ambient temperature ferrite process.

3.7 ACTIVE TREATMENT – SUMMARY

Active treatment technology may be a necessity, in instances where the scale of the problem is such as to make passive treatment unrealistic, as, for example, at Wheal Jane in Cornwall (see section 5.35), or it may be selected for other reasons, for example lack of suitable land for passive treatment, economic issues such as the value of recoverable materials, or political considerations. It may be a long-term process, for example at operating mines or in cases of severe contamination, or it may be envisaged as a short-term, quick-fix solution which in time can be superceded by a passive system as the change in rebound minewater chemistry permits, e.g. Ynysarwed in South Wales (see case study 5.38). There are also many instances where active treatment may be followed by a passive, polishing step, for example Monktonhall (case study 5.21), Old Meadows (case study 5.25) and Polkemmet (case study 5.27).

Table 3.5. Comparison of general characteristics of active treatment process types

	Selectivity	Purity of product	Degree of neutralisation	Nature of sludge	Complexity of process	Flow of minewater treatable
pH modification	Mostly non-selective	Not pure	Good	Variable stability and density	Simple to complex	Good flow
Ion exchange	Highly selective	Very pure	None	Concentrated solution	Complex	Limited flow
Biology-based treatments	Partially selective	Pure	Reasonable	Sulphide sludge	Simple to complex	Reasonable flow
Other adsorption treatments	Partially selective	Not pure	Some	Labile sludge	Simple	Reasonable flow
Electro-chemical treatments	Mostly selective	Pure	None	Concentrated solution	Complex	Limited flow
Physical process technology	Highly selective	Pure	None	Crystalline	Complex	Limited flow

Approaches to active treatment are many and varied. Table 3.5 summarises the sub-division of treatment types described in this chapter, and presents some of their general characteristics. Initial choice of an active treatment technology may be guided by these general considerations. For example, selectivity and purity of product may be an important consideration if there is a substantial loading of a potentially marketable product, such as zinc or copper, in the minewater. Recovery of such a product would have economic considerations in that its value could be offset against the operational costs of the treatment process.

The degree of neutralisation may be important when the minewater is particularly acidic and where a discharge consent pH value has to be met (discharge consent levels are generally pH 6 to 9).

The nature of the sludge will be an important consideration if there are waste disposal issues to address, for example if there is no on-site disposal facility, the volume and toxicity of the sludge will have considerable cost implications in terms of transport and the type of landfill site that will accept the sludge.

The complexity of the process that can be accommodated will, in part, be determined by the remoteness of the site and the availability of labour – a more complex process will demand more intensive management and monitoring, and a swifter response time in the event of process failure.

Table 3.6. Comparison of individual technologies in terms of process end-products

Technology name	Clean water produced?	Volume of sludge	Toxicity of sludge	Stability of sludge	Reclaimable products?	Other comments
pH modification						
Lime	Yes	High	?	?	No	Tried and tested technology
High density sludge	Yes	Low	Low	Good	Yes	Process can be modified to suit particular minewater
Unipure	Yes	Low	?	Good	?	
Keeco KB-1	Yes	Low	Very low	Good	Yes	
Aquafix	Yes	High	?	?	Possibly	Suitable for remote sites
Hazleton MAXI-STRIP	Yes	Low	?	?	Possibly	
MET HARDTAC	Yes	Low	?	?	Possibly	
Virotec Bauxsol	Yes	High	Low	?	?	Might also be suitable for passive systems
Mintek Savmin	Yes	?	?	?	Yes	Complex process
Hydrometrics CESR	Yes	?	?	?	?	Used for high-sulphate waters
Ion exchange						
Chemeffco GYP-CIX	Yes	Low	Low	?	Yes	
KAD	Yes	?	?	?	Yes	
Dynaphore Forager	Yes	?	?	?	Possibly	Might also be suitable for passive systems
Biology-based treatments						
Paques THIOPAQ	Yes	Low	Low, as long as kept anaerobic	Good in anaerobic conditions	Yes	
NTBC Biosulphide	Yes	Low	Low, as long as kept anaerobic	Good in anaerobic conditions	Yes	
AlgaSORB	Yes	Low	Treatment required prior to disposal	?	?	
Ligand-grafted polymer	Probably	Low	?	?	Yes	

Table 3.6. (continued)

Technology name	Clean water produced?	Volume of sludge	Toxicity of sludge	Stability of sludge	Reclaimable products?	Other comments
Other adsorption treatments						
Nalco–NALMET	Yes	?	?	?	Yes	
IBC – MRT	Yes	Low	?	?	Yes	
Enviro-Clean	Yes	?	?	?	?	Material needs to be regenerated in USA
Selentec–MAG*SEP	Yes	?	?	Good	Yes	
Selective Ion Recovery	Yes	?	?	Good	?	
Electrochemical treatment						
Geokinetics–RCE	Yes	Low	?	?	Yes	
Geokinetics–EIX	Yes	Low	?	?	Yes	
Aprotek–INCA	Yes	Low	?	?	Yes	
Physical process technology						
Ionics–RCC Brine Concentrator	Yes	Low	?	?	Yes	
USF Memcor	Yes	Low	?	?	?	
Ambient Temperature Ferrite process	Yes	Low	Low	?	?	

The quantity of wastewater treatable by a particular process is obviously an important consideration in order to ensure year-round compliance with discharge consents; a process that was unable to cope with peak flows could result in substantial penalties for non-compliance.

In addition to the general characteristics listed in table 3.5, more detailed factors concerning the end-products of the treatment process need to be considered. For example the purity of the treated water will determine whether it is suitable for potable or industrial use; in the latter case this may represent a potential source of income to offset against operating costs, or re-use of process water on the site may decrease the amount of metered water supply that has to be used. The exact nature of the sludge has implications for disposal as considered above, and the possibility of reclaiming marketable end-products from the treatment process has implications of cost benefit and sustainability which may make one process more viable than another. The factors vary with the different proprietary techniques, although the information is often not available and would have to be obtained from the process-providers on a site-specific basis, probably after on-site treatability tests had been carried out. Table 3.6 lists the various active treatment techniques considered in this chapter, and presents answers to these considerations where they are available.

4

Passive treatment

Passive treatment systems are obviously a highly desirable option for treatment of contaminated mine water since, ideally, they do not require ongoing input of material and financial resources as is the case for active treatments. Passive treatments may act on a purely chemical basis, but they are more normally based on biological systems, which may in time become self-sustaining and thus conform to the ideal of passive treatment.

4.1 CHEMICAL PASSIVE TREATMENT

Chemical passive treatments generally comprise a means of neutralising acidity and precipitating metals without the active addition of lime or other alkaline material. This was achieved by Lapakko and Antonson (1990) by allowing contaminated minewater to flow through a bed of limestone. They concluded that this is a feasible method for treating relatively low-volume discharges of acidic mine drainages that have a low iron content. The limestone bed improved the quality of the water so as to be suitable for discharge to receiving water courses and gave indications of having reasonable longevity, although data were only available from a 6 month trial. However, dissolution of the limestone, and clogging of the pore spaces between limestone particles by the precipitated

Melanie Brown, Bob Barley and Harvey Wood. ISBN: 1 84339 004 3

metals, mean that this type of system will have a finite life and thus does not truly conform to the ideal of a passive treatment, since a self-sustaining balance will never be achieved.

4.1.2 Anoxic limestone drains

The reason that the above system could only be applied to drainages with low iron content is that precipitation of iron hydroxide, due to neutralisation, would rapidly coat the limestone particles, thus preventing any further contact between the limestone and the acidic drainage. This problem was overcome by workers at the Tennessee Valley Authority (TVA, Turner and McEvoy 1990, Brodie *et al.* 1993) by constructing anoxic limestone drains (ALD). In these systems the crushed limestone is placed in an underground channel that is subsequently capped with impermeable clay to prevent the ingress of oxygen. Ideally, all iron in the water entering the ALD is in the reduced, ferrous, state. In the anoxic environment of the drain, the ferrous iron does not oxidise and ferric hydroxides do not form. The limestone, rather than becoming armoured with ferric precipitate, therefore dissolves, raising the pH of the mine drainage and adding alkalinity. The following reactions are assumed to occur within the ALD (Brodie *et al.* 1993):

$$CaCO_3 + 2H^+ \rightarrow Ca^{2+} + H_2CO_3 \tag{4.1}$$

$$CaCO_3 + H_2CO_3 \rightarrow Ca^{2+} + 2HCO_3^- \tag{4.2}$$

$$CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3^- \tag{4.3}$$

Equation 4.1 reacts limestone with acidity (at $pH < 6.4$) to form free calcium and carbonic acid. The reaction of carbonic acid with limestone (equation 4.2) produces bicarbonate alkalinity. When the production of alkalinity has raised the pH to > 6.4, equation 4.3 becomes the major reaction and bicarbonate becomes the dominant dissolved CO_2 species.

The effluent from the drain then enters a settling pond, and/or constructed wetland, where metal oxidation, hydrolysis and precipitation reactions occur in a strongly buffered solution. ALDs have proved to be a very effective passive treatment tool and are now widely used either alone or in conjunction with constructed wetlands (Nairn *et al.* 1991, Hedin and Watzlaf 1994, Cambridge 1995). However, the mine water to be treated must conform to certain criteria in order to be suitable for passage through an ALD. These criteria are:

- low dissolved oxygen content;
- all iron in the ferrous state;
- low aluminium content.

Failure of the mine water to comply with any of these criteria will lead to rapid malfunctioning of the ALD, due either to ferric hydroxide precipitation causing armouring of the limestone or to aluminium precipitate clogging up the pore spaces (Hamilton *et al.* 1994, Wildeman *et al.* 1997, Robbins *et al.* 1999). Robbins *et al.* (1999) also noted that low flow rates through an ALD were more likely to give rise to blockage within the ALD due to the lack of a 'flushing' effect that higher flows might provide. They also investigated the microbiological component of ALDs and showed that the processes are not entirely chemical, but have a biological input as well. An ALD flow collection system in Pennsylvania has shown signs of becoming blocked, presumably by aluminium precipitates (influent Al concentrations are in the region 20–30 mg/l), despite the use of coarse-sized limestone (7.6 to 10 cm) to try to prevent this occurring (Rose *et al.* 2001). Provision of pumps at the ALD outlet to dislodge and flush the precipitate have proved successful at this site. The ALD at Wheal Jane, in Cornwall, repeatedly and regularly became blocked with aluminium precipitates and was eventually by-passed to enable the passive pilot plant to operate without the need to regularly back-flush the ALD (see case study 5.35).

A recent development in the USA has adapted a method commonly used to flush wastes in animal barns to the passive, periodic flushing of precipitates from ALDs and vertical flow alkalinity producing systems (SAPS or RAPS) (Vinci and Schmidt 2001).

The technique uses devices known as automatic dosing siphons, which provide an automated, periodic flushing mechanism. There are two distinct phases in the operation of these siphons: a fill phase and a drain phase. Thus the nature of flow through the ALD or SAPS is changed from continuous plug flow to a more intermittent batch-type flow. This change in the nature of flow has implications for the design criteria of passive systems, but early results from test sites using these systems suggest that these design issues can be successfully resolved (Vinci and Schmidt 2001). These authors note that ongoing monitoring of the test systems is required to assess further the applicability of this technology to passive minewater treatment.

Although the ALD technology has been shown to be an effective way of adding alkalinity and raising the pH of acidic mine drainage, and is certainly much cheaper than active lime dosing, the fact that the limestone dissolves means that it is not a self-sustaining system. Most ALDs have been designed to have a lifespan of about 20 years, after which time it would be necessary to dig up the drain and replace the limestone or divert the minewater flow to a newly constructed ALD. Attempts at predicting longevity of passive, alkalinity-producing systems, such as ALDs, by calculating the rate of calcite dissolution have been made by Rose (1999) and Watzlaf *et al.* (2000).

Sizing criteria for ALDs have been proposed (Hedin and Watzlaf 1994) based on limestone dissolution rate and minimum retention time, which ideally should be in the region of 15 hours (Hedin and Watzlaf 1994). However, the calculation of retention time requires a value to be placed on the void space within an ALD and recent work, which involved on-site measurement of pore space (Cravotta and Trahan 1999) suggests that the value of 50% taken by Hedin and Watzlaf (1994) and determined by Cravotta and Trahan (1999) in the laboratory is a considerable over-estimate. The actual value of pore space determined *in situ* was 14%, which would give a substantially different value for the calculated retention time than if a value of 50% was used.

Another use for ALDs has recently been investigated by Nuttall and Younger (2000), who considered the efficacy of ALDs for removing zinc from minewater. Geochemical modelling work suggested that zinc removal by precipitation as zinc carbonate ($ZnCO_3$, smithsonite) was a feasible treatment option. This led to laboratory and pilot-scale test work that gave zinc removal rates of approximately 50% and 22% respectively (see Nenthead case study 5.23). The lower removal rate achieved in the field was attributed to lower temperatures and larger size limestone used in the ALD.

An ALD intended to remove zinc was incorporated into a pilot-scale passive treatment system at an active mine in the Scottish Highlands (Jarvis and Younger 2000; see Foss, Aberfeldy case study 5.14). Limited alkalinity generation due to low temperatures was thought to be responsible for the low removal rates achieved (10 to 15%), and it was suggested that increased retention time, and/or burying the treatment units for insulation purposes, would significantly enhance zinc removal performance.

An ALD to receive zinc-rich minewater from a lead-zinc-flourspar mine has recently been constructed at Tailrace Level, Frazer's Grove in County Durham (see case study 5.34).

4.1.2 Open limestone channels

Recent work has revisited the idea of using open limestone channels for minewater treatment, particularly for use when the water chemistry does not conform to the criteria necessary to make ALDs a viable option.

Ziemkiewicz *et al.* (1997) carried out a series of laboratory and field studies looking at the removal of acidity facilitated by passing acid minewater through open limestone channels. They showed that armoured limestone (limestone that has been coated by metal, particularly iron, hydroxide precipitates) was only 2 to 45% less effective at neutralising acidity than unarmoured limestone. These workers claim that open limestone channels may be a viable passive treatment option, particularly on sites where it is possible to construct the channels on steep slopes (> 20%) and where flow velocities are high. Both these attributes help keep metal hydroxide precipitates in suspension, thereby avoiding blocking the pore spaces in the limestone channel. Alternatively, where such slopes are not possible, periodic, gentle agitation should be sufficient to dislodge precipitated coatings (Rose and Lourenso 2000). The latter workers attribute part of the success of open limestone channels at removing acidity to the precipitation of iron hydroxysulphates within the channel, a process that is not dependent upon the limestone.

Cravotta and Trahan (1999) consider using oxic limestone drains (OLDs) to add alkalinity and remove metals simultaneously, as opposed to ALDs that are intended only to add alkalinity, with metal removal being achieved subsequently in a settling pond or wetland. The OLDs constructed and studied in the field had the same construction as ALDs, but the influent minewater contained 1 to 4 mg/l oxygen as well as ferric iron and aluminium (i.e. minewater chemistry that is considered not to be suitable for ALD treatment). Over a period of a year, pH was increased from less than 4 in the influent to between 6.2 and 7 in the effluent. Over 95% of influent iron and aluminium was removed; manganese, copper, zinc, cobalt and nickel were also removed by co-precipitation with hydrous iron and aluminium oxides in the downflow section of the OLD where pH was over 5. Limestone dissolution continued despite coatings of hydrous oxides on the limestone, particularly at the inflow end of the OLD where the pH was lowest. Removal of accumulated precipitates could be achieved by incorporating a perforated-pipe subdrain into the OLD. These workers conclude that by combining neutralisation and hydrolysis reactions within an OLD, less land area would be required for treatment than if a conventional ALD/oxidation-settling pond system was used. They also note, however, that much further work needs to be carried out (for example, on rates of limestone dissolution and the effects of hydrolysis products on limestone dissolution, co-precipitation and sorption of trace metals) before the design of OLDs can be optimised.

4.1.3 Other passive chemical treatment systems

Other materials for use in passive chemical treatment have been investigated: for example Bauxsol, which is a modified bauxite processing residue (see Chapter 3) and could be used in a similar manner to limestone in an open channel; Forager Sponge, which is an open-celled cellulose sponge containing a polymer (see Chapter 3), and could be used in a well or trench to intercept contaminated groundwater; and reactive barriers, which can be either essentially biological or chemical, or both (see section on migration control).

4. 2 BIOLOGICAL PASSIVE TREATMENT

The only truly self-sustaining, and therefore strictly passive, treatment systems are those with a biological component. In such systems, after the initial provision of suitable conditions, the fauna and flora achieve a dynamic, sustainable balance and long-term treatment can occur. Such systems have the attraction not only of longevity, but also, if their inception was sufficiently considered, of being natural systems that work in harmony, not in conflict, with, natural forces. The concept of humans working in harmony with nature for the mutual benefit of both has been called ecological engineering and was first described by Odum in the early 1960s (Odum 1962).

Ecological engineering, or the building of sustainable and self-designed ecosystems, is becoming a useful paradigm in ecology for dealing with all manner of environmental problems, among which is the amelioration of contaminated minewater (Mitsch 1993). An ecological engineering approach to minewater treatment has obvious financial attraction, since, if indeed such a self-sustaining treatment system is achievable, the need for ongoing expenditure on active treatment is avoided. There are also other benefits of this approach, which are summarised as (Mitsch 1991):

- the restoration of ecosystems that have been substantially disturbed by human activities such as environmental pollution, climate change or land disturbance;
- the development of new sustainable ecosystems that have human and ecological value;
- the identification of the life support value of ecosystems will ultimately lead to their conservation.

Minewater pollution can be tackled on the basic premise that sulphide weathering and acid generation are natural processes that are exacerbated, but not caused, by mining activity (Kalin *et al.* 1989, Kalin 1991, Kalin 1992). Since the formation of contaminated minewater is a natural phenomenon, it should be possible to ameliorate it by natural means. Construction of passive treatment facilities requires thorough consideration not only of immediate ecological implications, but also of the disciplines of microbial biogeochemistry (which examines microbially catalysed reactions and their kinetics in mineral cycles) and geomicrobiology (which examines the role of microbes in forming ore bodies and the role they currently play in rock and soil weathering, sediment formation and transformation). A suitably considered approach to passive biological treatment should ultimately lead to the formation of new ore deposits, not environmental hazards of the future.

The important point to note about the concept of ecological engineering, which has been briefly introduced above, is that it is an approach which has due regard for the intricacies and complexities of natural systems. All too often, constructed wetlands, or other biological passive treatment systems, are designed and constructed by engineers who do not have this perception and thus the systems do not achieve a self-sustaining status (McGuinness *et al.* 1996).

Kalin (2001) states that consideration should be given to environmental conditions outside the immediate area of the mine site, as well as the local conditions, in order to ensure that an ecologically integrated and sustainable treatment system is established.

4.2.1 Constructed wetlands

4.2.1.1 History and occurrence

Wetlands have developed their role in the natural world as purification systems, providing a buffer zone between the terrestrial and aquatic environments. They receive, hold and recycle nutrients and other components mobilised from upland regions. They also delay the incursion of these into the marine or freshwater body with which the wetlands are associated. Thus, for as long as mankind has discharged sewage and other wastewaters into the environment, wetlands have been involved in the cleaning of this water, with a greater or lesser degree of intention. It is known that ancient Chinese and Egyptian cultures used wetlands for disposal of wastewater (Brix 1994), and that, historically, wetlands have developed in ditches or topographical depressions into which wastewaters have been discharged. The first documented evidence of the use of constructed wetlands, as opposed to naturally occurring wetlands, for water treatment dates from 1904. This handwritten document (a transcript of which is given in Brix 1994) describes the disposal of wastewaters from individual dwellings to a specially prepared area planted with water-loving plants. In the 1950s and 1960s, work by Seidal (reviewed in Brix 1994) demonstrated the effectiveness of bulrushes (*Schoenoplectus lacustris*) for removing organic, inorganic and microbiological contaminants from wastewaters. This pioneering work led to the development, during the 1970s and 1980s, of reed bed technology to treat organic wastewaters both in Europe and the USA.

The current status of constructed wetlands for wastewater treatment is of great interest and development, evidence of which can be seen in the number of books collating contributions from authors on a global scale (Reddy and Smith 1987, Reed *et al.* 1988, Hammer 1989, Cooper and Findlater 1990, Moshiri 1993, Kent 1994, Kadlec and Knight 1995, Cooper *et al.* 1996, Nuttall *et al.* 1997, IWA 2000).

The role that wetlands play in the treatment of contaminated minewater, and the potential of exploiting this natural attribute for human advantage, was first recognised during the late 1970s and early 1980s in the USA. Two independent studies were carried out to assess the adverse effect that acid mine drainage was having on natural wetland vegetation (Huntsman *et al.* 1978, Wieder *et al.* 1982). These studies found that there was a lack of adverse effects and that, in fact, natural treatment of the minewater was occurring. Both these studies were of *Sphagnum*-dominated bogs, which were found to be raising pH and removing sulphate and iron from coal mine drainage.

These observations led to the construction of pilot-scale wetlands and then full-scale field trials to explore the remedial action that was occurring in *Sphagnum* bogs (Spratt and Wieder 1988, Dietz and Unz 1988, Hedin and Hyman 1989). However, these trials were not successful and attention was turned to the use of emergent macrophytes, principally *Typha*, instead of *Sphagnum*.

Initially the emphasis was on drainage from coal mines and was carried out by the US Bureau of Mines (e.g. Kim *et al.* 1982, Kleinmann 1989), Tennessee Valley Authority (e.g. Brodie *et al.* 1988, Brodie 1990), University of West Virginia (Sencindiver and Bhumbla 1988) and Villanova University, Pennsylvania (Wieder 1988, Wieder *et al.* 1989). In the 1980s over 140 wetlands were constructed in the eastern United States alone to receive drainage from coal mines (Wieder 1989). There is no up-to-date survey information on the number of constructed wetlands currently operative in the USA, although there is an active and ongoing programme of site clean-up operations at abandoned mines, which includes wetland systems where appropriate (Shea 2000). A review of constructed wetlands for AMD treatment (Gazea 1996) includes reference to many examples in the USA.

Work on effluent from metal mines has been carried out by the Colorado School of Mines (Emerick *et al.* 1988, Howard *et al.* 1989). This work initially comprised observation of natural wetlands with metal contaminated minewater input and led to the construction of a research facility at Big Five Tunnel in the Idaho Springs mining district, Colorado (Howard *et al.* 1989, Wildeman and Laudon 1989).

In Canada, awareness of the potential of wetlands for amelioration of contaminated minewater has been growing over the past decades, a major impetus to research having been provided by the formation of the MEND (Mine Environment Neutral Drainage) and BC AMD (British Columbia Acid Mine Drainage) programmes in 1989 and 1986 respectively (Filion *et al.* 1990). The current MEND programme, MEND 2000, is a 3 year programme that started in 1998, and has the key aim of technology transfer, to provide state-of-the-art information and technology developments to users (Tremblay 2000). A review of natural wetlands that receive and ameliorate minewater (Sobolewski 1999) is based largely upon wetlands in Canada, although reference is also made to natural wetlands in the USA and Australia.

On a global scale, the prospect of wetlands providing a passive, 'walk-away' treatment option has ensured a keen interest in this technology, and numerous research initiatives have been undertaken. In South Africa, there were reported to be about thirteen constructed wetlands for mine effluent treatment in 1992 (Fourie *et al.* 1992) and research into the potential of passive treatment for mine drainage and a number of other industrial discharges is continuing (Wood 1989, Wood 1990). In China, the concept of ecological engineering is well established; combined water treatment systems, which often include fish production ponds as well as areas of emergent macrophytes and water hyacinth, are commonly implemented (Shijun and Jingsong 1989, Jingsong and Honglu, 1989, Lan *et al.* 1990, Tang, 1993).

In Australia, research has been carried out on both constructed and naturally occurring wetlands associated with mining operations. These wetlands are favoured by the year-round water supply resulting from pit dewatering and are thus able to flourish in areas in which rainfall is too infrequent to support wetlands (Dunbabin and Bowmer 1992, Noller *et al.* 1994). The latter workers found numerous examples of wetlands at mine sites in Northern Australia that were effectively removing a suite of metal contaminants, including As, Cd, Cu, Fe, Mn, Ni, Pb, U and Zn. There is also active research into acid mine drainage prediction and prevention strategies being carried out by the Australian Mineral Industries Research Organisation (AMIRA, Greenhill 2000).

In the eastern European countries, constructed wetlands are seen as an attractive and economical option for treating contaminated minewater. In Bulgaria, bench-scale and pilot-scale tests have been carried out (Groundev *et al.* 1994, Gaydardjiev pers. comm.), on effluent from a uranium mine and a copper smelter, respectively. However, the results from the latter trials have been inconclusive to date (Gaydardjiev pers. comm.).

In the UK, wetland treatment for domestic wastewater treatment is well-established (Cooper and Boon 1986, Cooper and Hobson 1989, Hiley 1990, Collett *et al.* 1993, Gray and Biddlestone 1995, Nuttall *et al.* 1997). There are currently a growing number of constructed wetlands for treatment of minewater discharges in the UK, a review of which is one of the primary objectives of this book.

4.2.1.2 Components

In essence, wetlands can be considered to comprise the following five components (Hammer and Bastian 1989):

- substrates (which may have widely varying rates of hydraulic conductivity);
- plants adapted to water-saturated anaerobic conditions;

- a water column (water flowing in or above the substrate);
- vertebrates and invertebrates;
- aerobic and anaerobic micro-organisms.

This list of components applies to natural wetlands and should apply to constructed wetlands if the goal of a self-sustaining ecosystem is to be achieved. There is a great temptation, in the design and construction of treatment wetlands, to simplify the natural complexity and to attempt to impose a degree of control over the vastly complex processes and interactions that occur naturally (McGuinness *et al.* 1996). Hammer and Bastian (1989) stress that natural wetlands are biologically complex and cover large land areas, and that care must be taken in reducing them to minimal components and treatment areas. Such loss of biological complexity might prevent the achievement of a balanced, self-sustaining ecosystem which is the ultimate aim of passive treatment. Wetzel (1993) also sounds a note of caution regarding the need for species diversity and the importance of designing treatment wetlands to mimic as closely as possible natural integrated systems. Work carried out by Sistani *et al.* (1999) on wetlands constructed on coal mine spoil suggests that they developed the botanical and biogeochemical characteristics of natural wetlands within 3 to 4 years, with the exception of reduced accumulation of organic matter.

Despite the recognised need to mimic natural wetland ecosystems as closely as possible, a certain degree of simplification is inevitable and research into the relative importance of each of the components in treating contaminated minewater, and the effect of contaminated minewater on these components, has attempted to identify the most important features to include in constructed wetlands (Brown 1996). Thus the main thrust of research into wetlands for contaminated minewater treatment should be the identification of the key ameliorative processes so that constructed wetlands can be designed to optimise the conditions in which these processes occur.

4.2.1.3 Processes

There are two principal aims of minewater treatment:

- to neutralise acidity;
- to precipitate out metals.

In essence, a constructed wetland provides an environment in which processes can occur to achieve these objectives. Of the components listed above, it is probably the microbial population that is of primary importance, with the substrate and plants providing nutrients and an increased surface area for bacterial attachment and colony development. The existence, and in some instances the juxtaposition, of aerobic and anaerobic environments is a feature of wetlands that enables habitation by a population of microbes with diverse needs, but each with a role to play in contaminated minewater amelioration.

It is currently understood that the two most important processes occurring in constructed wetlands are:

- oxidation reaction;
- reduction reactions.

Other processes that are considered to be of minor importance (Wildeman *et al.* 1991) are:

- exchange of metals by an organic-rich substrate;
- adsorption of metals by ferric hydroxides;
- metal uptake by living plants;
- filtering suspended solids and colloidal matter from water;
- neutralisation and precipitation through the generation of ammonia and bicarbonate by bacterial decay of organic matter;
- adsorption or exchange of metals on to algal materials.

An additional mechanism that was found to be of major importance in a semi-natural wetland receiving minewater from a metalliferous mining area is adsorption of metals on to accumulated plant debris (Brown 1996).

4.2.1.4 Oxidation reactions

Oxidation of ferrous to ferric iron, followed by hydrolysis, is a major mechanism of iron removal from minewater due the low solubility of ferric hydroxide. The oxidation of ferrous to ferric iron is chemically a slow process, but iron-oxidising bacteria such as *Thiobacillus ferrooxidans* can dramatically increase the rate of this reaction. This oxidation and precipitation process forms the basis of many conventional, active treatment methods (section 3.1). However, the hydrolysis of ferric iron to give ferric hydroxide (see equations 4.4 and 4.5), releases H^+ ions into solution. If there is insufficient alkalinity in the minewater to buffer the increase in H^+ ions released as a result of hydrolysis (Hedin *et al.* 1994), the pH will fall, which is contrary to the aims of minewater treatment. Also, lower pH is not conducive to ferric hydroxide precipitation (ferric iron is soluble at pH values of less than about 3.5), so this series of reactions is self-limiting.

$$Fe^{2+} + \tfrac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \tfrac{1}{2}H_2O \qquad (4.4)$$

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^+ \qquad (4.5)$$

Various studies have been carried out to determine the importance of biotic factors in the oxidation of ferrous iron; some recent ones are considered below.

Kirby *et al.* (1999) carried out field studies and modelling work to assess the role of micro-organisms in ferrous iron oxidation. They found the following relationships between pH and ferrous iron oxidation rate:

pH 2.8 to approximately pH 5 – ferrous iron oxidation rate is negatively correlated with pH, and is catalysed by bacteria;

pH 5 to pH 6.4 – ferrous iron oxidation rate appears to be primarily abiotic and is positively correlated with pH;

pH above 6.4 – ferrous iron oxidation appears to be independent of pH.

Thus, in acidic minewater treatment, biotic oxidation of ferrous iron is more rapid at lower pH. A design consideration could therefore be to utilise this characteristic in the treatment process, by, for example, incorporating aerobic settling ponds before any pH modification is carried out. Substantial oxidation of ferrous iron from a net acid minewater in a settling pond was noted by Rose *et al.* (2001), and the high removal rates of iron in the first part of the lime free aerobic reed bed at Wheal Jane (see case study 5.35) is presumably due to this biotic catalylic effect.

The abiotic nature of ferrous iron oxidation at pH values greater than 5 was demonstrated in the field by Athay *et al.* (2001).

Seasonal effects on iron oxidation were investigated by Perry (1997). The minewater in this study was of neutral pH, so the process would be primarily abiotic. The principal conclusion of this worker was that the seasonal effects could be related to pH, which in turn was a function of dissolved CO_2, as carbonic acid, in the minewater. This concentration was higher in the winter, which caused a lower pH and slower rate of oxidation. A design consideration arising from this work is that in bicarbonate buffered, neutral minewaters, promotion of CO_2 degassing is of major importance to promote oxidation of ferrous iron. Thus the effectiveness of the often-applied strategy of aerating a net-alkaline minewater may be due more to CO_2 degassing (with attendant pH rise) than to introduction of oxygen into the water. The importance of early and vigorous aeration is thus emphasised (Watzlaf *et al.* 2001).

When ferric iron precipitates as hydroxides or oxyhydroxides, the solid is quite gelatinous. The scavenging properties of these precipitates is well known (Benjamin and Leckie 1981, Stumm and Morgan 1981, Johnson 1986), and a useful co-benefit of water quality improvement by precipitation. The metals removed from solution by co-precipitation with Fe and Al hydroxides includes As, Cd, Cu, Pb and Zn.

Manganese also undergoes oxidation reactions and co-precipitates with iron oxyhydroxides, but the geochemistry is more complex and not fully understood (Hedin *et al.* 1994). The oxidation of manganese is strongly pH dependent and the abiotic oxidation of Mn at pH < 8 is very slow. Although the oxidation and precipitation of Mn is accelerated in the presence of MnO_2 and FeOOH, this effect is not noticed if there is ferrous iron present since the ferrous iron reduces oxidised forms of Mn. Even allowing for the fact that biotic processes may increase rates of Mn oxidation in the field as compared to those rates achieved in the laboratory (Hedin *et al.* 1994), removal of Mn occurs at much slower rates than does the removal of Fe. Successful Mn removal was achieved at a constructed wetland receiving coal combustion by-product leachate in Pennsylvania, which was attributed to the relatively high pH of the influent water (average pH 7.2) (Ye *et al.* 2001). These workers found little seasonal variation in the treatment efficiency of the wetland. Conversely, monitoring of a newly constructed wetland in Ontario showed strong seasonal effects in both Fe and Mn retention, with the wetland acting as a source of these elements during the winter (Goulet and Pick 2001).

If Mn is a contaminant of particular concern, special features are often incorporated into passive treatment systems to attempt to remove it (for example at Wheal Jane, see case study 5.35). At a former coal mine site in Pennsylvania, limestone cells are incorporated into a passive treatment scheme that includes ALDs and aerobic wetland cells before the limestone cells (Robbins *et al.* 1999). A complex, multi-species ecosystem of bacteria, cyanobacteria, diatoms, green algae and fungi effectively oxidise and immobilise Mn such that discharge limits are met. The researchers identified twelve different strategies being used by the biotic community to achieve this oxidation.

Research carried out at the Tennessee Valley Authority Constructed Wetlands facility in Alabama investigated the removal of Mn in limestone gravel systems (Sikora *et al.* 2000). It was found that removal rates of between 1 and 17 g/m^2.day were achieved, which are substantially greater than removal rates generally achieved in surface flow aerobic wetlands. This is attributed to the greater surface area available for microbial attachment. It is recommended that in order to achieve successful Mn removal, the influent water should be well oxygenated, have less than 2 mg/l Fe and a pH of greater than 6.8. These workers suggest using a Mn loading rate of 5 to 10 g/m^2.day to design limestone gravel beds for Mn removal or, preferably, to base the design size on the biofilm area, i.e. the surface area of the gravel media. In this case, a loading rate of 0.05 to 0.1 g/m^2.day is recommended, where the area refers to the biofilm area.

Oxidation is thus a most important mechanism for the removal of iron from solution, and to a lesser extent for the removal of other metals as well. However, the gelatinous nature of the precipitate means that large volumes are deposited, which could rapidly smother plants and clog up flow paths. Periodic maintenance to remove accumulated precipitate is thus likely to be needed. Kalin (2001) recommends always using an oxidation pond prior to wetland treatment to precipitate out hydroxides and/or other secondary mineral products.

4.2.1.5 Reduction reactions

In the absence of oxygen, sulphate-reducing bacteria (SRB) such as *Desulfovibrio* oxidise organic matter using sulphate as a terminal electron acceptor, thus reducing it to sulphide, which is substantially hydrolysed as free hydrogen sulphide gas (see section 3.3 for further consideration of SRB). This hydrogen sulphide then either bubbles away as a gas or combines with cations and is retained in the wetland as sulphides (including iron monosulphide, FeS), polysulphides (including pyrite, FeS_2) and elemental sulphur (Hedin *et al.* 1988). Since FeS is more readily oxidised than FeS_2, promoting conditions in which FeS_2 formation occurs will be more efficacious in immobilising metals within the wetland substrate. However, analytical work carried out at the pilot-scale wetland at Big Five Tunnel in Colorado revealed that very little diagenesis of FeS to FeS_2 had occurred in the first season of operation (Wildeman *et al.* 1991). The presence of FeS_2 in the form of autochthonous pyrite framboids was demonstrated at a semi-natural wetland receiving metal-contaminated minewater in Cornwall, UK (Brown 1996), and experimental work at the former Bell Copper Mine in Canada showed that copper was retained as sulphides, particularly when low strength minewater was applied (Sobolewski 1996).

The reduction process also produces alkalinity, which counteracts acidity and thus reduces pH.

The fact that SRB facilitate both the precipitation of metals and the production of alkalinity means that both treatment aims are met. Design criteria for constructed wetlands should be such as to optimise conditions for these anaerobic bacteria.

It is also important to ensure re-oxidation of sulphides is prevented; this can be achieved by maintaining several centimetres of water over the substrate and minimising turbulence in the wetland (Fourie *et al.* 1992). Fluctuating water levels are likely to give periodic aerobic conditions in the substrate that will increase the risk of oxidation and mobilisation of the metal sulphides (Hedin *et al.* 1989).

Optimisation of conditions for SRB has been the focus of much research. Of primary importance seems to be a near neutral pH (Barnes *et al.* 1992). According to Hedin *et al.* (1989), the most common SRB, *Desulfovibrio*, are not active at pH values of less than pH 5. Also a plentiful supply of organic matter and sulphate are essential. Such conditions occur naturally in estuarine and marine sediments (Jørgensen and Fenchel 1974, Jørgensen 1978, Parkes *et al.* 1993), though rarely in freshwater wetlands where sulphate is often the limiting factor (McIntyre and Edenborn 1989). However, sulphate is not scarce in contaminated minewater so, by ensuring organic matter is available, sulphate reduction should occur in wetlands receiving mine drainage water (Herlihy and Mills 1985, Herlihy *et al.* 1987). Kalin (2001) summarises the optimal conditions for the anaerobic amelioration of minewater as:

- the presence of an organic sediment with proper microbial flora;
- the presence of anaerobic conditions necessary for sulphate-reducing and iron-reducing bacteria;
- the proper mix of carbon sources for short-term and long-term microbial growth.

The rate of SRB activity is highly variable in the heterogenous environment of wetland substrates and varies seasonally, due primarily to temperature effects (McIntyre and Edenborn 1989, Wildeman *et al.* 1992, Gammons *et al.* 2000, Kalin 2001). However, work carried out in a constructed wetland in Ontario found that SRB populations were greatest during the winter months when the water temperature was around 1°C (Fortin *et al.* 2000).

SRB are an extremely diverse and poorly understood group of bacteria (Singleton 1993). Research is constantly challenging previously held views of their taxonomy, physiology and ecology (Smith 1993). For example, it has been demonstrated that sulphate reduction can occur in the presence of oxygen and is not a strictly anaerobic process as was previously thought (Canfield and Des Marais 1991, Fründ and Cohen 1992). Other work suggests that SRB can directly reduce ferric to ferrous iron, whereas it was previously thought that this occurred indirectly as a result of H_2S generation (Coleman *et al.* 1993).

The above discussion of oxidation and reduction reactions highlights the two distinctive sets of processes occurring in wetlands. This has led to the division of constructed wetland types into aerobic and anaerobic. In simple terms, aerobic wetlands are recommended for treatment of net alkaline minewaters and anaerobic or compost wetlands are recommended for treatment of net acidic minewaters (Hedin *et al.* 1994). While this is a useful rule-of-thumb for selection and design purposes, the *caveats* mentioned in the introduction to this section on constructed wetlands should be borne in mind, i.e. the inherent loss of ecological complexity and reduction in sustainability of treatment invoked by over-simplification of treatment wetlands. McGuinness *et al.* (1997) caution that the attempt to divide wetlands into anaerobic and aerobic types has a disruptive effect on the microbial ecology and carbon cycle, which may potentially affect the efficacy of the wetland to treat minewater. Also, the natural development and complexity of ecological systems mean that it is not possible to design and maintain a wholly aerobic or anaerobic wetland. There will be micro-environments representing a whole spectrum of redox conditions in any wetland, whether constructed or natural (e.g. Eger 1994).

4.2.1.6 Longevity

Since constructed wetlands have been applied to minewater treatment for less than 20 years it is uncertain how long they can be expected to function effectively. Designers of wetlands in the USA proposed that the longevity of constructed wetlands could be considered to be several decades (Hedin 1996, Williams and Stark 1996), but with no examples as old as this it is impossible to be more definite about this suggestion. Studies of natural and semi-natural wetland areas that receive minewater inflow may help address the question of longevity of treatment facilitated by wetlands. Sobolewski (1999) reviewed monitoring data for natural wetlands at mine sites throughout the world, although with emphasis on Canadian examples. This review concluded that constructed wetlands are a viable option for long-term treatment of minewater and that limitations in using them do not lie with the wetlands themselves, but our knowledge of them. The case was made that as wetlands reach maturation, they acquire characteristics that enhance, rather than diminish treatment efficiency (for example, the formation of a detrital layer and the evolution of diagenetic processes). It was proposed that properly designed and managed constructed wetlands could perform significantly better than their natural counterparts. A comparison of two constructed wetlands and one natural wetland in the southeastern USA revealed that the removal efficiency of metals was greater in the constructed wetlands and that the natural wetland was approaching equilibrium in terms of import and export of metals, although the age of the natural wetland was not reported (Mays and Edwards 2001).

A study of a semi-natural wetland area in a former tin and copper mining district of Cornwall that received minewater contaminated with, principally, copper and zinc, demonstrated that amelioration of the minewater was still occurring after at least 50 years, but that at times the wetland area could act as a source, as well as a sink, of metals (Brown 1996). Beining and Otte (1996) present results from a marsh that receives runoff from a former lead–zinc mining area in County Wicklow and demonstrate that lead, zinc, arsenic and cadmium are still being actively retained in the marsh sediment, after over 100 years of receiving metal-contaminated minewater. They thus suggest that the effective lifetime of treatment wetland can be considered to be in the order of centuries rather than decades.

Study of a 6 year old passive treatment system in Tennessee indicates that the ALD has a remaining 6 years of useful life, the two in-series settling tanks will require emptying after 8 years and 60 years respectively, and the polishing wetland should continue to be effective indefinitely (Schmidt and Stearns 2001).

In contrast to the above findings, a series of experiments carried out on greenhouse mesocosms over a period of 4 years, suggested that treatment efficiency decreased with time (Sexstone *et al.* 1999). However, this might be more an illustration of the difficulties of scaling up from small-scale experiments to full-scale field trials, and the inadvisability of over-simplification, as discussed previously, than an indication of the situation for full-scale wetland systems. Another series of mesocosm experiments using aged wetland sediments also concluded that the metal-retaining capabilities of wetland sediments is essentially dynamic; wetlands that currently act as sinks for metals may act periodically, and perhaps on a long-term basis, as sources of these same metals (Stoddern 2002).

An investigation into the longevity of organic substrates as media for sulphate reducing bacteria and thus as a treatment mechanism for acidic minewater showed that the rate of sulphate reduction declines with time, and that the effective treatment lifetime of such systems may only be about 20% of the lifetime calculated from the total carbon in the substrate (Eger and Wagner 2001). Lack of a sufficiently diverse microflora to facilitate breakdown of organic matter to forms suitable for use by the sulphate-reducing bacteria may be a major factor in this reduced life expectancy (McGinness *et al.* 1997).

4.2.1.7 Sizing

In order for constructed wetlands to be a solution to any occurrence of contaminated minewater, it is desirable to stipulate design criteria. This will facilitate design and construction to suit a specific case without the need to carry out expensive and time-consuming pilot-scale trials in each instance. Design criteria for wetlands to treat domestic wastewaters are well established and allow calculation of the necessary treatment area per head of population (Reed *et al.* 1988, Cooper and Hobson 1989). However, due to the comparative infancy of applying the wetland treatment concept to minewater, available guidelines are somewhat more tentative and less reliable. The formulation of design criteria is not helped by the vast range of water chemistries that can be encountered in mine effluent.

The earliest attempt at design criteria was made by the US Bureau of Mines (USBM) in the mid-1980s (Kleinmann, as reviewed by Wildeman *et al.* 1994). This was very much a ‘rule of thumb’ approach based on the USBM available at that time, and stated that the area requirement was:

$$200\ \text{ft}^2 \text{ wetland for each gallon/minute of flow (US gallons)} \tag{4.6}$$

This guideline was intended for flows of 5 to 10 gallons/minute, water with a pH of 4 or above and Fe and Mn concentrations of less than 50 mg/l and 20 mg/l respectively (all units quoted from original publication). It was assumed that surface processes predominated in the treatment process. However, it was found that this guideline was based primarily on hydrological, and not on removal, considerations.

In 1988, the Tennessee Valley Authority (TVA) proposed design guidelines that took the water chemistry into account (Brodie *et al.* 1988). These guidelines stated that to achieve discharge concentrations of less than 3 mg/l Fe and 2 mg/l Mn, the following areas were required to cope with particular influent Fe and Mn concentrations:

If influent pH < 5.5, area of 2.0 m^2 required per mg of Fe, 7.0 m^2 per mg of Mn.
If influent pH > 5.5, area of 0.75 m^2 required per mg of Fe, 2.0 m^2 per mg of Mn.

Wieder *et al.* (1989) notes that the TVA approach gives rise to area requirements that are between 2 and 74 times greater than those derived from the USBM approach.

Based on a review of data from 48 constructed wetlands in Pennsylvania, Hellier (1989) proposed the following, empirically derived, surface-area-to-flow-rate guideline:

$$20 \text{ m}^2 \text{ of wetland per m}^3\text{/day of flow} \tag{4.7}$$

However, Hellier noted that this was very much a preliminary proposal and that more data were required to propose design criteria with confidence.

In 1990, guidelines were proposed that were based on area adjusted loading and removal factors, which took actual removal efficiency into account (Hedin and Nairn 1990, Hedin *et al.* 1994, Wildeman *et al.* 1994). These guidelines were calculated in terms of grams of Fe removed per square metre per day (GMD) and involved the following calculations:

$$\text{Fe GMD} = \frac{\text{g Fe in/day} - \text{g Fe out/day}}{\text{area}} \tag{4.8}$$

or, assuming flow into wetland equals flow out of wetland:

$$\text{Fe GMD} = \frac{1.44 \times (\text{conc. inflow} - \text{conc. outflow})}{\text{area}} \tag{4.9}$$

where flow is in l/min, concentrations are in mg/l, area is in m^2, 1.44 is the conversion factor to convert minutes to days and milligrams to grams).

On the basis of performance calculations of this nature, carried out on data from three wetlands in Pennsylvania, Hedin and Nairn (1990) proposed a model for the iron removal capacity of constructed wetlands, in which area requirements were calculated on loading-based criteria rather than flow-based criteria. It was found that in instances where the flow was greater than 50 l/min and the Fe concentration was greater than 50 mg/l, the loading-based method stipulated considerably larger wetlands than the flow-based approach.

The currently widely-accepted design and sizing guidelines for constructed wetlands to treat coal mine drainage are presented by Hedin *et al.* (1994) in a publication written for the US Bureau of Mines. These recommendations base sizing on the desired area adjusted removal, $g/m^{-2}.day^{-1}$ (equivalent to GMD introduced above) and are presented in table 4.1.

However, despite the current acceptance of the sizing criteria proposed by Hedin *et al.* 1994, there is an increasing awareness that this might not be the most appropriate method of determining the required wetland size, and that a method that considers the kinetics of removal processes may be more suitable (Tarutis *et al.* 1999). This is considered further below in the section on recent developments in constructed wetland design and implementation.

All the above guidelines are derived from results from surface flow wetlands where oxidation is the dominant process.

Table 4.1. Recommended sizing for passive treatment systems (Hedin *et al.* 1994)

	AML criteria[a] $g/m^{-2}.day^{-1}$		Compliance criteria[b] $g/m^{-2}.day^{-1}$	
	Alkaline	Acid	Alkaline	Acid
Fe	20	N/A[c]	10	N/A
Mn	1.0	N/A	0.5	N/A
Acidity	N/A	7	N/A	3.5

[a] AML criteria: Abandoned Mine Lands criteria, intended for remediation schemes where strict compliance is not a necessity.
[b] Compliance criteria: for remediation where US federal effluent limitations for coal mine drainage must be met (max. 6.0 mg/l Fe and 4.0 mg/l Mn on any one day, with a 30 consecutive days max. value of 3.0 mg/l Fe and 2.0 mg/l Mn).
[c] N/A: not applicable.

In 1991, Perry and Kleinmann proposed a sizing criterion in terms of net acidity of the influent, thereby incorporating the effect of pH, ferrous iron, ferric iron and aluminium into a single, easily measurable parameter. This guideline stated that the area requirement was:

$$200\text{–}500\ \text{m}^2 \text{ per kg/day net acidity} \tag{4.10}$$

and assumed that sulphate reduction was the dominant process.

However, it was found that when applied to treatment of metal mine drainage, which typically has a more complex suite of contaminants than coal mine drainage, none of the above criteria were satisfactory (Wildeman *et al.* 1991). This led to formulation of sizing guidelines based on reaction rates and volume, rather than the area of the wetland.

The basis of this alternative approach to formulating sizing guidelines is the maintenance of suitable conditions in the substrate to encourage SRB activity. This essentially involves not overwhelming the micro-environment established by the anaerobic bacteria by overloading the system with acidic and oxygenated minewater. The limiting reagent concept is used to determine how much water can be treated, instead of using loading criteria (Wildeman *et al.* 1991). The principle of the limiting reagent concept is to ensure that sulphide, produced by SRB metabolism, is always in excess as compared to the metal contaminants flowing through the system. Studies by the USBM wetlands group (Hedin *et al.* 1989, McIntyre and Edenborn 1989) suggest that a reasonable figure for sulphide generation by a consortium of SRB in an organic substrate is:

300 nanomole sulphide/cm^3.day (equivalent to 0.3 moles/m^{-3}.day^{-1}) (4.11)

This value, the volume of the wetland cell, and the concentration of metals in the mine drainage, are used to set the flow rate through the wetland cell. Experience has shown (Wildeman *et al.* 1992) that there is also a minimum surface area for a given flow to avoid excessive stress, in terms of low pH, to the SRB at the inflow end of the wetland. This minimum area is calculated as:

800 ft^2/gallon/minute (units as original publication) (4.12)

The above values are used for sizing of anaerobic wetland cells.

As a result of consideration of available design criteria, various attempts have been made to formalise the design and decision-making process by constructing flow charts for the selection of a passive mine drainage treatment system or combination of systems. Three examples are presented by Hedin *et al.* (1994), Wildeman *et al.* (1994), McCleary and Kepler (1994), and are reproduced in figures 4.1, 4.2 and 4.3. These flow charts include the provision of anoxic limestone drains (ALD), where applicable.

The above discussion has followed the development of design criteria for constructed wetlands, and has shown how the size of wetland cells can be calculated on the basis of flow rate, water chemistry and reaction mechanisms. It has also been shown that different methods of calculations are used for sizing aerobic and anaerobic cells. However, Brodie (1990) cautioned that due to the interrelationships of many factors of a wetland area (hydrology, alkalinity of minewater discharge, depth, area hydraulics, substrate, vegetative and microbial species and numbers), it may be imprudent to suggest treatment area guidelines until considerably more knowledge has been gained.

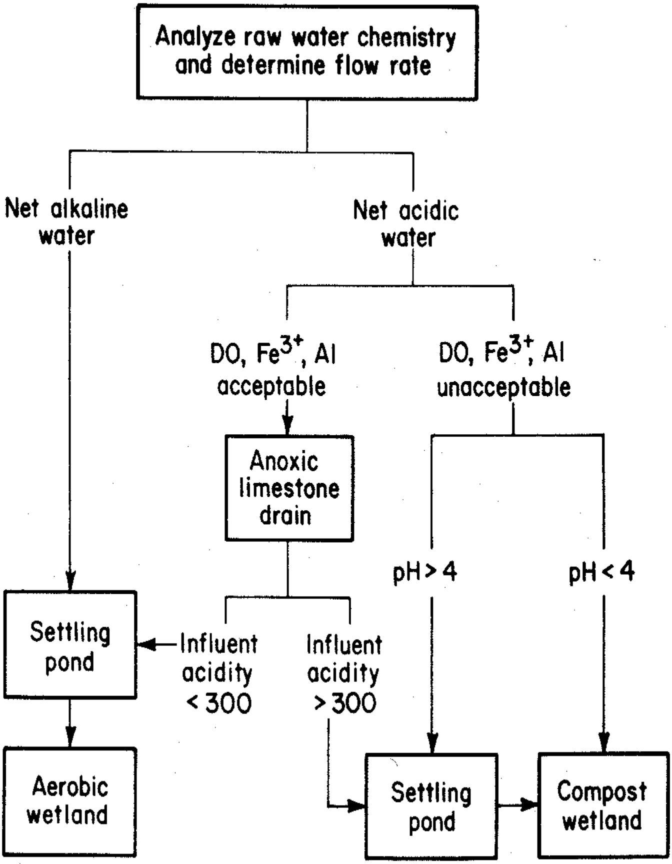

Figure 4.1. Flow chart showing chemical determinations necessary for the design of passive treatment systems (from Hedin *et al.* 1994).

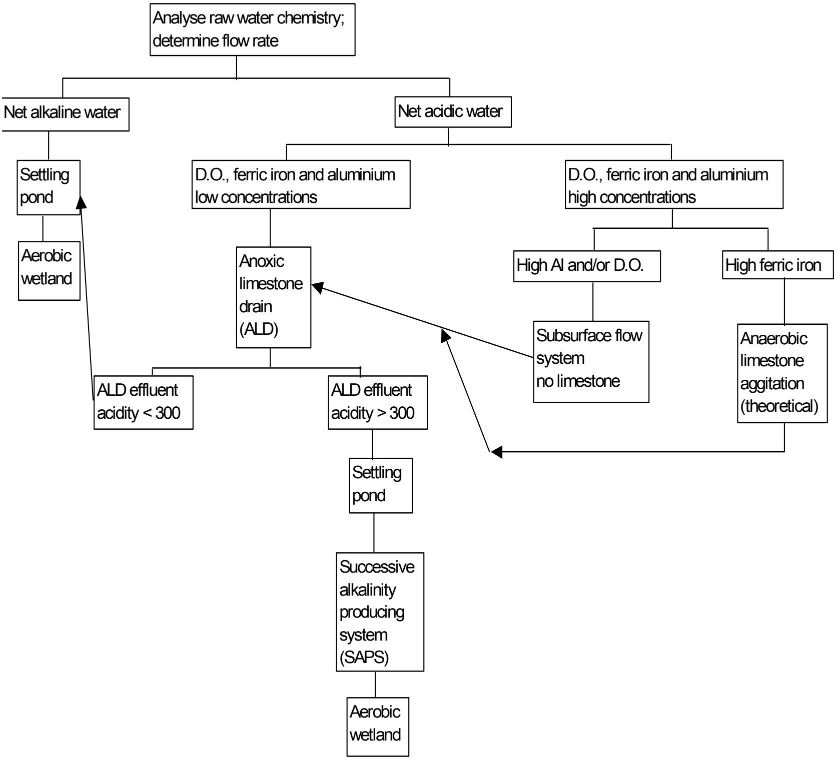

Figure 4.2. Flow chart showing decision making process for design of passive treatment systems (from McCleary and Kepler 1994).

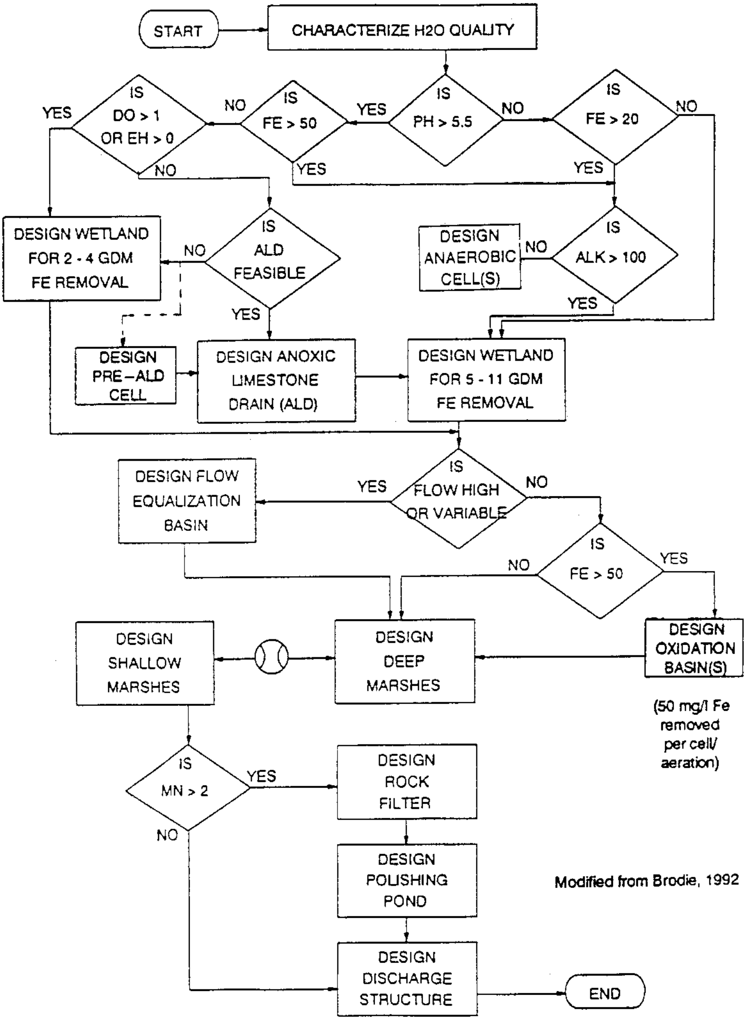

Figure 4.3. Flow chart showing decision making process for design of passive treatment systems (from Wildeman *et al.* 1994).

4.2.2 Other passive biological systems

Constructed wetlands are the most widely used passive biological systems, but there are other ways in which the action of living organisms can be utilised passively for minewater treatment.

Algae show significant levels of bioaccumulation and this has lead to research and development of algal-based passive treatment systems for minewater. The tendency of algae to accumulate metals was noticed by workers in the 1980s (Foster 1982, Kepler 1988), although at that time the algal biomass in most wetlands was considered to be very limited, so that its contribution to metal removal would rarely reach significant levels (Kleinmann 1989). This view, however, has been somewhat altered by subsequent work. Knight (1992) considered that algae could be encouraged within a wetland by incorporating areas of open and deeper water. This has been shown to be successful at a base metal mine in Northern Australia, where channels supporting copious growth of algae are playing a substantial role in water quality improvement by removing manganese and zinc (Jones *et al.* 1995). The final stage of the pilot passive treatment plant at Wheal Jane in Cornwall comprises rock filters that were specifically designed to encourage algal growth to remove manganese from the minewater (see case study 5.35). Kalin (1989) goes so far as to say that algae are an essential part of a self-maintaining biological treatment system, playing a vital role in metal accumulation and nutrient cycling.

Microbial mats, comprising mixed populations of heterotrophic bacteria, blue-green algae and filamentous green algae, have been investigated as a means of removing metals, principally iron, manganese and zinc, from minewater (Duggan *et al.* 1992, Phillips *et al.* 1994, Bender *et al.* 1994). The intricate juxtapositioning of oxic and anoxic microenvironments is thought to be an important feature of microbial mats. The mechanisms involved in metal immobilisation by microbial mats are thought to include surface binding to the mat or to mat exudates trapped within the porous media on which the mat is established, precipitation of the metals with anions present in the oxic/anoxic zones, mat mediation of the water conditions in favour of metal-oxide precipitation and active transport of the metals into the cell (Bender *et al.* 1994). Bender and Phillips (1995) devised a proprietary method whereby microbial mats, comprising consortia of specially cultured blue-green algae and bacteria, have been successfully used to remove cadmium, cobalt, chromium, copper, iron, manganese, lead and uranium from contaminated water. Various means of using microbial mats were investigated by these workers, including free-floating mats, mats immobilised on glass wool layered in baffle tanks, mats immobilised on floating glass-wool balls and mats established on the surface of acid mine drainage in field ponds by application of silage. The pilot-scale work carried out by Bender and Phillips (1995) suggested that microbial mats are a versatile remediation tool that can be adapted to treat many different combinations of contaminants.

The potential of using microbial mats for minewater treatment has recently been investigated at a Rio Tinto owned mine in Indonesia (Bernoth *et al.* 2000). There is an active lime treatment followed by polishing ponds at this site, but manganese concentrations are not sufficiently lowered by this process – discharge from current treatment has from 5 to 18 mg/l of Mn. Laboratory and pilot-scale tests were carried out using locally sourced bacteria and blue-green algae in floating mat and attached configurations. Average removal rates of 91% and 84% for floating and attached respectively were achieved at a flow rate of 6 l/min, and 88% and 64%, for floating and attached, at 10 l/min. It was considered, as a result of these trials, that use of this passive biological method would be suitable for full-scale remediation of Mn at this site. Despite the fact that the floating mat configuration gave better removal rates, it was considered that an attached configuration would be easier to engineer.

Various other recent work on passive biological systems can be found in the literature. For example, Ibeanusi and Wilde (1998) carried out batch tests in tanks using microbial cultures of specific strains of *Bacillus* and *Pseudomonas* bacteria to treat coal pile run-off.

Two proprietary biological treatment processes are reported by Skousen *et al.* (1996), as follows.

The Lambda Bio-Carb process is a bioremediation system utilizing site-indigenous, mixed micro-organism cultures selected for maximum effectiveness (Davison 1993). On a field site in Pennsylvania using this bioremediation process, Fe in acid minewater was decreased from 18 mg/l to < 1 mg/l, Mn declined from 7 mg/l to 2 mg/l, and pH increased from around 6.0 to 8.0.

The Pyrolusite process uses selected groups of micro-organisms growing on limestone to oxidize Fe and Mn into their insoluble metal oxides (Vail and Riley 1997). On a field site in Pennsylvania using a limestone bed inoculated with micro-organisms, Fe was decreased from 25 mg/l to < 1 mg/l, Mn went from about 25 mg/l to < 1 mg/l, while pH and alkalinity in the effluent were increased.

Sulphate-reducing bacteria can be used in passive bioreactors to treat minewater (see also the discussion of SRB in active bioreactors – the distinction between active and passive is by no means clear cut, since a certain degree of upkeep and maintenance will be required even for the so-called ‘passive’ reactors). Duc *et al.* (1998) carried out experimental work on substrate evaluation for passive anaerobic systems, and found that fresh and aged cow manure, individually or mixed with cellulosic waste, is effective at removing ferrous and ferric iron, zinc and manganese from acidic solutions (pH = 2.6). Despite the low pH of the influent, it seems that suitable conditions were established in the manure-based substrate for the successful establishment and metabolism of sulphate-reducing bacteria, with the pH of the effluent being raised to 7.5 or above. By contrast, use of cellulosic waste (pine wood chips, chestnut wood chips or hay) alone did not raise the pH or remove metals nearly as effectively, presumably because there was no initial inoculation of SRB, which would have been provided by the manure. In another investigation of the use of SRB in passive bioreactors, weather pine sawdust was used as the cellulosic substrate and mud from a pond near the mine site as the source of SRB (Thompson *et al.* 2000). Pinto *et al.* (2001) report on investigations into the suitability of locally available organic substrates for use in an anaerobic treatment system for minewater from a gold mine in Brazil. Substrate performance was evaluated by monitoring arsenic and sulphate removal, and pH/Eh variation. Mixtures containing iron filings (zero valent iron, see discussion of reactive barriers), alfalfa, manure and hay gave the best results. These workers consider that it is important to have sulphate reduction occurring in association with the reductive action of the zero valent iron, so that the iron is immobilised in the substrate as sulphide, rather than mobilised as ferrous iron. Although the precise mechanisms of arsenic removal in this instance are not fully understood, it seems that the removal of arsenic is more effective in an organic substrate containing zero valent iron than in an organic substrate alone.

Zaluski *et al.* (2000) studied three SRB passive field-bioreactors at an abandoned mine site at Butte, Montana, intended to mitigate acid minewater emanating from a waste rock pile. Each bioreactor contained separate chambers of an organic source, limestone and cobbles, with different configurations. They noticed strong temperature effects on SRB activity (two reactors were below ground and one was above), and noted that in the short timespan of the monitoring (8 months) adsorption to the organic matter was a mechanism of primary importance although there was also evidence of SRB activity.

Recent work has attempted to establish sizing criteria for gravel beds treating acid mine drainage by sulphate reduction, by investigating the rate of SRB activity in such beds (Lyew and Sheppard 1999) and establishing values for ratios incorporating four system parameters: the void volume of the bed, the surface area of the bed, the total surface area of gravel in the bed and the volume of minewater applied to the bed. Although only carried out at a laboratory scale, the authors suggest that the results could be scaled up for use in sizing gravel bed systems for treating minewater in sequential batch reactors, in holding ponds or lagoons or in constructed wetlands.

Reactive barriers often have a biological component, as for example, the reactive barrier recently constructed at Renishaw Park, South Yorkshire (case study 5.29).

4.2.3 Recent developments in constructed wetlands for minewater treatment

Kinetics

Section 4.2.1.7, on sizing, introduced the concept of area-adjusted removal. This is one of three measures of treatment performance that have routinely been applied to assess the effectiveness of treatment of minewater by constructed wetlands (Tarutis *et al.* 1999). The other two measures are treatment efficiency and first-order removal.

Treatment efficiency is a relative measure of performance and is calculated using the following equation:

$$\text{Treatment efficiency (\%)} = \frac{C_{in} - C_{out} \times 100}{C_{in}} \tag{4.13}$$

where C_{in} is the influent concentration of the contaminant of interest and C_{out} is the effluent concentration. Although this gives a useful indication of how well the wetland is performing, it cannot be used alone to provide wetland sizing criteria.

Area-adjusted removal was proposed as an absolute measure of performance, thereby providing a means of calculating the required size of wetland. It is calculated using the equation 4.14 (which is the same as the equations given above for GMD):

$$\text{Area-adjusted removal (g/m}^{-2}\text{.day}^{-1}\text{)} = \frac{C_{in} - C_{out} \times \text{flow}}{\text{area}} \tag{4.14}$$

Rearrangement of the equation gives the equation for determining the size of wetland required for a particular inflow and outflow concentration and flow:

$$\text{Area} = \frac{(C_{in} - C_{out)} \times \text{flow}}{k_0} \tag{4.15}$$

where k_0 is the rate constant for zero-order kinetics (Tarutis *et al.* 1999). Values for k_0 are suggested by Hedin *et al.* (1994) and are shown in table 4.1 (k_0 is the same as area-adjusted removal). Zero-order kinetics assumes that the rate of removal of the contaminant of interest (e.g. iron) is concentration-independent. In other words, use of area-adjusted removal as the criterion by which wetland size is determined does not distinguish between a high flow/low concentration and a low flow/high concentration input to the wetland. Hedin *et al.* 1994, while recommending the use of area-adjusted removal for sizing wetlands, note that iron removal rates may be a function of iron concentration, and that as concentrations get lower, the size of the system necessary to remove a unit of iron gets larger (that is, they note the possibility that zero-order kinetics might not be the most appropriate criterion to use).

Experience in the domestic wastewater treatment industry has shown that first-order kinetics is a more appropriate measure of performance for assessing removal rates of suspended solids, biochemical oxygen demand, nutrients and pathogens (Kadlec and Knight 1995), and that this can also be used to calculate the required size of treatment wetlands for these contaminants. First-order kinetics assumes removal is concentration-dependent and *does* distinguish between high flow/low concentration and a low flow/high concentration input to the wetland.

The oxidation of ferrous iron has been shown to follow first-order kinetics at constant pH and oxygen levels, and iron precipitation and sedimentation rates have also been shown to be first-order (Stumm and Morgan 1996). The rate of iron sulphide formation has been shown to follow first-order kinetics at constant pH and dissolved sulphide concentrations (Rickard 1974, Pyzik and Sommer 1981). Thus, it seems likely that removal of iron, and perhaps other contaminants as well, in a constructed wetland occurs by a first-order process, whether oxidative or reductive processes are occurring. It would therefore seem logical to use consideration of first-order kinetics for sizing such wetlands.

The equation for determining wetland size according to first-order kinetics is as follows (Tarutis *et al.* 1999):

$$\text{Area} = \frac{\text{flow}}{k_1 \ln(C_{in} / C_{out})} \tag{4.16}$$

where k_1 is the first-order rate constant and has units of $m.day^{-1}$, assuming concentration is expressed as g/m^{-3}, flow as $m^3.day^{-1}$ and area as m^2.

In order to apply first-order kinetics to wetland sizing, values for k_1 are needed. Tarutis *et al.* (1999) report on monitoring of 35 natural (volunteer) wetlands receiving coal mine drainage in western Pennsylvania from which values for k_1 for Fe and Mn removal were derived (values were also derived for removal efficiency and k_0). These data are reproduced in table 4.2.

Table 4.2 shows that values for k_1 determined in the study carried out by Tarutis *et al.* (1999) are variable, and they note that the values were obtained from consideration of minewaters with highly variable chemistry and from very diverse wetland types. These workers highlight the need for more work to compile values for k_1 and recommend that in the meantime the median values should be used as these give a more conservative estimate of wetland size.

Table 4.2. Descriptive statistics for treatment efficiency, area-adjusted removal and first-order removal of iron and manganese for 35 natural wetlands (Tarutis *et al*. 1999)

Performance measure	Mean ± std. deviation	Median	Range
Fe efficiency (%)	64 ± 28	72	-32 to 99
Fe area-adjusted removal, k_0 ($g/m^{-2}.day^{-1}$)	6.5 ± 8.8	3.5	-7.7 to 45
Fe first-order removal, k_1 ($m.day^{-1}$)	0.29 ± 0.34	0.18	-0.061 to 1.8
Mn efficiency (%)	16 ± 31	6.8	-130 to 98
Mn area-adjusted removal, k_0 ($g/m^{-2}.day^{-1}$)	0.73 ± 1.8	0.24	-1.2 to 13
Mn first-order removal, k_1 ($m.day^{-1}$)	0.057 ± 0.11	0.011	-0.043 to 0.73

The selection of appropriate criteria to determine the size of a constructed wetland needed to treat any particular minewater problem is thus a matter of ongoing research. Where sufficient data have been obtained for the sites considered in this project, values for k_1 have been calculated, but it should be noted that much more rigorous and detailed research is required to define appropriate values for k_1 with confidence.

The main practical implication of the zero-order/first-order debate is the fact that application of zero-order kinetics to design of treatment wetlands for Fe removal tends to lead to *undersized* wetlands in instances where, in terms of input to the wetland, flows are high and concentrations are low.

Successive Alkalinity Producing Systems (SAPS)

The use of anoxic limestone drains (ALDs) to add alkalinity to minewater by passive means has been discussed previously. However, this discussion also highlighted the limitations of ALDs; the presence of ferric iron, dissolved oxygen or aluminium in the minewater has serious implications for the operation of the ALD and can cause it to cease functioning effectively. Kepler and McCleary (1994) addressed this problem and suggested a wetland configuration that enables passive alkalinity addition without the constraints on water chemistry imposed by ALDs.

Successive Alkalinity Producing Systems (SAPS), or Reducing and Alkalinity Producing Systems (RAPS), as they have recently been termed (Watzlaf *et al.* 2000), effectively combine attributes of compost (anaerobic) wetlands and ALDs in a vertical flow configuration. This means that the land area requirement is reduced (as compared to an ALD/compost wetland in succession with horizontal flow), and that the armouring and blocking problems encountered in ALDs may be overcome.

Figure 4.4 shows the configuration of a typical SAPS system, and its mode of operation may be described as follows. The ponded water provides hydraulic head to 'drive' the system, in other words to make the water flow vertically down through the underlying substrate. The organic compost layer strips oxygen out of the water, due to the oxygen demand associated with bacterial decomposition of organic matter. The anoxic conditions that are thus created will favour establishment of sulphate-reducing bacteria, whose metabolism produces alkalinity. The alkalinity produced by SRB activity within the organic layer accounts for an average of 28% of alkalinity produced by the whole SAPS system (Watzlaf *et al.* 2000).

The lowest layer of substrate comprises limestone, which will add further alkalinity to the minewater due to dissolution, and should not become armoured with ferric precipitates since the minewater has passed through the reducing environment of the compost layer. Kepler and McCleary (1994) recommend that as many SAPS as necessary to achieve the desired level of treatment are constructed in succession, with surface-flow, compost, vegetated wetland areas in between to aid in suspended solids removal and to provide a sustainable supply of oxidisable organic matter to the SAPS units. Watzlaf *et al.* (2000) caution that SAPS should not be viewed as stand-alone treatment systems, but rather as a component of an integrated system. The use of a pond/wetland before the SAPS to precipitate iron and other settleable solids, and a pond/wetland after the SAPS for iron oxidation, precipitation and settling is advised.

SAPS have been successfully used at many sites in the USA to treat acidic minewater, often as part of an integrated system (e.g. Skousen *et al.* 1999, Watzlaf *et al.* 2000, Brookens *et al.* 2000, Rose *et al.* 2001). The amount of acid neutralisation potential afforded by existing SAPS systems ranges from 35 to over 400 mg/l $CaCO_3$ (Watzlaf *et al.* 2000). Despite the successful SAPS that are reported in the literature, a note of caution is sounded by Brookens *et al.* (2000). These authors report on a SAPS in Maryland which is successfully reducing metal concentrations and increasing pH and alkalinity, but the effluent from the treatment system has shown varying degrees of acute and chronic toxicity in standard toxicity tests. The causes of this toxicity have not been definitely identified, although aluminium is a possible culprit.

Modifications of the standard SAPS as described have been documented, for example the use of hydrated fly ash as an additional source of alkalinity to those provided by the compost and limestone (Nairn *et al.* 1999, Nairn *et al.* 2000) and the use of a mixed compost/limestone layer, rather than two distinct layers (Watzlaf 1997, Thomas and Romanek 2001). SAPS have also been successfully used in conjunction with ALDs to renovate a failed constructed wetland in Kentucky, USA

(Barton and Karathanasis 1999). An adaptation of SAPS has been proposed for use with high-aluminium minewater: called an ‘Aluminator’, it is essentially a SAPS with a shut-off valve on the effluent pipe and sufficient fall below the effluent pipe to allow occasional flushing of the system to remove aluminium hydroxides from the limestone gravel (Kepler and McCleary 1997).

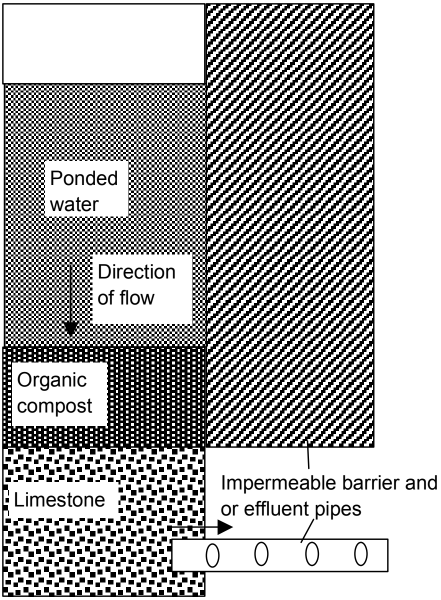

Figure 4.4. Typical cross-sectional view of a successive alkalinity producing system (SAPS) treatment component (Kepler and McCleary 1994).

An adaptation of the conventional SAPS/RAPS has been trialled at full-scale in Alabama, USA for treatment of runoff from a coal pile storage area. This system incorporates the ability to partially recirculate treated minewater through the RAPS, and has shown potential to enhance the basic RAPS design by moderating the pH of the influent minewater and reducing the contaminant loading prior to the RAPS component (Garrett *et al.* 2001). This reduction in loading to the RAPS could theoretically increase the effective lifetime of the RAPS from an initial design life of about 14 years, to over 50 years. There are also indications of enhanced manganese removal in this treatment system, compared to a conventional RAPS, although work is currently in hand to investigate this further.

Attempts to define design criteria for SAPS have been made using both field and laboratory studies (Jage *et al.* 2000). These workers concluded that factors controlling generation of alkalinity include residence time, temperature and influent water chemistry. There are currently, however, no definitive guidelines for designing and sizing SAPS systems in terms of these factors. Research carried out to determine the influence of system design and influent minewater chemistry on net alkalinity generation by SAPS revealed that residence time in the limestone layer shows a strong, positive logarithmic relationship with alkalinity generation, and that influent Fe and non-Mn acidity concentrations both show strong positive linear relationships with alkalinity generation (Jage *et al*. 2001). These results indicate that alkalinity generation in the limestone occurs most rapidly in the first few hours of minewater–limestone contact, with additional residence time yielding diminishing gains in treatment, and that dissolution of limestone is greater under more acidic conditions.

There is increasing interest in the use of SAPS in the UK and Europe. Bench-scale tests were carried out at an abandoned mine in Slovakia that investigated various configurations of organic matter and limestone (Gusek *et al.* 2000). The most successful configuration consisted of an upper layer containing 50% limestone, 30% old sawdust, 10% mouldy hay and 10% manure, and a lower layer comprising 20% limestone, 60% old sawdust, 10% mouldy hay and 10% manure. Sulphate reduction rates in excess of 0.3 moles/m^{-3}.day^{-1} (see sizing criteria above) were achieved and the researchers concluded that sulphate reduction, rather than limestone dissolution, was the most important alkalinity generating mechanism.

A pilot-scale integrated passive treatment system has been constructed at a Spanish mine site. This system comprises a SAPS system followed by an ALD, an oxidation cascade, a sedimentation pond and an anaerobic substrate wetland (Ordonez *et al.* 2000). Preliminary results from this system suggest that it would successfully treat the minewater in question for all contaminants except sulphate; work is continuing to refine and improve the system and to progress to full scale.

In the UK there are some recently constructed SAPS and SAPS-type systems, for example at Pelenna in South Wales, Bowden Close in County Durham and Foss, Aberfeldy in Scotland (see case studies in Chapter 5).

An important design consideration for SAPS-type systems is that there is sufficient hydraulic head to ‘drive’ the system. If the site contours do not allow a sufficient depth of water on top of the substrate layers, a SAPS system may not be a viable option; for example, at Quaking Houses in County Durham, a SAPS system would have been desirable but was not possible on the available site due to lack of head, so a compost wetland was constructed instead (see case study 5.28). Lack of hydraulic head could be one of the reasons for the current difficulties being experienced at the Garth Tonmawr site of the Pelenna project (see case study 5.26).

The importance of using SAPS as part of an integrated system rather than as a stand-alone system is illustrated at the Whitworth A site of the Pelenna project, where ochre build-up on the surface of the SAPS is compromising the efficiency of the system and may substantially reduce its effective lifetime (Dey and Williams 2000, Rees *et al.* 2001, see case study 5.26). The importance of incorporating settling ponds prior to SAPS to remove some of the iron has been noted at various sites (Garrett *et al.* 2001, Rose *et al.* 2001).

Modelling

Interest has been increasing in recent years in the development of computer models of constructed wetland treatment systems, which could then be used as predictive and design tools for future constructed wetlands. The task of modelling is made difficult by the inherent complexity of natural systems, and the fact that the precise mechanisms and rates of remediation processes are not always known. In effect, computer models and simulations are an extension and development of the sizing criteria considerations discussed above, and with ongoing research and refinement such models may form the basis of future design work, rather than the calculations based on contaminant removal currently used.

Models for wetland ecosystem function are well developed (Mitsch *et al.* 1988, Duever 1988), as are models for domestic wastewater treatment wetlands (Lantzke *et al.* 1999, Wynn and Liehr 2001). However, models for wetland treatment of minewater are less well-developed, perhaps due to the fact that there has been a tendency to treat constructed wetlands as a ‘black box’ solution to the minewater problem, with investigation into actual mechanisms taking second place to routine monitoring of treatment efficiency.

Models that have been developed often concentrate on just a few parameters or wetland processes, rather than try to incorporate the whole spectrum of ecosystem functions. For example, Kirby *et al.* (1999) focused on iron oxidation and attempted to quantify the relative contributions of biotic and abiotic factors in this process. They used seven factors in their model (inflow ferrous iron concentration, pH, temperature, *Thiobacillus ferrooxidans* and oxygen concentration, flow rate and pond volume). They found that the biotic contribution to iron oxidation was strongly pH dependent, as indeed are iron oxidation rates themselves. The predicted values of iron concentrations and oxidation rates agreed well with field-measured values of these two parameters. From pH 2.8 to approximately pH 5, ferrous iron oxidation rates are negatively correlated with pH (i.e. slower oxidation at higher pH) and catalysed by *T. ferrooxidans*. From pH 5 to 6.4, ferrous iron oxidation appears to be primarily abiotic and is positively correlated with pH. Above pH 6.4 oxidation appears to be independent of pH, with little or no increase in oxidation rate due to increase in pH. These authors stress the importance of a thorough understanding of iron oxidation processes in order to achieve optimum iron removal in a passive treatment system. For example they note that in instances of minewater with very low pH (pH 2 or lower) it might be better to make use of the very rapid oxidation of ferrous iron that occurs at this pH before raising the pH to precipitate out metal hydroxides as would conventionally be done.

The rate of iron oxidation in an alkaline minewater was modelled by Watzlaf *et al.* (2001). These workers concluded that aeration of alkaline minewater is of primary importance, and that the consequent pH increase due to loss of CO_2 was probably more important in accelerating iron oxidation than an increase in dissolved oxygen in the minewater.

Other examples of models with a specific focus have been developed by Wood and Shelley (1999), who investigated the bioavailability of metals in constructed wetland sediments receiving stormwater runoff using a system dynamics approach; Lung and Light (1996) who simulated the fate and transport of copper introduced into a wetland ecosystem; and Flannagan *et al.* (1994) who developed a model to predict metal retention in a constructed wetland receiving coal mine drainage using primarily physical and chemical, rather than biological, functions. This model was used in the design of a constructed wetland, which was subsequently built, thus allowing the model to be validated using data for the first year of operation of the wetland (Mitsch and Wise 1998). The data for actual iron retention gave a reasonable match with predicted iron retention, the over-estimation of the model being attributed to different hydrological conditions in the actual wetland as compared to the model. These authors conclude that computer models can play a large role in the design of future wetland systems, but note that background hydrology and water quality data are essential for this to be successful.

Hellier (2000) developed an integrated design model for passive treatment systems for minewater that included vertical flow reactors, aerobic and anaerobic limestone channels and horizontal flow aerobic and compost reactors. However, this model is based entirely on chemical processes and thus does not take into account the contribution of biotic factors to treatment, which can be considerable.

An outcome of the River Pelenna Restoration Scheme (Edwards *et al.* 1997) was a computer-aided design package to assist in the design and operation of passive treatment facilities for minewater, developed at the University of Newcastle (James *et al.* 1997). This package has four interconnected modules, with the following functions:

- minewater – provides background notes on minewater chemistry and sampling statistics, plus assistance in characterising minewaters and setting up sampling programmes;
- river – provides a dynamic model of water quality for simulating the impact of minewaters on streams;

- treatment – provides an interactive computer-aided design package for design and costing treatment facilities;
- management – provides a basic package for the interpretation of operating data.

The treatment module is currently limited to passive treatment options and offers a choice of anoxic limestone drains, aerobic wetlands, anaerobic wetlands, successive alkalinity producing systems and sedimentation tanks. Standard design criteria as proposed by Kepler and McCleary (1994) for SAPS, and Hedin *et al.* (1994) for aerobic/anaerobic wetlands and ALDs are used, although these can be over-ridden if required.

The current LINK project at Wheal Jane mine in Cornwall is attempting to identify and quantify geochemical and microbiological processes operative in the amelioration of acidic, metal-contaminated minewater, with the aim of formulating a computer model that will incorporate all the biotic and abiotic processes. The model currently incorporates 22 variables and 23 mass balance equations, giving a total of 506 equations to run the model. The work on this model is continuing and it will be refined and validated during the course of the project.

4.3 PASSIVE TREATMENT – SUMMARY

Passive treatment is a highly desirable strategy, particularly for minewater discharges from abandoned mines since, by definition, the ongoing operating and maintenance costs should be minimal. The ideal of passive treatment is, however, difficult to achieve, and it is very unlikely that a totally maintenance-free, walk-away treatment option can ever be attained.

Passive treatment may be chemical or biological in essence. Chemical passive treatment involves addition of material to neutralise acidity and precipitate out metals without the necessity of regular manual or automated intervention to achieve this. Examples of passive chemical treatment systems are anoxic limestone drains (ALDs), open limestone channels and some types of permeable reactive barriers. There are various problems to overcome, such as blocking of flow paths through ALDs and coating of limestone in open channels, and it is essential to use these treatment options on a strictly site-specific basis, as some water chemistries are simply not suitable for this method of treatment. Various innovative approaches are currently being trialled, for example the use of siphons for flushing deposits from ALDs (Vinci and Schmidt 2001) and the use of ALDs to treat zinc-rich minewater (Nuttall and Younger 2000).

Due to the inherently finite nature of chemical passive treatment systems – i.e. treatment longevity will depend upon the amount of material initially placed in the treatment system – biological treatment systems are the only type of systems that have the potential of becoming truly self-sustaining.

Constructed wetlands have been used for minewater treatment for about 20 years, initially in the USA but latterly attracting worldwide attention and interest. It is important to understand the fundamental processes and mechanisms operative in natural wetlands in order to harness these capabilities in constructed minewater treatment wetlands. Microbial processes are of prime importance in both oxidation and reduction reactions. Design concepts for constructed wetlands have evolved to promote aerobic or anaerobic conditions, dependent upon the target metal species. However, natural heterogeneity and complexity will make achievement of wholly aerobic or anaerobic conditions very unlikely and, arguably, also undesirable.

The potential longevity of constructed wetlands for minewater treatment is currently not known, with some studies suggesting sustainable and perhaps even enhanced treatment with time, and others warning of diminishing treatment and perhaps the possibility of the wetlands turning into a source of metals rather than a long-term sink.

Sizing criteria for constructed wetlands have been proposed by various workers over the years. The currently accepted standard was proposed by Hedin *et al.* in 1994 and is still widely used for sizing constructed wetlands for treatment of ferruginous minewaters. There is, however, ongoing debate about the most appropriate sizing criteria to use; a key feature is a proposal that removal kinetics should be considered (Tarutis *et al.* 1999). This debate is likely to continue for the foreseeable future and research efforts, entailing rigorous monitoring and data collection from as many constructed wetlands as possible, need to be focused on this topic. Sizing criteria for anaerobic, compost wetlands are based on the sulphide generation potential of sulphate reducing bacteria.

Other passive biological treatment systems include the use of algae, microbial mats, sulphate-reducing bacteria and various proprietary processes using microbial consortia.

Successive Alkalinity Producing Systems (SAPS), or Reducing and Alkalinity Producing Systems (RAPS), are a comparatively recent development in which vertical flow through compost and limestone layers introduces alkalinity to minewater. There is ongoing development and refinement of SAPS to optimise their performance, for example by using recirculation of minewater (Garrett *et al.* 2001) and the use of a passive siphon device to give pulsed flow through the system (Vinci and Schmidt 2001). Current indications are that SAPS function best as a component of an integrated system, tailored to the particular site and minewater chemistry, rather than as a stand-alone treatment system.

Modelling of minewater systems is a subject of current research and development, with the ultimate aim of an interactive computer-based design package.

5

Existing sites in the UK – Case studies

INTRODUCTION

In order to allow a comparison to be made between various treatment methods currently in use throughout the UK for minewater, all sites where treatment is being carried out have been identified and as much information as is available/accessible has been obtained. Due to the ongoing work of the Coal Authority in addressing minewater issues, there are schemes at various stages of planning, design and construction. The following sites are not included in the case studies: Lindsay Drift, South Wales (under construction during 2001); Six Bells, South Wales (under construction during 2001); Deerplay, Lancashire (under construction during 2001); Ewan Rigg, Cumbria ('low-tech' scheme using existing wet meadow area); Whittle, Northumberland (under construction during 2001); Silkstone, South Yorkshire (to be commissioned during 2001–2).

The following schemes are scheduled for construction during 2002–3: Fordell, Scotland; Morlais, Wales; Bridgewater Canal, England; Cuthill, Scotland; Frances, Scotland and Bates, England.

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Types of sites

The types of sites where minewater issues are encountered fall into a number of categories about which some generalisations can be made, although it should be stressed that each minewater is unique, being a function of the local geology, hydrogeology, geochemistry, meteorology, topography, biology and anthropogenic factors. The following are some general points about the characteristics of minewater from various sources.

Deep and shallow coal minewater

- Water from deep and shallow mines may be pumped or gravity flow.
- Minewater usually emerges at surface due to rebound of the water table on cessation of mine dewatering operations.
- Many mines have interconnected underground workings, so it is not always obvious where an outbreak will occur nor easy to predict timing or quality of the initial outbreak.
- The water quality when the mine was actively working and being dewatered is not necessarily an indication of the rebound minewater quality, due to rebound causing flushing of oxidation products from exposed rock surfaces during initial flooding.
- There is likely to be a ‘first flush’ effect due to the above-mentioned mobilisation of oxidation products, followed by a steady decline to lower contaminant loadings.
- Minewater may be net alkaline or net acidic (the former is more common in South Wales and Scotland).
- There may be stratification within the mine workings as the water table rebounds, so sampling down-shaft may not give a true picture of contaminant loading to expect when surface discharge commences.
- Minewater discharge comprises one or more discrete sources.
- If it is a pumped discharge, there is an ongoing cost implication, but also more control over the input flow that is put into the treatment system.
- If it is a gravity discharge, flow is likely to be more variable due to response to rainfall.
- Minewater originating in shallow workings will probably be less saline than that from deep workings but the iron concentration may be high. There may be a dilution effect from surface water infiltration.
- Deep minewater may have elevated concentrations of ammonia (e.g. the Nottinghamshire coalfield) presumably due to breakdown of proteinaceous materials in anaerobic conditions (presence of ammonia in shallow-source minewater is likely to be associated with materials used for blasting).

- There are not many measures that can be taken to implement source control although some *in situ* measures may be possible in shafts.
- Minewater discharge from abandoned coal mines is likely to be the responsibility of the Coal Authority, since no-one is legally responsible for discharges issuing from mines abandoned prior to January 2000.
- There is a national priority list that is constantly reviewed and updated by the Environment Agency and the Coal Authority.

Metal mines

- Many, but not all, of the features of underground coal mines also apply to metal mines.
- Minewater chemistry will depend on the mineralogy of the deposit and host rock and can be more complex, and potentially toxic, than coal minewater.
- There is no public body, equivalent to the Coal Authority, with a remit to implement remediation strategies for minewater from metal mines.

Spoil heap drainage

- Levels of contamination can be high and prolonged – low pH is typical, and other metals are likely in addition to iron, e.g. Ni, Zn, Cu, Al.
- Spoil heap drainage is likely to be very acidic, even in areas where minewater issuing from underground mines is net alkaline.
- Discharge is likely to be a diffuse source.
- Volumes of minewater will be very much influenced by rainfall, with flushing of contaminant likely in heavy rainfall events, thus giving a decrease in water quality rather than a dilution effect.
- Source control measures can be taken, e.g. capping, lime spreading.
- There is a possibility of carrying out a washing exercise to reclaim coal, although this may cause other problems due to the chemical reagents used in this process.
- Some remedial work may give an initial decline in the quality of the run-off water.
- Sites are likely to be the responsibility of the Local Authority or current landowner, not the Coal Authority.
- In theory, legal responsibility for the discharges can be allocated and therefore redress sought.
- There may be issues associated with the new contaminated land legislation.
- There is no national priority list or nationally agreed strategy to deal with the problem.

Opencast coal

- Working sites will, by law, have a water management policy.
- Restoration of decommissioned sites can be planned to minimize the potential for contaminated minewater generation.
- Opencast mining may involve removal of shallow mine workings and restoration of sites on which spoil heaps are situated, thereby making an important contribution to remediation of historical pollution;
- Alternatively opencast workings may provide a conduit whereby water enters old underground workings thus exacerbating pollution from these old workings.
- Strict planning conditions can be imposed on working sites.
- Restoration of opencast sites can have a positive benefit to the water environment, e.g. restoration of wetland and water features for habitat.

Quarries

- Water quality depends on the mineralogy of the deposit and the host rock.
- There is a lot of exposed rock surface, so any potential for acid generation will be amplified.
- Water management regimes can be implemented in working quarries to minimise any adverse effects on water quality.
- Abandoned quarries may form isolated bodies of water (pit lakes) which can be treated as separate entities and which do not impact the wider water environment.

Assessment criteria

By their very nature, the sites included in this study are diverse, and available data are variable in both quantity and quality. For some sites only the very minimum amount of data, to comply with discharge consent, have been collected, whereas for other sites a regular and thorough monitoring regime has been implemented. In cases where there is no formal discharge consent in place, there may be no actual water quality data at all. For the purposes of this project, to allow comparisons to be made between treatment methods and process designs, it has been necessary to attempt to standardise the data set for each site. Where possible, therefore, the following information has been compiled for each site and presented in case studies 5.1 to 5.38:

- pre-treatment water chemistry/volume and treatment requirements;
- site history and design criteria;
- monitoring data;
- comments on design;
- costs.

If sufficient data have been obtained (i.e. capital and operating costs), a cost benefit analysis has been performed, based on the cost per cubic metre of minewater treated, the cost per kilogram of iron removed and the construction cost per square metre of reed bed (these figures are presented in the relevant sections of Chapter 6 and summarised in Chapter 7). This allows comparison of sites on a ‘value for money’ basis, although it should be borne in mind that every case study has its own set of constraining circumstances and judgement of schemes cannot be made on the basis of cost alone: for example, in situations where there is no alternative to pumping, the costs will be substantially higher than for a gravity flow system, and lack of suitable available land in the immediate vicinity of the discharge can have considerable cost implications.

It was not always possible to obtain figures for the capital and operating costs of treatment schemes; the figures presented are mostly for Coal Authority schemes. The following points should be noted about the quoted figures for Coal Authority schemes:

- Capital costs are for contractors costs only and do not include consultancy fees, site investigations and project management costs.
- Operating costs similarly do not include consultancy/management fees to oversee operations.
- Operating costs do not include pump repairs/replacements. There is a separate budget for this of approximately £200,000 p.a..
- Other ad hoc items are also excluded from the operating costs, e.g. dayworks for improvements and repairs, dealing with vandalism, additional reed planting.

Attention has also be paid, throughout the project, to ‘best design’ specifications, i.e. notes have been made on a site-by-site basis of the design specification of various components of the treatment system, and an assessment made of their suitability, robustness and maintenance requirements. These notes draw on objective engineering considerations as well as more subjective observations made by site maintenance staff and other personnel involved with the treatment schemes on a day-to-day basis. It should be noted that it was not possible to visit some sites, particularly in the north-east of England, due to the outbreak of foot and mouth disease. The information presented on these sites is that which it was possible to obtain without actually visiting the site.

To aid identification, the National Grid Reference (NGR) for the site is given at the beginning of each case study.

5.1 ACOMB, NORTHUMBERLAND

NGR NY926663

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Acomb is net alkaline discharge from a former drift coal mine, with possible underground connections to old metal mines in the north of the area. Iron concentration is in the region 40 to 65 mg/l, and there have been reports of elevated concentrations of lead, copper, nickel and zinc although there are no data available to quantify these claims. Manganese concentration is up to 1 mg/l. The average flow is in the region of 10 l/s. Discharge occurs to the Red Burn, approximately 1000 metres above its confluence with the River Tyne.

The treatment requirement of the Acomb discharge is aeration and settlement to remove the iron from solution.

Site history and design criteria

Two coal seams have been worked at Acomb in the 19th and 20th centuries, with closure of the workings in 1952. The discharge occurs to the east of Acomb Drift via an old pipe and is known to have been present prior to reclamation of the colliery area in 1989.

A feasibility study was carried out during 2000 to identify the most appropriate means of remediating the discharge and to identify a suitable area of land on which to install the treatment scheme. The area of wetland required to treat the discharge was calculated to be between 0.27 and 0.56 hectares, using the sizing criteria recommended by Hedin *et al.* 1994 for net alkaline discharges.

The scheme was designed by IMC Consulting Engineers and construction work commenced on site in August 2001. Minewater is pumped from the top of the drift to an aeration tower. The treatment scheme comprises two irregularly shaped lagoons, of approximately 2.5 m depth and total area of approximately 750 m^2, with marginal planting of aquatic macrophytes (*Phragmites australis, Iris pseudoacorus, Typha angustifolia, Typha latifolia* and *Phalaris arundinacea*). The lagoons operate in parallel and are followed by two reed beds operating in series. The reed beds are planted with *Typha latifolia, Iris pseudoacorus,* and *Scirpus lacustris.* The total area of the reed beds is in the region of 1200 m^2. Landscaping of the site includes woodland, shrub and waterside marginal planting, together with fencing, footpaths and picnic areas.

Target Environmental Quality Standards for the recipient watercourse have been set by the Environment Agency: iron in the receiving watercourse to be less than 1 mg/l and pH between 6 and 9. This translates into a discharge Fe concentration of 2 mg/l when there is in the region of 18 l/s flow in the receiving stream, and 1 mg/l when there is no flow in the stream.

Monitoring data

There are no data for the effectiveness of the treatment scheme as it is currently under construction. Table 5.1.1 presents data for the quality of the raw minewater.

Table 5.1.1. Water quality of Acomb minewater discharge for period August 2000 to July 2001

n = 8	Average value mg/l	Load kg/day
Flow	10.9 l/s	
pH	6.6	
Alkalinity	239	
Sulphate	550	
Ferrous iron	22.1	
Total iron	32.9	30
Manganese	0.6	

Costs

The construction costs at Acomb are in the region of £350,000.

5.2 ALLANTON, SCOTLAND

NGR NS853575

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Allanton is net acidic spoil heap leachate, with a concentration of iron in the region of 35 mg/l. No records of the flow volume are available.

Site history and design criteria

The spoil heaps at Allanton were processed by coal-washing techniques in the 1950s, and there have been numerous minewater outbreaks in the area. The current treatment involves passing the minewater through lagoons, from which there is one main discharge. There are also numerous contaminated seeps in the area which do not pass through the lagoons.

Figure 5.2.1. The main minewater discharge at Allanton.

Figure 5.2.2. One of the settlement lagoons at Allanton.

Figure 5.2.3. The outflow from the settlement lagoons.

Monitoring data

There are no monitoring data available, although it is understood that the minewater contains in the region of 35 mg/l Fe. The discharge from the settlement lagoons appears to be fairly clear (see figure 5.2.3).

5.3 ASPATRIA, CUMBRIA

NGR NY164440

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Aspatria is acidic spoil heap drainage which requires alkalinity addition in order to precipitate out the metals. The pollution of the receiving watercourse (Sandwith/West Newton Beck) has been evident for at least 25 years, causing severe chemical and biological degradation of this stream.

Site history and design criteria

As mentioned above, the discharge at Aspatria (Carr Wood) is historic and has caused serious pollution in both water quality and aesthetic terms. In 1990, a major pollution incident was recorded when Economic Forestry Group (EFG) drained part of the spoil heap, releasing significant amounts of silt and acid minewater to the receiving stream. This caused severe discolouration of the stream through the village of West Newton and there are unsupported accounts of ochre deposits being evident all the way to the coast at Allonby.

Various remedial measures were tried, which included use of cut-off ditches to try to divert clean water away from the site, installation of settlement ponds, diversion of drainage to an existing wetland area and isolation of the newly installed drainage system from the Sandwith Beck. These measures initially worked well. However, in 1997 the local farmer carried out some drainage work which resulted in another release of acidic minewater to the stream, similar to, although not as serious as, the incident that occurred in 1990.

The treatment afforded by these measures are becoming less effective with time: the wetland vegetation appears to be deteriorating and the drainage ditches are becoming blocked with ochre and silt, so that most of the flow is diverting to the stream rather than passing through the wetland. Sandwith Beck now regularly becomes affected by ochre staining in winter months.

There are plans to install a pilot treatment system based on a compost-type wetland at the site, but all work has been put on hold due to the foot and mouth outbreak (that commenced in February 2001).

5.4 BENHAR BING, SCOTLAND

NGR NS893625

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Benhar Bing is acidic and metal-rich spoil heap drainage (pH 2.7, 47.7 mg/l total Fe and 49.2 mg/l total Al – Heal and Salt 1999). Water treatment using a constructed wetland system has been implemented as part of a wider reclamation scheme of the whole bing.

Site history and design criteria

Benhar Bing is an ironstone mine spoil heap, covering an area of 5 ha. Mining ceased in the early 20th century but the bing has remained mostly unvegetated due to the highly acidic and infertile nature of the spoil. Reclamation of the bing, carried out by Central Scotland Countryside Trust, began in 1996 with regrading of the bing to be visually more attractive and to make it suitable for tree planting (Heal and Salt 1999). The top 300 mm of the spoil was treated with peat, dried pelleted sewage, coarse limestone chips and ground limestone, and planted with alder and birch in May 1997. Drainage from the bing was collected in drainage ditches and diverted through a settling pond and a 0.4 ha surface flow constructed wetland. The wetland comprises 16 cells, divided by straw bale dykes and has a design water depth of 300 mm. The planting substrate comprises a 500 mm layer of mushroom compost over a 100 mm layer of crushed limestone. *Typha latifolia* were planted in August 1997, although on a site visit in July 2001, there were very few of this species apparent, with naturally colonised *Juncus* being the most prevalent macrophyte.

Monitoring of the site has been carried out by the Universities of Stirling and Edinburgh, and data for the period June 1997 to June 1998 have been published (Heal and Salt 1999).

Figure 5.4.1. Benhar Bing settlement lagoon.

Figure 5.4.2. Straw bales between cells.

Figure 5.4.3. Outlet v-notch from Benhar Bing constructed wetland.

Monitoring data

Table 5.4.1. Water chemistry of constructed wetland inflow and outflow

n = 13 June 1997 – June 1998	Median inflow mg/l	Median outflow mg/l	Treatment efficiency % (average values)*	Area adjusted removal $g/m^2.day$ (median values)*
pH	2.7	2.8		
Fe	226	119	21	0.8
Mn	66.4	56.0	18	0.01
Al	67.4	43.6	29	
Sulphate	2416	1830	20	-7.1
Acidity	519	298	33	

* Values taken from published information (Heal and Salt 1999).

Table 5.4.1 presents the data for the wetland performance that were published in 1999 (Heal and Salt 1999). These authors also consider the effect that the reclamation scheme as a whole had on water chemistry, and conclude that there has been an improvement in average throughflow water chemistry but that the effect is spatially variable. However, the reclamation has resulted in an increase in the concentration of Fe and Mn in the surface water drainage, even after passage through the constructed wetland system. (Before reclamation the median Fe and Mn concentrations in the surface water below the bing were 47.7 mg/l and 18.8 mg/l respectively, whereas post-reclamation values were 226 g/l and 66.4 mg/l.)

Comments on design

This wetland treatment system was implemented as part of an overall reclamation strategy. The spoil heap runoff is highly acidic, and concentrations of Fe and Mn have increased substantially as a result of the reclamation. This highlights the point that when planning spoil heap reclamation work, the possibility of a worsening of water quality should be incorporated into the water treatment provision. The low pH and high metal content of the bing run-off water makes it a difficult candidate for a purely passive treatment: in order to achieve more satisfactory removal of contaminants, supplementary measures such as alkali addition would need to be considered.

5.5 BLAENAVON, GWENT

NGR SO242085

Pre-treatment water chemistry/volume and treatment requirements

The minewater discharge at Blaenavon is from deep coal mine workings. It issues from an adit that is also the emergency escape route from the Big Pit Mining Museum. The minewater has a neutral pH but does not appear to have excess alkalinity (SRK 1994). Iron concentration in the minewater is low (generally less than 7 mg/l), although historic data indicate intermittent flushing events of high concentration. Approximately 1 km length of the receiving stream is affected with ferruginous discoloration. The treatment system being constructed for this discharge comprises a surface flow constructed wetland.

Site history and design criteria

The minewater discharge at Blaenavon issues from an adit known as River Arch. It joins with a minor run-off flow approximately 30 m downstream of the adit portal and with the culverted River Lwyd a further 400 m downstream (SRK 1994). During mining operations at Big Pit, pumped minewater discharged via the River Arch adit: the reported volume of discharge is between 2.5 and 3.5 Ml/day (30 to 40 l/s). Mining ceased at Big Pit in 1980. Flow from the adit is now gravity flow from two underground watercourses, draining from the north and from the south. The drainage from the north may be affected by opencast workings (SRK 1994).

The treatment scheme at Blaenavon was constructed during 2001. It comprises three reed bed cells, gravity fed by a pipe that runs from approximately 100 m inside the adit. The stream was diverted to enable the reed beds to be accommodated in the narrow valley.

Monitoring data

The scheme was still in its construction stage at the time of compilation of this report and as such there are no data of the performance of the treatment system.

Figure 5.5.1. The diverted stream at Blaenavon, showing ferruginous impact (prior to diversion of minewater through treatment system).

Figure 5.5.2. First reed bed cell during construction, showing inlet structures, lining material, bank stabilisation material and topsoiling in process.

Figure 5.5.3. View of the reed bed construction, from the third cell looking ‘upstream’, showing how the scheme has been fitted into the narrow valley.

Comments on design

There was confined space to accommodate this scheme and the cells have been designed to follow the course of the valley; the completed scheme should therefore be visually pleasing and have a relatively natural appearance. Minimal use of concrete should also add to the aesthetic appeal of the completed scheme.

Costs

Construction costs are reported to be in the region of £300,000, although the project was still ongoing at the time of compilation of this report. Operating costs are projected to be £1560 p.a.

5.6 BOWDEN CLOSE, COUNTY DURHAM

NGR NZ184358

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Bowden Close is acidic spoil heap drainage that is causing severe ecological damage to the receiving stream (Willington Burn, Jarvis & Younger 1997). Treatment is required to remove Fe, Mn, Al and acidity. In order to achieve this passively, an experimental vertical flow SAPS (RAPS) was installed prior to implementation of a full scale system.

Site history and design considerations

The Bowden Close Colliery and Cokeworks was abandoned in the 1960s. Restoration of the site was carried out by Durham County Council in the mid 1970s although no measures were taken to minimise infiltration and throughflow of water through the spoil heaps. As a result, acidic metalliferous minewater began issuing from a number of points, causing deterioration in the quality of the nearby stream. Work carried out in 1998–99 identified three discharges of minewater from the site (Younger 2000a). A pilot scale SAPS (RAPS) was installed to investigate the possibility of achieving passive treatment of the minewater. Two of the discharges were diverted through the pilot-scale system, with the third larger and more contaminated discharge being left until the full-scale system is installed.

It was deemed that there was sufficient hydraulic head on the site to facilitate the use of a Successive Alkalinity Producing System (SAPS), or Reducing and Alkalinity Producing System (RAPS) as they are sometimes now called. A short section of the Wilmerton Burn was culverted and the pilot-scale treatment system built in an L-shape over the culvert.

The pilot-scale RAPS unit that was installed had a surface area of 128 m^2, and it comprised a 500 mm layer of 13 mm diameter single-size limestone gravel overlain by 500 mm depth of horse manure and straw. The water depth was typically a few centimetres.

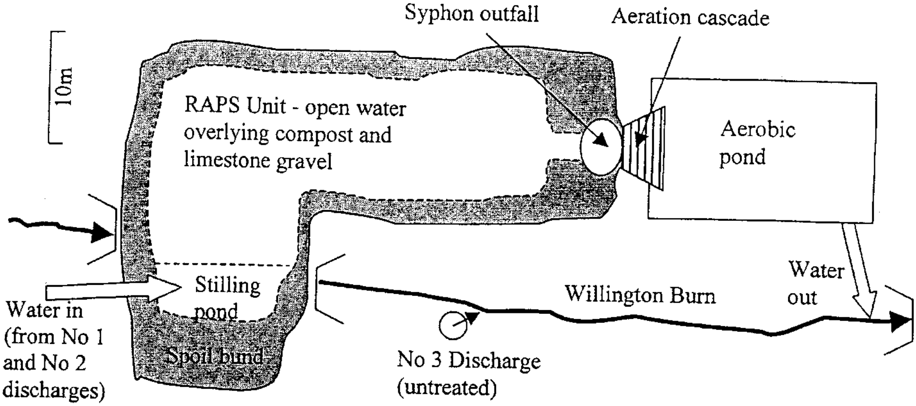

Figure 5.6.1. Sketch plan of Bowden Close treatment scheme (after Younger 2000).

Monitoring data

After initial installation, testing and modification of the pilot-scale system, the performance was monitored from March to September 2000. These data are summarised in table 5.6.1. Additional monitoring was also carried out on a daily and hourly basis to assess the consistency of treatment efficiency. It was found that efficiency was maintained on a daily basis, and that there was a slight decline in efficiency for a few hours at night. It is suggested that this latter trend may be due to photolytic oxidation reactions or temperature effects on treatment mechanisms. It was concluded as a result of the pilot study that adequate treatment capacity could be provided by a system of this sort and thus plans were made to scale up the pilot-scale system to full scale.

Table 5.6.1. Water quality data for pilot-scale treatment system at Bowden Close

n = 6, (Younger 2000)	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe	15.9	1.0	93.7	1.51	11.8
Mn	1.30	0.49	62.3	0.08	0.67
Al	5.29	2.12	59.9	0.32	2.5
Alkalinity	24	129			
pH	6.4	7.2			

The proposals for the full-scale system at Bowden Close are for a RAPS approximately 2.5 m deep and with a surface area of 2000 m^2 followed by a small aerobic reed bed. The full-scale system will receive all three acidic discharges.

Comments on design

The pilot-scale system functioned well for the short time for which it was operational. It is to be hoped that the full-scale system will perform equally well. The high concentration of aluminium in the minewater is a cause for concern, as there is a possibility that this will cause blockage of the system; the problems of Al blockage in anoxic limestone drains is well demonstrated at Wheal Jane (see case study 5.35) and it is not yet known whether similar blockage may occur in the limestone layer of RAPS. There is also the possibility that the surface of the RAPS may become ‘blinded’ with ochre, as has been observed at Pelenna phase III (case study 5.26), which is probably exacerbated by the lack of a settling pond prior to the RAPS.

Costs

The pilot-scale system cost in the region of £20,000; it is estimated that the full scale-system will cost £65,000.

5.7 BULHOUSE, YORKSHIRE

NGR SE214029

Pre-treatment water chemistry/volume and treatment requirements

Minewater is pumped from the Bulhouse adit discharge point near the river to an old quarry void where treatment is facilitated by aeration and settlement. Treatment is necessary to prevent adverse impact on the River Don, and also on the Little Don from the Sheephouse Wood adit, due to Fe loading. A length of up to 6 km of the river was affected.

Pumped volumes are very variable, ranging from 20 l/s to 80 l/s. Iron concentration is in the range 30 to 50 mg/l and pH is between 4.4 and 5.8.

The design, construction and commissioning of the Bullhouse treatment project was incorporated into a 2 year research project carried out by workers at Imperial College. During this research, it was noted that the minewater tends to be net-acid, and that aeration and settlement treatment would lead to a lowering of the pH and thus self-inhibition of the iron removal process. However, it was felt that the large volume available for the settlement lagoon in the Bulhouse Quarry would provide a sufficiently long residence time for full oxidation and settlement of ferrous iron to occur even under acid conditions (Laine & Dudeney 2000, Dudeney *et al.* 2000). It was noted that addition of alkali chemicals might be necessary on a flexible basis during times of peak flows or if the receiving stream showed periodic or sustained adverse effects from the minewater.

Site history and design criteria

Deep mining has been carried out in this area on a historical basis, and a system of drainage adits was constructed in the period 1870–1880 to dewater the workings. Deep mining ceased in 1918. Drift mining was carried out in the 1960s.

The current treatment scheme was a public/private sector partnership scheme (involving the Coal Authority, the Environment Agency, Barnsley Metropolitan Borough Council and Hepworth Building Products) and was completed in September 1998. Previous to this scheme, a study carried out in 1984 by the National Coal Board and South Yorkshire County Council recommended installation of hydrostatic dams underground in order to direct a single combined stream to Sheephouse Wood where it could be treated by lime addition and lagoon sedimentation (Dudeney 1997). In 1992 The NRA and BOC Foundation for the Environment co-funded a further study at Sheephouse Wood, in which pilot plant operations indicated that optimal pH increase by alkali addition, flocculation and sedimentation in two stages could produce water of acceptable quality for discharge, in simple equipment of relatively small volume appropriate to the local hilly terrain. This would have produced ochre sludge suitable for use by a local brick company.

The current scheme resulted from a feasibility study funded by the NRA in 1994 which was based on the impending completion of pipeclay production from the Bulhouse Quarry, operated by Hepworth Building Products. This quarry was identified as a suitable site for a sedimentation lagoon and polishing pond for minewater pumped from the Bulhouse adit.

Minewater from the discharge adit is diverted into underground storage tanks which are constructed in reinforced concrete and comprise twin tanks, each 15 m × 3 m × 4 m deep. The water is pumped from these tanks by two inverter-controlled 50 HP Isoglide pumps, each with a maximum capacity of 525 gpm (40 l/s). The tanks have approximately 2 to 3 hours holding capacity of inflow from the adit and provide a safeguard against overflow of untreated water to the river in the event of a pump failure. A third pump unit recirculates the sump contents to prevent ochre settlement in the pump suctions.

The minewater is pumped up the hill, along an old railway line, a distance of about 850 m, to a large settlement lagoon. The system comprises twin delivery pipelines, 160 and 200 mm in diameter, and a return-water pipeline 200 mm in diameter (Laine & Dudeney 2000). Hatch-box access points were placed at 100 m intervals to allow access for periodic jetting of ochre accretions within the pipes. Modifications were made to the system in September 2001 to allow ‘pigging’ of pipelines to address the problem of ochre accretion.

Aeration is facilitated by cascading the minewater down a 6 m deep reinforced-concrete manhole that has alternating platforms to divert and aerate the flow as it falls.

Minewater enters the settlement lagoon via twin-walled plastic pipework arranged to spread the flow over the full area of the lagoon. Intermediate manholes allow access for flow and water quality monitoring, and for the introduction of chemical treatment if this becomes necessary.

The settlement lagoon was constructed in a former quarry. The lagoon has a surface area of 7000 m^2, a depth of 8 m and a capacity in the region of 50,000 m^3 (Dudeney *et al.* 2000). Retention time of the minewater within the settlement lagoon is calculated to be approximately 16 days. The prevailing wind direction is across, rather than along the lagoon, so there is not a major problem with strong winds agitating the water surface and preventing adequate settling of the precipitated ochre. The success of the treatment process is attributed to the long retention time of the minewater within the settlement lagoon, which allows sufficient time for the precipitated ochre to settle out of suspension.

Minewater passes from the settlement lagoon to a small polishing pond (area 582 m^2, depth 1.5 to 2 m), from where it is pumped back down to the River Don at a point adjacent to the sealed former adit.

Figure 5.7.1. The Bulhouse lagoon.

Monitoring data

Table 5.7.1. Water quality data for the Bulhouse treatment scheme

n = 7	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe	51.8	2.5	95.2	170	24.3
pH	5.5	4.6			
Alkalinity	23	3			
Sulphate	658	655			

Laine & Dudeney (2000) report that successful treatment was achieved over the first year of operation, despite the net-acid nature of the minewater and the drop in pH that occurs in the lagoon. The River Don, into which the minewater is discharged, has sufficient alkalinity to counteract the low pH of the discharge and Environment Agency discharge consents for iron (< 1 mg/l) and pH (> 6.5) were consistently met downstream of a 200 m mixing zone.

Comments on design

The scheme is over-sized in terms of the depth of the lagoon, which has the advantage that there is adequate retention time for settling of the precipitated ochre to occur. The net-acid nature of the minewater means it is not ideally suited to passive aeration and settlement treatment, but the long retention time, and the buffering effect of the receiving stream, means that adequate treatment has been achieved to date without the need for addition of chemicals.

The aeration mechanism is simple and effective – the cascading minewater is repeatedly impacted on a surface ensuring small droplet size and efficient aeration.

Pumping costs are very high. Ongoing maintenance and regular jetting of the pipework will be necessary to prevent ochre build-up and subsequent restriction of flow (investigation of the 160 mm upflow pipe found an ochre accretion rate of 2 to 4 mm/month (Laine & Dudeney 2000)).

Costs

The Bulhouse minewater treatment scheme cost approximately £1.2 million. Most of this amount was accounted for by the cost of the pumps and pipework; the lagoon itself cost about £120,000. Operating costs are £43,404 p.a.

5.8 CAPHOUSE COLLIERY, WEST YORKSHIRE

NGR SE246162

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Caphouse Colliery, which is pumped from the Hope Shaft, is deep minewater without excess alkalinity (despite having underground connection to Woolley (case study 5.36) where the minewater is net alkaline). The iron concentration of the minewater fluctuates at around 50 mg/l. The treatment requirement is to add alkalinity and settle out the iron precipitate. The volume of water pumped at Hope Shaft is dependent, in part, on the volume pumped at Woolley. The water level has to be kept pumped down in order to maintain safe underground access at the Caphouse National Coal Mining Museum for England. Since 1998, pumping at Woolley has decreased and that at Hope has increased. The average pumped volume at Hope in 1999 was 2400 m^3/day (28 l/s) and in the first 3 months of 2000,the volumes were 2600 m^3/day (30 l/s) (ECS 2000). In early 2001, the average pumped volume at Hope was 3300 m^3/day (38 l/s).

Site history and design considerations

Caphouse Colliery closed in 1985, at which time coal was being transported underground to Woolley Colliery, 7 miles to the south. Subsequent to its closure, Caphouse was chosen as the preferred site for the Yorkshire Mining Museum (now the National Coal Mining Museum for England), which has an underground exhibition gallery and offers underground tours of the mineworkings. It is therefore imperative to control water levels within the underground workings. In 1997 there was a significant increase in minewater flows and levels within the Caphouse workings and the underground exhibition gallery was temporarily closed to the public. A scoping study was commissioned by the Museum and the Coal Authority to evaluate what actions could be taken to maintain the Museum and secure its future (ECS 2000). This study discussed the complexity of minewater movements in the Caphouse/ Woolley pond and put forward various options, together with a preferred option, for continued minewater management and treatment at the Hope Shaft.

Currently minewater is pumped at Hope Shaft, lime is added by an automated lime dosing plant and the water then passes over a cascade into two settling tanks before discharging to the Smithy Brook. Treated minewater constitutes most of the flow in the Brook, which is used by a local farmer for watering cattle. The settling tanks quickly fill with sludge and need to be emptied every 6 to 8 weeks. The tankered sludge is taken to sludge drying beds at Woolley.

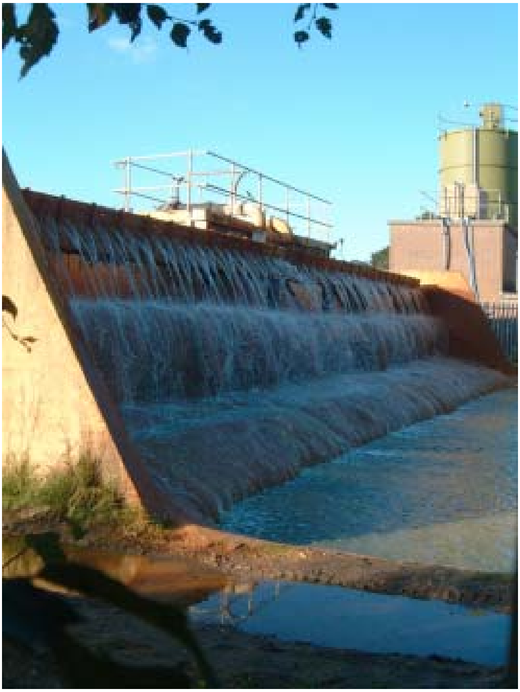

Figure 5.8.1. Aeration cascade and lime silo at Caphouse Colliery.

The current discharge consent for the Caphouse site is 10 mg/l of iron, although a new consent has been set by the Environment Agency which sets a limit of 3 mg/l Fe for flows up to 3000 m^3/day (35 l/s), dropping to 2 mg/l if the flow rises to 8000 m^3/day (93 l/s).

The preferred option identified in the scoping study is to upgrade the existing treatment facility at the Hope Shaft by the addition of two settlement lagoons, a balancing lagoon and two reed beds. This should enable increased flows to be treated to conform with the tighter discharge consent. Planning for improvements to the treatment system is currently underway (November 2001).

Monitoring data

The minewater pumped at Hope Shaft is sampled and analysed daily for pH, acidity, total and ferrous iron. In 1998 the total iron concentration was just above 60 mg/l, and it currently fluctuates at around 50 mg/l. There is no indication of any increasing or decreasing trends in the Fe concentration.

Treated minewater has an average total Fe concentration of 3.08 mg/l (October 1998 to January 2000, n=61).

Costs

Current operating costs are £205,080 per annum.

5.9 CRAIGENBAY QUARRY, GALLOWAY, SCOTLAND

NGR NX560790

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Craigenbay Quarry is acidic and contaminated with several 'List II' heavy metals. The minewater originates in a quarry that is used by Forestry Enterprise as a source of roadstone for forest roads in Scotland. The rock-type is a low-grade metamorphic slate that is sulphide-rich and has mineralisation associated with a nearby granite pluton. The main contaminants are Fe, which can reach concentrations of up to 40 mg/l and Al, which reaches concentrations of nearly 90 mg/l. The most contaminated water is below the settling lagoons and it is assumed that the source of this contamination is largely the stockpiles of weathered roadstone. Water issuing from the quarry itself has average Fe and Al concentrations of approximately 13 and 21 mg/l respectively. Mercury has been identified as a problem, and there is a small amount of gold associated with the mineralisation.

Flows are highly variable and alter rapidly with rainfall. Attempts to estimate the typical flow from the site affecting the burn by use of v-notches concluded that flows were typically between 0 and 3.8 l/s, and the value of 3.8 l/s was used for treatment design purposes.

The acidic nature of the discharge, and the requirement for a passive treatment system, meant that it was decided to install a SAPS system, which was intended to raise pH and remove Al before the flow discharged to a compost wetland.

Site history and design criteria

In 1995, serious pollution levels were recorded in the Shaws Burn, about 1 km downstream from Craigenbay Quarry. The need to install a treatment system was identified, and the desire to achieve treatment passively was expressed. The system that was designed made use of SAPS technology and sizing criteria as proposed by Hedin *et al.* (1994) for the subsequent wetland cells.

Construction work commenced in late summer 1997. An impermeable clay was found during site works and this was used to line the cells. The anaerobic wetland had a base layer of limestone with a covering of a mixture of soil and peat. The anaerobic wetland was planted with *Typha latifolia* and flooded with surface water and rainwater to allow the plants to become established prior to introduction of minewater in summer 1998.

Figure 5.9.1. The wetland area at Craigenbay.

Holding tanks were installed prior to the SAPS and wetland cell 1 to act as settling and water balancing tanks.

Cascades were incorporated into the channel between wetlands cells 2 and 3 to aid in oxygenation of the minewater. The final outflow from wetland cell 4 flows into an area of natural wetland that performs a ‘polishing’ function prior to discharge to the burn.

Monitoring data

Although it is understood that regular monitoring is carried out at Craigenbay Quarry, it has not been possible to obtain the resultant data. The Scottish Environment Protection Agency visit the site regularly but do not take water quality samples. Forest Enterprise provided the ‘ball park’ figures presented in table 5.9.1, but were not able to supply more detailed data.

Table 5.9.1. Water quality at Craigenbay Quarry

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
pH	3.0	5.7			
Fe	30	1	96.7	9.5	6.9

N.B. There is no regular monitoring at this site – the data values used in the table are 'best guess' values provided by Forest Enterprise so can only be regarded as approximations of the treatment performance of the system.

Comments on design

The system that has been installed at Craigenbay seems to be functioning effectively, although no detailed data are available to carry out an accurate assessment of performance. At the time of visiting the site, in July 2001, the SAPS had been excavated to facilitate the installation of a new membrane.

As with other ALDs and SAPS, the life expectancy of the system is uncertain, in view of the possibility of Al precipitates causing blockages.

In the long term, Forest Enterprise are considering expanding the quarry, which might exploit the sulphidic, mineralised material and thus remove the source of the contamination.

Costs

The total cost of the design and construction works at Craigenbay is quoted as £33,200 (Norton *et al.* 1998).

5.10 DALQUHARRAN, AYRSHIRE

NGR NS266017

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Dalquharran is acidic and highly ferruginous, issuing from a drift mine that was closed in the late 1970s. The current discharge consent is that discharge to the burn should have Fe concentrations not exceeding 350 mg/l and a pH of between 5 and 9. The strategy adopted, rather than a treatment system, is to use the 'dilute and disperse' principle, whereby minewater is retained within the mine workings and in a holding tank and released when flow in the burn and receiving river is high. There is a requirement for concentrations of Fe in the River Girvan, below the confluence with the burn, to remain below 1 mg/l.

Reed beds have been established on the site to receive excess flow from the drift discharge.

Site history and design criteria

The initial outbreak of minewater from the Dalquharran drift mine occurred in 1979 and caused a large scale pollution incident on the River Girvan, a prime salmon river. The initial minewater had a pH of 3 to 4, an iron concentration of 1300 mg/l and a flow rate of 26 l/s (Carter 1994). Fish life was destroyed along the 10 mile stretch of the river to the sea, and highly visible iron staining occurred along this length. The National Coal Board was successfully prosecuted by the Clyde River Purification Board for the incident.

The solution that was devised by Babtie Geotechnical, on behalf of the National Coal Board, was principally one of water management, whereby incoming, fresher minewater is intercepted by a horizontal relief borehole, which substantially decreased flows from the adit, and the deeper, more polluted minewater is allowed to issue from the drift mouth at a controlled rate (Carter 1994). Sealing of the shallow workings was also carried out in an attempt to reduce water inflow. The river level, depth of water in the tank, and discharge volume are monitored continuously and the data are relayed by telemetry to the offices of International Mining Consultants Ltd (IMC), the site managers. Release of minewater from the holding tank to the burn is initiated automatically when recorded river levels are sufficiently high to give adequate dilution. Minewater is stored within the mine waorkings and in a water holding tank; it is understood that there is up to 30 days' storage capacity in the mine workings.

Reed beds were added at the drift mouth in 1994 to receive any overflow from the minewater holding tank to try to prevent uncontrolled discharge to the burn.

Figure 5.10.1. Discharge point from minewater holding tank to the burn.

Figure 5.10.2. Holding tank, overflow to wetland and valve arrangement (in box) to allow water to be diverted to wetland when needed.

Figure 5.10.3. General view of the wetland area and the minewater holding tank.

Monitoring data

As mentioned above, there is continuous monitoring of various critical parameters to allow the release of minewater into the river at appropriate times. Water quality monitoring is carried out approximately monthly but the data set is not suitable to facilitate determination of the treatment efficiency of the reed beds, since flows into the reed beds are highly intermittent and are not recorded. Monitoring is carried out by both the site managers (IMC) and the Scottish Environment Protection Agency (SEPA), although the exact points at which each of these entities monitor vary. Table 5.10.1 presents summary data for the period from 1996 to 2000.

Table 5.10.1 Average water quality values at Dalquharran

1996 to 2000	Average values in holding tank mg/l	Average outflow from wetland mg/l	Average discharge to burn mg/l
pH	5.9	6.0	6.1
Fe	147 (range 38–342)	10.4	180 (range 125–227)

This is more of a water management solution to minewater pollution, rather than a treatment system, and as such, calculation of the removal of Fe is not relevant, nor is the data set adequate to allow it.

Comments on design

Dalquharran is the only abandoned mine in the UK where the original operator (the National Coal Board) was successfully prosecuted for *causing* poisonous, noxious or polluting matter to enter controlled waters (Lockhart v. National Coal Board 1981).

Interception of water entering the old workings decreased the volume of contaminated water substantially. A holding tank was installed at Dalquharran to store water until high river flows allowed its discharge without making a significant impact on the river (Water of Girvan). However, overflows during dry weather and discolouration still occurred, so wetland was put in place. The system thus operates on a principle of 'dilute and disperse', with discharge to river only occurring when river flow is high. The wetland provides a 'buffer' for use when the holding tank overflows and discharge to the river directly from the tank is undesirable.

Costs

There are no capital costs available for Dalquharran. Operating costs are £4,560 p.a.

5.11 DODWORTH, SOUTH YORKSHIRE

NGR SE314057

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Dodworth is acidic, metal-rich spoil heap leachate, with possible effects from deep minewater rebound as well (Bannister 1997). The flows are highly variable, dependent on rainfall. Periods of high flow correspond to higher concentration of contaminants in the leachate, which indicates a flushing rather than a dilution effect (Bannister 1998).

Metal contaminated water from the northern section of the site constitutes the start of a small watercourse that forms a tributary to Silkstone Beck. Due to the variability of flow, the stream becomes dry in periods of low rainfall.

The treatment requirement is to add alkalinity and precipitate out metals. Various remediation techniques have been used to effect improvement of the quality of water leaving the site (see site history, below) and a SAPS system is currently under consideration (Bannister pers. comm.).

Site history and design criteria

Tipping commenced at the Dodworth site in the 1860s and is thought to have been mostly washery waste tipped in shale bunds. The tip was extended northwards in the 1970s and is currently over 800 m long with an approximate area of 29.5 ha. Tipping ceased in 1984 and the site was purchased from British Coal by Barnsley Metropolitan Borough Council (BMBC) in 1989. Extensive remediation works were carried out by BMBC, including reshaping, capping with clay, tree planting and establishment of agricultural grassland. In 1993 a two-stage trial reed bed was constructed which comprised an ALD and a reed bed. The area of the reed bed was about 50 m^2. There were considerable problems with vandalism on the site but nevertheless the reed bed successfully removed Fe and Al from the leachate (Bannister 1997).

The water treatment system was extended in 1995 by the addition of a lagoon, with an area of approximately 500 m^2, comprising two compartments divided by gabion baskets filled with lump limestone, and in 1996 by the construction of two reed beds with a combined area of 1500 m^2. The layout of the treatment system is shown in figure 5.11.1 – one reed bed is upstream of the lagoon, and the second reed bed is downstream. A perimeter ditch collects leachate from the northern section of the tip, and the wetland complex receives leachate (and surface runoff during storm events) from this ditch as well as diffuse seepage of leachate and runoff from the tip (Hine *et al.* 2000).

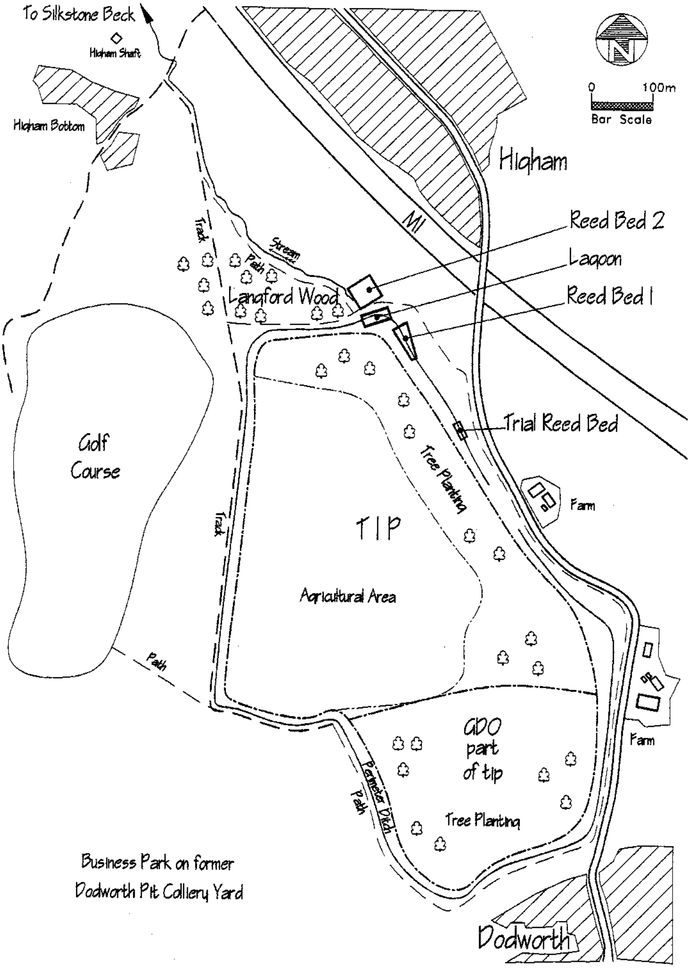

Figure 5.11.1. Site layout at Dodworth showing position of reed beds and settlement lagoon (from Bannister 1997).

Figure 5.11.2. Drainage ditch between trial reed bed and reed bed 1; note Al precipitates. Spoil heap is to the left of this photograph (photo courtesy of John Hine).

Other remedial work has been carried out on the site, comprising addition of chemical ameliorants (Bannister 1997). During the period January 1994 to February 1996, agricultural lime was added to the perimeter drainage ditch and the lagoon. Lime was also spread on the agricultural grassland that had been established on the tip. From January 1996, soda ash was used instead of lime, being added to the perimeter ditch and lagoon using gabion divisions and mesh boxes, respectively. In autumn 1997, an investigation into the use of bonemeal as an ameliorant was implemented (Bannister 1998).

The Environment Agency have stipulated the following Discharge Consent levels:

PH	6 to 9
Iron	4 mg/l
Nickel	800 µg/l
Solids	30 mg/l
Copper	100 µg/l
Zinc	1500 µg/l

Figure 5.11.3. Reed bed 1 at inflow; receives discharge from the drainage ditch (see figure 5.11.2) and also direct inputs from the tip (photo courtesy of John Hine).

Figure 5.11.4. Settlement lagoon prior to discharge to reed bed 2. Sodium carbonate is added; note bags of this on the dividing bund, and also note precipitation of Al following this addition (flow is from left to right) (photo courtesy of John Hine).

There is no consented level for Al, although this is probably the major contaminant of concern. Presumably it is reasoned by the EA that if sufficient alkalinity is added to the minewater to attain consent levels for the other parameters, then a suitable Al concentration will also be achieved (Bannister 1997).

Monitoring data

The tip leachate at Dodworth has been monitored by BMBC since 1993 (Bannister 1997) and has been the subject of detailed modelling and geochemical investigation by a PhD student at Leeds University (Hine *et al.* 2000). Tables 5.11.1 and 5.11.2 present summary data from two periods – 1997/8 and 2000.

Table 5.11.1. Water quality data for the reed beds at Dodworth for the period February 1997 to March 1998

n = 13	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %
Reed bed 1			
pH	4.5	4.0	
Fe	22	2.4	89.1
Al	43	35	18.6
Ni	0.6	0.7	-16.7
Zn	1.0	1.1	-10.0
Reed bed 2			
pH	3.8	4.1	
Fe	14	1.5	89.3
Al	60	72	-20.0
Ni	1.2	1.4	-16.7
Zn	1.9	1.6	15.8

N.B. No flow data are available for this period so no calculations of contaminant loading or retention are possible. Data supplied by BMBC.

Table 5.11.2. Water quality data for reed bed 2 at Dodworth for the period January to May 2000

n = 5	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
pH	4.4	4.4			
Fe	7.09	0.47	93.4	1.1	0.8
Al	64.7	59.6	7.9	0.8	
Cu	0.11	0.10	9.1	0.002	
Mn	37.9	33.7	11.1	0.7	
Zn	1.85	1.47	20.5	0.1	
$S0_4$	3382	3014	10.9	60.1	

Data from Hine *et al.* 2000, retained quantities based on an average flow of 1.87 l/s, although it must be stressed that flows are highly variable

Consideration of table 5.11.1 reveals that the reed bed cells were effectively removing Fe during the period in question but were less efficient at removing Al, Ni and Zn. In fact, the reed bed cells act as a net source of Zn and Ni. Examination of the raw data (not presented here) reveals that consented levels for Zn were exceeded in the final effluent, from reed bed 2, on eight of the 13 sampling occasions, and those for Ni on ten occasions. Aluminium concentrations in the effluent from reed bed 2 exceeded those in the influent on five out of thirteen occasions. Removal of Fe is, however, effectively achieved and on none of the thirteen sampling occasions was the consented level exceeded. Consented levels of pH were not achieved on any of the sampling occasions. These observations corroborate the remark made by Bannister (1997) that, with the exception of Fe, "the consent levels are unrealistically low and are unlikely to be met in anything other than the summer and early autumn months, no matter what is reasonably done."

Consideration of table 5.11.2 (and raw data which are not presented) reveals that the second reed bed continues to perform efficiently for Fe removal, with no instances of failure to comply with consented levels on any of the five sampling occasions. Consent levels for pH, however, are not achieved on any occasion, and Ni, Zn and Cu fail on three, three and two occasions respectively. Manganese is only decreased in concentration on passage through the reed bed cell on one occasion (in January 2000). On the other sampling occasions a negative removal efficiency is achieved (i.e. the reed bed cell acts as a source rather than sink for Mn).

The maximum removal efficiency for all the contaminants (i.e. considering combined Al, Cu, Fe, Mn, Ni and Zn concentrations) corresponds to the sampling period with the lowest flow, which indicates that low flow conditions and extended retention time within the reed bed cell give better removal of contaminants (Hine *et al.* 2000). Conversely, conditions of high flow and high concentrations of contaminants result in the least effective removal (Hine *et al.* 2000). It is also suggested by Hine *et al.* (2000) that remobilisation of metal colloids from the wetland substrate may occur in conditions of high flow.

Comments on design

The monitoring results reveal that, due to highly variable flows and contaminant concentrations, the current water treatment system does not always perform satisfactorily. The planned installation of a SAPS system should greatly improve the degree of treatment that can be achieved passively on this site. Addition of alkalinity should address the problems of low pH; Cu, Zn and Ni should be retained within the organic substrate as sulphides, and Al should also be effectively removed, although possible blocking of the system by Al-precipitates may be a problem.

This case study demonstrates the highly problematic nature of spoil heap leachate treatment. The flows and concentrations of contaminants tend to be much more variable than for underground minewater discharges, which in turn means that achieving consistently high standards of treatment is more difficult for spoil heap leachate than for deep minewater.

Costs

The reed beds were installed at a cost of £23.00/m^2, and the annual running costs are of the order of £20,000 per annum., which sum includes all expenses associated with the site: personnel time, consumables, monitoring, contribution to EA for monitoring etc. (Bannister pers. comm.).

5.12 EDMONDSLEY, DURHAM

NGR NZ231493

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Edmondsley is net alkaline, issuing from a former drift coal mine. The major contaminant of concern is iron and the treatment requirement is aeration and precipitation.

Due to available land constraints, minewater is pumped from the emergence point in the valley to the treatment site some 45 m higher up the valley side. For the period September 1999 to May 2000, the pumped flows, monitored on a monthly basis, varied between 3.2 and 6.6 l/s, with an average value of 4.4 l/s.

Site history and design criteria

The treatment wetland at Edmondsley was constructed in 1999 by the Coal Authority with consultants SRK (UK) Ltd and contractors Rockbourne Civil Engineering Ltd. It comprises three cells in series, with raw minewater input to each cell as well as flow between cells. The cells were planted with *Typha* and *Scirpus* in a compost substrate. Inlet to the first cell is through gabion baskets (see figure 5.12.1), and to the second and third cells via a plastic pipe along the top of the dividing walls. Flow between the cells is via pipes through the dividing walls, with the pipes extending the whole height of the walls to allow for build-up within the wetland cells (see figure 5.12.2). The outlet is similar to the between-cell arrangement, with a collecting channel (see figure 5.12.3). The treated water returns to the river by gravity flow in a pipe.

Figure 5.12.1. Inlet to first wetland cell, showing gabion baskets through which flow occurs. Note poor *Typha* growth and ochre accumulation. The *Scirpus* at the far end of the wall, middle left of the photograph, appear to be more healthy than the *Typha*.

Figure 5.12.2. Dividing wall between cells, showing pipe along top of wall for raw minewater input, and small pipes through wall for between-cell flow.

Figure 5.12.3. Outlet from the Edmondsley constructed wetland.

Monitoring data

Monitoring of the treatment system at Edmondsley has been carried out on a monthly basis since completion of the scheme in September 1999. Table 5.12.1 presents data for the first 9 months of operation.

Table 5.12.1. Water quality data for Edmondsley minewater treatment scheme for the period 8.9.1999 to 5.5.2000

n=9	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe (total)	13.4	0.6	95.5	4.9	1.5
Fe (sol)	6.6	<0.05			
Mn (total)	2.3	1.2	47.8	0.4	0.14
Mn (sol)	2.0	0.9			
pH	6.4	7.1			

The data presented in table 5.12.1 show that the scheme is working well for removal of iron, and is also removing about half of the influent manganese. Based on an average flow through the system of 4.4 l/s, the average hydraulic loading is 0.12 /day and the average retention time within the system is 3.4 days, assuming sequential flow through all the cells. These values suggest that the system is not being put under particular stress (the hydraulic loading is low, and the retention time high compared to other systems, see discussion in Chapter 7). Average area adjusted removal for Fe is 1.6 g/m^2.day, which is considerably lower than suggested 'design parameter' values (Hedin *et al.* 1994) of 10 to 20 g/m^2.day (depending on water chemistry). The system can thus be considered to be 'oversized' for the volume and concentration of the minewater that it is to treat, but this should mean that effective treatment will continue for a considerable period.

Comments on design

The fact that the minewater has to be pumped was, presumably, unavoidable in view of available land constraints and topography, but is unfortunate as it stops the overall treatment scheme treatment being essentially 'passive'. There is the possibility of ochre build-up in the pipes which may compromise the efficient running of the system, and have high maintenance cost implications.

Access to the pumps, in the valley, is difficult, which can cause problems when maintenance is required.

The gabion wall inlet arrangements are subject to blocking with ochre (an unblocking operation was carried out in September 2001), and the small pipes through the concrete dividing walls are very liable to become blocked with plant debris and require periodic clearing.

The *Typha*, especially in the first cell, do not appear to be thriving, but the *Scirpus* seem to be quite healthy. It can be expected that *Scirpus* may become the dominant plant species, replacing the *Typha*. There is also a certain amount of colonisation by other plants, notably *Juncus* and *Carex* species.

The fact that the scheme is on a hillside, rather than in a valley bottom, means that it is difficult to make it 'blend in' and look natural, although planting of some marginal species, such as willow and alder, on the banks would probably improve the aesthetic appeal of the scheme.

Costs

Capital costs for the scheme were £277,000, operating costs are £24,036 per annum.

5.13 FENDER, DERBYSHIRE

NGR SK364752

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Fender is net alkaline from deep and outcrop mines. It is both an historic discharge and also as a result of post-pumping water table rebound. The discharge on the bank of the river is collected and pumped to the wetland treatment scheme. The receiving stream (the Barlow Brook) was previously stained orange due to ochre precipitation, but since pumping started in July 2000, the stream bed has become considerably cleaner.

Iron in the discharge is approximately 10 mg/l and the pumped volume is in the region of 25–26 l/s. Treatment comprises aeration and settlement in an aerobic wetland.

Site history and design criteria

The treatment scheme at Fender was commissioned and financed by the Coal Authority. Pumping commenced in July 2000 and completion of the wetland treatment scheme was in August 2000. The scheme was designed SRK (UK) Ltd and the contractor was Rockbourne Civil Engineering Ltd.

The minewater discharge is pumped to the treatment wetland. The average flow is 25–26 l/s and the design flow is 111 l/s. The wetland comprises four cells in series, planted with *Phragmites*, *Juncus* and *Scirpus.*

The inflow to the first cell is through gabion baskets. At the time of the site visit (1.11.2000) the flow through the gabion baskets was much reduced from the design flow; it was establised that this was because the loading of Fe to this first cell had been much greater than expected and had resulted in the plants becoming smothered in precipitated ochre. The water level has been reduced and the first 5 metres of this bed are to be replanted, this time with *Scirpus* which shows greater tolerance to Fe loading.

Inflow to the remaining three beds is via plastic pipes along the top of the upper wall of each bed, which facilitates cascading and aeration. These pipes are used as a by-pass system to allow maintenance to be carried out on individual cells. The main flow path is between the cells via the overflow portals in the dividing walls and via a plastic perforated pipe that serves as an overflow from each cell and runs the length of the wetland.

Figure 5.13.1. Gabion baskets at the inlet to the first wetland cell. Note the heavy ochre precipitation and poor plant growth.

Figure 5.13.2. Inlet cascade to second wetland cell, being used to by-pass the first cell.

Outflow from the wetland system is via three concrete outflow structures in the fourth bed with subsequent discharge to the river. One of the outflow structures has a nearby vertical pipe arrangement as an emergency overflow.

Figure 5.13.3. One of the three outlet structures from fourth wetland cell.

Monitoring data

Monitoring of the treatment scheme at Fender began in September 2000 and has been carried out on a monthly basis, with a period from February to April 2001 when no monitoring was carried out due to site access restrictions caused by the foot and mouth disease outbreak. Inflow and outflow water quality is monitored. In addition, on three occasions, water has been sampled at each of the cell dividing walls.

Examination of table 5.13.1 shows that iron is being quite effectively removed, and that very little particulate Fe is exiting the treatment system. It is to be expected that as the plant growth becomes more dense, the retention of particulate Fe should improve due to better settling of fine solid material. The concentration of manganese is only decreased slightly by passage of the minewater through the system.

Graphical representation of the results of sampling at intermediate points through the system is presented in figures 5.13.4 and 5.13.5. There is an indication of improved treatment in June/July compared to May, which is probably due to increased plant growth as the growing season proceeds.

Table 5.13.1. Water quality data for Fender minewater treatment scheme for the period 21.9.2000 to 19.7.2001

n=10 (7 for Mn)	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (total)	9.53	1.85	80.6	16.6	2.8
Fe (sol)	3.85	0.49			
Mn (total)	1.08	0.84	22.2	0.5	0.09
Mn (sol)	1.01	0.77			
pH	6.6	7.2			

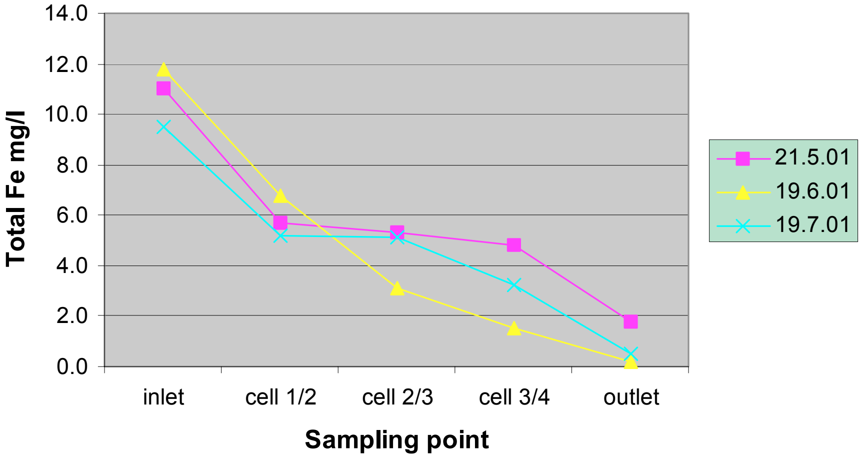

Figure 5.13.4. Total iron concentration at intermediate points through the Fender treatment scheme.

Figure 5.13.4 shows that the first cell retains the highest proportion of total Fe, with the remaining three cells retaining variable amounts on the different sampling occasions. Examination of figure 5.13.5, which is plotted with the same vertical axis scale to allow direct comparison, shows that soluble Fe in the inflow to the scheme is already quite low (i.e. a substantial proportion of the Fe is in particulate form as the minewater enters the wetland). By the exit of the first cell, the dissolved Fe has decreased still further, which indicates that effective aeration and precipitation occurs in the first cell. The gradual decrease in dissolved Fe concentration through the rest of the system is indicative of further aeration and precipitation.

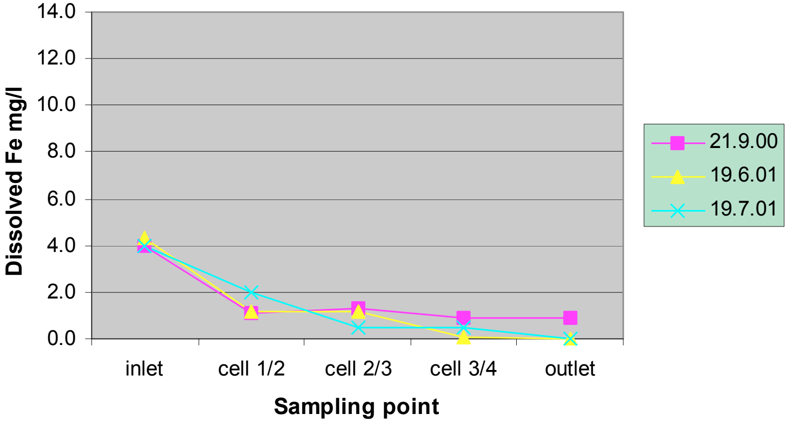

Figure 5.13.5. Dissolved iron concentration at intermediate points through the Fender treatment scheme.

Comments on design

The gabion inlet structure is liable to block with ochre, as are the perforations in the perforated pipe that runs the length of the scheme.

The fact that a large proportion of the precipitated iron is retained in the first cell suggests that it might be preferable for this to be a settling tank, without plants. This tank could then be periodically cleaned out, and if a use could be found for the ochre, it would not be 'contaminated' with organic matter, as would be the case with the current arrangement.

The pumphouse is in the industrial estate and is therefore easily accessed for maintenance.

Costs

Capital costs were £385,000, and operating costs are £21,204 per annum.

5.14 FOSS MINE, ABERFELDY

NGR NN814 545

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Foss barite mine is acidic and has elevated concentrations of iron, manganese, aluminium and zinc. The mine is still active and approximately 12.5 l/s is pumped during 7-hour weekday shifts. The current treatment method used for this minewater is the addition of lime and settlement in lagoons, which ensures the discharge meets stringent quality standards before discharge into the Frenich Burn, which drains north into Loch Tummel.

A pilot-scale passive treatment system was constructed on the site to investigate the possibility of using a passive treatment approach in the future. The requirements of the treatment system are to add alkalinity and remove metals, and the pilot system comprised SAPS, aerobic wetland cells and an ALD to try to achieve this.

Site history and design criteria

Foss barite mine is situated in the Scottish Highlands, some 100 km north of Edinburgh and at an elevation of approximately 690 m. This means that the climate is sub-arctic, with the mine having to close for between 2 weeks and 2 months each year due to heavy snowfall and sub-zero temperatures. Thus, any passive treatment that is installed will need to cope with the seasonal fluctuation in temperature to ensure effective year-round treatment.

Researchers at Newcastle University designed a system to address each of the contaminant problems in the Foss minewater (Jarvis & Younger 2001). Two SAPS (successive slkalinity producing systems), or RAPS (reducing and alkalinity producing systems), were used to strip out oxygen from the minewater and generate alkalinity. Aerobic wetland cells followed, with the purpose of removing Fe and Al as solid hydroxide precipitates. The final stage of the pilot system was an anoxic limestone drain (ALD) which was intended to remove Zn as smithsonite ($ZnCO_3$) as has been successfully demonstrated at Nenthead in the Pennines (Nuttall & Younger 2000a and 2000b; see case study 5.23).

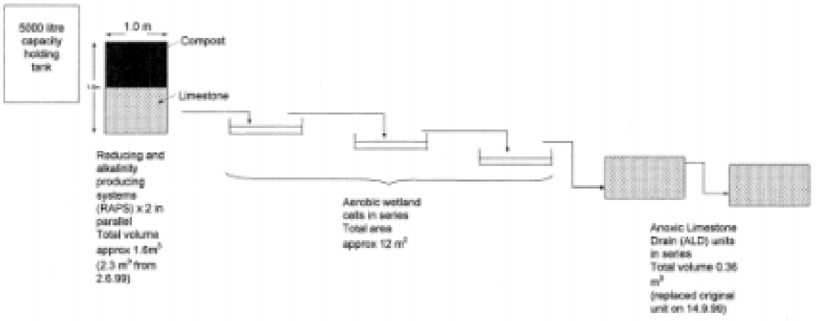

Figure 5.14.1. Schematic layout of pilot passive treatment scheme at Foss mine.

Monitoring data

The pilot scale system was monitored over a 3 month period, during which time eleven sets of samples were collected. These data are presented in table 5.14.1.

Table 5.14.1. Water quality data for pilot scale treatment system at Foss Mine, Aberfeldy

n = 11 during 9/9/1998 to 18/12/1998	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
pH	3.91	6.61		
Fe	12.62	0.62	95.1	0.031
Al	11.25	1.07	90.5	0.026
Mn	16.88	12.17	27.9	0.012
Zn	33.48	18.56	44.6	0.039

Data from Jarvis and Younger 2001. Calculation of retained quantity made using average flow value of 2 l/min (Younger pers. comm.)

Iron and aluminium are consistently removed by passage through the treatment system, with a treatment efficiency in both cases of over 90%. Jarvis and Younger (2001) state that closer examination of the raw data reveals that a significant proportion of the Fe is removed in the RAPS cells, which was evidenced by ochre formation on the surface of the RAPS. These workers suggest that iron is also removed *within* the RAPS by adsorption of ferrous iron to organic matter and precipitation of iron sulphides as a result of bacterial sulphate reduction. Although some Al was removed in the RAPS cells, it was predominantly removed in the aerobic cells, presumably as hydroxide.

Concentration of manganese was reduced by less than 30% by passage through the treatment system, with removal falling off with time; this suggests that sorption processes may be responsible for the immobilisation of Mn, and that sorption sites become saturated over time (Jarvis & Younger 2001).

Removal of Zn was moderately successful but very variable with time (Jarvis & Younger 2001). Most of the Zn appears to have been removed in the aerobic cells, which was probably by adsorption to organic matter, co-precipitation with Fe and Mn, or precipitation as sulphide in anaerobic micro-environments within the aerobic cells. Removal of Zn within the ALD was not as effective as had been anticipated; it is suggested this was due to low ambient temperatures having an adverse effect on the rate of alkalinity generation, which in turn restricted the amount of Zn precipitated as carbonate within the ALD. It is proposed that either burying the ALD to increase the ambient temperature, or increasing residence time of the water within the ALD may improve the Zn-removing performance.

Comments on design

This pilot-scale system used recently developed passive treatment technologies to investigate the possibility of treating an acidic minewater with two particular metal contaminants that are difficult to remove by passive means – zinc and manganese. The investigation raised some interesting issues which will, no doubt, be the subject of further research.

There are no current plans to install a full-scale passive treatment system at this site.

Costs

Since this was an experimental, pilot-scale system, the extrapolation of costs to a full scale is not relevant, nor have any budget costs been prepared for such an exercise.

5.15 GILMERTON BING, MID LOTHIAN

NGR NT297682

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Gilmerton Bing is acidic spoil heap leachate. The volume of minewater passing through the reed bed is not recorded. The iron concentration in the minewater is high, and very variable; SEPA data collected during the period 1995 to 1998 records Fe concentrations of between 13.3 mg/l and 1216 mg/l. Aluminium concentrations are also high and variable; recorded values vary between 4.9 mg/l and 178 mg/l.

In order to achieve satisfactory treatment of a minewater of this nature, it would probably be necessary to use active treatment, due to the very high concentrations of Fe and Al that are recorded periodically.

Site history and design criteria

Figure 5.15.1. Inflow pipe and terrace.

The wetland at Gilmerton Bing was put in place by the National Coal Board, although it has not been possible to find out much about the history of the site. The bing itself has been reclaimed for agricultural use, and presumably the wetland was made at the same time to attempt to address the minewater issues. The wetland comprises a piped inflow to a terrace and then a single wetland cell with two bunds extending most of the way across the wetland area to give a sinuous flow path. The wetland area is mostly free of vegetation, with some *Phragmites* growing near the inflow and outflow points. The outflow comprises gabion baskets across the width of the cell with a single underground piped discharge to a stream some considerable distance away.

Figure 5.15.2. The wetland cell at Gilmerton, looking towards the inflow and showing the reclaimed bing behind the wetland and the bunds extending across the width of the cell.

The wetland cell is approximately 30 m square. There is very distinct ochre colouration within the cell and presumably a considerable depth of ochre has accumulated. There does not appear to be any organisation or individual currently taking responsibility for this wetland, although SEPA do periodically carry out monitoring of the inflow and outflow water quality.

Figure 5.15.3. Gabion baskets at the outflow of the wetland cell.

Monitoring data

There are limited data that have been collected by SEPA. The inflow water quality was tested on fourteen occasions (on four of which there was no flow) during the period January 1995 to March 1998, and the outflow water quality was tested a further eleven times during the period May 1998 to September 2000. The water quality of both the inflow and outflow are highly variable. Average values are presented in table 5.15.1, and figure 5.15.4 presents the data for Fe graphically.

Table 5.15.1. Average water quality data for Gilmerton Bing wetland

	Average inflow mg/l (n = 10)	Average outflow mg/l (n = 22)	Treatment efficiency %
pH	3.8	3.0	
Fe	403	310	23.1
Al	72	88	-22.2

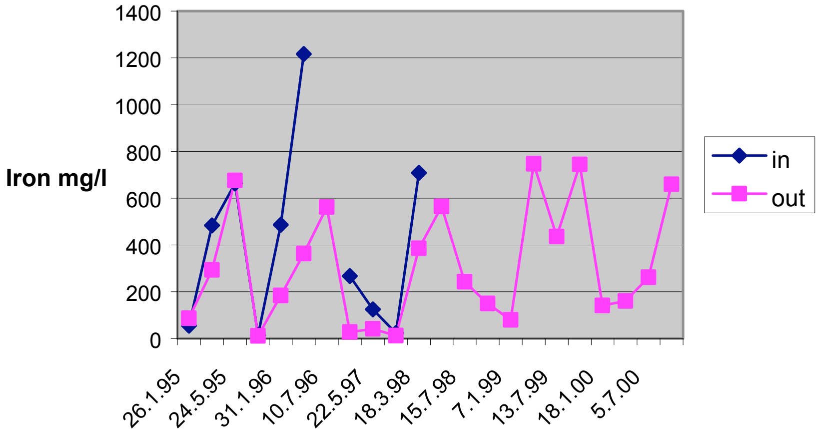

Figure 5.15.4. Iron concentration in inflow and outflow of Gilmerton wetland.

Examination of the average data presented in table 5.15.1 shows that treatment efficiency for Fe is low, which is confirmed by the plot of the whole data set shown in figure 5.15.4. However, the treatment efficiency can also be seen to be highly variable. Aluminium is not successfully removed – an average negative removal efficiency is derived from the raw data. Examination of the raw data, not presented here, shows that Al concentration in the outflow exceeded that in the outflow on five out of nine occasions.

It can thus be seen that the wetland at Gilmerton Bing is having very little positive impact on the minewater discharge. The highly contaminated nature of the minewater, and the high variability in water quality, would necessitate an entirely new scheme, possibly involving active treatment, if consistent and satisfactory treatment was required.

5.16 GWYNFI

NGR SS900968

Pre-treatment water chemistry/volume and treatment requirements

The minewater of the Gwynfi discharge is net alkaline, with a flow of up to 18 l/s, pH of 6 to 7 and an iron concentration of approximately 8 mg/l. The treatment requirement is to aerate the minewater to facilitate the oxidation and precipitation of iron.

Site history and design criteria

The Gwynfi minewater discharge is associated with the former Glyncorrwg and Corrwg Rhondda Colliery at Blaengwynfi. The discharge affects a 2 km length of the Nant Gwynfi, which is a tributary of the River Afan. There are two point sources of minewater, plus a further diffuse source (see figure 5.16.1). The topography of the area is hilly, with the point source discharges emitting from the side of a steep valley. The consequent limited land area available for reed bed creation has meant that the Nant Gwynfi had to be re-aligned as part of the construction work. The land on which the treatment scheme is situated has been acquired on a long-term lease from the Forestry Commission and the system is managed and maintained by the Coal Authority (Wardell Armstrong currently have the management contract).

The treatment system was designed to be a relatively simple means of substantially reducing the loading of iron entering the river while at the same time causing minimal disturbance to the environment. Flows are diverted from the Level B discharge to flow through the reed beds, and additional flow enters the reed beds from Point E (see figure 5.16.1). Discharge from reed bed 3 re-joins the untreated, by-pass flow before entering the realigned river.

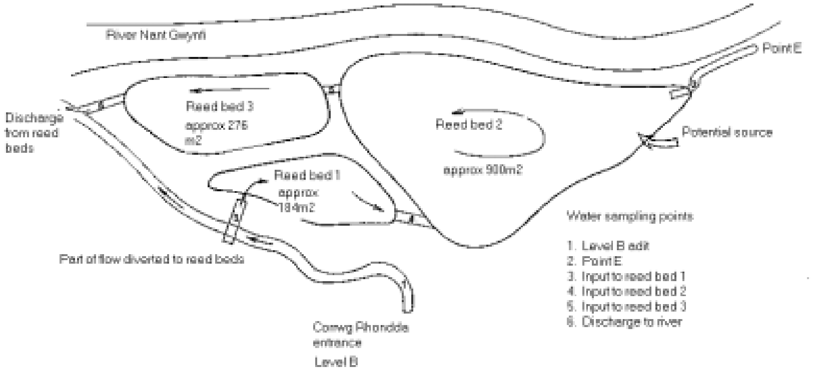

Figure 5.16.1. Schematic plan of Gwynfi reed bed.

Figure 5.16.2. Diversion of part of stream flow from level B to wetland.

Figure 5.16.3. Wetland area – input from level B is near the red lifebuoy, outflow point is off to the right of the picture. Note ferruginous staining of river from upstream, diffuse sources.

Figure 5.16.4. Outflow from reed bed 3 – by-pass flow from level B is to the right of the picture, the re-aligned river is to the left.

Monitoring data

Water quality monitoring is undertaken at Gwynfi on a monthly basis, at the sampling points shown in the schematic plan (points 1 to 6). At present no flow measurements are taken so it is not possible to carry out any calculations of treatment efficiency nor to access the loading of iron to the reed beds and the amount of iron retained.

Table 5.16.1 gives summary data for the period April 1999 to February 2001. The discharge Fe concentration was less than 0.6 mg/l on all but one occasion (a value of 7.63 mg/l was obtained in October 1999). If this one high value is discarded as an outlier, the revised average is 0.39 mg/l. A plot of the data, excluding the outlying value, is shown in figure 5.16.5. The rise in Fe concentration between input B and the input to bed 1 has been attributed to the different flow regimes at the respective sampling points, and revised instructions were issued to the site personnel carrying out sampling to try to minimise the discrepancies (Wardell Armstrong communication to Coal Authority).

Table 5.16.1 Summary of monthly data for Gwynfi for the period April 1999 to February 2001

n = 23	Input B		Input E		Input bed 1		Input bed 2		Input bed 3		Discharge	
	Fe mg/l	pH	Fe mg/l	pH	Fe mg/	pH	Fe mg/l	pH	Fe mg/l	pH	Fe mg/l	pH
Average	1.93	6.5	6.20	6.3	3.15	6.5	1.58	6.6	0.89	6.7	0.71	6.9
Max.	4.31	7.5	10.4	6.5	8.82	6.8	7.23	7.0	4.27	6.9	7.63	7.6
Min.	0.19	5.8	0.41	5.9	0.06	6.3	0.01	6.3	0.02	6.3	0.01	6.5

Figure 5.16.5 shows that a decrease in iron through the reed bed system occurs, representing a substantial improvement in water quality, although as mentioned above it is not possible to carry out further assessment of the data in terms of treatment efficiency, loading rates etc. due to the lack of flow monitoring data. Table 5.16.2 gives approximate figures for these values, based on the design flow rate of 10 l/s and taking the average water quality values of the most contaminated input (input E), although these can only be considered to be an approximation.

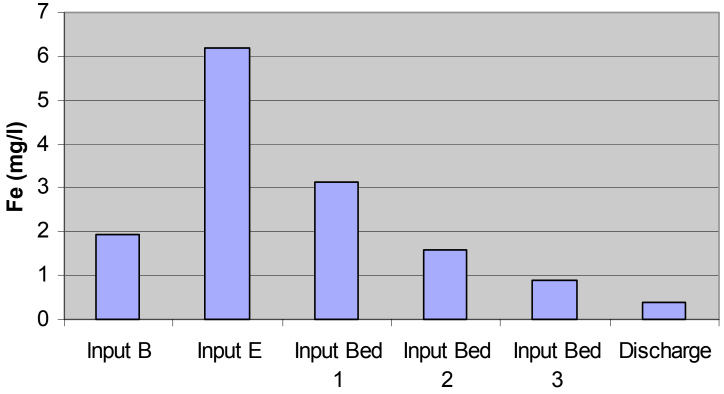

Figure 5.16.5. Plot of average iron concentrations through the Gwynfi reed bed system.

Table 5.16.2. Calculation of approximate values of performance, based on design flow of 10 l/s

n = 23	Average inflow mg/l	Average outflow mg/l*	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe	6.2	0.39	93.2	5.0	3.7
pH	6.3	6.9			

* excluding one outlying value.

Comments on design

The Gwynfi reed bed system has been designed to fit sympathetically into a confined space. There is insufficient land available to treat a larger flow of water and thus the river still has ferruginous staining, which is probably due to diffuse sources on the opposite bank of the river. The Coal Authority hope, in due course, to pipe more contaminated minewater into the wetland system if the wetland is considered to have the capacity. Successful treatment of water flowing through the reed bed system is achieved, with concentrations of Fe in the discharge of less than 1 mg/l.

The use of gabion baskets with locally sourced stone is visually pleasing, as are the informal shapes of the reed bed cells.

The use of terracing is successful for re-aeration of the minewater between the cells, and makes maximum use of the available land area, as well as being aesthetically pleasing.

Costs

Construction costs were £235,000, operating costs are £1,560 per annum.

5.17 KAMES, MUIRKIRK

NGR NS685262

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Kames is net alkaline, issuing from deep underground coal mine workings. There are two shafts; one is capped and has four pipes to allow outflow of water, and the other, a former air shaft, has been backfilled. The capping and pipes were installed by the Coal Authority a year or so before the main treatment scheme was constructed. Iron concentration in the minewater is in the region of 12 mg/l. The flow is up to 25 l/s – the treatment system has been designed to cope with a flow of up to 36 l/s.

Treatment of the minewater is by aeration and settlement, followed by surface flow aerobic wetland, with re-aeration between in-series wetland cells.

Site history and design criteria

The receiving watercourse (Garpel Water/River Ayr) has been impacted by minewater since the early 1970s. The current treatment system was designed by Babtie Environmental Ltd. and was constructed in 2000, with minewater being introduced to the system for the first time in September 2000.

Minewater flows from the capped mineshaft, down a stepped aeration cascade, to two settlement tanks, which operate in parallel. Outflow from the tanks remains in two separate streams, each of which passes into a series of two wetland cells. There is a concrete weir/cascade structure between each of the sets of two cells to facilitate re-aeration of the minewater. The outflows from the wetland cells recombine in a concrete channel and discharge to the receiving watercourse. Individual cells can be isolated for maintenance purposes.

The wetland cells were planted with *Typha* and *Phragmites*, but due to low water levels after planting, and perhaps also due to exposure, the cells have become colonised by Reed Canary Grass and the planted reeds have been effectively smothered. However, there are signs, as at August 2001, that the reeds are recovering. There are straw bunds within the cells to encourage utilisation of the full wetland area and a grassed bank separates the two parallel wetland systems.

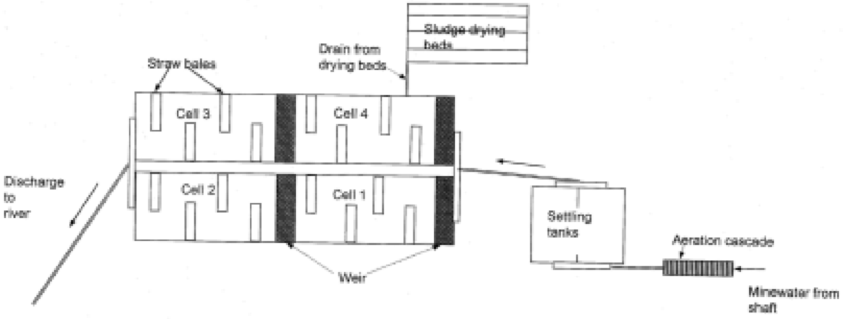

Figure 5.17.1. Schematic representation of the Kames treatment scheme.

Concrete sludge-drying beds have been constructed near the settlement tanks and it is anticipated that the tanks will require emptying annually. The system has been designed for a 20 year life.

Figure 5.17.2. Settlement tanks at Kames – two tanks in parallel. The stepped aeration channel is to the left of the picture and the fenced area beyond the tanks is the location of the shaft.

Figure 5.17.3. Between-cell re-aeration structure.

Figure 5.17.4. General view of the wetland cells at Kames, looking up towards the inflow point – the cascade is at the right of the picture, just in front of the cars. Top concrete structure is the sludge-drying beds, the other concrete structures as figure 5.17.2. Straw bales can be seen in the foreground.

Figure 5.17.5. Outflow to river.

Monitoring data

The system has been monitored since completion in September 2000, with an enforced break in monitoring in March and April 2001 due to the outbreak of foot and mouth disease. Data for the period October 2000 to June 2001 are presented in table 5.17.1. The data for the two parallel streams have been aggregated, and the intermediate sampling points, between the upper and lower wetland cells, which have been sampled four times, are considered over two different time periods as there is a marked change in the effluent quality in 2001 compared to 2000.

Data presented in table 5.17.1 show that very little Fe is being removed in the settling tanks (raw minewater has an average of 12.7 mg/l Fe and influent to the first wetland cells has an average of 12.4 mg/l Fe, which is equivalent to a daily retention of 0.4 kg/day). The reason for this is not clear, although perhaps the residence time of the aerated minewater within the settling tanks is insufficient; at an average flow of 16.7 l/s, and given a tank volume of 360 m^3, average retention time is calculated as 0.2 days.

Examination of the raw data (not presented here) shows that effective aeration and precipitation of Fe is achieved by the initial cascade, and that ferrous iron is mostly below detection limit in all samples except the raw minewater.

Table 5.17.1. Water quality data for the Kames minewater treatment scheme

Raw minewater: Fe 12.7 mg/l, pH 6.4

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
First cells					
Fe	12.4	7.6	38.7	6.9	1.7
pH	7.1	7.5			
Second cells (October and December 2000)					
Fe	8.7	4.7	46.0	5.8	1.4
pH	7.7	7.7			
Second cells (May and June 2001)					
Fe	6.5	0.2	96.9	9.1	2.3
pH	7.5	7.7			

Calculations of loadings based on an average flow of 16.7 l/s, n = 4.

The performance of the reed bed cells, particularly the lower two cells, shows indications of changing with time, but a longer monitoring period will be needed to confirm this. On the two sampling occasions in 2001, the lower two cells had a treatment efficiency, for Fe removal, of 96.9 %, as compared to 46% during October and December 2000. This could be indicative of the maturation of the wetland during early summer, with the increased plant growth providing an efficient means of settling out the particulate iron. This high treatment efficiency suggests that the dominant vegetation (Reed Canary Grass) forms an effective 'filtration mechanism' for particulate Fe. It thus seems that the 'invasion' by Reed Canary Grass and the temporary 'checking' of the planted species (reed growth was showing a resurgence in August 2001) should not be viewed as a problem. In fact, the self-colonised species may be more suitable to cope with the exposed conditions at this site.

Comments on design

The treatment system at Kames seems to be functioning well, although there are limited data currently available. The fact that most of the iron is being retained in the reed beds rather than in the settling tanks may present a problem long-term, in that it will be more difficult to remove accumulated ochre from the reed beds than from the settling tanks, and the resultant material will not have a potential market as it will not constitute 'pure' ochre.

Aesthetically, the reed beds and grassed banks form an attractive feature, although the concrete inflow and outflow structures are quite visually imposing, as are the settling tanks and sludge-drying beds. However, from a functional point of view the settling tanks and sludge-drying beds will be easy to clean out.

The Kames minewater no longer has an adverse effect on the river: monitoring data for upstream and downstream of the discharge point gives average values of 0.5 mg/l and 1.0 mg/l respectively. It is understood that fish have returned to the river since the treatment has been in place.

Costs

The construction costs at Kames were £622,000 and operating costs are £4,080 per annum.

5.18 KIMBLESWORTH, COUNTY DURHAM

NGR NZ260475

Pre-treatment water chemistry/volume and treatment requirements

Kimblesworth is one of nine pumping stations currently operated by the Coal Authority in County Durham in order to lower the water level. The pumped minewater is net alkaline; it has Fe concentration of between 0.82 and 3.47 mg/l (mean 1.87, n = 21) and manganese concentration of between 0.34 and 0.78 mg/l (mean 0.63, n = 21) (Jarvis & Younger 2001). The pumped minewater is currently discharged to a tributary of the River Wear, at a rate of approximately 9.1 Ml/day, and although the Fe concentrations are not particularly high, it nevertheless causes ochre staining of the river bed (Jarvis and Younger 2001).

An investigation was carried out by researchers at Newcastle University into the possibility of using a high surface-area media reactor as an alternative to the conventional treatment for net-alkaline minewaters (aeration and settlement in tanks and constructed wetlands).

Site history and design criteria

Two pilot-scale reactors were constructed at the Kimblesworth pumping station and were continuously fed with minewater for the period April 1999 to August 1999. The reactors were constructed from pre-cast concrete, with metal framework and a ladder to provide access to the top of the reactors. The general layout and dimensions of the reactors is shown in figure 5.18.1. The high surface-area media used were commercially available plastic trickling filter media, which are widely used in the water industry. Since these media are thoroughly characterised in terms of their specific surface area, porosity etc. they are very suitable for use in an experimental situation. Reactor A has a larger media, with an approximate surface area of 124 m^2; reactor B has a smaller media with a total surface area of approximately 178 m^2. Flow of minewater into the reactors was through a 20 mm internal diameter PVC flexible pipe with ball valves to control flow at the influent points to the reactors.

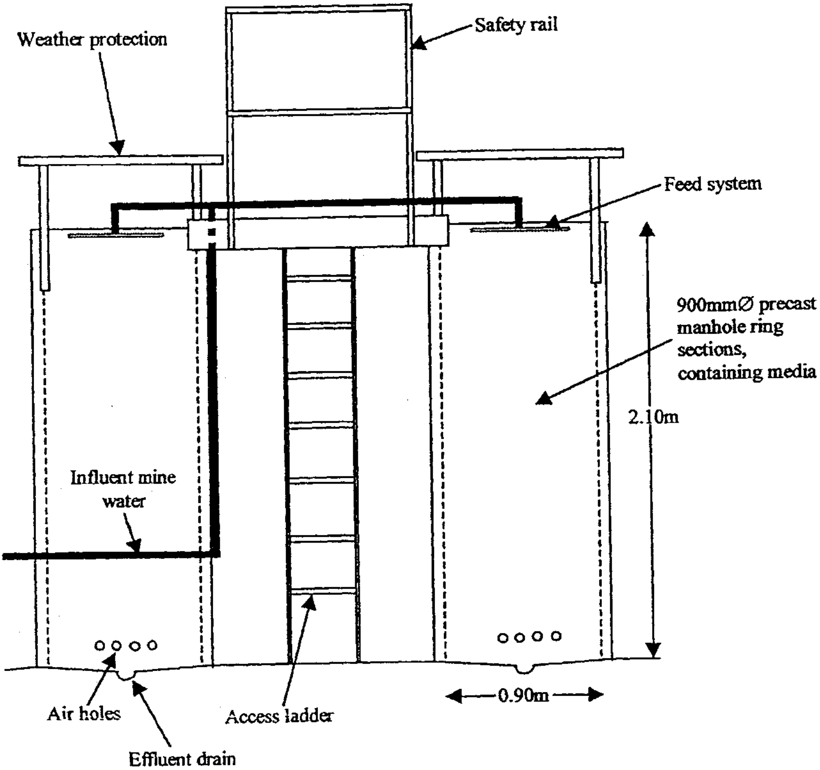

Figure 5.18.1. Schematic diagram of the pilot-scale reactors at Kimblesworth (from Jarvis and Younger 2001).

Monitoring data

Table 5.18.1 presents summary data for the experimental period from which it can be seen that both reactors achieve a significant reduction in iron and manganese concentrations in the minewater.

Table 5.18.1. Water quality data for pilot-scale treatment scheme at Kimbleworth

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained g/day	Volume adjusted removal g/m^3.day
Fe reactor A (n = 16)	1.43	0.41	71.3	4.9	3.0 mean and 10.8 at highest loading rate
Fe reactor B (n = 16)	1.43	0.38	73.4	4.2	2.6 mean and 12.3 at highest loading rate
Mn reactor A (n= 16)	0.67	0.53	20.9	0.7	0.4
Mn reactor B (n = 16)	0.67	0.47	29.9	0.8	0.5
pH (n = 17)	6.95				
Alkalinity (n = 16)	758				
Sulphate (n = 19)	308				

Data from Jarvis and Younger 2001. Calculations of retention based on average flows of 3.34 l/min to Reactor A and 2.80 l/min to reactor B.

Discussion of the results presented in Jarvis & Younger (2000) notes that iron removal rates increase linearly with loading rates up to 14 g/day, and that at low loading rates (0.5 to 0.9 g/day) up to 50% of Mn was removed. Reactor B, with the higher surface area, performs significantly better than Reactor A in terms of both Fe and Mn removal. Residence time in the reactors is as little as 70 sec. The combination of short residence time and efficient removal of Fe at low influent concentrations (approximately 1.5 mg/l) strongly suggests that these reactors may be an effective passive alternative to aerobic wetlands, particularly at sites where topography and/or land area are restrictive (Jarvis & Younger 2001).

Comments on design

This was an investigation into an innovative means of removing iron and manganese from a net-alkaline minewater which might be useful in instances where the more conventional means, such as aeration cascades, settlement tanks and aerobic wetlands, are not appropriate. Research work is continuing to investigate other media and to derive design criteria for optimisation of such reactors.

There is no information available on the longevity of the reactors, for example, how quickly they become blocked by ochre accretion, how the media is regenerated/removed, how saturated media is disposed of.

5.19 MAINS OF BLAIRINGONE, CLACKMANNAN, FIFE

NGR NS980973

Pre-treatment water chemistry/volume and treatment requirements

The minewater issuing from the drift adit at Mains of Blairingone is net alkaline and ferruginous. The iron concentration in the minewater has declined since 1994 when the problem with the discharge was first noticed: at that time the Fe peaked at 118 mg/l. During the period September 1997 to January 2001, the average Fe concentration has been 12.5 mg/l and the average pH of the discharge is 6.9. The flow was monitored at intervals for the period April 1994 to September 1997 and averaged 7.8 l/s, with a range of 0.09 to 17.0 l/s (n = 35).

The treatment requirement of the minewater is aeration to oxidise the ferrous iron and settlement to allow the precipitated ferric iron to settle out of solution.

Site history and design criteria

In February 1994 a public complaint alerted the Forth River Purification Board to a pollution incident on the River Devon, which was at the time a Class I watercourse designated as a Salmonid Fishery. Examination of the river revealed a stretch of about 2 km, below the confluence with a tributary draining a 19th century drift mine adit at Mains of Blairingone Farm, was stained with ochre precipitate. The tributary itself was severely affected by ferruginous deposits for a length of 0.8 km and supported no invertebrate life. As a result of this pollution, the River Devon was downgraded to a Class II watercourse (Roxburgh & Marsden 1997).

Investigation of the local coal mining, geology and hydrogeology revealed that the origin of the ferruginous minewater outbreak may have been associated with the operational opencast coal mine site at Lambhill, to the south-east of Blairingone. Although the operators of the opencast site, British Coal, would not accept responsibility for the outbreak, negotiations led to the formation of a treatment system using a natural wetland area, in collaboration with the landowner. An area of approximately 6.2 ha, that already supported wetland vegetation, was converted to a treatment area by the construction of bunds. The affected tributary was diverted to the wetland area by a new pipe, capable of carrying twice the maximum recorded flow. A cascade was incorporated at the inlet to the wetland to aid aeration of the minewater, and a deeper area of water was created at the diagonally opposite outlet end of the wetland. Installation of a v-notch meant that flows could be recorded.

The discharge is not subject to a discharge consent, but treatment has proved to be very successful, with the River Devon now reinstated as a Class I river, with no visible iron-staining.

Various academic studies have been carried out on the wetland area. In particular an MSc student at Stirling University carried out an investigation to assess the potential for the longer-term remobilisation of iron from the wetland (thesis entitled 'The dynamics of iron and manganese storage in a wetland constructed to treat mine drainage in the Devon Catchment'). This work concluded that the high iron content within the sediment is mostly held in a mineral form and is therefore not likely to be mobilised and released into the water. A BSc dissertation studied the performance of the wetland during a storm event (dissertation entitled 'The effectiveness of a constructed wetland in removing iron from mine drainage during rainfall events').

Figure 5.19.1. The inflow cascade to the Mains of Blairingone treatment wetland.

Figure 5.19.2. General view of the wetland area, showing areas of deeper water, and species diversity.

Figure 5.19.3. Outflow from the Mains of Blairingone treatment wetland.

Monitoring data

The raw minewater has been monitored since 1994, and the outflow from the wetland has been monitored since the minewater was diverted through the wetland in 1995. Figure 5.19.4 shows graphical representation of this data, from which can be seen the decrease in iron concentration in the minewater over time.

The consistently high degree of treatment afforded by the wetland can also be seen, with only 2 occasions when there was a high concentration of Fe in the effluent from the wetland (January 1997 and October 1998). In the October 1998 sample the suspended solids concentration was a factor of ten higher than usual, so it seems likely that some ochre was disturbed during sampling, giving an abnormally high value for Fe. The reason for the high Fe concentration in January 1997 (when it was over 40 mg/l) is not clear.

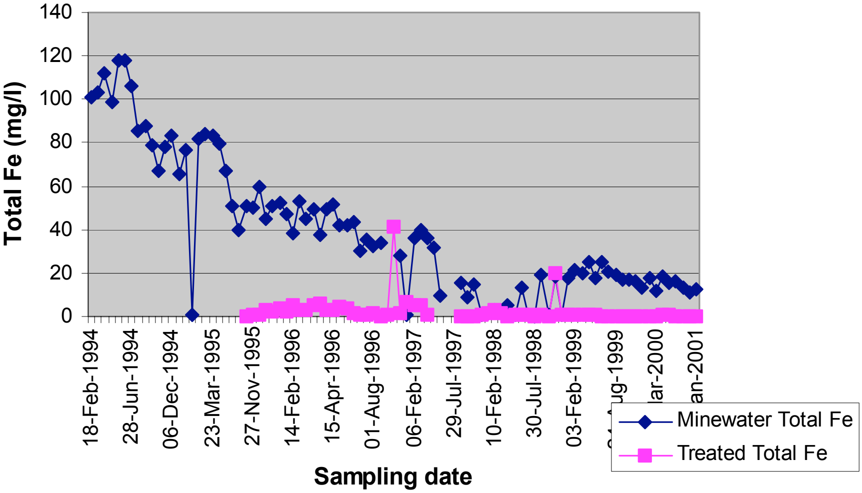

Figure 5.19.4. Graph showing change of Fe concentration in raw minewater with time, and Fe in treated minewater.

In order to give a measure of treatment performance in recent times data collected since September 1997 have been used, after which time the decrease in Fe concentration seems to have levelled off somewhat. These data are summarised in table 519.1.

Examination of table 5.19.1 reveals that the treatment efficiency afforded by the wetland is high. The figure for area adjusted removal can only be considered to be an approximation, since the exact area of the wetland is not known, and a study of flow paths revealed that only about one third of the wetland actually receives minewater flow due to the formation of preferred pathways for water flow across the wetland. Thus the calculation of area adjusted removal has been made based on an area of approximately 2 ha, but it is not known how appropriate this figure is as an indication of the area of wetland actively involved in water quality improvement.

Table 5.19.1. Water quality data for the treatment wetland at Mains of Blairingone for the period September 1997 to January 2001

n = 35	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
pH	6.9	7.4			
Fe (total)	12.5	0.5	96.0	8.1	0.4
Fe (sol)	11.3	0.1			
Alkalinity	102	95			
Aluminium	1.1	0.2	81.8	0.6	0.03

Peak Fe concentration (in 1994) was 118 mg/l. Outlying value of 20.2 mg/l Fe in treated minewater in October 1998 rejected from data set.

Comments on design

This wetland has very high aesthetic and ecological value, and is also working very efficiently. It must be noted that the area of the wetland is large in relation to the loading of contaminants, principally iron, that it is receiving. However, this is a good example of the desirability, where possible, of creating as large an area of wetland as possible/feasible if the aesthetics and ecology are deemed to be important. This wetland also demonstrates that it is ecologically acceptable to use a natural wetland for treating minewater in certain instances. Even where it is not possible to use pre-existing wetland areas, or where there are constraints on size, certain features of this wetland, such as the use of marginal planting (for example of willows and alders) and the incorporation of areas of deeper water could be used in constructed wetlands to increase their aesthetic and ecological value.

Costs

It has not been possible to determine the costs of this scheme, but they can be assumed to be relatively low, both in terms of construction and maintenance.

5.20 MINTO, FIFE

NGR NT205950

Pre-treatment water chemistry/volume and treatment requirements

The minewater issuing from the former mineworkings at Minto Colliery is net alkaline and flows from an old shaft. Iron concentration in the minewater has an average value of 11.0 mg/l with a maximum of 18.0 mg/l. Flows generally vary from 15 to 45 l/s, with a maximum of 60 to 75 l/s and an average of 37.8 l/s.

The treatment requirement of this minewater is aeration and settlement.

Site history and design criteria

Minewater discharge issuing from a shaft of the former Minto Colliery had a highly visible and biological impact on the River Ore due to the precipitation of ochre on the river bed. A feasibility study commissioned by the Coal Authority concluded that a passive treatment system would successfully remediate the minewater and 3.5 acres of land was purchased from Fife Council. The scheme was constructed between October 1997 and July 1998.

Minewater issuing from the mineshaft is diverted to an aeration cascade, piped for a distance of about 20 m and split into two streams in a valve chamber. Flow enters two parallel reed bed cells via valve arrangements, is recombined at the outflow from these cells, then split into two again to pass through two further parallel reed bed cells. The cells are planted with *Typha*, and the water depth can be altered by adding plastic pipe to alter the height of the outflow points. Problems with plant establishment were experienced initially due to flooding of the cells being caused by blocking of the perforated outfall pipe. Replanting was carried out during 2000 and the outfall pipe arrangement was changed to overcome the problems of blocking and flooding.

Figure 5.20.1. Minewater aeration cascade below issue point – the mineshaft is within the metal-fenced area.

Figure 5.20.2. Inlet valve to reed bed.

Figure 5.20.3. General view of reed bed cell showing inlet valves.

Monitoring data

The water quality at Minto is monitored on a monthly basis at the shaft emission and the outflow from the treatment system. Data for 1999 to 2000, and separate data for 2001, are presented in table 5.20.1.

Table 5.20.1. Water quality data for Minto constructed wetland for the period February 1999 to September 2000, and for two sampling occasions in 2001

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe total n = 22	11.0	4.3	60.9	34.7 at 60 l/s 21.9 at 37.8 l/s	3.5 at 60 l/s 2.2 at 37.8 l/s
Fe sol n = 15	3.1	0.0			
Fe total Aug/Sept 2001, n = 2	13.9	0.54	96.1	69.3 at 60 l/s 49.6 at 37.8 l/s	6.9 at 60 l/s 4.4 at 37.8 l/s
PH n = 19	6.8	7.7			
Alkalinity n = 3	670	620			
Sulphate n = 3	1370	1330			

Consideration of table 5.20.1 reveals that, although the concentration of Fe was being decreased by passage through the constructed wetland, the efficiency during 1999 and 2000 was not as high as other similar systems achieve, and the outflow still had an average concentration in excess of 4 mg/l. The soluble Fe in the outflow was generally below detection limit; this means that it is particulate Fe, that had failed to settle and be retained in the reed bed cells, that was exiting the system. The reason for this could be the relatively high average hydraulic loading to the system (between 0.33 and 0.52 m/day at flows of 37.8 and 60 l/s respectively) and low average retention time (approximately 9 to 14 h). These operating conditions may not have allowed sufficient time for all the particulate Fe to settle out of suspension. Also, the initially poor growth of plants due to flooding problems would have meant the particulate matter was less likely to be retained within the wetland cells.

The marked improvement in treatment efficiency on the two sampling occasions for which data are available in 2001 may be due to improved plant establishment and growth as the wetland matured, with a denser covering of vegetation giving better retention of particulate Fe in the wetland cells. Ongoing monitoring will reveal if this improvement in treatment efficiency is sustained, which will be a strong indication of the importance of dense, healthy and mature plant growth within minewater treatment systems.

Comments on design

The aeration cascade, which is really just an open channel, seems to be effective at introducing air into the minewater. There have reportedly been blocking problems with the pipe that takes minewater into the wetland after the cascade and an open channel for the whole distance might have been preferable.

The inlet valves, while allowing further aeration, seem to be unnecessarily complex and liable to become blocked with ochre.

The plant growth seems to be rather poor in some of the cells, which may be due to higher than ideal water levels in the cells due to flooding. Initial planting of a more diverse species mix might have helped to overcome this problem as different species have different tolerance of water depth. However, the outlet pipe arrangement was altered in 1999/2000 to overcome flooding problems.

Costs

The capital costs of this scheme were £260,000, and operating costs are £2,460 per annum.

5.21 MONKTONHALL, MID LOTHIAN

NGR NT323703

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Monktonhall is alkaline, ferruginous deep minewater pumped from the shaft. Pumping is carried out to keep water level in the shaft at about 100 m down; pumped volumes vary seasonally, but the average pumped volume is 98 l/s. Two pumps are installed and pumping with one or both is carried out for as many hours per day as necessary to maintain the required water level in the shaft. Iron concentration in the minewater is around 50 mg/l.

Treatment requirement for alkaline minewater is usually aeration and settlement, but due to the large volumes to be treated at Monktonhall, an active treatment is used (lime and flocculant addition) prior to settlement in lagoons followed by reed beds.

Site history and design criteria

The current treatment system at Monktonhall was completed in July 1998. Minewater is pumped from the shaft to a treatment building, where lime is added via a screw hopper and flocculant is added as a trickle from a pipe.

Pre-treated minewater flows to two settling lagoons, in parallel, and from these to a collection chamber, where the water is pumped to two reed bed cells, also in parallel. The reed bed cells are planted with *Typha,* with *Scirpus* planted near the outlets. Outlet from the reed beds cells is via a concrete channel structure and there is a long pipe to the discharge point.

Four drying beds, constructed of shale and clinker, have been constructed for future use.

Figure 5.21.1. Lime dosing system, showing dosing tank and screw hopper for introduction of lime. The pipe in the wall of the tank is the outflow to the settling tanks.

Figure 5.21.2. Settling lagoon, showing inlet pipe. The fence around the pumping station and shaft area can be seen in the middle distance, beyond the lagoon.

Figure 5.21.3. Reed bed cell, showing inlet pipe arrangement.

Figure 5.21.4. Outlet from reed bed cell.

Monitoring data

Tables 5.21.1 and 5.21.2 present data for the Monktonhall treatment system. It can be seen that the system works very effectively; the reed beds show a high degree of efficiency, despite high hydraulic loads (1.7 m/day) and low retention time (0.2 days). The reasons for this are not clear, although the following could be factors:

- Addition of flocculant to the minewater improves the settling characteristics of the precipitated solids, making them less fine and more likely to be retained in the reed beds.

- The very limited data set (with only one set of figures for reed bed influent and effluent water quality) may not give the true treatment efficiency.
- Calculations involving flow have been carried out using a value of 98 l/s, which is quoted as the average pumped volume. However, it is probable that the pumped volume is often less than this (at the time of the site visit in July 2001, it was approximately 25 l/s) and so calculated hydraulic loading rates are unrealistically high and retention times low.
- Pumping is only carried out for a few hours per day, as necessary. Therefore the considerable periods each day when there is little or no flow through the system will give good conditions for the suspended solids to settle.

Table 5.21.1. Water quality data for Monktonhall treatment system (whole system)

	Average inflow mg/l n = 2	Average outflowmg/l n = 4	Treatment efficiency %	Retained kg/day
Fe	50.9	3.8	92.5	399
pH	6.8	7.7		
Alkalinity	447	352		
Sulphate	1400	1420		

Table 5.21.2. Water quality data for reed beds

n = 1	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe	12.4	0.66	94.7	99.4	19.6
pH	7.7	7.8			
Alkalinity	352	333			
Sulphate	1420	1440			

Comments on design

The treatment system at Monktonhall treats a large volume of minewater effectively. The reliance on pumps means that breakdowns can cause problems; it is understood that the small pumps in the collection chamber between the settling lagoons and the reed beds do not always perform adequately, which causes backing up of water in the settling lagoons and in the treatment tank. There are insufficient data to carry out reliable calculations regarding treatment performance.

Costs

The construction costs at Monktonhall were £297,000 and operating costs are £97,260 per annum.

5.22 NAILSTONE COLLIERY, LEICESTERSHIRE

NGR SK430085

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Nailstone Colliery is acidic spoil heap leachate with high concentrations of iron and aluminium. The flows are highly variable with direct rainfall effects. High flows often have higher contaminant concentrations rather than there being a dilution effect.

The treatment requirements are to raise the pH and settle out the metals. Also site remediation would help to lessen the generation of acidic leachate: there are current proposals to cap the exposed spoil on the site with a 1500 mm thick cap of mercia mudstone which would effectively isolate the acid-generating spoil from rainfall.

Site history and design criteria

Nailstone Colliery closed in the 1980s. The current site is approximately 50 ha in extent and comprises exposed spoil heaps (approximately 20 ha) and capped, but unrestored, settling bays. Subsequent to closure of the colliery there was some coal-washing activity on the site.

There are currently three discharge consents in place relating to run-off from the site, which related to the treatment regime that has subsequently been modified. This original scheme comprised lagoons that were filled with a 0.5 m depth of limestone and manure mixture, through which the water flowed, to have the same effect as a type of SAPS. There were two lagoons on the northern boundary and one open ditch followed by a lagoon on the southern boundary. This system successfully raised the pH by one or two units (with influent pH sometimes being as low as pH 2), but was only really effective when flows were low; when rainfall gave increased site run-off, with increased concentrations of contaminants, the system was unable to cope.

The system has now been modified to bring as much of the runoff as possible to the southern boundary of the site, and the discharge point has been moved downstream to effectively give one discharge rather than the previous three. Balancing lagoons have been installed in the centre of the site, with valves to control the flow, in an attempt to regulate the flow of water that needs to be treated.

Figure 5.22.1. SAPS in foreground, balancing pond beyond

Figure 5.22.2. Water accumulating from the spoil heap.

Figure 5.22.3. Dosing system for sodium carbonate.

Sodium carbonate is added, to raise the pH, at about six places on the site. A ‘low tech’ approach to this addition was used as a temporary measure, but it seems to be working well so there are currently no plans to upgrade it to a more automated system. The current approach is to add granular Na_2CO_3 by suspending hessian bags in catch pits (see figure 5.22.3), which comprise plastic drums with 300 mm pipework taking water into and out of the drum. The water then flows to settlement lagoon to allow the precipitated solids to settle out of suspension.

The dosing stations need to be inspected twice a day, which is carried out by locally based labour.

Monitoring data

No monitoring data are currently available.

5.23 NENTHEAD, CUMBRIA

NGR NY781436

Pre-treatment water chemistry/volume and treatment requirements

There are numerous minewater discharges in the Nent Valley, in the North Pennine Orefield, which are net alkaline and have elevated concentrations of zinc, in the region of 3–8 mg/l. The minewaters also contain lead, cadmium and arsenic. The treatment requirement is thus to provide suitable conditions to cause the precipitation of Zn, and preferably the other contaminants as well, in order to protect the river from these ecotoxicological elements.

Site history and design criteria

The Nent Valley has experienced over two centuries of intensive mining for lead and zinc and the legacy of this is diffuse and point source pollution of the River Nent; there are five disused mine adits that discharge to the river and also runoff from spoil heaps and contaminated land in the valley. The minewater is hard and net alkaline due to interaction with calcareous carboniferous rocks. Although lime dosing would be a suitable means of treating the minewater, it would be preferable to find a passive method that would not incur the ongoing costs of active chemical addition. It was not considered that a compost wetland would be suitable for removing Zn from net alkaline minewater (Nuttall & Younger 2000a and 2000b) and research was carried out at Newcastle University to find an alternative passive treatment method.

The treatment method was selected on the basis that Zn forms a stable carbonate mineral (smithsonite, $ZnCO_3$) over a narrow pH range (approximately pH 7.5 to 8.0). Geochemical modelling was carried out to determine the range of conditions that would need to be created in order to favour the precipitation of smithsonite, and thus the removal of Zn from solution. It was found that the optimum pH would be pH 8.2. In order to raise the pH to this value, it was decided to use an anoxic limestone drain (ALD), since the closed-system conditions in such a reactor allow higher pH to be obtained than in an equivalent open system (Nuttall & Younger 2000a and 2000b, Freeze & Cherry 1979).

Laboratory testing of the concept was carried out using a sealed 5 litre polyethylene reaction vessel to simulate an ALD, with a constant feed of raw minewater. The system was run from the end of March 1998 to the beginning of October 1998, a total of 105 days.

Data obtained from the laboratory test work were used to design a pilot-scale field reactor to treat a small proportion of the minewater discharging from one of the abandoned mines (Caplecleugh). The reactor comprised a double thickness 2 m × 5 m heavy duty polyethylene bag into which was placed 4 tonnes of limestone gravel. Pipes were installed to act as inflow and outflow channels, a tarpaulin was placed over the top and the bag was covered with material previously removed from the site (Nuttall & Younger 2000a and 2000b). This reactor was monitored regularly and ran for a period of approximately 3 months.

Monitoring data

Tables 5.23.1 and 5.23.2 present average data from the laboratory and pilot-scale tests.

Table 5.23.1. Average water quality data for the laboratory tests

n = 10	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained g/day
Zn	5.54	2.70	51	0.013

Table 5.23.2. Average water quality data for the pilot-scale reactor at Nenthead

n = 8	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained g/day
Zn	7.33	5.76	21	2.5
pH (target pH adjustment)	7.5	8.2		

The difference in performance between the laboratory and pilot-scale tests (51% and 21% treatment efficiency, respectively) has been surmised as being due to the difference in temperature between the laboratory (20°C) and the field (≤ 10°C), and the difference in limestone particle size between the laboratory system (10–15 mm) and the field system (35 mm) (Nuttall & Younger 2000a and 2000b).

This was an experimental approach that showed promising results, and work is continuing on refining and improving the treatment potential of such a system.

5.24 OATLANDS, PICA, CUMBRIA

NGR NX021213

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Oatlands is acidic spoil heap drainage. The flow is in the region of 1 to 2 l/s, although exact data are not available.

Treatment requirement is to add alkalinity and precipitate out the metals.

Site history and design criteria

The wetland treatment system at Oatlands was designed by ARM Ltd for the Environment Agency. The spoil heap drainage water collects in a ditch which is vegetated with *Typha* and *Juncus*. At the time of the site visit (August 2001) the ditch had a thick accumulation of ochre and in several places flow in the ditch was overtopping and decanting into the stream (River Keekle).

Figure 5.24.1. Oatlands wetland area when newly planted (photo courtesy of ARM Ltd).

At the end of the ditch the flow is collected into a plastic pipe system that distributes it to three discharge points into the reed bed, which is a monoculture of *Typha*. There is a single discharge point from the reed bed.

The Environment Agency have now returned the reed bed to the land's former owners (a local farmer) on the understanding that remedial works will be undertaken on the spoil heap by its owners (Cumbria County Council), which should entail capping the spoil heap, thereby tackling the problem at source.

Monitoring data

Monitoring data have been supplied by the EA and are presented in table 5.24.1.

Table 5.24.1. Water quality data for Oatlands wetland for the period 11.12.1997 to 22.2.2001 (monthly readings)

n=38	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %
Fe (total)	53.4	10.0	81.3
Al	6.2	5.0	
Sulphate	911	796	
pH	5.0	3.9	

Alkalinity readings are also taken and are generally below the detection limit of 5 mg/l, although the inlet value has an average value of 46.4 mg/l from sixteen readings (the other 22 values were below detection limit) and the outlet has an average value of 26.5 mg/l from eight readings (with 30 values being below detection limit).

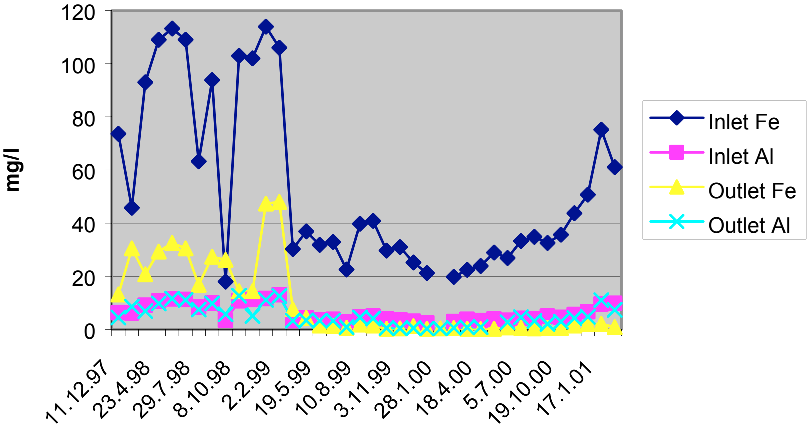

Figure 5.24.2. Iron and aluminium data for Oatlands wetland.

There has also been considerable temporal variation, particularly in Fe and Al concentrations, which is shown graphically in figure 5.24.2. It can be seen from figure 5.24.2 that since April 1999 there has been a substantial decrease in the concentration of Fe in the water flowing into the wetland and very low concentrations of Fe in the outflow. Inlet and outlet Al concentrations have also decreased since this date. The reason for the abrupt change in water quality is not known, although there is an indication that concentrations were rising again during 2000 and into 2001. Treatment efficiency for Fe, however, remains good, despite the increase in inflow Fe concentrations.

5.25 OLD MEADOWS, BACUP, LANCASHIRE

NGR SD867238

Pre-treatment water chemistry/volume and treatment requirements

The minewater outflow at Old Meadows is a drainage sough into workings from a shaft mine which emits from an adit on the banks of the River Irwell. Prior to treatment being implemented there was a visual impact, due to iron ochre staining, for a distance of 5 to 6 km downstream. The minewater contains average concentrations of 30 mg/l iron and 3 mg/l manganese. The average flow from the adit is 49 l/s.

The treatment requirements are to add alkalinity and precipitate out the iron and manganese.

Site history and design criteria

The treatment scheme at Old Meadows was constructed in 1998. There was no suitable land available near the adit except for approximately 3 ha of privately owned agricultural land on the sides of the valley. The steep hillside gradients caused design and construction difficulties, and made it harder to integrate the scheme into the landscape. The general layout of the site is shown schematically in figure 5.25.1.

The Old Meadows adit used to have to be cleared of ochre by boat. The adit is now sealed so such cleaning is no longer possible and there are periodic flushes of ochre from the underground workings. Water flows along two gravity mains from the adit to a pumphouse a short distance downstream. This pipeline does not tend to have problems of ochre accretion, presumably because most of the ochre is in solution and the dissolved oxygen in the minewater is low. There is a storage sump of approximately 8000 litres capacity below the pumphouse to allow for pumping ‘down-time’ during maintenance operations. From the pumphouse the water is pumped up the hillside via a 150 mm rising main to the caustic soda dosing plant. There are two pumps, each with a 40 l/s pumping capacity. The flow is monitored continuously by computer which can be remotely accessed by the operating contractors. The rising pipeline has to be jet-cleaned annually due to ochre build-up within the pipe.

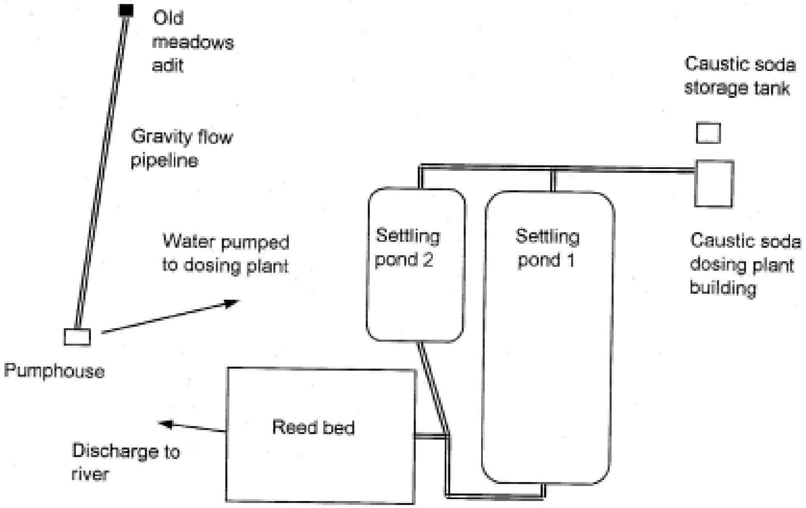

Figure 5.25.1. Schematic layout of the Old Meadows treatment scheme.

Caustic soda (20% solution) is added from a bulk tank, outside the treatment building, to a caustic dosing tank, which is followed by a reaction tank. The amount of caustic soda added is adjusted automatically depending on the pH of the raw minewater, and the pH is raised to about pH 7. The dosing tank has a capacity of 1 m^3, and there are four pumps. Approximately 1 tonne of caustic soda is used each day, and there needs to be a delivery of the 20% caustic solution by tanker about once a week (less in summer when flows and therefore caustic requirements are lower).

The pre-treated minewater then enters the settling lagoons. There are two lagoons, in parallel, with the second lagoon being about half the size of the first one. Currently most of the flow goes to the first lagoon – only a small flow is passed to the second lagoon to keep the pipes from blocking. The lagoons are clay lined. The sludge that is removed from the lagoons will be taken to sludge drying beds at Deerplay, which is further up the valley.

Figure 5.25.2. Inflow to first settling lagoon at Old Meadows, showing ochre build-up.

Figure 5.25.3. View of first settling lagoon, with the active dosing plant housed in the building on the right.

Figure 5.25.4. Inlet to reed bed cell.

Figure 5.25.5. Outflow from reed bed cell.

Flow from the settling tanks then enters the reed bed cell via a concrete distribution channel. The reed bed was lined with clay and planted with a mixture of *Typha*, *Juncus, Iris* and *Scirpus*. Outflow from the reed bed is into a concrete channel, from where the treated water flows down a pipe to be discharge just below the pumphouse.

Monitoring data

The raw minewater and treated effluent at Old Meadows have been monitored on a weekly basis since the scheme's completion in 1999, and intermediate points of the system, i.e. inlets and outlets to individual component parts, have been monitored monthly. Summary data of the whole system monitoring is presented in table 5.25.1, of the reed bed in table 5.25.2, and figure 5.25.6 shows the data from intermediate monitoring points graphically.

Table 5.25.1. Water quality data for the Old Meadows treatment scheme: whole treatment system, for the period June 1999 to October 2000

n = 41 (n = 63 for Fe)	Average inflow mg/l	Average outflow g/l	Treatment efficiency %	Retained kg/day
Fe (total)	30.7	1.3	95.8	124.5
Fe (sol)	13	0.3		
pH	6.3	7.8		
Mn	3.0	1.8	40.0	5.3
Alkalinity	42	121		
Acidity	47	1.8		
Sulphate	515	503		

Table 5.25.2. Water quality data for the Old Meadows treatment scheme: reed bed, for the period March to August 2000

n = 6	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (total)	1.7	1.0	41.2	3.0	1.6
Fe (sol)	0	0			
pH	7.6	7.9			
Mn (n = 2)	2.6	2.5	4	0.4	0.2
Alkalinity	150	150			
Acidity	4.5	1.7			
Sulphate	500	500			

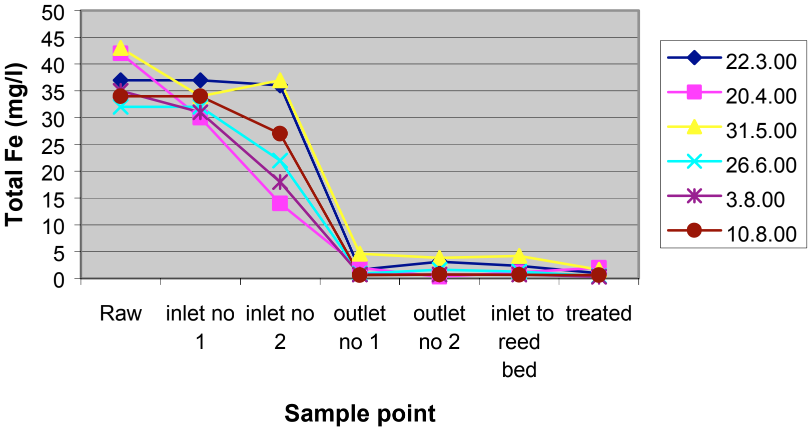

Figure 5.25.6. Monitoring data from intermediate sampling points.

Consideration of the data presented in tables 5.25.1 and 5.25.2 shows that the treatment system works effectively, with Fe removal efficiency being on average greater than 95%. Figure 5.25.6 shows that most of the Fe is removed in the first settling lagoon and that the reed bed is acting as a final polishing step.

Comments on design

The treatment system at Old Meadows is effective. The fact that no land was available in the valley floor means that ongoing pumping costs are unavoidable, and the net acid nature of the minewater means that an active scheme is needed to achieve satisfactory treatment. The reed bed acts as a final polishing mechanism, and provides wildlife habitat – there are reported to be Canada geese and ducks that use the reed bed in the spring.

The steep approach road to the site makes access for the delivery tankers difficult in severe winter weather, which calls for careful planning to avoid requiring deliveries in the severest weather.

Costs

The construction costs at Old Meadows were £949,000 and the operating costs are £132,696 per annum.

5.26 PELENNA, SOUTH WALES

NGR	Whitworth 1	SS798968
	Garth Tonmawr	SS815971
	Phase III	SS799972

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Pelenna is acidic, from deep and drift mines. There are six discharges, of which five require treatment. The River Pelenna, a tributary to the River Afan that enters the sea at Port Talbot, has been affected by minewater discharges for more than 25 years leading to severe discoloration of approximately 7 km length of the river by iron and poor survival of fish and other river life. Two tributaries to the River Pelenna are affected – the Nant Gwennffrwd and the Blaenpelenna. There are four discharges affecting the former and two affecting the latter (see figure 5.26.1). The total flow from all six discharges is in the region of 34 l/s and represents a total loading of iron into the river of 93 kg/day (Ranson *et al.* 1998).

Treatment requirements of the minewater are to raise the pH and remove the iron. The techniques used demonstrate an evolution of methods used to passively treat acidic minewater as ideas have changed and knowledge has increased.

Site history and design criteria

The problems of minewater impact on the River Pelenna have long been recognised but the situation gained greater attention in the late 1980s as production in the South Wales coalfield declined. Survey work funded by the BOC Foundation for the Environment carried out in 1992 identified the six separate discharges for which treatment was considered and of which it was decided to treat five. A feasibility study commissioned by the NRA in 1993 concluded that passive treatment would be appropriate, using a range of techniques.

Funding was obtained from a variety of sources, including the European Union LIFE fund, the Welsh Development Agency and the BOC Foundation for the Environment. The project has been managed in partnership by Neath Port Talbot County Borough Council and the Environment Agency.

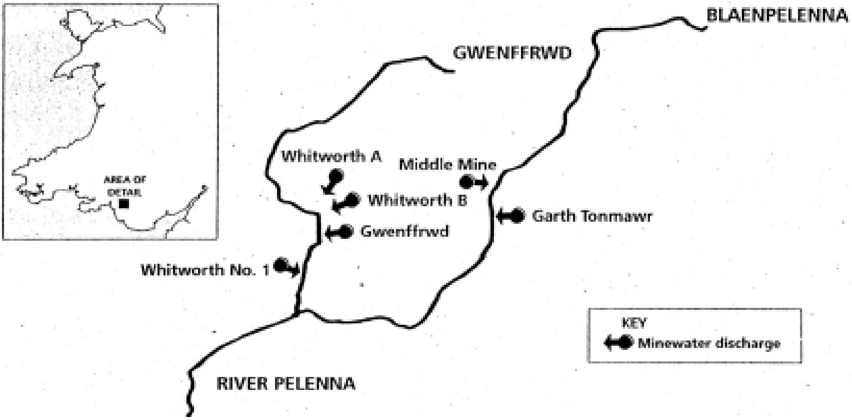

Figure 5.26.1. Location plan of the Pelenna discharges (from Edwards *et al.* 1997).

A requirement of the LIFE funding was that the remediation work should constitute a demonstration project of European significance, to show the effectiveness of wetland treatment as a low cost, sustainable means of improving watercourses contaminated by coal mine effluent, to assess and develop opportunities to enhance the conservation aspects and to disseminate information on the project (Ranson *et al.* 1998).

The development of the treatment systems at Pelenna was planned to be phased over a period of 5 years. Work started in January 1994 and the final stage was completed in August 1999. The demonstration role of the project was reinforced by incorporating lessons learned in each phase into subsequent phases.

Associated and additional work

In addition to the main restoration project at Pelenna, the Environment Agency and its predecessors have carried out or initiated several projects, including:

- fisheries, biological and water quality studies (Edwards 1995);
- acidification studies (Acer Environmental Ltd 1995);
- sediment studies (Edwards & Maidens 1995);
- investigations into natural wetlands (Collins 1995);
- ecological/conservation studies (Pryce Consultant Ecologists 1994, Roberts 1996);
- minewater treatment computer aided design package (James *et al.* 1997).

There are also numerous undergraduate and postgraduate studies that have been carried out, or are in process, studying such aspects as the sediments, ecological surveys, sources of alkalinity generation and modelling. Additional studies have been carried out, mostly by graduate students, on the Whitworth 1 system. These include detailed monitoring of the biological and chemical processes occurring in one of the wetland cells (cell 3) as a PhD project at Leeds University; a study of enzyme activity around root systems in cell 4 as a PhD project at the University of Wales, Bangor; analysis of substrate composition in July 1996, as part of an MSc project at the University of Wales, Aberystwyth.

Whitworth 1

Phase I of the wetland development was constructed at the outflow from the East End abandoned colliery, known as Whitworth 1. The flow to be treated was 3 to 4 l/s, with an iron concentration of approximately 20 mg/l. A number of demonstration/experimental features were incorporated: surface flow wetlands, subsurface flow wetlands, different substrate types (mushroom compost and bark mulch) and different plant species (nursery grown *Typha* and locally sourced *Juncus*).

The wetland comprises four cells, with flow configuration, substrate type and plant species as shown in the schematic diagram (figure 5.26.2)

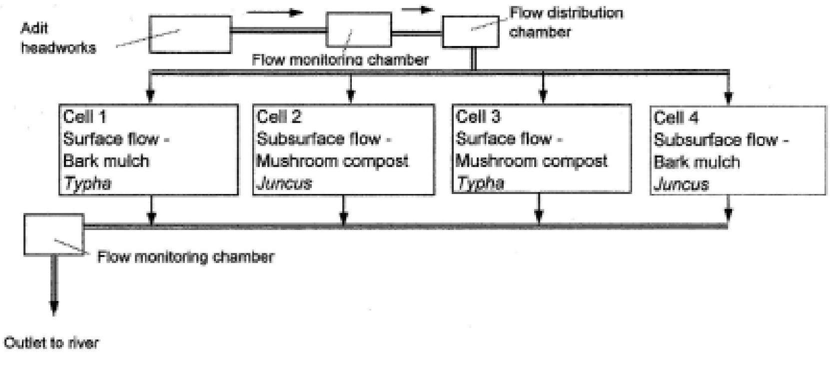

Figure 5.26.2. Schematic diagram of the Whitworth 1 treatment scheme.

The wetland cells are constructed from pre-cast concrete with a geosynthetic base liner (bentonite sandwich). Flow into the cells is by gravity from the adit, via a flow monitoring chamber and a flow distribution chamber. Flow into the wetland cells is via an open channel with periodic openings allowing flow onto gabion baskets across the width of the cell (see figure 5.26.3). A four-way split using decanting columns divides the flow between four such channels, theoretically to provide equal flow to each cell, although this has not proved to be very successful. In fact cell 4 has received most of the flow on occasions.

The subsurface flow *Juncus* beds have baffles that are designed to force water down through the substrate, but they have proved to be insufficiently high to achieve the desired aim.

The Phase I system has performed satisfactorily overall, but some problems have been experienced with flow distribution to the wetland cells, blockages, uneven flow distribution and uneven performance in terms of iron removal.

Figure 5.26.3. Flow distribution system of Whitworth 1 wetland, showing open channel and gabion baskets.

Figure 5.26.4. Whitworth 1 wetland, showing flow distribution channels for cells (the water level in the cell in the foreground is higher than it should be).

Figure 5.26.5. Outflow structure – there are two of these valve regulators for each cell.

Monitoring data

Monitoring the performance of the Whitworth 1 treatment system has been carried out on a monthly basis since October 1995. The results for the period October 1995 to June 2001 are presented in table 5.26.1. The calculations of treatment efficiency, retention and area adjusted removal assume a flow of 3 l/s, evenly distributed between the 4 cells, although this ideal situation is rarely achieved due to problems of flow distribution. In reality cell 4 receives more flow than the other cells.

Statistical analysis of the whole data set, from 1995 to 2001, shows that at a significance level of 5%, Fe concentrations in the outflow from cell 1 are significantly lower than from cells 2 and 3, and that those from cell 4 are significantly less than from cell 2. Thus cells 1 and 4 appear to be performing better than cells 2 and 3. These two cells have a common substrate – bark mulch – but different flow regimes and vegetation. However, in reality, due to uneven flow distribution, cell 4 receives more flow than the other cells, and cell 1 probably receives less flow than the other three cells. If the area adjusted removal is recalculated assuming that cell 4 receives one third of the total flow (1 l/s) and cell 1 receives one sixth of the total flow (0.5 l/s), the respective values are 7.9 g/m^2/day for cell 4 and 3.9 g/m^2/day for cell 1. This indicates that cell 4 performs the most efficiently and that cell 1 is perhaps the least efficient. Cell 4 comprises subsurface flow through bark mulch, with *Juncus* vegetation.

Table 5.26.1. Monitoring data for the Whitworth 1 treatment scheme for the period October 1995 to June 2001

n = 42	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Cell 1 pH	6.0	6.1			
Cell 1 Fe	24.2	3.96	83.6	1.3	5.8
Cell 2 pH	6.0	5.9			
Cell 2 Fe	24.2	6.24	74.2	1.2	5.2
Cell 3 pH	6.0	6.0			
Cell 3 Fe	24.2	4.4	81.8	1.3	5.7
Cell 4 pH	6.0	6.1			
Cell 4 Fe	24.2	3.52	85.5	1.3	6.0
Whole system pH	6.0	6.1			
Whole system Fe (total)	24.2	4.3	82.2	5.2	5.7
Whole system Fe (dissolved)	22.5	3.3			
Whole system Mn	1.79	1.15	35.8	0.2	0.2
Whole system sulphate	338	292			

The above discussion serves to illustrate the difficulty of carrying out accurate comparison of wetland configurations at field scale due to practical problems such as flow distribution. At best, an indication of the optimal configuration can be obtained, but detailed statistical analysis of results is not relevant if ideal flow conditions are not achieved.

Comments on design

The main problems encountered were with flow distribution to the cells: deposition of iron oxyhydroxides in the pipes entering the system caused blockages, resulting in uneven flow distribution and precluding an accurate assessment of the relative performance of the four cells (Edwards *et al.* 1997). In extreme circumstances, a significant amount of untreated minewater by-passes the treatment system via the overflow. Recent maintenance has been carried out to try to rectify these problems, and the monitoring regime has been recently changed to measure flows for individual cells so that future assessment can be made of the performance of individual cells.

Spent mushroom compost was used, presumably, because it is the substrate most often used in constructed wetlands for coal mine drainage in the USA. However, it is not a readily available substrate in South Wales and subsequent phases took this into consideration when substrates were selected (Younger 1998a).

This treatment system was one of the first to be implemented in the UK and it was intended primarily for experimental purposes. Hence little attempt was made to blend it into the landscape, as has been increasingly the practice with more recent constructed wetland sites.

This site has now been receiving minewater for 6 years, and as such is a valuable site to continue studying since it may yield information on the longevity of constructed wetlands for minewater treatment; continued monitoring will reveal whether treatment efficiency improves, diminishes or remains stable with time. Certainly from the first 6 years' of data, treatment seems to have maintained the same level of efficiency throughout this period.

On a site visit in July 2001 it was noted that a number of Alder seedlings were growing in the reed bed cells, and it was noted that the current maintenance policy is to leave these rather than to try to remove them.

Whitworth A & B and Gwenffrwd

The treatment system for three discharges in the Gwenffrwd Valley is referred to as Phase III, although in reality it was completed before Phase II, the Garth Tonmawr system.

In the design of Phase III, particular attention was paid to minimising the impact of the scheme on the landscape. This was easier to achieve in Phase III than in Phases I and II due to less significant engineering constraints and greater land availability for Phase III. Thus, the layout and construction of the scheme has a more natural appearance, minimising the use of concrete and forming more natural shapes. The conceptual design was carried out by Paul Younger of Newcastle University and incorporated the concept of Successive Alkalinity Producing Systems (SAPS) that had been pioneered in the USA by Kepler & McCleary (1994).

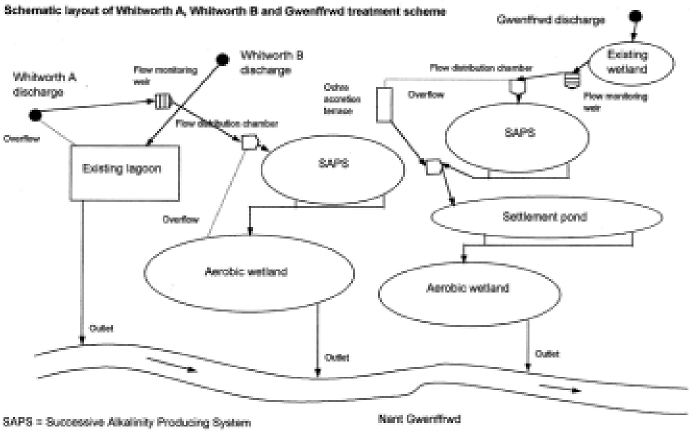

Figure 5.26.6. Layout of Pelenna Phase III treatment system.

Phase III essentially comprises three parallel systems (figure 5.26.6, Younger 1998):

- The existing Whitworth Lagoon was left to treat Whitworth B discharge alone, by aerobic wetland processes.
- Whitworth A discharge is treated by passing through a SAPS followed by an aerobic wetland before discharge to the river.
- Gwenffrwd discharge passes through an existing wetland area, through a SAPS, with overflow going to an ochre accretion terrace, through a settlement pond and finally an aerobic wetland before discharge to the river.

Banks and bunds throughout are reinforced with geotextiles to ensure slope stability while avoiding the need for excessive use of concrete. Because of the large flows and variable chemistry of Whitworth A and Gwenffrwd discharges, provision was made in the design to allow easy construction of interconnections between the two respective treatment systems at various points, should these ever be deemed desirable. In fact, soon after the system was first commissioned (in April 1998), the Gwenffrwd discharge diverted underground (in December 1998/January 1999) and flowed to the Whitworth B lagoon, joining with the Whitworth B discharge. This problem has only recently been rectified (in March 2001) by redirecting the Gwenffryd discharge, via an open channel, to the intended Gwenffryd treatment system.

Figure 5.26.7. Whitworth B settlement lagoon.

By-pass arrangements were incorporated to deal with high flows, when it would not be possible to achieve the desired 14 h retention time in the SAPS (as recommended by Hedin *et al.* 1994) for all the minewater. It was considered preferable to ensure any minewater passing through the SAPS would have a 14 h retention time, and would then mix with the untreated minewater that had bypassed the SAPS, hopefully introducing sufficient alkalinity to achieve adequate treatment. This remixing of treated and untreated minewater occurs in the aerobic wetland of the Whitworth A system and the settlement pond of the Gwenffrwd system.

Figure 5.26.8. Whitworth A SAPS, showing distribution channel and ochre build-up on the surface.

The ochre accretion terraces were incorporated into the Gwenffrwd system as a means of removing some of the ochre before the overflow minewater enters the settlement pond and aerobic wetland.

The aerobic wetland area made use of the flood plain of the river, although to increase retention time and to allow freeboard for ochre build-up, channel-edge bunds were constructed. Some small, pre-existing wetland areas within the flood plain were left intact to allow natural expansion of the existing flora into the newly flooded, bunded areas.

Figure 5.26.9. Aerobic wetland area adjacent to river.

Figure 5.26.10. Gwennffrwd system ochre accretion terrace.

Figure 5.26.11. Gwenffrwd system SAPS. Note less ochre build-up on surface due to lack of minewater flow to this system for the period end 1998 to early 2001/.

Figure 5.26.12. Gwenffrwd system settlement pond.

Monitoring data

Due to the problems with flow management, in particular the fact that the Gwenffrwd discharge diverted underground soon after the system was commissioned (system was commissioned in April 1998, diversion occurred in December 1998/January 1999) and flowed instead to the Whitworth B lagoon, there are only really reliable data for the Whitworth A system. A one-off sampling, carried out in 1998, before the Gwenffrwd underground diversion occurred, for inflows and outflows to each system, is given in table 5.26.2. The average flows reported in the same paper as these monitoring data (Younger 1998a) are:

Whitworth A	8.3 l/s
Whitworth B	0.6 l/s
Gwenffrwd	8.9 l/s

These values are used to calculate the net retention of Fe in the systems given in table 5.26.2.

The performance of the Whitworth A system can be assessed using Environment Agency monitoring data that has been collected since the system was commissioned in April 1998. Sampling was carried out at approximately fortnightly intervals until January 2000, and then at monthly intervals. These data are summarised in table 5.26.3. The average flow used for the calculations is 9.2 l/s (n = 71).

Table 5.26.2. Spot sampling for the Pelenna Phase III treatment systems, carried out in 1998 (Younger 1998a)

n = 1		Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
Whitworth A	Fe total	81.7	0.16	99.8	58.5
	Fe sol	81.7	0.09		
	pH	6.0	7.2		
Whitworth B	Fe total	5.38	1.46	72.9	0.2
	Fe sol	5.38	0.11		
	pH	6.4	6.6		
Gwenffrwd	Fe total	13.1	0.94	92.8	9.4
	Fe sol	12.9	0.21		
	pH	5.0	7.2		

Table 5.26.3 shows that the Whitworth A system is performing well for iron removal. Aluminium concentrations are very low (average 0.016 mg/l) and therefore problems of blockage due to Al precipitation should not be encountered, although, as discussed below, accumulation of oxidised Fe on the surface of the SAPS may have a detrimental effect on performance. Aluminium concentration in the combined Gwenffrwd/Whitworth B discharge is higher (average 1.5 mg/l for Gwenffrwd and 0.3 mg/l for Whitworth B) and therefore might cause problems with blocking the SAPS.

Table 5.26.3. Whitworth A treatment system monitoring data, April 1998 to June 2001

	Average inflow mg/l (n = 114)	Average outflow mg/l (n = 26 for SAPS, 50 for final discharge)	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
SAPS					
Fe	59.7	8.5	85.8	36.7	20.1
Mn	0.85	1.0			
Sulphate	334	327			
pH	5.8	6.6			
Wetland					
Fe	8.5	1.7*	80.0	5.4	1.2
Mn	1.0	0.5			
Sulphate	327	297			
pH	6.6	6.6			

* excluding one outlying value of 28 mg/l recorded in July 1998.

Iron removal within the treatment systems was shown to be good when spot sampling was carried out in 1998 (average 90% removal, Dey & Williams 2000), although the removal mechanisms are not those that were intended. The intended treatment mechanisms of a SAPS are to provide extra alkalinity to buffer minewater as precipitation and hydrolysis of iron causes a fall in pH, and to promote sulphate reduction, thereby enhancing the removal of metals as sulphide. However, work carried out by Dey and Williams (2000) suggests that sulphate reduction is not occurring; iron is being removed by aerobic precipitation of oxyhydroxides rather than by anaerobic precipitation of sulphides. The precipitation of iron oxyhydroxides has an autocatalytic effect, thereby promoting further iron removal, but this has adverse effects on the SAPS, in effect causing 'blinding' of the surface with iron precipitates and limiting infiltration of minewater through the substrate. Preliminary work (Dey & Williams 2000) suggests that the SAPS substrate could become effectively impermeable in as little as 4 years. An alternative undesirable scenario proposed by these workers is that the iron sludge surface will become exposed, giving the appearance of a large ochre pad posing a dust problem on dry days.

Comments on design

Deposition of ochre on the surface of the Whitworth A SAPS has been identified as a problem. If there had been sufficient land area, it would have been advantageous to have a settlement pond before the SAPS to allow Fe precipitation and accumulation prior to introducing the minewater to the SAPS.

In addition to the observations made above regarding the deposition of ochre, and the deleterious effect this might have on the longevity of the SAPS systems, there have been problems with pipe blockages causing system failure and the ingress of untreated minewater to the river. These pollution incidents, ironically, have probably been all the more noticeable to members of the public because the river has become so much cleaner as a result of the ongoing remediation works. (It was reported on the Environment Agency website in September 1999 that trout had returned to the Gwenffrwd tributary of the River Pelenna after more than 30 years, and that there were also improvements to the Blaenpelenna downstream of the Garth Tonmawr treatment scheme).

A problem with some of the pipe blocking incidents is that access was needed to the flow distribution chambers to clear the blockage, and only personnel with 'working in confined spaces' certification were able to carry out this work. The delay in rectification of the problem is an indication that such items as site maintenance health and safety issues, as well as simplicity of design, should be fully considered at the design stage in an attempt to foresee and prevent such problems in the smooth operating of the scheme.

Another problem that became apparent in discussing the Pelenna project with various members of the project team is that, while funding was acquired for the design and implementation of the schemes, insufficient allowance was made for ongoing monitoring and maintenance. This precluded the opportunity of modifying the scheme to optimise its operation, or even of carrying out sufficient routine maintenance, which is particularly vital in the early stage of establishing a 'passive' treatment system.

Garth Tonmawr

Phase II of the Pelenna scheme was completed in August 1999. This scheme comprises a series of wetland cells of different types based on the same principles as SAPS (see figure 5.26.13). The Garth Tonmawr minewater discharge is only slightly acidic, but it still requires added alkalinity to buffer the fall in pH that accompanies the precipitation and hydrolysis of iron.

The treatment scheme is designed to accept a flow of up to 20 l/s, with overflow going direct to the river. The iron concentration is in the region of 30 mg/l, giving a loading of about 50 kg/day Fe.

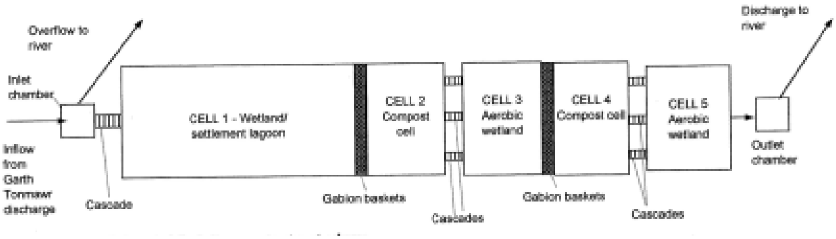

Figure 5.26.13. Layout of Garth Tonmawr treatment scheme.

There are five cells in series which have the following composition and function:

Cell 1: aerobic wetland and settlement lagoon, which has become naturally colonised with wetland vegetation. The function of this cell is to facilitate aeration of the minewater with resultant precipitation and settling of iron (see Figure 5.26.14);

Cell 2: compost cell with underlying limestone layer. The function of this cell is to strip out oxygen, by downward flow of the minewater through the compost layer, and to add alkalinity by flow through the underlying limestone layer. Oxygen stripping is necessary to prevent precipitation of iron solids within the limestone layer and subsequent armouring of the limestone. The design hydraulics for this cell were for the water level to be at the surface of the compost, but impeded flow through the compost has meant that the cells are flooded to the full depth of the retaining walls;

Cell 3: aerobic wetland, to facilitate precipitation of iron hydroxide/oxyhydroxide, with alkalinity gained from the previous cell providing buffering and thus preventing a drop in pH;

Cell 4: compost cell with underlying limestone layer, as cell 2;

Cell 5: aerobic wetland, as cell 3.

Figure 5.26.14. Garth Tonmawr – Cell 1, aerobic wetland and settlement lagoon, with natural colonisation.

Figure 5.26.15. Garth Tonmawr – cell 5, aerobic wetland in the foreground, looking ‘upstream’. Flow from cell 4 to cell 5 is occurring over the top of the dividing wall, rather than through the pipes through the wall, at the top of the cascades, as intended

Monitoring data

Monitoring started at the Garth Tonmawr treatment site in March 1999. The data are presented below in two summary tables. Table 5.26.4 shows the Fe loading to each cell and the calculated values of treatment efficiency, Fe retention and area adjusted removal based on an average flow of 22.6 l/s, and table 5.26.5 shows the performance of each cell in terms of other parameters in addition to Fe which allows some interpretation of how the system is functioning.

Table 5.26.4. Summary data for Fe and pH for the period March 1999 to June 2001

n = 27 for Fe, 17 for pH		Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Cell 1	Fe	28.8	11.2	61.0	34.3	13.8
	pH	5.42	4.3			
Cell 2	Fe	11.2	3.5	68.8	15.1	15.6
	pH	4.3	6.5			
Cell 3	Fe	3.5	3.1	12.9	0.9	0.9
	pH	6.5	6.2			
Cell 4	Fe	3.1	0.8	74.8	4.5	4.5
	pH	6.2	6.6			
Cell 5	Fe	0.8	2.0	-158.4	-2.4	-2.5
	pH	6.6	6.5			
Whole system	Fe	28.8	2.0	93.1	52.4	8.2
	pH	5.4	6.5			

Table 5.26.5. Summary data for pH, Fe, Mn, Al, sulphate and alkalinity for the period March 1999 to June 2001

n = 17 for pH, 27 for other parameters	Cell 1 inlet mg/l	Cell 1 outlet mg/l	Cell 2 outlet mg/l	Cell 2 over-flow* mg/l	Cell 3 outlet mg/l	Cell 4 outlet mg/l	Cell 4 over-flow* mg/l	Cell 5 outlet mg/l
pH	5.4	4.3	6.5	3.8	6.2	6.6	6.5	6.5
Fe	28.81	11.23	3.50	0.34	3.05	0.77	0.22	1.99
Mn	0.66	0.64	0.60	0.63	0.63	0.43	0.40	0.52
Al	0.31	0.19	0.05	0.25	0.12	0.01	0.10	0.07
Sulphate	258	245	264	230	259	267	238	264
Alkalinity	30.0	10.8	353	0	27	57	34	37

* overflow is the flow over the walls comprising surface water runoff from cell 2 to 3 and cell 4 to 5, monitoring has been carried out since November 2000, n = 8.

The data presented in tables 5.26.4 and 5.26.5 show that the Garth Tonmawr system is functioning well as a minewater treatment system, even though it is not strictly functioning as intended: the compost cells are meant to be flooded only to the top of the substrate, with all flow occurring through the substrate, rather than the current situation in which they are flooded right up to the top of the retaining walls, with a proportion of flow occurring over the top of the walls. In order to further study the current mode of operation, the Environment Agency is now monitoring the flows over the top of the walls as well as through the substrate.

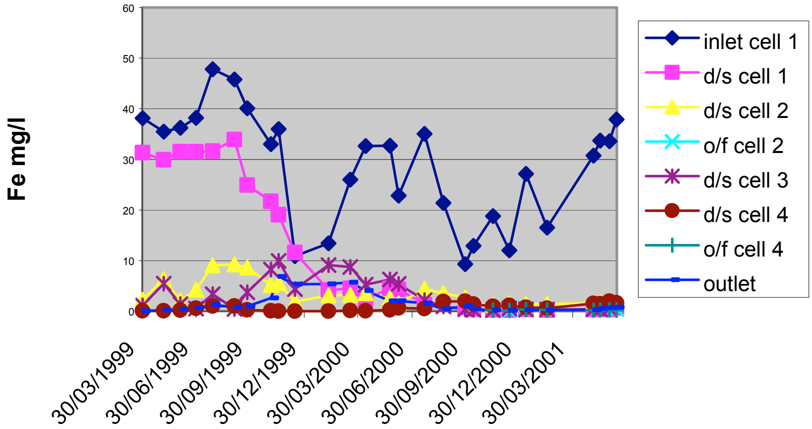

Figure 5.26.16. Plot of Fe concentrations through the Garth Tonmawr treatment system. (d/s = outlet from cell. o/f = overflow over the wall between cells).

Table 5.26.4 shows the overall treatment efficiency of the system to be over 90%. It can be seen, however, that some cells are working better than others. Cell 1 is retaining 65% of the Fe, based on these average figures, and the subsequent aerobic cells are performing less satisfactorily. Cell 3 is retaining only a small proportion of the Fe (less than 2%) and cell 5 is actually releasing more Fe than it is receiving, thereby acting as a net source of Fe rather than a sink. This indicates that Fe that has previously been deposited in cell 5 is being remobilised, although the reason for this is not clear. Examination of the raw data provided by the EA, not presented in this report, shows that an average of 80% of the Fe in the outflow from cell 5 is in solution, so re-suspension of precipitated iron hydroxides is not the cause of this net source effect.

Examination of figure 5.26.16 reveals that there was a period between September 1999 and July 2000 when final outlet Fe concentrations (i.e. outflow from cell 5) exceeded those in the inflow to cell 5. During this period there are also other cells acting as net source of Fe, for example the outflow from cell 3 has a higher Fe concentration than the outflow from cell 2. However, since this time, final outlet concentrations have been consistently low. This might suggest that the apparent source-effect noticed in cell 5 on examination of average data, is in effect more likely to be a temporary effect occurring during the establishment phase of the system. Indeed, since July 2000, removal of Fe appears to have occurred predominantly in cell 1. Ongoing monitoring will reveal if this is the steady state of the system.

Table 5.26.5 allows some assessment to be made of the way in which the system is functioning. In cell 1, the first aerobic cell, over half the Fe precipitates out, with an accompanying drop in pH that can be expected due to the hydrolysis reactions (equation 1.3 in chapter 1). Some Al also precipitates out in cell 1. The outflow from cell 2 has a considerably higher pH and alkalinity than the inflow, which indicates that the compost and limestone layers are functioning as anticipated, as a way of adding alkalinity. Iron and Al are also removed in cell 2, presumably retained within the compost or limestone layers. The overflow from cell 2 has low alkalinity and pH, indicating that this is surface flow water from cell 1 that has effectively by-passed the treatment capacity of cell 2. The iron concentration in the overflow, however, is low, which indicates that further oxidation and hydrolysis of Fe has occurred in the surface flow water column of cell 2, which causes a further drop in pH. This precipitation of Fe from the surface flow might have implications for the longevity of the system, as accumulation of ochre on the surface of the compost may decrease the amount of flow through the compost, thereby hindering the alkalinity-addition mechanism.

In cell 3, the second aerobic cell, there is not a great deal of change in water quality except that the alkalinity added by cell 2 is nearly all removed. The second compost/limestone cell, cell 4, strips out further Fe and Al, although the alkalinity is not raised as much as by the first compost/limestone cell. The overflow from cell 4 suggests that some further aerobic precipitation of Fe has occurred in the water column. Cell 5, the final aerobic cell, shows signs of mobilisation of particulate Fe, as discussed above.

Comments on design

When this system was first commissioned, treatment efficiency was good – in excess of 98% removal of iron. However, the compost cells soon became flooded, so that much of the flow from the compost cells to the following aerobic wetland cell occurs by overflow over the top of the retaining wall, rather than through the pipes at the top of the cascades, as intended. Thus, flow through the compost and limestone layer, and the desired function of the compost cells, is not being fully achieved.

The reasons for the lack of flow through the compost are not known, although the following are possible explanations:

- iron concentration and loading are higher than anticipated;
- precipitated iron is blinding the surface of the compost;
- the underlying limestone is becoming blocked;
- three pipes between the cells are insufficient to allow adequate flow.

Costs

Capital costs for the various stages of the Pelenna treatment scheme were:

Whitworth 1 £214,314;
Garth Tonmawr £223,534;
Phase III £360,198.

5.27 POLKEMMET, WEST LOTHIAN

NGR NS933640

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Polkemmet has iron concentrations in the region of 40 to 50 mg/l and a pH of between pH 6 and 7. The current pumping capacity is 68 l/s. Treatment is carried out by a combination of active and passive methods to ensure consistent quality is achieved.

Site history and design criteria

Polkemmet Colliery closed in 1984 and pumping finally ceased at the pit in 1986. Monitoring in the area alerted the Scottish Environment Protection Agency (SEPA) and the Coal Authority that rising minewater within the former colliery workings was a major cause for concern and the site was targeted by SEPA as their priority site for action in Scotland. The Coal Authority commissioned a feasibility study and it was concluded that minewater could break out along coal outcrops during early 1998, posing a potential pollution hazard to the River Almond near Bathgate. An urgent contingency plan was implemented by the Coal Authority, working in conjunction with West Lothian Council and SEPA, which comprised pumping from Polkemmet shaft, chemically treating the water and discharging up to 100 l/s into the Cultrig Burn. This original system at Polkemmet comprised a cascade and settlement lagoon followed by a serpentine ‘coagulant’ channel and two further settlement lagoons in parallel. This system was modified in 1999 and now comprises a combination of active chemical dosing and settlement followed by polishing in a wetland area. The new scheme involved the construction of two new settlement lagoons and the incorporation of a wetland area. The layout of the two treatment schemes is shown schematically in figures 5.27.1 and 5.27.2.

Minewater is pumped from the No. 1 shaft at Polkemmet by one 144 HP, 415v, nominal 900 gpm (68 l/s) pump to ensure water level in the shaft is maintained at such a height as to prevent surface discharges occurring in the Mosside area into the River Almond. The pumped minewater is injected with 35% hydrogen peroxide solution and piped to newly constructed, clay-lined settlement lagoons. The water is split into two equal streams and a polymer flocculant is added in the flow-splitting chamber (the flocculant powder is mixed with water in a screw-fed hopper housed in the chemical dosing building and piped to the inlet to the settlement lagoons). There are four inlet pipes to each of the lagoons and six outlet pipes. Minewater then flows to a secondary lagoon (which was the main lagoon of the old system), from where it is piped to a wetland area approximately 200 m away. Most of the settlement of precipitated Fe seems to occur in the first two lagoons, and very little accumulation of Fe-solids is observed in the secondary lagoon.

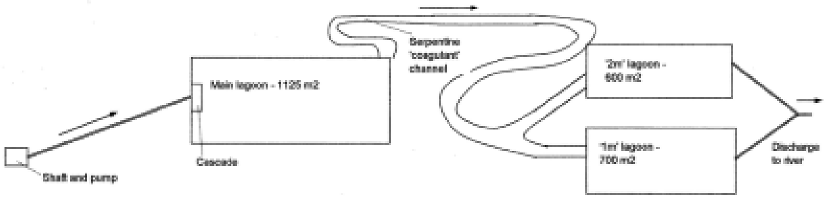

Figure 5.27.1. Polkemmet – old treatment scheme.

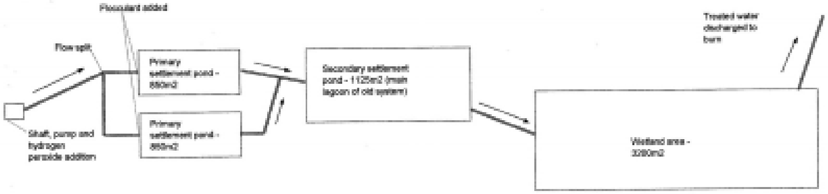

Figure 5.27.2. Polkemmet – new treatment scheme.

The wetland area was an area of pre-existing wetland which has been incorporated into the treatment scheme by the construction of bunds, of colliery spoil, around the perimeter of the wetland. The vegetation comprises principally *Typha* and *Juncus*, with alders along the margins.

The 'coagulant' channel and the two secondary lagoons of the original system are now used in the modified system for sludge storage.

Figure 5.27.3. Inflow zone to primary settlement lagoon, showing flow-splitting chamber and ochre accumulation.

Figure 5.27.4. Secondary lagoon of current treatment system (this was the main lagoon of the original treatment system – note disused aeration cascade).

Figure 5.27.5. Sludge-drying/storage lagoon (which was one of the secondary lagoons of the original system).

Figure 5.27.6. Wetland area incorporated as polishing mechanism into Polkemmet treatment scheme.

Monitoring data

The treatment scheme has been monitored since it was implemented in 1998. Data are available for the old system for the period January 1999 to November 1999, and for the new system from November 1999 to October 2000. Data for the old system include results from monitoring at intermediate points through the system but only data for raw minewater and inflow/outflow of the wetland area are available for the new system.

Table 5.27.1. Monitoring data for the old treatment scheme at Polkemmet for the period January to November 1999

n = 36	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Main lagoon Fe tot	33.1	20.9	36.9	71.7	63.7
Main lagoon Fe sol	4.1	0.6			
pH	6.6				
‘1m’ lagoon Fe tot	11.7	2.5	78.6	27.0	38.6
‘1m’ lagoon Fe sol	1.7	0.1			
‘2m’ lagoon Fe tot	11.7	2.6	77.8	26.7	44.6
‘2m’ lagoon Fe sol	1.7	0.1			
Final outflow pH	7.2				
Fe in burn upstream of discharge	2.4				
Fe in burn downstream of discharge	2.4				

Examination of table 5.27.1 reveals that iron in the inflow to the main lagoon of the old system was primarily in particulate form, and that the outflow has only 0.6 mg/l Fe in solution. Thus effective aeration was being achieved and the main task of the remaining treatment system was to facilitate settlement and retention of the particulate iron. The old system achieved effluent Fe concentrations of approximately 2.5 mg/l and there was no net impact on the water quality of the receiving burn.

The modified system incorporates a wetland as a polishing mechanism. Consideration of the data presented in tables 5.27.2 and 5.27.3 reveals that this polishing is effectively achieved, with most of the final 2.5 mg/l particulate Fe being retained in the wetland, to give an average effluent concentration of less than 0.3 mg/l. The effluent from the Polkemmet treatment scheme actually gives rise to improvement of the water quality of the receiving burn, presumably due to dilution effects.

Table 5.27.2. Summary water quality data for Polkemmet wetland for the period November 1999 to February 2000

n = 29	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Wetland Fe total	2.66	0.31	88.3	13.8	4.3
Wetland Fe sol	0.23	0.00			
Wetland pH	7.0	7.4			
n = 4					
Fe in burn upstream of treatment system	1.67				
Fe in burn downstream of treatment system	0.50				

Table 5.27.3. Summary water quality data for Polkemmet treatment system for the period March to October 2000

n = 31	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Raw minewater Fe total	48.9				
Raw minewater Fe sol	44.3				
Wetland Fe total	1.31	0.14	89.3	6.9	2.1
Wetland Fe sol	0.04	0.00			

Comments

The modified treatment system is working effectively, with the wetland providing a final polishing mechanism allowing a better water quality to be achieved than by the use of settlement lagoons alone.

The wetland is aesthetically attractive due to its natural appearance, with marginal alders playing a key role in this.

Costs

The construction costs of the Polkemmet treatment scheme were £198,000. Operating costs are £205,368 per annum.

5.28 QUAKING HOUSES, COUNTY DURHAM

NGR NZ190508

Pre-treatment water chemistry/volume and treatment requirement

The minewater at Quaking Houses is net acidic spoil heap leachate with elevated concentrations of iron, aluminium and manganese. The volume of leachate is highly variable, with rapid fluctuation during storm events.

The treatment requirement is to neutralise acidity and precipitate out metals. There was insufficient hydraulic head to facilitate use of a vertical flow, successive alkalinity producing system (SAPS), so a surface flow compost wetland was constructed.

Site history and design criteria

The source of the minewater at Quaking Houses is the 35 ha spoil heap of the abandoned Morrison Busty colliery, which closed in 1974. However, it was not until 1980, when road construction cut through the spoil heap, that pollution was noted in the Stanley Burn. After sustained pressure from the Quaking Houses Environmental Trust, the National Rivers Authority (now the Environment Agency) commissioned a pilot-scale study in 1995 to investigate ways of remediating the minewater discharge. A pilot-scale anaerobic wetland achieved satisfactory results during its 18 months of operation.

Funds were obtained, primarily from the Northumbrian Water Kick-Start Fund, in 1996 to construct a full-scale wetland treatment system on 500 m^2 of land provided by British Coal Property. Site surveys revealed that there was insufficient hydraulic head for a SAPS to be effective, so it was decided to construct a surface flow compost wetland. The land was found to be heavily contaminated, so the wetland was formed by building up the sides with bunds, rather than excavating into the contaminated material. Pulverised fuel ash (PFA) was used for the bunds, and a central weir was also constructed with PFA, covered with PVC to prevent erosion, to divide the wetland into two cells. The layout of the wetland is shown in figure 5.28.1.

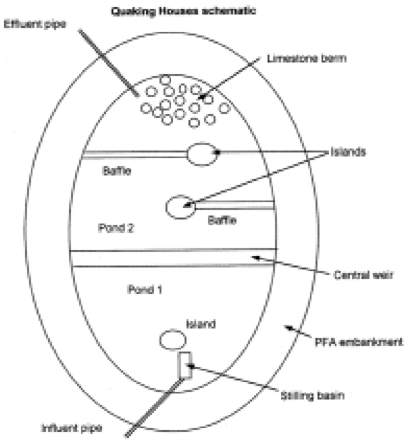

Figure 5.28.1. Layout of Quaking Houses treatment scheme.

A combination of limestone and three types of manure (cattle manure, municipal waste compost and horse manure) were used as the wetland substrate, which was placed to a depth of between 300 and 500 mm. A 100 mm diameter pipe forms the inlet from the culvert, which was the original discharge point of the minewater, to the wetland, and a 150 mm diameter pipe discharges treated minewater to the Stanley Burn.

Figure 5.28.2. Quaking Houses constructed wetland, showing wooden walkway and diversity of wetland species.

Monitoring data

The wetland was commissioned in November 1997 and has been regularly monitored (weekly or fortnightly), except when access was not possible due to the foot and mouth outbreak. Table 5.28.1 presents data collected during 1997–1998 (Jarvis & Younger 1999). Flow rates through the wetland are highly variable, ranging from 30 l/min to 420 l/min, with an average value of 127 l/min. Higher flows generally show a dilution effect in terms of Fe, Al and acidity concentration in the raw minewater but there appears to be no correlation between flow rate and removal of Fe and acidity. Removal of Al, however, decreases when flows are higher, which is probably due to a shorter retention time and disturbance of previously precipitated Al hydroxides on the substrate surface (Jarvis & Younger 1999). The removal mechanism for Fe appears to be both by aerobic precipitation of Fe hydroxides on the surface of the compost and by anaerobic precipitation of iron sulphides within the body of the compost.

Average acidity removal of over 10 g/m^2.day compares very favourably with the recommended design criteria of 7 g/m^2.day (Hedin *et al.* 1994; see table 2.7).

Table 5.28.1. Average water quality data for Quaking Houses constructed wetland

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (n = 65)	4.55	2.48	45.4	0.4	0.9
Al (n = 61)	6.6	2.4	63.0	0.8	1.7
Acidity (n = 32)	55.0	28.3	48.5	4.9	10.4
pH (n = 54)	6.16				
Mn (n = 65)	3.58				
Sulphate (n = 56)	845				

Comments

The Quaking Houses constructed wetland scheme is a good example of how such schemes can be integrated into the landscape and made to be attractive features. The incorporation of wooden walkways, and community involvement throughout the project, ensures that this is a valued amenity area as well as being a minewater treatment scheme.

Costs

The scheme was constructed for a sum of just under £18,000, which equates to £41/m^2 of wetland. A large proportion of these costs were associated with professional costs (designers and consultants) and plant hire/operation. Material costs were kept to a minimum by use of locally sourced materials and waste products. Maintenance costs are estimated to be in the region of £500 per annum (Younger pers. comm.).

5.29 RENISHAW PARK, SOUTH YORKSHIRE

NGR SK427377

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Renishaw Park is acidic spoil heap leachate. The pH is generally very low (about pH 3) and the Fe concentration is in the region 350 to 400 mg/l. Flows are generally low (approximately 1 to 2 l/s), but variable, dependent on rainfall. The treatment requirement is to add alkalinity and precipitate out the metal contaminants.

Site history and design criteria

The requirement on this site was to achieve passive treatment in a very limited land area – the available area measures approximately 40 m × 8 m. Thus, a two-stage permeable reactive barrier was installed. This comprises two trenches, the first of which is approximately 40 m long, 2.3 m deep and filled with a mixture of ornamental bark (60%), limestone (10%) and cattle manure (30%). Flow of groundwater from the spoil heap is directed down through this trench, then rises up through a second trench, approximately 25 m long, which is filled with limestone. A final trench, approximately 40 m long and 2 m wide, is planted with wetland plants and provides a polishing step to the treatment process.

Monitoring data

There are no point source flows at this site, and thus it has not been possible to collect monitoring data of the treatment process, although treatment seems to be successful as the impact on the receiving water course has been reduced.

5.30 SHILBOTTLE, NORTHUMBERLAND

NGR NU220077

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Shilbottle is acidic spoil heap leachate. The volume of minewater is highly variable, dependent on rainfall. There are about three point-source seeps of minewater issuing from the spoil heap as well as diffuse seepage along the toe of the spoil heap. The iron concentration in the leachate is up to 500 mg/l, and the pH is low.

The treatment requirement is to neutralise acidity and precipitate out metals. A permeable reactive barrier is planned to provide anaerobic conditions in which this can be achieved.

Site history and design criteria

The spoil heap of the former Shilbottle colliery, which is approximately 15 ha in extent, was reshaped as part of a land reclamation scheme in 1996. This scheme included the construction of a small constructed wetland to receive spoil heap leachate prior to its discharge into the Tyelaw Burn. There are reed beds on three levels, only the lower one of which receives leachate; the upper ones only receive rainwater, ditch drainage and agricultural field runoff. Thus, although the reed beds are well-established, and are an attractive feature, they are having little impact on the quality of minewater discharging to the burn.

Site investigations and research carried out by Newcastle University have indicated that a permeable reactive barrier (PRB) would be an effective means of intercepting and treating the contaminated leachate from the spoil heap. This scheme has been designed and was put out to tender in August 2001.

The reactive media used in the PRB was initially to have comprised a mixture of calcite limestone chips, green waste compost and cattle slurry screenings. However, due to the foot and mouth outbreak, and the requirement to minimise risk of transmitting the disease through road haulage of animal wastes, the tender documents included the provision of using just limestone and composted green waste. If cattle slurry is not used, provision has been made to inoculate the reactive mixture with a source of sulphate-reducing bacteria by incorporating some compost substrate from the Quaking Houses constructed wetland.

Figure 5.30.1. Aerial view of the Shilbottle constructed wetland and part of the spoil heap – the red-coloured area towards the left is the lower wetland cell which receives most of the spoil heap leachate (photo courtesy of Aln Elliot).

Figure 5.30.2. Lowest reed bed cell at Shilbottle.

Figure 5.30.3. Second reed bed cell, with reclaimed spoil heap in background.

Figure 5.30.4. Drainage ditch along toe of spoil heap. The bushes mark the course of the burn and the bright green area beyond this is the site for river realignment and PRB construction.

The installation of the PRB will entail realigning the Tyelaw Burn, which currently flows along the toe of the spoil heap. The burn will be moved approximately 12 m out from the toe to give space for a haulage road, the PRB and wetland/settling pond areas.

The PRB will be a minimum of 3 m wide, and 2.5 to 3 m deep and will be capped with an impermeable cover layer, such as compacted clay. Permeable filter drainage composite material (geotextile) will be placed on either side of the reactive mixture to provide a separation/filtration medium between the reactive mixture and the *in situ* material. On the down-gradient side of the PRB, treated groundwater will be collected in a constructed horizontal drainage path comprising mixed size limestone (minimum diameter 300 mm, maximum diameter 800 mm), through which it will flow into a settling pond/wetland area. These newly constructed wetland areas will discharge to the lowest cell of the existing reed bed system before final discharge to the Tyelaw Burn.

5.31 SKINNINGROVE, CLEVELAND, NORTH YORKSHIRE

NGR NZ713195

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Skinningrove is net alkaline, ferruginous and saline, sourced from two former ironstone mines. The flows from these two mines are each in the region of 24 l/s. The treatment requirement is to oxidise the minewater and facilitate the precipitation and settlement of iron hydroxides. However, the lack of any suitable flat land in the vicinity of the discharges on which to construct a wetland meant that alternative ways of achieving treatment had to be considered.

Site history and design criteria

The Kilton Beck was affected with ferruginous minewater from two former ironstone mines – Carlin How Mine, which closed in 1954, and Loftus Mine, which closed in 1958. Both mines are flooded and give rise to minewater discharge from a shaft and adit respectively.

In the 1970s the local council undertook measures to lessen the ochre staining on the river bed by installing a pipe to take the two discharges out to sea. However, after two days, multiple leaks occurred in the pipe and no further action was taken to remedy the situation.

In recent years efforts to deal with the pollution have been renewed, and a voluntary grouping, including local residents, the Loftus Development Trust and Newcastle University has obtained funding to implement a treatment scheme. Although the minewater would be suitable for treatment by means of an aerobic wetland, there was no suitable flat land available on which to construct such a scheme. Thus, a novel method was subjected to trial at pilot scale and has since been implemented at full scale. This method is termed the surface catalysed oxidation of ferrous iron (SCOOFI) and entails sorption of ferrous Fe onto a surface, where it is catalytically oxidised to ferric Fe and undergoes hydrolysis to become a coating of ochre on the original surface. This ochre layer attracts further ferrous Fe, and so the process of Fe-removal from solution continues. The pilot-scale scheme used gabion baskets filled with locally sourced blast furnace slag, and the full-scale scheme uses boxes packed with sheeted plastic filter media.

Figure 5.31.1. Newly constructed full-scale reactor at Skinningrove.

Monitoring data

The pilot-scale system operated for a period of 6 months from October 1999 and achieved 50% iron removal (influent iron concentration was in the region of 15-18 mg/l). The pilot plant provided design and sizing criteria for the full-scale system, which has been constructed but is not as yet (September 2001) receiving minewater.

5.32 ST HELEN AUCKLAND, COUNTY DURHAM

NGR NZ 199268

Pre-treatment water chemistry/volume and treatment requirements

The minewater at St Helen Auckland is net alkaline and mildly ferruginous (average Fe concentration is approximately 3 mg/l). The discharge has been occurring since 1979, and has typical flow of 2000 m^3/day (23 l/s).

The minewater is suitable for treatment in a surface flow aerobic wetland.

Site history and design criteria

St Helen Auckland Colliery worked from 1830 to 1926. However, minewater discharge at surface only occurred in 1979 following closure of Fishburn Colliery and the subsequent rebound of minewater in the region. By this time the former colliery site had been redeveloped and the former St Helen Auckland Engine Shaft had a building over it. Consequently the initial discharge of minewater occurred into this building and was quickly diverted to a nearby drain. This drain had formerly carried only seasonal flow of surface runoff, and discharged to a small ditch which ran through land belonging to Brusselton Farm, finally emptying into the River Gaunless. The greatly increased flow in this ditch, up to 2000 m^3/day (23 l/s) resulted in the field being almost permanently flooded with minewater.

The landowner tried to petition the National Coal Board and the Northumbrian Water Authority to take action to divert the minewater off his land, but he was unsuccessful in getting any positive response until after the formation of the Coal Authority in 1994. The Coal Authority negotiated with the owners of the St Helen Auckland Engine Shaft (English Partnerships) that they would co-fund, with the Coal Authority, the implementation of a solution to the problem. The Environment Agency also became involved in the selection of a suitable solution and it was decided, with advice from Newcastle University, to divert the minewater into an area of existing wetland. Minor earthworks were carried out to create a clay bund around the wetland, and pipework was installed to distribute the minewater along one edge of the wetland, and to collect treated water for discharge into the River Gaunless. A deep French-drain was also constructed around the perimeter of the bunded wetland area to prevent any seepage of minewater back into the previously affected field. Minewater was diverted into the wetland area in August 1999 and has been monitored by Newcastle University to ascertain its effectiveness.

Figure 5.32.1. General view of the St Helen Auckland wetland area (photo courtesy of Paul Younger).

Monitoring data

Monitoring data for a short period after the wetland system was commissioned are presented in table 5.32.1. It can be seen that the wetland performs well as a treatment system and that concentrations of iron in the final discharge are low.

Table 5.32.1. Average water quality data for St Helen Auckland wetland

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
Fe (n = 4)	2.53	0.2	92.1	4.6
Alkalinity (typical)	500	480		
pH (typical)	6.3	6.7		

Comments on design

This was a relatively ‘low-tech’ solution to a longstanding minewater discharge problem which seems to be functioning well.

5.33 TAFF METHYR, SOUTH WALES

NGR SO170020

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Taff Methyr is net alkaline, issuing from deep mines of the Taff Methyr Colliery, which was the most recent deep mine closure in South Wales. Discharge occurs from two shafts, and has a total flow in the region of 60 l/s. A spot sampling in June 2000 gave results of 2.2 mg/l Fe in the South Shaft emission and 14 mg/l Fe in the North Shaft emission. Sampling of the combined raw minewater in December 2001 gave a result of 7.7 mg/l Fe with pH of 7.3.

The treatment requirement of the minewater is for aeration and settlement to facilitate oxidation and precipitation of the iron.

Site history and design criteria

The minewater treatment scheme at Taff Merthyr is part of a larger £12 million reclamation scheme of three former coal mines: Taff Merthyr, Deep Navigation and Trelewis Collieries. This overall scheme is known as the Taff Bargoed Millennium Park and is intended to be a community park with fishing lakes and canoe slaloms set in a landscaped and conservation-orientated park. It is being carried out under the auspices of Merthyr Tydfil County Borough Council.

Closure of the mines in the 1990s resulted in minewater rebound and resultant pollution of a 3 km stretch of the Bargoed Taff, a tributary of the River Taff. The discharges were ranked high on the Environment Agency priority list in view of the serious environmental impact, and the major social and economic benefits that would accrue from remediation.

The constructed treatment wetland forms an integral part of the overall reclamation project and occupies an area of approximately 3 ha.

The treatment process entails aeration and settlement followed by a series of reed beds, with further settling tanks prior to discharge to the river. The initial settlement process is achieved in two concrete tanks in parallel. These tanks are approximately 80 m long and it is estimated that they will need emptying of sludge once every 5 years. There are no sludge drying facilities on site; when a tank needs to be emptied, the excess water will be pumped into the other tank and the sludge in the tank to be emptied will be allowed to dry for a period before removal. The parallel arrangement of the tanks allows one tank to be temporarily taken ‘off line’ in this manner.

Figure 5.33.1. Sludge settling tanks at Taff Merthyr.

There are sixteen wetland cells, with a fall of approximately 1 m between cells which facilitates re-aeration of the minewater by means of cascades. The cells are lined with bentonite liners and 400 mm of topsoil has been added. Nursery-grown *Phragmites* and *Typha* have been planted in a 70:30 mix. The total area of the wetland cells is 1.8 ha.

Channels between the cells are constructed of concrete. The inlet to the cells comprises a concrete channel across the width of the cell with openings to allow flow into the cell. The openings are levelled with neoprene flanges, which can more easily produce a perfect level than concrete alone. Outlets from the wetland cells comprise openings in the concrete wall across the width of the cell with vertical bars to allow infill of opening as sediment accumulates – a depth of 1 m accumulation can be accommodated.

Figure 5.33.2. Concrete structure between consecutive cells at Taff Merthyr, showing change in height between cells.

Figure 5.33.3. Concrete structure between cells.

Figure 5.33.4. Outfall from Taff Merthyr treatment system to the river.

As a temporary measure, prior to establishment of the reed beds, filtration was used at the outflow of the final settlement tanks to retain suspended material within the tanks. Initially a wool-based filtration material was used, as had been the case at Woolley Colliery, but it was found that this became blocked almost immediately. Subsequently a geotextile membrane was used which gave satisfactory performance and needed to be changed approximately monthly.

Monitoring data

Sampling in December 2001 gave a result of 7.7 mg/l Fe in the raw minewater (pH 7.3), and 0.09 and 0.10 mg/l Fe in the two discharges (both at pH 8.0).

Costs

The construction costs at Taff Merthyr were in the region of £1,625,000 and operating costs are £18,888 per annum.

5.34 TAILRACE LEVEL, DURHAM (FRAZER'S GROVE)

NGR NY896440

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Tailrace Level is net alkaline with Zn concentrations of up to 40 mg/l. The volume that was pumped while the mine was working was 1895 m^3/day (22 l/s) (Johnson & Younger 2000).

The treatment requirement is to remove the Zn by passive means for which an anoxic limestone drain has been installed.

Site history and design criteria

Frazer's Grove Mine is a fluorite/lead/zinc mine in the North Pennine Orefield. It actually comprises four interlinked sets of underground workings. Closure of the mine, and consequent cessation of pumping, occurred in 1999. Detailed geochemical and hydrogeological modelling was carried out to predict the possible impacts that minewater rebound would have on the environment (Younger 2000b). Early indications were that the water quality, when discharge occurred at surface from Tailrace Level, would not pose an environmental threat, due to the presence of a thick limestone layer (the Great Limestone). However, due to the presence of previously undocumented Zn and Mn-rich acid-generating shales, and various underground blockages and stratification issues (Johnson & Younger 2000), discharge of contaminated minewater occurred in August 1999 which had an adverse effect on Rookhope Burn. Prior to discharge from Tailrace Level, the mine owners, Durham Industrial Minerals Ltd, in partnership with the Environment Agency and with advice from Newcastle University, had constructed two lagoons to attempt to prevent pollution and to protect the stream. The lagoons were treated with limestone, and straw bales were added.

During November 1999, work started on converting the lagoons into anoxic limestone drains in an attempt to provide a more robust treatment method by encouraging the action of zinc-removing bacteria within the ALDs.

Figure 5.34.1. Frazer's Grove mine.

Monitoring data

There are currently no monitoring data available for either stage of the treament scheme at Tailrace level, although current indications are that the ALDs have become blocked or coated with iron precipitates and are thus not functioning effectively. The predicted long-term Zn concentration in the discharge is 4 mg/l, which it is anticipated will be achieved within the first 20 months following initial discharge (Johnson & Younger 2000).

5.35 WHEAL JANE, CORNWALL

NGR SW775428

Pre-treatment water chemistry/volume and treatment requirements

Wheal Jane minewater is an acidic, metalliferous pumped discharge. The current pumping capacity at the main shaft (No. 2 shaft) of the former tin mine is 330 l/s. Pumping is carried out to keep the water level in the shaft below the level of Nangiles Adit, from which minewater discharges by gravity flow to surface if the level in the shaft rises above the adit. This gravity discharge occurs periodically, particularly in the winter months when rainfall is very high.

The treatment requirements of the minewater are neutralisation of the acidity and precipitation of metals. The bulk of the treatment is carried out in a newly constructed active treatment plant at the main mine site, situated next to the No. 2 shaft. There is also an experimental passive system that was constructed in the Carnon valley below the mine; this is currently being run as a wetland research project.

Site history and design criteria

Wheal Jane lies within the Gwennap Mining District in Cornwall. Although metal extraction at the current Wheal Jane site was most recently carried out from 1979 to 1991, mining has been carried out in the vicinity of the Wheal Jane site since the 18th century, if not for considerably longer. There is thus an extremely complex array of mine workings, tailings dams and rock dumps within the environs of the current mine site. There is no documentation of the more ancient mine workings, so their full extent is not known.

The intention to cease mining at Wheal Jane was announced in early 1991, and in March 1991, the pumping operation to dewater the mine ceased. The various political and financial constraints affecting the cessation of mining and pumping are explained in a report issued by the mine operators, Carnon Consolidated Ltd (1992). The mine was formally abandoned under the Mines and Quarries Act (1954) in September 1991.

Contingency plans were drawn up to deal with the flow of minewater that was expected to occur from the Wheal Jane adit, as the water level in the mine rose. This adit was the lowest natural exit point for discharge water from the mine (10.48 m A.O.D [Above Ordnance Datum]). Discharge from this adit commenced in November 1991, and control and treatment measures were implemented. These measures consisted of pumping the discharge, adding lime and allowing settlement of the precipitate in the tailings dam, which had been used for this purpose during the period of mining. A period of strong winds in early January 1992 meant that the settlement process in the tailings dam was adversely affected. Pumping and discharge of water to the dam were therefore temporarily suspended, and sealing of the concrete plug that had been installed in the Wheal Jane adit portal meant that the underground water level rose. It was anticipated that the next adit discharge would occur from the Mount Wellington adit (17.0 m A.O.D). However, on 13 January 1991, failure of a plug on an adit intermediate in height between the Wheal Jane adit and the Mount Wellington adit caused millions of gallons of highly contaminated minewater to be released into the Carnon River. Previous examination of the plug sealing the Nangiles adit had shown no indication of weakness. When the highly acidic minewater entered the more saline sea water in Restronguet Creek, ferric iron precipitation caused orange discolouration of the water. A plume of this opaque material affected an estimated area of 6.5 million m^2 (RCS 1992) of Restronguet Creek, Mylor Creek, Carrick Roads and parts of Falmouth Docks (see figure 5.35.1). This was a highly visible incident that attracted international media coverage.

Pumping from Wheal Jane adit to the tailings dam recommenced on 21 January 1992. Pumping and treatment was carried out by Carnon Consolidated Ltd, with financial assistance from the NRA. Pumps were installed in the main shaft (shaft No. 2), to give a pumping capacity of 3.3 million gallons per day (145 l/s) (Hamilton *et al.* 1994), which was adequate to cope with discharge during dry periods (in the order of 2 million gallons per day) but unable to cope with peak flows of up to 9.3 million gallons per day (400 l/s). Installation of more pumps during 1995 allowed the pumping capacity to be increased to approximately 6 million gallons per day (260 l/s).

However, despite the efficacy of the treatment regime (Cambridge 1995), it could only be considered as a temporary solution due to the high cost (almost £2 million in the first year), and low density of the sludge that is produced. The latter consideration gives rise to disposal problems, since the tailings dam has a finite capacity, and under the treatment regime in operation at that time, the dam would have become full by the end of the year 2000 (NRA 1996).

Figure 5.35.1. Aerial photograph of Restronguet Creek and Carrick Roads showing the plume of ochre.

In December 1992, the government approved an £8 million scheme to develop a long-term treatment solution. In the announcement of this project (NRA 1992b), the problem at Wheal Jane was described as unique, with its combination of high and variable flow, extreme acidity and high concentration of metals. It was also stressed that this is a long-term problem. The criteria for a long-term solution were summarised as (Hamilton, pers. com.):

- reasonable capital cost;
- minimal operational cost;
- early implementation;
- phased transition from existing treatment;
- viable sludge disposal;
- achievement of water quality standards.

The key elements of the 3 year study to assess various treatment options were identified as (NRA 1992b):

- flow control, to limit the quantity of contaminated water needing treatment;
- primary treatment to reduce acidity – methods to include anoxic limestone drains and organic slurry ponds;

- secondary treatment to achieve the controlled deposition of metals – methods to include oxidation ditches, and possibly sludge recirculation systems and sludge drying;
- tertiary treatment to provide final 'polishing' – methods to include constructing artificial marshes to test different species of aquatic plants.

Pilot passive treatment scheme

In February 1994, the consultants appointed by the NRA to carry out the investigation, Knight Piésold and Partners, announced their proposed schedule for finding a long-term treatment solution (NRA 1994b). This included reference to trials of active techniques to remove metals physically or chemically from the minewater, and an outline of the pilot wetland treatment plant to be constructed in the Carnon Valley. The pilot plant was constructed during 1994 and completed in November of that year.

Figure 5.35.2. Aerial photograph of the Wheal Jane mine site (top), tailings dam and the pilot passive treatment scheme in the Carnon valley, taken soon after completion of the pilot passive treatment scheme in 1995.

The proposed pilot wetland treatment plant consists of three schemes, which are identical in their main treatment methods, but differ in pre-treatment techniques (see schematic representation in figures 5.35.3 to 5.35.5). The main treatment consists of a series of aerobic cells, followed by an anaerobic cell then a rock filter. The aerobic cells comprise plants, mostly *Phragmites australis* and *Typha latifolia*, planted into an essentially inert substrate (South Crofty tailings from the processing mill at

Wheal Jane), with a free water surface and channels between consecutive cells. These cells are designed to facilitate aluminium and iron precipitation as hydroxides. Co-precipitation of arsenic should also occur. The anaerobic cell in each scheme is filled with a mixture of cattle manure and sawdust, and covered with a vegetated soil cap. Sulphate-reducing bacteria are expected to be active in these cells and lead to the precipitation of copper, zinc and cadmium sulphides. The rock filter comprises a cell filled with a rock/gravel substrate, on which it is hoped that algae will become established to create a high pH for manganese precipitation.

The three different pre-treatment stages comprise:

(1) lime dosing and a sludge channel;
(2) anoxic pond and an anoxic limestone drain (ALD);
(3) no pre-treatment.

Modifications were carried out to the three systems during the winter of 1996–7. Prior to the modifications, the components of each system were operated in-series with minewater flowing consecutively through the pre-treatment (where applicable), aerobic cells, anaerobic cell and rock filter to the discharge point. The modifications were intended to increase flexibility of operation, thereby allowing more investigative work to be carried out. They allowed the aerobic cells and anaerobic cells to be operated independently of each other (in parallel) so that flow could occur directly into the anaerobic cells, by-passing the aerobic cells; the facility to lime dose the ALD System influent water prior to the ALD was also incorporated, to attempt to remove aluminium in the sludge channel and reduce blockages of the ALD.

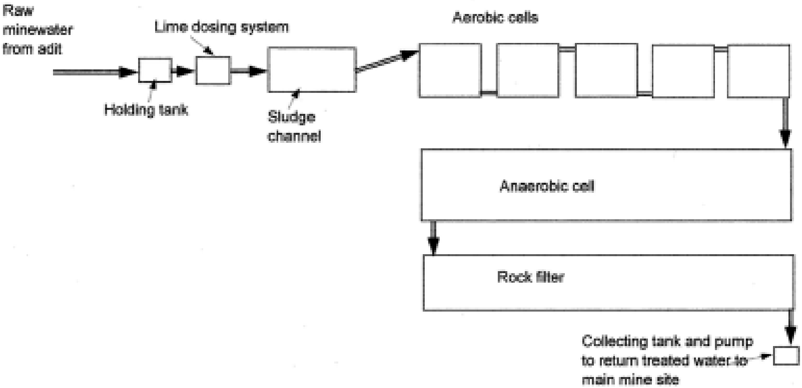

Figure 5.35.3. Layout of lime dosed system.

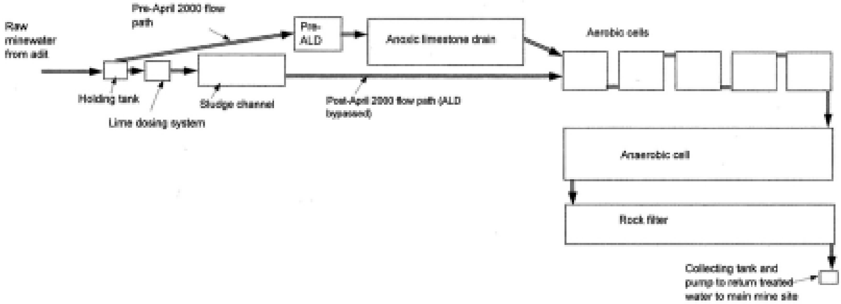

Figure 5.35.4. Layout of ALD system.

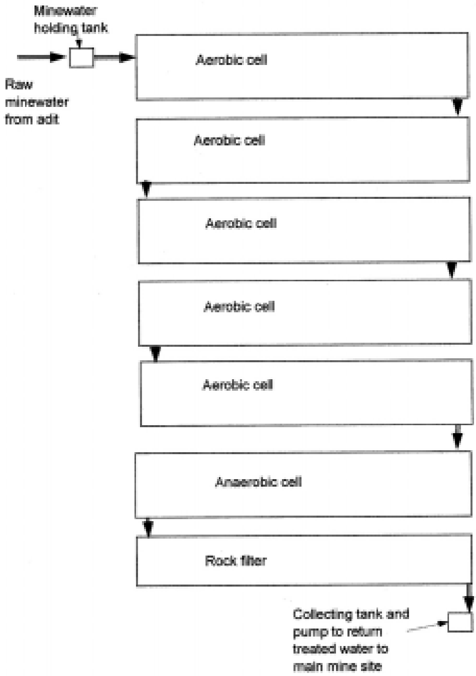

Figure 5.35.5. Layout of lime free system.

Figure 5.35.6. Concrete channel connecting consecutive aerobic cells.

Figure 5.35.7. Anaerobic cell during its construction showing organic material.

Figure 5.35.8. Rock filter, with anaerobic cell and aerobic cells to the right.

The pilot passive treatment system received flows, from Wheal Jane adit, of between 0.1 to 0.6 l/s, with short periods of flows up to 1.9 l/s in the part of the system with no pre-treatment of the minewater. This represented a tiny fraction of the total volume of minewater requiring treatment (less than 1%). Preliminary evaluation of results (Dodds-Smith *et al.* 1995) indicated that the anoxic pond, ALD, and aerobic cells performed well. However, the report for the 3 year investigation (NRA 1996) stated that the passive treatment trials were inconclusive and that further testing would be required to confirm the efficacy of this type of system for long-term use at Wheal Jane. The final report on the pilot passive treatment system (Environment Agency 1998) concluded that such a system was not a viable option for treating all the Wheal Jane minewater to an environmentally acceptable standard and that the scale and cost of a suitably sized wetland would make this type of treatment neither feasible nor competitive with chemical treatment. It also seemed that the improvement in water quality that could be expected, with passage of time from the initial flooding of the mine workings, had not been as substantial as might have been expected, based on other documented cases (Robinson pers. com.), thus necessitating the continued use of an active treatment system.

The pilot passive treatment plant is currently being run as a research project under the LINK Scheme, in which a consortium of academic institutions is carrying out research, with input from industry and funded by the DTI and the Environment Agency. The objectives of the current study are to gain a fuller understanding of the microbiological, geochemical and ecological functioning of the pilot plant, and to incorporate all the resultant data and knowledge into a computer model of the system.

Active treatment scheme

In June 1996, a six party consortium, lead by Knight Piésold, were commissioned by the Environment Agency to undertake an appraisal and selection of a long-term treatment option. This study included consideration of the water quantity and quality, a technical and economic assessment of the different treatment technologies, and the use of a cost-benefit approach in order to establish the most appropriate treatment option (Environment Agency 1999). This study identified an oxidation and chemical neutralisation system to be the most cost-effective way of treating minewater from Wheal Jane, a recommendation that was endorsed by the Environment Agency and the Department of Environment, Transport and the Regions in October 1998.

A design, build and operate contract was awarded to Hyder (now United Utilities) in 1999 and the new active treatment plant became operational in October 2000.

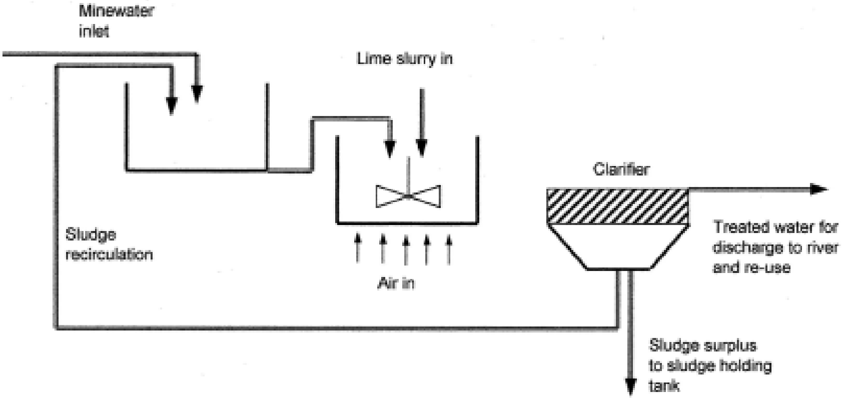

Figure 5.35.9. Schematic of active treatment system.

The active system comprises a system of tanks in which successive stages of the treatment are carried out. Minewater is pumped from the shaft to the treatment system. Also water from the toe drain and supernatant of the tailings dam and the pilot passive treatment plant is treated in this system.

In the shaft there are 6 pumps each with a 55 l/s capacity, to give a current maximum of 330 l/s from the shaft. There is pipework to add two extra pumps: the maximum capacity of the system is 440 l/s. The current discharge consent is for 350 l/s to the river (treated water is also used for mixing lime slurry, carrying polymer and washing down).

Two identical process streams operate in parallel to allow for closure and maintenance of one stream when necessary.

The principle of the treatment process is that recirculation of sludge allows the formation of larger particles which therefore have better settling characteristics than the rather 'fluffy' precipitate typical from lime dosed treatment alone.

Pumped minewater enters the first tank where recirculated sludge is added at a ratio of between 50:1 and 25:1. Mixers within the tank keep the minewater and sludge agitated. Addition of the sludge raises the pH to about 8.5.

The minewater/sludge mixture enters the second tank by flowing under the dividing wall. In the second tank aeration is carried out by an air diffuser at the base of the tank with a vertical turbine paddle above to break up the bubbles. Lime is added in this tank from a constantly recirculating loop dosing system in the form of a 5% lime slurry. Approximately 18 tonnes/day of lime is used. Vigorous agitation in this tank prevents the formation of a 'fluffy' flocculant. The operating pH is 9.25.

An offtake pipe in the corner of the second tank takes the minewater/lime/sludge mixture, by gravity flow, through a pipe with a mesh to break up the flow before addition of a polymer flocculant (Magnafloc 10 is currently used at a dosing rate of 2.5 g/l). The mixture then flows up a pipe where the flow is split into two and directed into two parallel clarifiers (for each stream).

Figure 5.35.10. First tank of the treatment process, in which recirculated sludge is added.

Figure 5.35.11. Second tank, in which lime slurry is added and aeration carried out.

Figure 5.35.12. Clarifiers, at the final stage of treatment process – the central channel divides the two parallel streams of process water. The tailings dam is visible in the distance.

The clarifiers comprise lamellar plates that give a total surface area of about 50 m^2 for each tank, thus providing a much increased settling area. Solids settle out in the clarifiers and are drawn off at the base by a rotating rake. The solids are then either recirculated to the first tank, or piped to the sludge holding tank. Supernatant water decants over the overflow launders into a central channel and flows to the treated water holding tank.

Figure 5.35.13. View of whole treatment plant, with lime storage silos in the background, sludge addition and lime addition/aeration tanks in the middle and clarifiers in the foreground.

Sludge in the sludge holding tank settles to over 30% solids in 5 hours and can go up to 40% solids. It is then piped to the tailings dam. Stability tests carried out on the sludge reveal that it is relatively stable in terms of mobility of metals it contains, with a low permeability and a high pH micro-environment with some residual lime. Filter press trials have shown that a 70% solids sludge could be obtained if the tailings dam were no longer available and off-site disposal had to be considered.

Water from the treated water holding tank is either used for wash down, lime mixing or to carry the polymer flocculant, or discharged direct to the river.

Retention times within the system are approximately 30 min in the tanks and 25 min in the clarifiers.

Monitoring data

Pilot passive treatment system

Table 5.35.1. Summary data from the Wheal Jane Pilot Passive Treatment Plant for the period June 1995 to January 1998

	Median inflow mg/l June 1995 to Jan 1998 (whole monitoring period)	Median outflow mg/l Jan 1997 to Jan 1998 (post-modifications)	Treatment efficiency %	Retained kg/day[1]	Area adjusted removal g/m^2.day[1] (aerobic cells)
Lime dosed system					
pH	3.8	3.4			
Total Fe Aerobic cells only Whole monitoring period	142	31	78.0	2.9	2.2
Total Fe Whole system	142	23	83.8		
Al	50	26	48		
Mn	23	19	17		
As	2.68	0.03	98.8		
Zn	75	50	33		
Cu	0.41	0.27	34		
Sulphate	1742				
ALD system					
pH	3.9	3.8			
Total Fe Aerobic cells only Whole monitoring period	149	17.5	88.3	3.4	2.6
Total Fe Whole system	149	5	96.6		
Al	50	5	90		
Mn	23	15	35		
As	2.57	0.007	99.7		
Zn	75	23	69		
Cu	0.42	0.05	88		
Sulphate	1720				

Table 5.35.1. (continued)

	Median inflow mg/l June 1995 to Jan 1998 (whole monitoring period)	Median Outflow mg/l Jan 1997 to Jan 1998 (post-modifications)	Treatment efficiency %	Retained kg/day[1]	Area adjusted removal g/m^2.day[1] (aerobic cells)
Lime-free system					
pH	4.0	3.2			
Total Fe Aerobic cells only Whole monitoring period	140	34.5	75.4	6.4	1.3
Total Fe Whole system	140	28	79.3		
Al	50	39	22		
Mn	23	19	17		
As	2.49	0.019	99.2		
Zn	72	54	25		
Cu	0.40	0.29	27		
Sulphate	1694				

[1] Assuming flow rate of 0.3 l/s into lime dosed and ALD systems (which is less than design flow rate, but was the flow rate for most of the post modification period) and 0.7 l/s into lime-free system. In reality, flow rates were varied throughout the operational period.

The conclusions drawn as a result of data analysis and interpretation carried out by the consultants operating the pilot system are given in the following sections (taken from the final report of the project, Environment Agency 1998). Comments and observations additional to the reported conclusions are added in italics.

Influent minewater quality

The aerobic cells received lower than expected influent iron loads per unit area due to the lower influent iron concentrations. (*The pilot plant was designed to receive a predicted influent iron concentration of 250 mg/l, but in fact the influent iron concentration had a median value of 140 to 150 mg/l.*) There was a strong positive correlation between influent iron load and iron removal per unit area in g/m^2.day. As iron load increased, there was a tendency for percentage iron removal to decrease, reflected by a weak to moderate negative correlation. The lower than expected influent iron concentration, therefore, limited the investigation of these relationships at higher influent loads.

In the anaerobic cells, metal removal is nominally dependent upon the rate of sulphide generation. In this context, the lower than expected influent zinc concentration (*median value of 76 mg/l as opposed to a design value of 250 mg/l*) should not have affected anaerobic cell performance and the higher sulphate concentrations (*median values of 1753 mg/l as opposed to a design value of 1000 mg/l*) would be expected to have provided a greater driving force for the sulphidisation. A relationship was not established between influent metal loads and percentage and unit volume removal rates. This was because the influent flow rates were limited by the need to maintain viable bacterial communities within the cells.

Pre-treatment

At an influent pH less than approximately 5, a form of pH elevating pre-treatment may be beneficial.

Passing pre-treated minewater directly into an anaerobic cell proved beneficial to anaerobic cell performance.

The addition of bicarbonate alkalinity and increase in pH as the minewater passed through the ALD was presumed to be responsible for the more abundant and luxurious reed growth observed in the ALD system. (*It is also probable that mobilised organic matter from the pre-ALD, which comprised a mixture of cattle manure and sawdust, provided nutrients and a source of organic carbon to the reeds in the ALD aerobic cells, thus improving their growth and health.*)

pH elevating pre-treatment did not result in enhanced iron removal in the aerobic cells, compared to the system without pre-treatment, at comparable influent iron loads.

Lime dosing pre-treatment has the disadvantage of ongoing management and maintenance requirements.

The use of the ALD is applicable where the aluminium concentration in the influent minewater is approximately 5 mg/l or less; at higher concentrations, precipitation of aluminium hydroxide may reduce the flow rate through the cell. (*The pre-ALD became blocked in November 1997 and was by-passed. The ALD itself became blocked at approximately monthly intervals during the first 6 months of the LINK research project, necessitating back-flushing with river water to restore flow through the ALD. The decision was made to discontinue use of the ALD as this regular back-flushing was preventing a steady-state situation becoming established in the aerobic cells, which in turn was proving detrimental to the planned research programme.*)

Aerobic treatment

In all three systems iron removal (allowing for dilution/evaporation) ranged from 51% to 98% or 0.2 to 5.5 g Fe/m^2.day with arsenic removal close to below detection limits.

For the lime-free system over a limited period iron removal per unit area in the first of the five aerobic cells was found to be significantly higher than in subsequent cells, with a median of 10 g Fe/m^2.day.

A system of aerobic cells configured in parallel would be capable of treating a proportionately higher flow rate and would remove a greater amount of iron per unit area, although at the risk of higher iron concentration in the treated effluent.

At comparable influent iron loads in each system the use of pre-treatment did not result in higher removal rates.

Sizing an aerobic system for iron removal should be based upon the relationship between influent load and removal per unit area (*i.e. assuming zero-order removal kinetics, as discussed by Hedin et al. 1994, rather than first-order kinetics, as proposed by Tarutis et al. 1999 – see Chapter 2*).

The actual contribution of the reeds themselves in enhancing metal removal, in the short term, is inconclusive, while in the longer term, it is anticipated that the presence of reeds would be beneficial as a source of organic matter that may encourage bacterial processes associated with metal removal. (*Work being carried out as part of the LINK project disputes this view. The 'beneficial' bacteria in terms of iron oxidation are autotrophic, acquiring their required carbon by photosynthesis. The presence of a carbon source, in the form of reeds and their associated by-products, may provide a competitive advantage to heterotrophic bacteria which may out-compete the autotrophs and thus have a detrimental effect on the amount of iron immobilised through oxidation*).

Reed growth would be likely to benefit from an organic-rich substrate.

The main limit to iron removal within the aerobic cells was reduction in pH resulting from ferric iron hydrolysis.

An aerobic cell with an organic-rich substrate incorporating limestone is preferred due to enhanced metal removal expected from limestone dissolution and bacterial sulphate reduction processes.

Anaerobic treatment

Sulphate removal ranged from zero to 0.57 moles/m^3.day, although generally removal was much lower than this maximum value, at around 0.08 moles/m^3.day.

Target metal removal (Zn, Cd, Cu and Fe) was lower than sulphate removal at approximately 0.04 moles/m^3.day, with minewater pH increasing from approximately pH 3.0 – pH 4.0 to pH 6.0 across the cells.

Zinc removal ranged from zero to 100% (below detection limits) with removal generally above 60% while cadmium and copper were, for most of the time, removed to below detection limits.

Receiving raw minewater directly the lime-free system anaerobic cell successfully removed a median of 80% zinc, 33% iron, with arsenic, cadmium and copper all being removed to below detection limits.

Overall the low pH of the aerobic cell effluent was detrimental to the anaerobic cell performance, thus emphasising the benefit to be derived from passing pre-treated minewater directly into an anaerobic cell.

To achieve the maximum removal of as wide a range of metals as possible in a given land area, minewater passed directly into an anaerobic system would be the preferred configuration (with pre-treatment to pH 5 or 6, if necessary).

It is anticipated that the inclusion of limestone in the anaerobic cell substrate would be beneficial, leading to higher and more sustained levels of metal removal, although in the short term the results are inconclusive.

Rock filter treatment

At times, high levels of manganese removal were observed in the rock filters, with enhanced removal coinciding with a higher influent pH. Sustained manganese removal, however, was not consistently demonstrated.

The rock filters proved to be efficient at reducing the BOD of the anaerobic cell effluent.

Active treatment

The current active treatment system has been operational since October 2000. Table 5.35.2 summarises the treatment performance of this system.

Table 5.35.2. Summary data for active treatment plant performance

	pH	TSS mg/l	Fe mg/l	Zn mg/l	Mn mg/l	Cu mg/l	Ni mg/l	Cd mg/l
Influent	3.8	3	159	44	5	0.4	0.4	0.056
Effluent	9	5	1.3	0.4	0.2	< 0.01	< 0.1	< 0.001

Costs

Passive treatment

Design and construction costs were £1,300,000, maintenance costs are £100,000 per annum.

5.36 WOOLLEY COLLIERY, WEST YORKSHIRE

NGR SE312113

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Woolley is net alkaline deep pumped minewater which requires aeration and settlement to remove the Fe loading. The ammonia–nitrogen concentration is also of concern. When the other pumping stations in the region closed the pumping was centralised at Woolley. There was a 2 year gap between cessation of pumping for mining and resumption of pumping for environmental reasons. The resumption of pumping in 1991 increased the rate of discharge to the River Dearne from 40 l/s to 200 l/s. The capacity of the treatment system had been increased to deal with the higher flows and poorer water quality but this proved insufficient. A considerable impact was made on the river, with a pronounced ochre plume being apparent 4 km downstream. The Fe concentration in the raw minewater has fallen from about 100 mg/l in the early 1990s to the current level of about 15 mg/l with a pH of around pH7.

The consented discharge limits are as follows:

Suspended solids	80 mg/l
Rate	750 m^3/hr (208 l/s)
Total iron	15 mg/l until April 1995 5 mg/l from May 1995 to April 1996 3 mg/l from May 1996

Site history and design criteria

After pumping of an extensive area of underground workings was centralised at Woolley in 1991, the existing treatment facility needed upgrading to cope with the increased flow, which became about five times the flow for which the system was originally designed. To comply with the staged discharge consent the existing system of concrete settlement tanks followed by earth-bunded lagoons was improved by the following means:

- Improvements were made to the cascade aeration spillways at the inlet to the concrete settlement tanks.

- Flow to the concrete settlement tanks was divided and the first two tanks run in parallel rather than in series.
- The dividing banks of the earth-bunded lagoons were removed to give improved hydraulic flow, larger surface area and improved sedimentation. The pond banks were also raised to increase storage volume and to increase the head to facilitate gravity flow through the system.
- A temporary filtration system was installed to remove fine suspended solids from the minewater by means of filter fabric straining walls.
- A wetland area was established (in 1995) to provide a polishing facility in the short term and to replace the function of the filtration system in the longer term. Flocculant addition was also used pending the establishment of the reed bed.

Possible future improvements (Laine 1997) include the provision of a large lagoon to provide initial settlement and a large sludge storage capacity for at least 40 years of ochre, improved pumping to make use of 'off-peak' tariff and reduce power costs, and a wetland area to achieve reduction in the concentration of ammonia in the discharge from the current treatment system (the latter feature is currently on an experimental scale). The capacity of the various components of the treatment system are shown in figure 5.36.1.

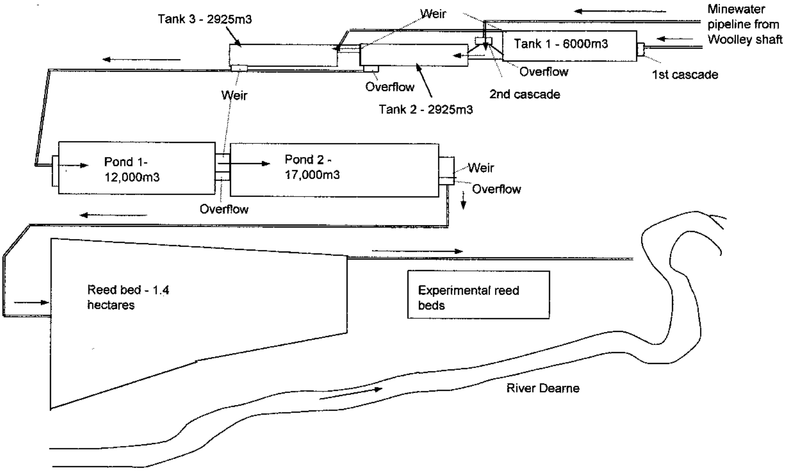

Figure 5.36.1. Layout of treatment scheme at Woolley.

Figure 5.36.2. Aeration cascade at inlet to second settlement tank – the overall height of the cascade structure is about 2 m.

Figure 5.36.3. Outflow from third settlement tank.

Figure 5.36.4. Inlet structure to first lagoon – see text for explanation.

The treatment system at Woolley has incorporated features that were already constructed (the settlement tanks and lagoons) and as such was not specifically designed for the function that it is now performing. In terms of design criteria, the settlement tanks are of a far higher engineering specification than would be affordable for a purpose-built system constructed today. Nevertheless, the sizing and operational parameters of the system provide much useful information that can be applied to the design of future treatment systems.

The inlet structure to the first settlement lagoon (figure 5.36.4) comprises timber ‘steps’ to improve aeration of cascading minewater and vertical pipes to hold and dispense flocculant bricks. Neither of these two facilities are now in use – the aeration step was found not to be necessary as nearly all the iron is in suspension by this stage (see monitoring data) and it was considered inadvisable to use the cascade mechanism as this would break up the flocculated particles; the addition of flocculant was found to be no longer necessary as adequate settling and filtration is being achieved in the lagoons and wetland without the need for flocculant.

Figure 5.36.5. Weir structure and by-pass channel between first and second settlement lagoons – the vertical posts and wire mesh structures were incorporated to hold the filtration material, which is no longer necessary

Figure 5.36.6. View from outlet from second settlement lagoon back to disused aeration structure to first lagoon. Note colonisation of vegetation (*Phragmites*) around the edge.

The outlet structure from the first lagoon leading to the second lagoon, and from the second lagoon, had framework incorporated so that a wool-based filtration material could be used to trap suspended solids (figure 5.36.5). However, after the initial ‘start-up’ period, and once the wetland area was established, it was found that the use of this filtration device was no longer necessary.

The wetland area was established to provide, primarily, long-term removal of fine suspended solids. The wetland was established on arable land between the colliery site and the River Dearne. The land available restricted the size of the wetland that could be established to just about 1.4 ha (simulation studies of the wetland’s performance showed a requirement of up to 2 ha to achieve the required discharge quality, Laine 1997). Relatively little grading was required to achieve the required land profile, which reduced earthmoving costs and helped to preserve a good soil structure in which to plant.

An earth berm was constructed around the wetland area and concrete was only used for the outlet to the discharge ditch and for six adjustable weirs to allow water level to be controlled. The inlet to the wetland area comprises a simple pipe distribution system to spread the flow across the 110 m width of the wetland. This consists of four 300 mm plastic pipes with tee-fittings arranged across the top of the wetland area.

Construction of the wetland area commenced in May 1995 and planting was virtually completed within 2 months. A mixture of species was selected to provide wildlife habitat, create visual diversity and tolerate the prevailing conditions. These species comprised:

Scirpus lacustris (True Bulrush)
Typha latifolia (Cattail)
Typha angustifolia (Lesser Cattail)
Iris pseudocorous (Flag Iris)
Phragmites australis (Common Reed)
Phalaris arundinacea (Reed Canary Grass)
Juncus effusus (Soft Rush)

Species were planted in blocks to reduce dominance problems – the blocks of species compete with one another rather than individual members of species competing within a full-mixed planting, which is likely to lead to a single species becoming dominant. *Juncus effusus* was planted at the inlet and *Scirpus lacustris* near the outlet, based on accumulated knowledge of the growth habits and tolerances of these species – *Juncus* is tolerant of fluctuations in water chemistry and *Scirpus* is an aggressive species and thus would prevent *Phragmites and Typha* encroaching on, and blocking, the outlets.

Figure 5.36.7. Plastic pipes at inlet to wetland area.

The concentration of ammonia in the raw minewater is about 3.5 mg/l; while this is not currently a parameter subject to discharge consent, the Coal Authority is keen to find a passive means of reducing the ammonia concentration to prevent adverse impact on the River Dearne and to meet any future discharge limits that may be set. The main wetland area has little effect on the ammonia concentration. Consequently, a research wetland area has been established to examine different wetland configurations that would be more effective at removing ammonia. This experimental wetland area comprises four subsurface-flow gravel beds, each being 20 m long, 14 m wide and 1 m deep and filled with pea gravel to a depth of 0.4 m. Various planting regimes and ammonia dosing regimes have been applied to the different beds to investigate the removal of ammonia. The research work is continuing but to date encouraging results have been obtained (Demin 2000).

Figure 5.36.8. Outlet structure from main wetland area.

Figure 5.36.9. First cell of experimental wetland area from which plants have been removed to allow investigation of plants and algae in the ammonia removal process.

Monitoring data

There is a comprehensive monitoring record of water quality at the Woolley water treatment site, including total and soluble iron at each intermediate site in the process for the period of one year (1999). There are also spot sampling data available for August and September 1996 and April 1998. Thus it is possible to carry out various manipulations of these data to assess removal rates in various parts of the treatment system and how these rates have changed with time.

Table 5.36.1 shows the monitoring data for total and soluble Fe at the various sampling points throughout the treatment system.

Table 5.36.1. Summary monitoring data and calculations for Woolley treatment system

			Aug 1996	Sept 1996	April 1998	1999 Average values n = 13
	Average flow	l/s	110	116	174	162
TANK 1 60% of flow	Fe (total) in	mg/l	45	50	21.6	12.7
	Fe (sol) in	mg/l	45	50	20.3	11.8
	Fe (total) out	mg/l	13.6	13.3	10.3	8.6
	Fe (sol) out	mg/l	3.9	8.1	2.1	1.4
	Fe retained	kg/day	179.1	220.7	101.9	34.9
	Area adjusted Fe retention	g/m^2.day	111.9	137.9	63.7	21.8
TANK 2 40% of flow	Fe (total) in	mg/l	45	50	31.5	17.9
	Fe (sol) in	mg/l	45	50	25.9	16.8
	Fe (total) out	mg/l	14.3	12.8	11.6	7.1
	Fe (sol) out	mg/l	5.2	10.9	3.0	1.3
	Fe retained	kg/day	116.7	149.1	119.7	61.1
	Area adjusted Fe retention	g/m^2.day	110.5	141.2	113.3	57.9
TANK 3 Flow recombines	Fe (total) in	mg/l	13.9	13.1	10.8	8.0
	Fe (total) out	mg/l	13.0	10.4	11.5	7.5
	Fe (sol) out	mg/l	2.7	8.6	2.0	2.0
	Fe retained	kg/day	8.4	27.1	-10.2	7.0
	Area adjusted Fe retention	g/m^2.day	7.9	25.6	-9.7	6.7

Table 5.36.1. (continued)

			Aug 1996	Sept 1996	April 1998	1999 Average values n = 13
	Average flow	l/s	110	116	174	162
POND 1	Fe (total) in	mg/l	13.0	10.4	11.5	7.5
	Fe (sol) in	mg/l	2.7	8.6	2.0	2.1
	Fe (total) out	mg/l	2.5	2.6	6.8	4.7
	Fe (sol) out	mg/l	1.6	2.2	0.6	1.2
	Fe retained	kg/day	99.8	78.2	71.0	39.5
	Area adjusted Fe retention	g/m^2.day	33.3	26.1	23.7	13.2
POND 2	Fe (total) in	mg/l	2.5	2.6	6.8	4.7
	Fe (sol) in	mg/l	1.6	2.2	0.6	1.2
	Fe (total) out	mg/l	2.7	1.9	3.9	2.9
	Fe (sol) out	mg/l	1.3	1.8	0.7	1.3
	Fe retained	kg/day	-1.9	7.0	43.9	25.1
	Area adjusted Fe retention	g/m^2.day	-0.5	1.9	11.7	6.7
WETLAND	Fe (total) in	mg/l	2.7	1.9	3.9	2.9
	Fe (sol) in	mg/l	1.3	1.8	0.5	1.0
	Fe (total) out	mg/l	0.4	0.5	0.5	0.3
	Fe (sol) out	mg/l	0.3	0.4	0.2	0.2
	Fe retained	kg/day	21.9	14.0	52.5	36.3
	Area adjusted Fe retention	g/m^2.day	1.6	1.0	3.7	2.6

Raw minewater

The Fe content of the raw minewater can be seen to have decreased considerably, from 45–50 mg/l in 1996 to 10–20 mg/l in 1998/9. This is a more rapid improvement in water quality than was expected (Laine 1999) and in fact represents a return to the Fe concentration in the minewater during the working life of the mine – a situation that was not expected to occur for many years.

The reason for the difference in values for Fe content of the inflow to the first and second settlement tanks is not known – one would expect the water quality to be the same in each of these streams, since they are both sourced from the main pumping station and the flow is split between the first two tanks.

Most of the Fe in the raw minewater is in solution, although there is considerable variation, from all of the Fe being in solution (0% in suspension) to up to 80% being in solution (20% in suspension). Once again, the reason for this variation is not known as there is no discernible seasonal trend, correlation between pumped volumes and proportion of Fe in solution, or even correlation between the proportion of Fe in solution in the two different inflow streams even though they both come from the same single pumped source.

Aerated minewater

The first step in the treatment of the minewater is aeration. At the inlet to the first and second settlement tanks there is a structure to cascade the minewater (see figure 5.36.2). This aeration step, followed by passage through a settlement tank, is very efficient at precipitating the Fe out of solution, as can be seen from the monitoring data for the outflows from the first two tanks. The maximum precipitation achieved is over 99% of the total Fe and the average is just over 80% for both tanks (these figures for percentage precipitation represent the proportion of total Fe in suspension).

The graphs in figure 5.36.10 show the proportion of Fe in suspension, for the period July 1999 to January 2000, as the minewater flows through the system. It can be seen that after an initial increase as a result of aeration in tanks 1 and 2, the proportion of particulate Fe varies as the minewater passes through the system. This could be due either to previously precipitated Fe going back into solution, or remobilisation of particulate Fe that had previously settled out of suspension. The latter scenario is more likely since the excess alkalinity in the minewater buffers pH and therefore there is no drop in pH which would cause particulate Fe to redissolve. In addition, the total amount of Fe is decreasing as the minewater passes through the system, so the proportion of dissolved and suspended Fe will be affected by this.

Generally, the outflows from tanks 1 and 2 have over 60% of the Fe in suspension, so the principal treatment requirement of the minewater is to settle out this suspended Fe and retain it in the settlement tanks and lagoons.

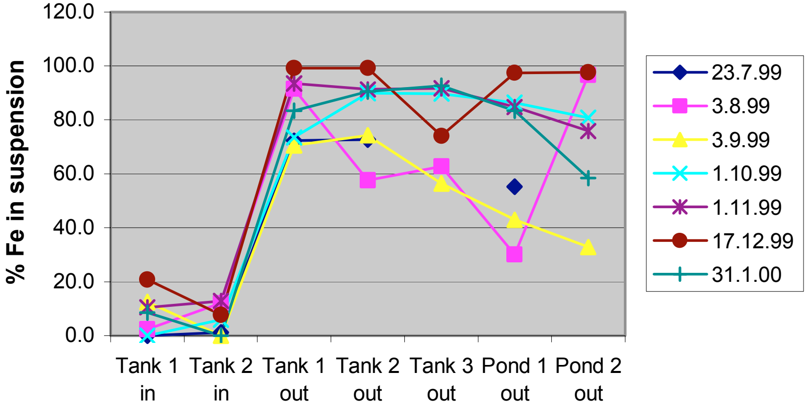

Figure 5.36.10. Percentage of iron in suspension at various sampling points through the Woolley treatment system, for the period 23 July 1999 to 31 January 2000.

Iron loading through the treatment system

Data for Fe concentration at the various monitoring points through the system can be combined with the flow rate through the system to give a value for Fe loading at the various points, expressed as kg/day. Calculation of Fe loading into and out of each part of the treatment system allows the amount of Fe retained in each part to be determined, expressed once again as kg/day. It should be noted that these values will only be indicative of the Fe retention that is actually occurring, since the flow rate figure is an average for the period of a month, and the concentrations of Fe are determined on one occasion each month.

The values for Fe retained in each part of the system can be assessed in various ways, to provide a measure of how well the treatment system is functioning.

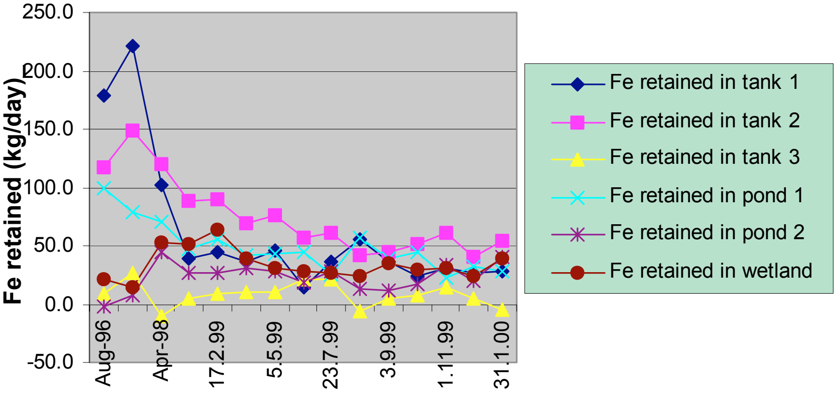

Figure 5.36.11. Amount of iron, expressed as kg/day, retained in component parts of the Woolley treatment system.

The graphs in figure 5.36.11 show the amount of Fe retained, on a daily basis, in each part of the treatment system. It can be seen that in 1996, when influent Fe concentrations were higher, greater retention of Fe was achieved, particularly in the first two settling tanks. During 1999 there has tended to be a levelling off in the amount of Fe retained. Settling tank 2 generally retains the most Fe, and settling tank 3 the least (sometimes a negative amount so that the tank is acting as a source of Fe rather than retaining it). The other components of the system show an increasing similarity, over time, in terms of the amount of Fe being retained.

The graphs in figure 5.36.12 show a plot of the percentage of Fe entering each part of the system that is retained in that part. It can be seen that the wetland area consistently retains in the order of 90% of the Fe in the influent water and that the third settling tank performs the least well, occasionally acting as a source of Fe rather than retaining it. Settling tank 2 generally retains about 60% of the influent Fe, and the other components of the system tend to retain between 30 and 50% of the influent Fe.

A plot of the performance of each component of the system in terms of the percentage of total Fe entering the complete treatment system retained in each part is shown in figure 5.36.13

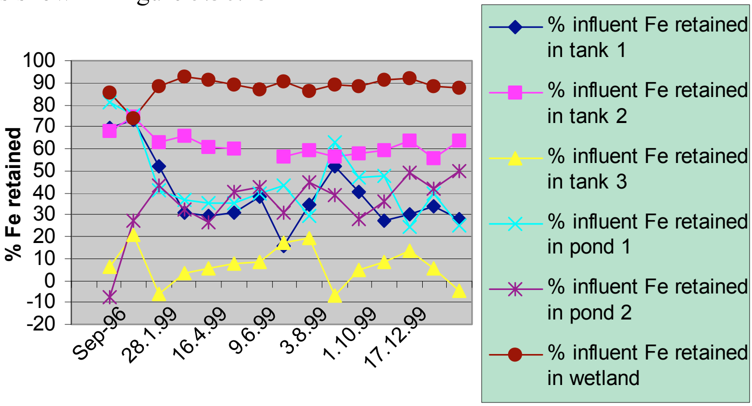

Figure 5.36.12. Percentage of influent iron loading retained in each part of the Woolley treatment system.

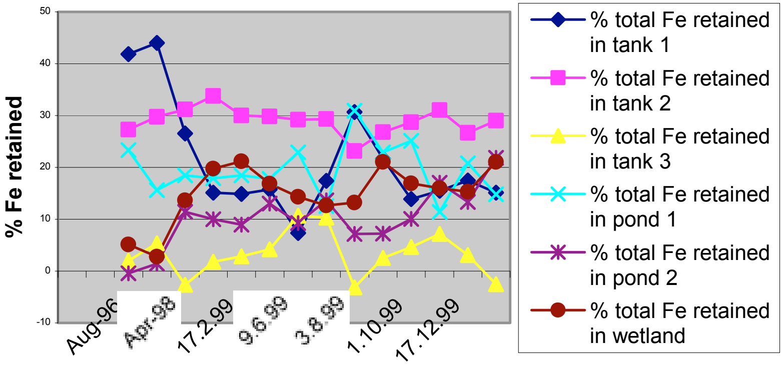

Figure 5.36.13. Percentage of total iron loading entering the Woolley treatment system retained in each part.

The foregoing presentation of data from the Woolley treatment system can be summarised as follows:

- Effective aeration is achieved by the cascades and the primary role of the rest of the treatment system is to facilitate settlement of the particulate iron.
- There is considerable seasonal variation in the degree of settlement of particulate Fe that is achieved through the system. Further study would be required to ascertain reasons for this, but it may be due to weather conditions, for example heavy rainfall and strong winds causing turbulence and discouraging settlement.
- Tank 2 consistently retains the most Fe and tank 3 consistently retains the least. The reason why tank 2 outperforms tank 1 is not clear, since they both receive similar hydraulic loading (approximately 5.3 m/day), and tank 1 actually has a longer theoretical retention time than tank 2 (0.71 days and 0.52 days respectively – see site summary 6.36). The poor performance of tank 3 is probably due to the recombination of flows from tanks 1 and 2, to give a very high hydraulic loading (13.25 m/day) and low retention time (0.21 days).
- The other component parts of the system (i.e. tank 1, ponds 1 and 2 and the wetland) each retain similar quantities of Fe.
- The wetland consistently retains about 90% of influent Fe loading, and in the region of 10 to 20% of the total Fe loading entering the treatment system.
- Overall, the treatment system functions effectively (average 98% treatment efficiency for Fe) and each component part, except tank 3, plays a significant role in treatment.

Costs

There are no figures for the construction costs of the system as it used many pre-existing features, such as the settlement tanks and lagoons, which were modified for the current water treatment process. The wetland construction costs were approximately £5/m^2.

Operating costs are £100,308 per annum, largely due to electricity costs for pumping.

Approximately 13 m^3 of sludge was initially produced per day (Laine 1997) – about 75% of which settles in the concrete tanks (tanks 1, 2 and 3). These tanks needed emptying about every 18 months. Sludge is removed by suction tankers (it is about 4% solids at this stage) to permeable burnt shale lagoons – after a month of reasonable weather, the sludge in these drying lagoons is about 20% solids and can be handled by excavator. Sludge accumulation is now, in 2001, considerably less and it is anticipated that the tanks will need emptying every 5 to 7 years.

5.37 WREN'S NEST, DUDLEY

NGR SO935919

Pre-treatment water chemistry/volume and treatment requirements

The minewater at Wren's Nest was a discrete body of water enclosed in a cavern, under pressure, of a limestone mine. The main contaminant of concern was sulphide; this was in solution but there was concern that instability in the ground would lead to its release as hydrogen sulphide gas, perhaps affecting a nearby canal tunnel which is used by tourist boats. Hydrogen sulphide was at concentrations of approximately 35 mg/l and the volume of the water body was in the region of 40,000 m^3.

The treatment requirement was to remove the H_2S and thus the potential hazard. A pump and treat scheme was necessary, with the cavern being grouted as the water was removed, so that the ground was stabilised.

Site history and design criteria

Biffa entered into negotiations with Dudley Metropolitan Borough in mid-1998 to devise a scheme to safely remove the contaminated water from the flooded, abandoned limestone mine. The overall concept was to pump, treat and re-circulate the water through the cavern so that by a combined process of displacement and dilution the contaminant concentrations would be progressively reduced until the water quality was such that discharge to sewer would be consented. A computer model of the proposed flushing system was constructed, which incorporated the three dimensional shape of the mine, determined by ultrasonic scans, the flow paths of water and the contamination profile through the body of water. The cavern was long and narrow, with higher concentrations of sulphide at one end and alkalinity at the other.

The actual treatment process comprised injecting lime into the minewater as it came up the pipe – this kept the pH high so that the sulphide stayed in solution as the pressure dropped. Next, iron chloride was added, to drop iron sulphide (Fe_2S) out of solution. A dissolved air flotation unit (DAF) was then used to separate the sulphidic sludge for disposal to landfill, with the treated water being discharged to sewer (after suitable quality had been achieved by the recirculation process).

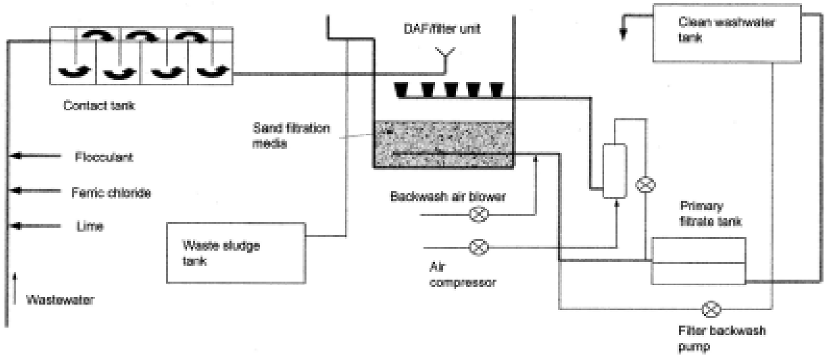

Figure 5.37.1. Schematic of Wren's Nest treatment scheme.

The treatment plant was constructed on site, with the layout being such as to spread the load as much as possible due to the danger of subsidence. An onsite laboratory was incorporated into the design to allow ongoing testwork and monitoring to be carried out.

When treatment started it was found that there was a strong buffering effect, which meant that considerably more lime was used than anticipated. This in turn meant that it was hard to meet the EA consent level of 15 mg/l suspended solids in water being returned to the cavern and the DAF unit was undersized, so the rate of treatment was less than the proposed 100 m^3/day. However, the initial process problems were soon resolved and the target treatment rate was met. The whole treatment of the water body took 21 weeks. It was found to be necessary to recirculate between one and a third to one and a half times to achieve satisfactory treatment. The sulphide concentration in the raw minewater fell over the treatment period, perhaps due to turbulence and mixing. During the final 14 days of treatment, the sulphide concentration in the minewater was negligible and the main treatment required was removal of suspended solids. Treatment was completed in October 1999, at which time the cavern had been emptied of water, all water had been treated to the required standard and the cavern had been filled with grout to prevent subsidence. The treatment plant was removed from site and the site was reinstated.

Monitoring data

This treatment scheme was a one-off pump and treat exercise on which were imposed strict consent limits, which had to be achieved. The summary data, of water quality and consent limits, are presented in table 5.37.1.

Table 5.37.1. Summary water quality data for the Wren's Nest treatment scheme

	Initial water quality mg/l	Discharge consent (for recirculation to cavern) mg/l
H_2S	Up to 35	< 0.2
SO_4	Up to 15	
pH	7	6–9
Suspended solids		< 15
Fe		< 1

Comments on design

This scheme was designed to meet a specific water quality and geotechnical problem and was carried out over a short period of time, thus making it unlike the majority of minewater treatment schemes, where the treatment needs to be carried out over a period of years or decades, not months as in this instance. However, this scheme is a good example of how a specific scheme can be designed for a specific minewater chemistry, and illustrates the importance of carrying out a thorough characterisation of the minewater chemistry prior to planning a treatment scheme.

5.38 YNYSARWED, SOUTH WALES

NGR	Treatment plant	SN807016
	Wetland	SN804011

Pre-treatment water chemistry/volume and treatment requirements

The minewater flowing from the Ynysarwed adit of Lower Ynysarwed Mine is net acidic resulting in pH levels as low as 3.5 following oxidation and hydrolysis of iron in the receiving watercourse (the Neath Canal). Peak flows of 36 l/s were discharged, with over 400 mg/l iron, in early summer 1994. Iron concentrations had declined to approximately 150 mg/l in early 2000, and the long-term value of the fully recovered minewater is predicted to be in the region of 30 mg/l (Younger *et al.* 1996, unpubl., quoted in Ranson *et al.* 2000).

The treatment requirement of this minewater is to raise pH and precipitate out iron. After a process of balancing costs, risks and benefits of the various options (Ranson & Edwards 1997), Neath Port Talbot Borough Council and the project partners (Environment Agency, Welsh Office and the Coal Authority with technical input from Steffen, Robertson & Kirsten (UK) Ltd) identified the preferred treatment option as a combined active and passive system.

Site history and design criteria

The Ynysarwed discharge emerges from Ynysarwed adit which is associated with the workings of the Lower Ynysarwed Mine. Although this mine closed in the 1930s, there was no significant, polluting discharge until the closure of a mine in an adjoining valley in 1991 (the Blaenant Colliery in the Dulais Valley), which resulted in a substantial outflow of contaminated minewater in spring 1993. The minewater discharged into the Neath Canal and initially affected a 12 km length of the canal. Measures were taken to manage the flow regime of the canal; the polluted water was contained in a 7 km stretch before discharge into the River Neath estuary. The discharge was ranked as the worst affected in the National Minewater Ranking List (April 1998) not then receiving treatment.

A project team was formed comprising Neath Port Talbot Borough Council, the Environment Agency, the Welsh Office and the Coal Authority, and steps were taken to secure funds and identify a long-term solution for the problem. A number of constraints were imposed on the scheme by, for example, significant organisational change, a time limited staged grant, land availability and uncertainties of operational funding, consultation and democratic decision making. Conflicts of interest, particularly between angling and industrial interests, increased the complexity of the project (Ranson & Edwards 1997).

The treatment option that was identified (by Hyder Consulting Ltd.) was an active chemical treatment with lime and flocculant dosing, sludge dewatering and disposal off-site, together with a constructed wetland to provide final polishing of the treated minewater prior to its discharge to the River Neath. The River Neath had not previously been affected by any minewater discharge and was of excellent quality, which meant that a stringent Discharge Consent was imposed; the requirements specified a water quality, on dilution in the receiving water course, ten times better than the European Inland Fisheries Advisory Council's (EIFAC) recommended standard for salmonid fisheries (1.0 mg/l dissolved iron). Thus the target to achieve was set at a discharge concentration of 5 mg/l iron and a concentration after dilution of 0.1 mg/l iron.

The design of the treatment option took into account the long-term predicted decline in iron concentration in the minewater discharge from its peak of 400 mg/l to around 30 mg/l. It is anticipated that when the iron concentration has fallen to this predicted residual level, precipitation of iron can be achieved without generating excess acidity, thereby removing the necessity to dose with lime; it therefore should be possible to discontinue the active treatment after about a decade, and achieve satisfactory treatment in the aerobic wetland.

The active treatment plant was constructed (by Birse Construction) in a very confined location between the main road and the Neath Canal, meaning that a pipeline had to be installed across the road to conduct flows from the existing adit. The plant was designed for a flow of 36 l/s and a maximum iron concentration of 250 mg/l. A concrete pumping sump at the head of the process pumps minewater to an above-ground aeration tank, which is 6 m high and fitted with a mechanical surface aerator. This facilitates aeration of the minewater and releases any methane and hydrogen sulphide gas that may be in the minewater, thereby ensuring no hazardous gases enter the process building.

Lime slurry is added to the aeration tank, and a polymer flocculant is added before the minewater enters settling tanks containing lamellar plate separators. Sludge is automatically drawn off at the bottom of the tanks, pumped to a sludge-holding buffer tank and then dewatered by a centrifuge before being discharged to skips. The skips are taken away and emptied every 2 days. The solids content of the sludge is in the region of 30 to 40% solids, depending on the operational pH of the process – current operating pH is pH8, which results in a sludge with approximately 30% solids. The sludge produced each day contains approximately 1.5 tonnes of iron and is currently taken to landfill, although it could theoretically be re-used if a suitable market/end-user could be identified.

A lagoon of 200 m^3 capacity was constructed to provide a temporary storage facility in the event of power failure or for when the plant is shut down for routine maintenance. It is estimated that this lagoon would provide approximately 2 h storage for untreated water or 2 days' storage for sludge.

Figure 5.38.1. Channel taking pumped minewater from treatment plant to wetland (minewater is piped for most of the 1 km distance, with an open channel taking part of the water to the one of the wetland cells).

A 1 km pipe takes treated water, which generally has iron concentrations of 3 to 5 mg/l, to the constructed wetland. The wetland is approximately 1 ha in area and has been constructed to comprise natural shapes. There are two wetland cells, and a liner with under-drainage has been used. Alluvial soil from the site was used as the planting medium and earth bunds were constructed around the wetland and extending into the wetland area to create a sinuous flow path. Existing trees have been retained where possible to help to integrate the wetland area into the landscape. This aim has also been achieved by minimal use of concrete: the inlet structure comprises a cascade incorporating natural stone, and the outlet structure is a pipe through the earth bund to a concrete discharge portal on the river bank.

Figure 5.38.2. The wetland area at Ynysarwed.

Figure 5.38.3. Earth bund separating wetland (right) from the river (left).

6

Existing sites in the UK – Site summaries

INTRODUCTION

This chapter presents summary information for each of the sites considered in more detail in Chapter 5. The information is presented in a standard format to allow quick assessment and comparison of sites to be carried out. Treatment criteria have been calculated using the average values of available data. It must be stressed that these are only guideline values, since the performance of treatment systems will vary both temporally and spatially. Also, in most cases sampling of influent and effluent were made during the same sampling exercise, thereby representing a ‘snapshot’ assessment. In treatment systems where there is a distinctly pulsed flow, the influent and effluent samples are not directly comparable and such a snapshot approach may provide a misleading picture of the treatment performance; for example if a pulse of highly contaminated water happens to be entering the wetland cell at the time of sampling and there has been a considerable period between such pulses, the effluent sample is likely to have a lower contaminant loading than if it were sampled after the appropriate

Minewater Treatment: Technology, Application and Policy.
Melanie Brown, Bob Barley and Harvey Wood. ISBN: 1 84339 004 3

period to represent the retention time of the ‘pulse’ within the cell. However, with the data set available, it is not possible to make allowances for retention time in the comparison of influent and effluent water quality, so this limitation of the calculated performance values should be borne in mind.

When values for capital and operational costs were available a cost benefit analysis has been performed, based on the cost per cubic metre of minewater treated, the cost per kilogram of iron removed and the construction cost per square metre of reed bed. The treatment costs (i.e. for Fe removal and volume of water treated) are calculated on the basis of a 10 year period. Thus, capital costs are spread over the 10 years, and annual operating costs incorporated to obtain an annual cost figure. The sites for which such information was available were mostly those operated by the Coal Authority; the figures used in the calculations are presented in the relevant case studies in Chapter 5, the introduction to which includes comments on the compilation of these figures.

A summary of operating parameters, treatment efficiency and costs is included in Chapter 7.

6.1 ACOMB, NORTHUMBERLAND

NGR NY926663

Minewater type:	Net alkaline drift minewater
Type of system:	Settlement lagoons and reed beds
Year of completion:	Construction commenced August 2001
Owner/Consultant/Contractor:	Coal Authority IMC Consulting Engineers
Pre-treatment, if any:	Aeration

Construction

Area of component parts: (approx. areas scaled from plans)	Lagoons – 750 m^2 Reed beds– 1200 m^2
Layout:	Minewater is pumped from the drift to an aeration tower, and then flows by gravity to two lagoons operating in parallel, followed by two reed beds operating in series
Substrate:	Imported topsoil
Vegetation:	Marginal planting to lagoons and reed beds: *Phragmites australis* (30%) *Iris pseudoacorus* (25%) *Typha angustifolia* (20%) *Typha latifolia* (15%) *Phalaris arundinacea* (10%) Reed beds: *Typha latifolia* (50%) *Iris pseudoacorus* (40%) *Scirpus lacustris* (10%)
Inlet/outlet structures:	Reinforced concrete

Treatment criteria

Average flow/pumped volume:	10.9 l/s
Average hydraulic loading:	Lagoons – 1.26 m/day

Reed beds – 0.78 m/day

Average retention time:

Lagoons – 2 days
Reed beds – 0.4 days

Table 6.1.1. Water quality of Acomb minewater discharge for the period August 2000 to July 2001

n = 8	Average value mg/l	Load kg/day
Flow	10.9 l/s	
pH	6.6	
Alkalinity	239	
Sulphate	550	
Ferrous iron	22.1	
Total iron	32.9	30
Manganese	0.6	

Costs

Construction costs – approximately £350,000

6.2 ALLANTON, SCOTLAND

NGR NS853575

Minewater type: Acidic spoil heap leachate

Type of system: Settlement ponds

Year of completion: Not known

Owner/Consultant/Contractor: North Lanarkshire Council
Babtie Group

Pre-treatment, if any: None

Construction

Layout: Settlement ponds

Treatment criteria

No data are available

6.3 ASPATRIA, CUMBRIA

NGR NY164440

Minewater type:	Acidic spoil heap drainage
Type of system:	Compost wetland
Year of completion:	Due to be constructed in 2001
Consultant:	IMC Consulting Engineers
Comments	Work was due to start on this site, with pilot-scale trials, in 2001, but the foot and mouth outbreak (that started in February 2001) caused this to be delayed

6.4 BENHAR BING, LANARKSHIRE

NGR NS893625

Minewater type:	Acidic spoil heap drainage
Type of system:	Settlement pond and surface flow constructed wetland
Year of completion:	August 1997
Owner/Consultant/Contractor:	Reclamation carried out by Central Scotland Countryside Trust in consultation with Stirling and Edinburgh University
Pre-treatment, if any:	None

Construction

Area of component parts:	Area of settling pond – not known Area of wetland – 0.4 ha
Depth, if known:	Design depth – 300 mm
Layout:	Runoff from bing collected in trenches, diverted to settlement pond then to wetland, which comprises 16 cells divided by straw bales
Substrate:	100 mm layer of crushed limestone overlain by 500 mm layer of spent mushroom compost
Vegetation:	*Typha latifolia* planted initially *Juncus effusus* has colonised
Inlet/outlet structures:	Plywood v-notch weirs

Treatment criteria

Average flow/pumped volume:	Not known
Average hydraulic loading:	Not known
Average retention time:	Not known

Table 6.4.1. Summary data for Benhar Bing constructed wetland performance

n = 13 June 1997 – June 1998	Median inflow mg/l	Median outflow mg/l	Treatment efficiency % (average values)*	Area adjusted removal g/m^2.day (median values)*
pH	2.7	2.8		
Fe	226	119	21	0.8
Mn	66.4	56.0	18	0.01
Al	67.4	43.6	29	
Sulphate	2416	1830	20	-7.1
Acidity	519	298	33	

* values taken from published information (Heal and Salt 1999)

Costs

None available

References/sources of information

Heal & Salt 1999

6.5 BLAENAVON, GWENT

NGR SO242085

Minewater type:	Neutral pH deep coal minewater
Type of system:	Surface flow wetland
Year of construction:	2001
Owner/Consultant/Contractor:	Coal Authority SRK Alun Griffiths
Pre-treatment, if any:	None

Construction

Area of component parts:	Approx. 4000 m^2
Layout:	Three reed bed cells in series
Substrate:	Topsoil
Vegetation:	*Typha*
Inlet/outlet structures:	Minewater is piped from within the adit to a distribution chamber. Three concrete channels facilitate flow into first cell. Three concrete channels between first and second cells and four between second and third cells. Discharge pipe to stream

Treatment criteria	No data currently available
Costs	Construction costs – £301,000 Operating costs – £1560 p.a. Cost per m^2 of reed bed – £75
References/sources of information	SRK 1994

6.6 BOWDEN CLOSE, COUNTY DURHAM

NGR NZ184358

Minewater type:	Acidic spoil heap leachate
Type of system:	Pilot-scale RAPS
Year of completion:	September 1999 – pilot-scale April 2001 – full-scale construction commences
Owner/Consultant/Contractor:	Durham County Council Paul Younger Nuwater Consulting Services Ltd
Pre-treatment, if any:	Stilling pond at inlet to RAPS system and overflows to prevent excessive flow to system

Construction

Pilot-scale system

Area of component parts:	128 m^2
Depth, if known	Water depth typically a few centimetres
Layout:	L-shaped RAPS unit receiving two discharges by gravity flow, followed by an aeration cascade and aerobic pond
Substrate:	0.5 m thick layer of 13 mm diameter single-size limestone gravel, overlain by 0.5 m thick layer of horse manure and straw
Vegetation:	None
Inlet/outlet structures:	Inlet: piped discharges into ‘stilling pond’ Outlet: from RAPS, siphon outfall; from aerobic pond, pipe to stream

Treatment criteria

Data for pilot-scale system

Average flow: 70 l/min (1.17 l/sec)

Average hydraulic loading: 0.79 m/day

Average retention time: 1.3 days, assuming 1 m depth

Table 6.6.1. Water quality data for pilot-scale treatment system at Bowden Close

n = 6 (Younger 2000)	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe	15.9	1.0	93.7	1.51	11.8
Mn	1.30	0.49	62.3	0.08	0.67
Al	5.29	2.12	59.9	0.32	2.5
Alkalinity	24	129			
pH	6.4	7.2			

Costs

Design and construction – £20,000 for pilot scale, approximately £65,000 for full scale

Comments

Daily and hourly sampling also carried out

Acidity removal mean value – 25 g/day.m^2 (equivalent to 25 g/day.m^3, since depth is 1 m): this value used to calculate size of full-scale system which will incorporate the third, larger and more contaminated discharge

References/sources of information

Jarvis and Younger 1997
Younger 1995
Younger 2000

6.7 BULHOUSE, YORKSHIRE

NGR SE214029

Minewater type:	Acidic deep and drift minewater – pumped
Type of system:	Aeration and settlement, in an old quarry
Year of completion:	September 1998
Owner/Consultant/Contractor:	Public/private sector partnership – EA, CA, Barnsley MBC, Hepworth Building Products
Pre-treatment, if any:	Aeration

Construction

Area of component parts:	Lagoon – 7000 m^2 Polishing pond – 582 m^2
Depth, if known:	Lagoon – 7.5 to 8.5 m Polishing pond – 1.5 to 2m
Layout:	Single settling lagoon in a former quarry followed by polishing lagoon
Substrate:	N/A
Vegetation:	N/A
Inlet/outlet structures:	Inlet to lagoon is via an aeration manhole – pumped minewater falls down manhole, deflected by fins within the manhole to aid aeration Outlet by pipes to polishing lagoon then pumped back down to river at abstraction point

Treatment criteria

Average flow/pumped volume:	Variable: 20–80 l/s, average 40 l/s

Average hydraulic loading:

Lagoon – 0.49 m/day
Polishing pond – 5.94 m/day

Average retention time:

Lagoon – 16 days
Polishing pond – 0.3 days

Table 6.7.1. Water quality data for the Bulhouse minewater treatment scheme

n = 7	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Ar re g/
Fe	51.8	2.5	95.2	170	24
pH	5.5	4.6			
Alkalinity	23	3			
Sulphate	658	655			

Costs

Construction costs – £1,200,000
Operating costs – £43,494 p.a.
Cost per kg Fe removed – £2.63/kg
Cost per m^3 water treated – £0.13/m^3

References/sources of information

Dudeney 1997a
Dudeney 1997b
Dudeney, Tarasova, Chan & Demin 2000
Laine 1998
Laine & Dudeney 2000

6.8 CAPHOUSE, WEST YORKSHIRE

NGR SE246162

Minewater type:	Acidic deep minewater
Type of system:	Lime dosing and settlement
Year of completion:	Modified system to be installed 2001–2003
Owner/Consultant/Contractor:	National Coal Mining Museum for England, Coal Authority Consultant for modified system – Wardell Armstrong
Pre-treatment, if any:	Lime dosing

Construction

Area of component parts:	Current system: 2 settling tanks, approx. 1040 m^2 in total Proposed system (subject to final design): Additional treatment facilities: 2 × 2000 m^3 settling tanks 1 × 1500 m^3 balancing tank 2 × 1500 m^2 reed beds 2 × 1250 m^3 drying out ponds
Layout:	Subject to final design
Substrate:	Subject to final design
Vegetation:	Subject to final design
Inlet/outlet structures:	Subject to final design

Treatment criteria

Average flow:	38 l/s

Average hydraulic loading:

Existing lagoons – 3.16 m/day
Proposed reed bed – 1.09 m/day

Average retention time:

Proposed reed bed – 0.3 days

Table 6.8.1. Average water quality data for Caphouse (existing system)

n = 61	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe	50.0	3.07	94.0	154.3	148.4

Costs

Current operating costs – £205,080

References/sources of information

ECS Engineering Services Ltd.2000

6.9 CRAIGENBAY QUARRY, GALLOWAY, SCOTLAND

NGR NX560790

Minewater type: Acidic open pit and stockpile drainage

Type of system: SAPS and planted, anaerobic wetland cells

Year of completion: Summer 1998

Owner/Consultant/Contractor: Forest Enterprise, Peter Norton Associates

Pre-treatment, if any: SAPS acts as pre-treatment to wetland cells

Construction

Area of component parts: Total area (SAPS and wetland cells) – 1371 m^2

Depth, if known: Water depth in wetland cells – 10 to 20 cm

Layout:
Holding/collecting ponds
SAPS
Wetland cells
Area of natural wetland
Discharge to river

Substrate:
SAPS – limestone at base covered by approx 1 m depth of organic material and 1 m depth of water
Wetland cells – 95% pure limestone (40 mm size) – 150 mm depth at base covered by 1.0 m deep soil/peat mix

Vegetation: In wetland cells – *Typha*

Treatment criteria

Average flow/pumped volume: 3.8 l/s – flows highly variable, dependent on rainfall

Average hydraulic loading: Whole system – 0.24 m/day

Average retention time: Whole system – 0.6 days

Table 6.9.1. Water quality at Craigenbay Quarry

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
pH	3.0	5.7			
Fe	30	1	96.7	9.5	6.9

N.B. There is no regular monitoring at this site – the data values used in the table are 'best guess' values provided by Forest Enterprise so can only be regarded as approximations of the treatment performance of the system.

Costs

Design and construction for whole project – £33,200 (about £24/m^2)

Comments

No reliable data currently available

References/sources of information

Norton, Norton and Tyrell 1998

6.10 DALQUHARRAN, AYRSHIRE

NGR NS266017

Minewater type:	Acidic deep mine drift drainage
Type of system:	Aerobic wetland/flow control
Year of completion:	September 1994
Owner/Consultant/Contractor:	Constructed by British Coal/Babtie, managed by IMC
Pre-treatment, if any:	Flow interception; holding tank to regulate flow

Construction

Area of component parts:	Approx. 1400 m^2 wetland
Depth, if known:	Generally about 300 mm but variable according to how much flow is diverted through reed beds
Layout:	Minewater is stored underground in mineworkings and holding tank receives flow from the drift entrance. Stored water is discharged to the burn when flow in the burn is high. Overflow from holding tank flows through reed beds
Substrate:	Soil
Vegetation:	*Phragmites, Typha, Iris, Phalaris, Eleocharis.* More recently *Scirpus* and *Juncus* have been added
Inlet/outlet structures:	Inlet – overflow from minewater holding tank; outlet – pipe to burn

Treatment criteria

Average flow/pumped volume:	Average discharge to burn (Jan 1996 to May 1997) was 9.2 l/s, maximum discharge 54 l/s

Flow to wetland is intermittent and not recorded

Average hydraulic loading: Not known

Average retention time: Not known

Historical data:

1979 – Fe conc 1300 mg/l
1980 – Fe conc 880 mg/l
1989 – Fe conc 326 mg/l
1993 – Fe conc 223 mg/l

Table 6.10.1. Average water quality values at Dalquharran

1996 to 2000	Average values in holding tank mg/l	Average outflow from wetland mg/l	Average discharge to burn mg/l
pH	5.9	6.0	6.1
Fe	147 (range 38–342)	10.4	180 (range 125–227)

Discharge consent limits are 350 mg/l Fe, based on dilution, and pH range 5.0 to 9.0. No calculations on wetland performance are possible as flows into and out of wetland are not recorded.

Costs

Construction costs not available; operating costs are £4560 p.a.

Comments

This system operates on a principle of 'dilute and disperse', with discharge to river only occurring when river flow is high. The wetland provides a 'buffer' for use when the holding tank overflows and discharge to river directly from the tank is undesirable.

References/sources of information

Carter 1994
Ross 1995
Marsden, Kerr, Holloway and Wilbraham 1997

6.11 DODWORTH, SOUTH YORKSHIRE

NGR SE314057

Minewater type:	Acidic spoil heap leachate with, possibly, deep minewater rebound as well
Type of system:	Part of spoil heap reclamation scheme Trial reed bed Settlement lagoon, lime/soda ash addition, reed beds
Year of completion:	Trial reed bed – 1993 Current treatment system – autumn 1996
Owner/Consultant/Contractor:	Barnsley Metropolitan Borough Council Yarningdale Nurseries (funding – DOE Derelict Land Grant, English Partnerships)
Pre-treatment, if any:	Agricultural lime spread on tip site, around perimeter ditch, in and around the stream and on and around the settlement lagoon Soda ash added by way of gabion baskets in lagoon and in perimeter ditches

Construction

Area of component parts:	Reed beds – 1500 m^2 Lagoon – approx. 500 m^2
Layout:	One reed bed upstream of lagoon, one downstream
Substrate:	Topsoil from site enriched with bark compost
Vegetation:	*Typha latifolia*

Treatment criteria

Table 6.11.1. Water quality data for the reed beds at Dodworth for the period February 1997 to March 1998

n = 13	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %
Reed bed 1			
pH	4.5	4.0	
Fe	22	2.4	89.1
Al	43	35	18.6
Ni	0.6	0.7	-16.7
Zn	1.0	1.1	-10.0
Reed bed 2			
pH	3.8	4.1	
Fe	14	1.5	89.3
Al	60	72	-20.0
Ni	1.2	1.4	-16.7
Zn	1.9	1.6	15.8

N.B. No flow data are available for this period so no calculations of contaminant loading or retention are possible. Data supplied by BMBC.

Data collected in 2000:

Average flow/pumped volume: 1.87 l/s

Average hydraulic loading: 0.11 m/day to reed beds

Average retention time: 2.8 days in reed beds

Table 6.11.2. Water quality data for reed bed 2 at Dodworth for the period January to May 2000

n = 5	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
pH	4.4	4.4			
Fe	7.09	0.47	93.4	1.1	0.8
Al	64.7	59.6	7.9	0.8	
Cu	0.11	0.10	9.1	0.002	
Mn	37.9	33.7	11.1	0.7	
Zn	1.85	1.47	20.5	0.1	
$S0_4$	3382	3014	10.9	60.1	

Data from Hine *et al.* 2000, retained quantities based on an average flow of 1.87 l/s, although it must be stressed that flows are highly variable

Costs

Construction costs for the reed beds – £23.00/m^2; operating costs – in the region of £20,000 p.a.

Comments

Main problem seems to be Al, though there is no discharge consent for this. Main failure of discharge consent seems to be pH (consent 6 to 9, discharge from 2nd reed bed 4 to 5)

Discharge consent levels:

pH	6 to 9
Fe	4 mg/l
Ni	0.8 mg/l
SS	30 mg/l
Cu	0.1 mg/l
Zn	1.5 mg/l

Performance data figures are based on mean quality data – more detailed consideration of data is included in case study 5.11

References/sources of information

Bannister 1997
Bannister 1998
Hine, Morris and Raiswell 2000

6.12 EDMONDSLEY, DURHAM

NGR NZ231493

Minewater type:	Net alkaline drift minewater, pumped
Type of system:	Aerobic wetland
Year of completion:	September 1999
Owner/Consultant/Contractor:	Coal Authority SRK Rockbourne Civil Engineering Ltd
Pre-treatment, if any:	None

Construction

Area of component parts:	Wetland cell 1 – 1165 m^2 Wetland cell 2 – 1200 m^2 Wetland cell 3 – 870 m^2
Depth, if known:	Average water depth – 400 mm
Layout:	Three cells, in series, with flow between cells and raw minewater input to each cell
Substrate:	Compost
Vegetation:	*Scirpus* *Typha*
Inlet/outlet structures:	Inlet – cell 1 – gabion wall with flow through it Inlet - cells 2 and 3 – pipe along top of dividing wall (approx. 200 mm diameter) with regularly spaced holes to emit minewater (about 6 holes across width of cell) Flow between cells – approx. 30 mm diameter pipes spaced at approx. 150 mm in concrete wall, for whole height of wall to allow approx. 20 years worth of build-up in cells Outlet – small pipes set in concrete

wall as for flow between cells, discharging into concrete channel and gravity flow back to river (via small amenity pond)

Treatment criteria

Average flow/pumped volume:

4.4 l/s for period 8.9.1999 to 5.5.2000, n = 9

Average hydraulic loading:

0.12 m/day

Average retention time:

3.4 days, assuming sequential flow through all cells

Table 6.12.1. Water quality data for Edmondsley minewater treatment scheme for the period 8 September 1999 to 5 May 2000

n=9	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe (total)	13.4	0.6	95.5	4.9	1.5
Fe (sol)	6.6	< 0.05			
Mn (total)	2.3	1.2	47.8	0.4	0.14
Mn (sol)	2.0	0.9			
pH	6.4	7.1			

Costs

Construction costs – £277,000; operating costs – £24,036 p.a.
Cost per kg Fe removed – £28.93/kg
Cost per m^3 water treated – £0.37/m^3
Cost per m^2 reed bed – £86/m^2

Comments

Difficult to access pumps.
Treatment site is 300 m away from discharge and 45 m higher, which was necessitated by terrain in vicinity of discharge

6.13 FENDER, DERBYSHIRE

NGR SK364752

Minewater type: Net alkaline deep and surface minewater

Type of system: Aerobic wetland

Year of completion: August 2000

Owner/Consultant/Contractor: Coal Authority
SRK
Rockbourne Construction

Pre-treatment, if any: Aeration cascade

Construction

Area of component parts: 0.6 ha reed bed

Depth, if known: 300 mm

Layout: 4 cells in series

Substrate: Imported soil/compost

Vegetation: *Phragmites, Juncus, Scirpus*

Inlet/outlet structures:

Inlet to 1st cell – gabion baskets
Inlet to other cells – pipe along top of dividing wall (approx. 200 mm diameter) with regularly spaced holes to emit minewater (3 across width of cell). These pipes are for by-pass flow to facilitate maintenance on individual cells. Flow between cells through 3 adjustable height portals in dividing walls, and via a plastic perforated pipe which serves as an overflow from each cell and runs the length of the wetland.
Outflow – 3 concrete structures comprising wire grid and adjustable height (by addition of boards to 4th

side of concrete chamber). Also emergency overflow – vertical pipe approx 600 mm higher than current water level

Treatment criteria

Average flow/pumped volume: 25–26 l/s

Average hydraulic loading: 0.36 m/day

Average retention time: 0.8 days

Table 6.13.1. Water quality data for Fender minewater treatment scheme for the period 21.9.2000 to 19.7.2001

n=10 (7 for Mn)	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (total)	9.53	1.85	80.6	16.6	2.8
Fe (sol)	3.85	0.49			
Mn (total)	1.08	0.84	22.2	0.5	0.09
Mn (sol)	1.01	0.77			
pH	6.6	7.2			

Costs

Construction costs – £385,000; operating costs – £21,204 p.a.
Cost per kg Fe removed – £9.85/kg
Cost per m^3 water treated – £0.08/m^3
Cost per m^2 reed bed – £64/m^2

6.14 FOSS, ABERFELDY

NGR NN814 545

Minewater type:	Acidic pumped drainage from active deep barite mine (in a pyritic and sphaleritic ore body)
Type of system:	Pilot scale – SAPS, aerobic wetland, anoxic limestone drain
Year of completion:	August 1998
Owner/Consultant/Contractor:	MI Drilling Fluids UK Ltd Pilot-scale system designed by Newcastle University
Pre-treatment, if any:	Current ‘active’ system involves lime addition and settlement in lagoons – up to 12.5 l/s pumped during 7 hour weekday working shifts

Construction

(of pilot-scale system)

Area of component parts:	SAPS – 1.6 m^2 (2.3 m^2 from 2.6.99) Aerobic cells – 12 m^2
Depth, if known:	SAPS – 1 m deep Aerobic cells, approx. 300 mm
Layout:	Two SAPS in parallel Three aerobic wetland cells in series Two ALD units in series
Substrate:	SAPS comprise horse manure over limestone gravel Wetland cells have thin layer of compost ALDs contain limestone
Vegetation:	Locally-grown *Juncus* in the aerobic cells

Treatment criteria

Average flow: /pumped volume
1 to 3 l/min (base calculations on 2 l/min)

Average hydraulic loading:
To SAPS – 1.3 m/day
To aerobic cells – 0.22 m/day

Average retention time:
SAPS – 19 hours
Aerobic cells – 1.4 days

Table 6.14.1. Water quality data for pilot-scale treatment system at Foss Mine, Aberfeldy

n = 11 during period 9/9/98 to 18/12/98	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
pH	3.91	6.61		
Fe	12.62	0.62	95.1	0.031
Al	11.25	1.07	90.5	0.026
Mn	16.88	12.17	27.9	0.012
Zn	33.48	18.56	44.6	0.039

Costs

Not available

References/sources of information

Carter 1994
Younger 2000

6.15 GILMERTON BING, MID LOTHIAN

NGR NT297682

Minewater type:	Acidic spoil heap leachate
Type of system:	Wetland
Year of completion:	Not known
Owner/Consultant/Contractor:	Not known
Pre-treatment, if any:	None

Construction

Area of component parts:	Wetland approx. 30 m × 30 m
Layout:	Single wetland cell with cross-bunds
Substrate:	Not known
Vegetation:	Sparsely vegetated with *Phragmites*

Treatment criteria

Table 6.15.1. Average water quality data for Gilmerton Bing wetland

	Average inflow mg/l (n = 10)	Average outflow mg/l (n = 22)	Treatment efficiency %
pH	3.8	3.0	
Fe	403	310	23.1
Al	72	88	-22.2

6.16 GWYNFI, NEATH

NGR SS900968

Minewater type:	Alkaline drift mine discharge
Type of system:	Aerobic wetland
Year of completion:	August 1998
Owner/Consultant/Contractor:	Coal Authority Land leased from Forestry Commission Constructed by Llettyshenkin Construction Co. Managed by Wardell Armstrong
Pre-treatment, if any:	None

Construction

Table 6.16.1. Area of component parts of Gwynfi wetland (measured from plans)

	Area m^2	Depth m
Reed bed 1	184	0.3
Reed bed 2	900	0.3
Reed bed 3	276	0.3

Layout:

Part of flow from Corrwg Rhondda entrance (level B) of the drift mine is diverted into a surface flow wetland (reed bed 1). This overflows into reed bed 2, which also receives an additional minewater inflow.
Water from reed bed 2 overflows into reed bed 3 before discharging to the River Nant Gwynfi (realigned to accommodate scheme)
The reed bed cells are constructed using gabion baskets, which facilitate a degree of terracing to make maximum use of the limited land area available

Substrate: Natural soil material

Vegetation:
Reed bed 1 – sparsely vegetated, some *Juncus* around edge of cell
Reed bed 2 – predominantly *Juncus*
Reed bed 3 – predominantly *Typha*

Inlet/outlet structures: Concrete channels

Treatment criteria

Average flow: Up to 10 l/s

Average hydraulic loading:
Whole system – 0.64 m/day
Reed bed 1 – 4.7 m/day
Reed bed 2 – 0.96 m/day
Reed bed 3 – 3.13 m/day

Average retention time:
Whole system – 0.5 days
Reed bed 1 – 0.1 days
Reed bed 2 – 0.2 days
Reed bed 3 – 0.1 days

Table 6.16.2.Average water quality data for Gwynfi

n = 23	Average inflow mg/l	Average outflow mg/l*	Treatment efficiency %	Retained kg/day	Area adjusted removal $g/m^2.day$
Fe	6.2	0.39	93.2	5.0	3.7
pH	6.3	6.9			

* excluding one outlying value.
Sampling of minewater in 1993/4 gave the following results: sulphate 126 mg/l, alkalinity 30 to 50 mg/l.

Costs

Capital costs – £235,000
Operating costs – £1560.
Cost per kg Fe removed – £13.73/kg
Cost per m^3 water treated – £0.08/m^3
Cost per m^2 reed bed – £173/m^2

Comments

There is a limitation to accuracy of calculations of retention etc. due to lack of flow monitoring data and 2 minewater inputs at different points in the reed bed system. It is not possible to do calculations of Fe retention for individual beds due to lack of flow data and extremely variable data

References/sources of information

Ranson, Reynolds and Smith 2000
SRK 1994

6.17 KAMES, MUIRKIRK

NGR NS685262

Minewater type: Net alkaline deep minewater

Type of system: Aeration, filtration, settlement

Year of completion: 2000

Owner/Consultant/Contractor: Coal Authority
Babtie Group
Raynesway Construction

Pre-treatment, if any: Cascade aeration

Construction

Area of component parts: Area of settling tanks – 360 m^2
Total area of reed beds – 4000 m^2

Depth, if known: Settling tanks – 1 m
Reed beds – 500 mm

Layout: Cascade and spillway followed by two settlement pits (in parallel), then into wetland cells – four altogether, two in parallel and two in series, separated by a weir
Straw baffles within cells
Flows recombine at exit for discharge to the river

Substrate: 200 mm depth sand overlain by 400 mm depth topsoil

Vegetation: Planted with *Typha* and *Phragmites* but has been invaded with Reed Canary Grass

Inlet/outlet structures: Concrete, stepped aeration channel to settling tanks; concrete channel and weir structures (adjustable height by adding boards) to first wetland cells; concrete channel, weir and cascade structure between wetland cells;

concrete channel outlet structure with cascade to river

Treatment criteria

Average flow/pumped volume:**:** 16.7 l/s (n = 4)

Average hydraulic loading:
Settling tanks – 4 m/day
Wetland - 0.36 m/day

Average retention time:
Settling tanks – 0.2 days
Wetland - 1.4 days

Table 6.17.1. Water quality data for the Kames minewater treatment scheme

Raw minewater: Fe 12.7 mg/l, pH 6.4

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
1st cells					
Fe	12.4	7.6	38.7	6.9	1.7
pH	7.1	7.5			
2nd cells (October and December 2000)					
Fe	8.7	4.7	46.0	5.8	1.4
pH	7.7	7.7			
2nd cells (May and June 2001)					
Fe	6.5	0.2	96.9	9.1	2.3
pH	7.5	7.7			

Costs

Capital costs – £622,000
Operating costs – £4080
Cost per kg Fe removed – £19.95/kg
Cost per m^3 water treated – £0.13/m^3
Cost per m^2 reed bed – £156/m^2 (this is not a true reflection of wetland costs; much of cost was incurred by construction of the settlement tanks and sludge drying beds)

References/sources of information

Best and Aikman 1983
Carter 1994
Ross 1995

6.18 KIMBLESWORTH, COUNTY DURHAM

NGR NZ260475

Minewater type:	Net alkaline pumped minewater
Type of system:	Pilot-scale high surface area media reactor
Year of completion:	April 1999
Owner/Consultant/Contractor:	Pilot-scale tests carried out at Kimblesworth which is one of the 9 pumping stations in the Durham coalfield operated by the Coal Authority Research by Newcastle University
Pre-treatment, if any:	None

Construction

Area of component parts:	0.9 × 0.9 m each reactor
Depth, if known:	2m Volume of reactors – 1.62 m^3 each
Layout:	See schematic in case study
Substrate:	PVC Cascade Filterpak – plastic trickling filter media, 2 different sizes Reactor A has lower total media surface area than reactor B
Vegetation:	N/A
Inlet/outlet structures:	Inlet – 20 mm diameter pipe direct from rising main of pumping station, flow controlled by ball valves Outlet – collection drain which takes water back to original channel

Treatment criteria

Average flow/pumped volume:
Reactor A – 3.34 l/min
Reactor B – 2.80 l/min

Average hydraulic loading:
Reactor A – 5.94 m/day
Reactor B – 4.98 m/day

Retention time:
Reactor A — 70 to 170 sec (at flow rates 12 and 1.48 l/min respectively)
Reactor B – 110 to 300-460 sec (at flow rates 12 and 0.8 l/min respectively)

Table 6.18.1. Water quality data for pilot-scale treatment scheme at Kimblesworth

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained g/day	Volume adjusted removal g/m^3.day
Fe Reactor A (n = 16)	1.43	0.41	71.3	4.9	3.0 mean and 10.8 at highest loading rate
Fe Reactor B (n = 16)	1.43	0.38	73.4	4.2	2.6 mean and 12.3 at highest loading rate
Mn Reactor A (n= 16)	0.67	0.53	20.9	0.7	0.4
Mn Reactor B (n = 16)	0.67	0.47	29.9	0.8	0.5
pH (n=17)	6.95				
Alkalinity (n=16)	758				
Sulphate (n=19)	308				

Costs

Since this was an experimental, pilot-scale system, it is not particularly relevant to extrapolate costs to a full scale, nor have any budget costs been prepared for such an exercise

Comments

This technique is promising for sites where land availability or acquisition issues make the use of an aerobic wetland problematic There is no information available on the longevity of the reactors – how quickly they become blocked by ochre accretion, how the media is regenerated/removed or disposal of saturated media.

References/sources of information

Jarvis and Younger 2001

6.19 MAINS OF BLAIRINGONE, CLACKMANNAN, FIFE

NGR NS 980973

Minewater type:	Drift mine adit, possibly draining backfilled opencast workings. Net alkaline and ferruginous
Type of system:	Natural wetland into which minewater flow has been diverted
Year of completion:	1995
Owner/Consultant/Contractor:	Minewater discharge diversion carried out by site owner, in consultation with SEPA and Scottish Coal
Pre-treatment, if any:	Cascade to aid aeration

Construction

Area of component parts:	Total wetland area – 6.2 ha
Depth, if known:	Up to 1 m around outflow
Layout:	Survey of flow distribution found poor dispersion within the wetland, with evidence of flow over only 33% of the area and most flow following a channelled path across the area
Substrate:	Existing wetland soil
Vegetation:	Existing wetland vegetation
Inlet/outlet structures:	Minewater discharge ditch intercepted and diverted to natural wetland area

Treatment criteria

Average flow/pumped volume:

Average flow from adit for period April 1994 to September 1997 – 7.8 l/s, n = 35, range 0.09 to 17.0 l/s

Average hydraulic loading:

0.03 m/day, assuming mean flow over 1/3 of the area

Average retention time:

9.2 days assuming average depth of 0.3 m and mean flow

Table 6.19.1. Water quality data for the treatment wetland at Mains of Blairingone for the period September 1997 to January 2001

n = 35	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
pH	6.9	7.4			
Fe (total)	12.5	0.5	96.0	8.1	0.4
Fe(sol)	11.3	0.1			
Alkalinity	102	95			
Aluminium	1.1	0.2	81.8	0.6	0.03

Peak Fe concentration (in 1994) was 118 mg/l. Outlying value of 20.2 mg/l Fe in treated minewater in October 1998 rejected from data set.

Comments

The wetland at Mains of Blairingone is aesthetically very attractive, as it is essentially an area of natural wetland that has been used for minewater treatment. There is thus a high species diversity within the wetland and marginal species such as willow and alder help to blend the wetland area with the surrounding woodland

References/sources of information

Carter 1994
Henton 1998
Marsden, Kerr, Holloway and Wilbraham 1997
Roxburgh and Marsden 1997
Younger 2000

6.20 MINTO, FIFE

NGR NT205950

Minewater type:	Deep mine shaft overflow, net alkaline
Type of system:	Aeration and aerobic wetlands, gravity flow
Year of completion:	July 1998
Owner/Consultant/Contractor:	Coal Authority Entec Kennedy Construction Currently managed by IMC
Pre-treatment, if any:	Aeration

Construction

Area of component parts:	Total land area – 3.5 acres Constructed wetlands – 10,000 m^2
Depth, if known:	200 mm
Layout:	Four wetland cells, 2 in parallel, 2 in sequence
Substrate:	Soil
Vegetation:	*Typha latifolia*
Inlet/outlet structures:	Distribution chamber, with valves controlling inflow Open vertical pipes control outflow – to distribution chamber and discharge to the burn

Treatment criteria

Average flow/pumped volume:	Average shaft discharge is 37.8 l/s, range is 15–45 l/s and maximum is approx. 60 l/s
Average hydraulic loading:	0.52 m/day at 60 l/s 0.33 m/day at 37.8 l/s

Average retention time:

(assuming 20 0mm average depth)

0.4 days at 60 l/s
0.6 days at 37.8 l/s

Table 6.20.1. Water quality data for Minto constructed wetland for the period February 1999 to September 2000, and for two sampling occasions in 2001

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe total n = 22	11.0	4.3	60.9	34.7 at 60 l/s 21.9 at 37.8 l/s	3.5 at 60 l/s 2.2 at 37.8 l/s
Fe sol n = 15	3.1	0.0			
Fe total Aug/Sept 2001, n = 2	13.9	0.54	96.1	69.3 at 60 l/s 49.6 at 37.8 l/s	6.9 at 60 l/s 4.4 at 37.8 l/s
pH n = 19	6.8	7.7			
Alkalinity n = 3	670	620			
Sulphate n = 3	1370	1330			

Costs

Construction costs – £260,000
Operating costs – £2460
Cost per kg Fe removed – £3.56/kg
Cost per m^3 water treated – £0.02/m^3
Cost per m^2 reed bed – £26/m^2

References/sources of information

Younger 2001

6.21 MONKTONHALL, MID LOTHIAN

NGR NT323703

Minewater type:	Deep minewater, pumped
Type of system:	Lime dosing followed by aerobic wetland
Year of completion:	July 1998
Owner/Consultant/Contractor:	Coal Authority Scott Wilson Managed by IMC
Pre-treatment, if any:	Lime dosing and flocculant addition

Construction

Table 6.21.1. Area of component parts

	Area m^2	Depth m	Volume m^3
Settling lagoon 1	2770	5	13850
Settling lagoon 2	1456	5	7280
Reed bed 1	3000	0.4	1200
Reed bed 2	2070	0.4	828
Sludge storage			
4 tanks – each	625	1.3	812.5

Layout:	Two settling lagoons Two reed beds
Vegetation:	*Typha latifolia* *Scirpus* near outlet
Inlet/outlet structures:	Pipes from lime dosing plant into settling lagoons Pipes out of settling lagoons to collection chamber Pumped to reed bed cells;inlet to reed bed cells is a pipe across width of the cells with openings

Outlets from reed beds cells are concrete channels.

Treatment criteria

Average flow/pumped volume:

98 l/s

Average hydraulic loading:

Settling tanks – 2.0 m/day
Reed beds – 1.7 m/day

Average retention time:

Settling tanks – 2.5 days
Reed beds – 0.2 days

Table 6.21.2. Water quality data for Monktonhall treatment system (whole system)

	Average inflow mg/l n = 2	Average outflow mg/l n = 4	Treatment efficiency %	Retained kg/day
Fe	50.9	3.8	92.5	399
pH	6.8	7.7		
Alkalinity	447	352		
Sulphate	1400	1420		

Table 6.21.3. Water quality data for reed beds

n = 1	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe	12.4	0.66	94.7	99.4	19.6
pH	7.7	7.8			
Alkalinity	352	333			
Sulphate	1420	1440			

Costs

Capital costs – £297,000 (for the reed beds)
Operating costs – £97,260.
Cost per kg Fe removed – £0.82/kg
Cost per m^3 water treated – £0.04/m^3
Cost per m^2 reed bed – £59/m^2

6.22 NAILSTONE COLLIERY, LEICESTERSHIRE

NGR SK430085

Minewater type:	Acidic spoil heap leachate
Type of system:	Settlement ponds, balancing ponds, sodium carbonate addition (redundant SAPS)
Year of completion:	Not known
Owner/Consultant/Contractor:	Viridor
Pre-treatment, if any:	None

Construction

Layout:	Collection ditches, balancing ponds, sodium carbonate addition in tubs, settlement ponds
Vegetation:	

Treatment criteria

No data available

6.23 NENTHEAD, CUMBRIA

NGR NY781436

Minewater type:	Zn-rich alkaline minewater from Pb–Zn mine
Type of system:	Pilot-scale, closed limestone dissolution reactor
Year of completion:	August 1998 – run for 3 months
Owner/Consultant/Contractor:	Research carried out by Newcastle University
Pre-treatment, if any:	None

Construction

Area of component parts:	A 2 m × 5 m heavy duty polyethylene bag was used, filled with 4 tonnes of limestone
Substrate:	Limestone
Vegetation:	None
Inlet/outlet structures:	Pipes inserted into polyethylene bag to provide inlet and outlet

Treatment criteria

Average flow/pumped volume:	65 l/hr
Average retention time:	14 hours

Table 6.23.1. Average water quality data for the pilot-scale reactor at Nenthead

n = 8	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained g/day
Zn	7.33	5.76	21.4	2.5
pH (target pH adjustment)	7.5	8.2		

Costs

Pilot-scale tests only, so costs not applicable to possible future full-scale application.

Comments

Research into using ALD to remove Zn as zinc carbonate by raising pH from 7.5 to 8.2

References/sources of information

Nuttall 1997 and 1999
Nuttall and Younger 1999, 2000a and 2000b
Younger 1998

6.24 OATLANDS, PICA, CUMBRIA

NGR NX021213

Minewater type: Acidic spoil heap leachate

Type of system: Wetland

Year of completion: Not known

Owner/Consultant/Contractor: Environment Agency
Site now returned to former land owners

Pre-treatment, if any: None

Construction

Layout: Single reed bed cell with three inlet points and one outflow point

Vegetation: *Typha*

Treatment criteria

No data are available

6.25 OLD MEADOWS, BACUP, LANCASHIRE

NGR SD867238

Minewater type:	Acidic drift mine discharge
Type of system:	pH adjustment with NaOH, settling lagoons, reed bed
Year of completion:	September 1999
Owner/Consultant/Contractor:	Coal Authority IMC Thysson Construction Operated by ECS
Pre-treatment, if any:	NaOH (20% solution

Construction

Area of component parts:	Pond 1 – 2910 m^2 Pond 2 – 1515 m^2 Reed bed - 1825 m^2
Depth, if known:	Pond 1 – 4 m Pond 2 – 2 m Reed bed – 0.5 m
Layout:	2 settling ponds in parallel, followed by reed bed
Substrate:	Soil
Vegetation:	*Typha* – 4680 no *Juncus* – 1280 no *Iris* – 260 no. *Scirpus* – 1280 no
Inlet/outlet structures:	Concrete channels

Treatment criteria

Average flow/pumped volume:	49 l/s (n = 25)
Average hydraulic loading:	Pond 1 – 1.45 m/day Pond 2 – 2.79 m/day Reed bed - 2.3 m/day
Average retention time:	Pond 1 – 2.7 days Pond 2 – 0.7 days Reed bed - 0.2 days

Table 6.25.1. Water quality data for the Old Meadows treatment scheme – whole treatment system

n = 41 (n = 63 for Fe)	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
Fe (total)	30.7	1.3	95.8	124.5
Fe (sol)	13	0.3		
pH	6.3	7.8		
Mn	3.0	1.8	40.0	5.3
Alkalinity	42	121		
Acidity	47	1.8		
Sulphate	515	503		

Table 6.25.2. Water quality data for the Old Meadows treatment scheme – reed bed

n = 6	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (total)	1.7	1.0	41.2	3.0	1.6
Fe (sol)	0	0			
pH	7.6	7.9			
Mn (n = 2)	2.6	2.5	4	0.4	0.2
Alkalinity	150	150			
Acidity	4.5	1.7			
Sulphate	500	500			

Costs

Capital costs – £949,000
Operating costs – £132,696 p.a.
Cost per kg Fe removed – £5.01/kg
Cost per m^3 water treated – £0.15/m^3

Comments

Only suitable land available for the scheme was approx 3 ha of agricultural land with steep gradients

References/sources of information

Laine 1998

6.26 PELENNA, SOUTH WALES

NGR	Whitworth 1	SS798968
	Garth Tonmawr	SS815971
	Phase III	SS799972

Minewater type:	Acidic discharges from deep and drift mines – six discreet sources of which five required treatment
Type of system:	Whitworth 1 – surface flow compost wetland Whitworth A & B and Gwenffrwd – SAPS and aerobic wetland Garth Tonmawr – aerobic/SAPS cells repeated in series
Year of completion:	Whitworth 1 – Oct 1995 Whitworth A & B and Gwenfwydd – April 1998 Garth Tonmawr – March 1999
Owner/Consultant/Contractor:	Whitworth 1 – Neath Port Talbot Borough Council, SRK Whitworth A & B and Gwenfwydd – Neath Port Talbot Borough Council, Paul Younger, Nuwater Consulting Garth Tonmawr – Neath Port Talbot Borough Council, SRK (Funding sources – EA, WDA, EU LIFE Programme, BOC Foundation for the Environment; actual amounts provided by funders were: EA – £200K, Neath Port Talbot Borough Council – £200K, WDA – £500K, EU LIFE – £500K
Pre-treatment, if any:	None

WHITWORTH 1

Construction

Area of component parts:	900 m^2 total
Depth, if known:	Assume 300 mm
Layout:	Four cells in parallel – two cells are surface flow, two cells are subsurface flow Constructed in precast concrete with a geosynthetic base liner
Substrate:	Mushroom compost (2 cells) Wood bark mulch (2 cells)
Vegetation:	*Typha* (2 cells) *Juncus* (2 cells)
Inlet/outlet structures:	Inlet – gravity flow from adit to a flow monitoring chamber with a v-notch, to a flow distribution chamber. Flow to each cell is via a perforated pipe across the width of each cell which discharge onto gabion baskets Outlet – valve structure (see figure 5.26.5 in case study 5.26)

Treatment criteria

Average flow/pumped volume:	3 l/s
Average hydraulic loading:	0.29 m/day
Average retention time: (assuming 300mm deep)	1.0 day

Table 6.26.1. Monitoring data for the Whitworth 1 treatment scheme

Monthly figures October 95 – June 2001	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Cell 1 pH	6.0	6.1			
Cell 1 Fe	24.2	3.96	83.6	1.3	5.8
Cell 2 pH	6.0	5.9			
Cell 2 Fe	24.2	6.24	74.2	1.2	5.2
Cell 3 pH	6.0	6.0			
Cell 3 Fe	24.2	4.4	81.8	1.3	5.7
Cell 4 pH	6.0	6.1			
Cell 4 Fe	24.2	3.52	85.5	1.3	6.0
Whole system pH	6.0	6.1			
Whole system Fe (total)	24.2	4.3	82.2	5.2	5.7
Whole system Fe (dissolved)	22.5	3.3			
Whole system Mn	1.79	1.15	35.8	0.2	0.2
Whole system sulphate	338	292			

WHITWORTH A & B AND GWENFWYDD (PHASE III)

Construction

Area of component parts:	Whitworth A SAPS – 1825 m^2 Gwenffrwd SAPS – 2425 m^2 Whitworth A wetland – 4500 m^2 Gwenffrwd settlement pond – 850 m^2 Gwenffrwd wetland – 2000 m^2
Depth, if known:	Assume aerobic wetland areas are 600 mm deep
Layout:	Whitworth A discharge and Gwenffrwd discharge both pass through similar, parallel systems comprising a SAPS (successive alkalinity producing system) and an aerobic wetland, which then discharge to the river. The Gwenffrwd system also has an area of existing wetland before the SAPS

and a settlement pond after the SAPS
Whitworth B discharge passes into an existing lagoon prior to discharge to the river

Substrate:

SAPS – 450 mm layer of bark mulch and cow manure/straw over a 600 mm layer of limestone
Aerobic wetland areas – existing substrate

Vegetation:

Natural colonisation of SAPS
Existing vegetation and natural colonisation in the wetland areas

Inlet/outlet structures:

Concrete distribution channels – recent modifications have attempted, where possible, to replace pipes with open channels to try to overcome pipe blocking problems

Treatment criteria

Average flow/pumped volume:

Whitworth A – 8.3 l/s
Whitworth B – 0.6 l/s
Gwenffrwd – 8.9 l/s

Average hydraulic loading:

Whitworth A SAPS – 0.39 m/day
Gwenffrwd SAPS – 0.32 m/day
Whitworth A wetland – 0.16 m/day
Gwenffrwd settlement pond – 0.9 m/day
Gwenffrwd wetland – 0.38 m/day

Average retention time:

Target retention time in SAPS – minimum 14 hours
Whitworth A wetland – 3.8 days (assuming 600 mm deep)
Gwenffrwd settlement pond – 1.1days (assuming 1 m deep)
Gwenffrwd wetland – 1.6 days (assuming 600 mm deep)

Table 6.26.2. Spot sampling for the Pelenna Phase III treatment systems, carried out in 1998 (Younger 1998a)

n = 1		Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
Whitworth A	Fe total	81.7	0.16	99.8	58.5
	Fe sol	81.7	0.09		
	pH	6.0	7.2		
Whitworth B	Fe total	5.38	1.46	72.9	0.2
	Fe sol	5.38	0.11		
	pH	6.4	6.6		
Gwenffrwd	Fe total	13.1	0.94	92.8	9.4
	Fe sol	12.9	0.21		
	pH	5.0	7.2		

Table 6.26.3. Whitworth A treatment system monitoring data, April 1998 to June 2001

	Average inflow mg/l (n = 114)	Average outflow mg/l (n = 26 for SAPS, 50 for final discharge)	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
SAPS					
Fe	59.7	8.5	85.8	36.7	20.1
Mn	0.85	1.0			
Sulphate	334	327			
pH	5.8	6.6			
Wetland					
Fe	8.5	1.7*	80.0	5.4	1.2
Mn	1.0	0.5			
Sulphate	327	297			
pH	6.6	6.6			

* excluding one outlying value of 28 mg/l recorded in July 1998

GARTH TONMAWR

Construction

Area of component parts:	Cell 1 wetland/settlement lagoon – 2480 m^2 Cell 2 compost cell – 970 m^2 Cell 3 aerobic wetland – 980 m^2 Cell 4 compost cell – 980 m^2 Cell 5 Aerobic wetland – 960 m^2
Depth, if known:	Cell 1 water depth 400 mm, soil depth 300 mm Cell 2 compost depth 400 mm, flooded to surface of compost Cell 3 water depth 500 mm, soil depth 300 mm Cell 4 as cell 2 Cell 5 as cell 3 N.B. these are the theoretical, design depths. In fact, depths in the compost cells, 2 and 4, are up to 1 m due to lack of flow through the substrate
Layout:	Cells in series – as above and figure 5.26.13
Substrate:	Cells 1, 3 and 5 – site sourced soil material Cells 2 & 4 – imported compost material
Vegetation:	No planting was carried out – cells have become naturally colonised with wetland plants where water depth allows.
Inlet/outlet structures:	Inlet to cell 1–3 m wide cascade Between cells 1 and 2, and 3 and 4 – gaps in brick walls Outlet to cells 2 and 4 – perforated pipe along width of cell, below surface of compost with 3 pipes through wall to

cascades as inlet to cells 3 and 5, although water levels are such that flow is also occurring over the top of the walls
Outlet from cell 5 – concrete channel to outlet chamber

Treatment criteria

Average flow/pumped volume:

22.6 l/s (since 1996, n = 62)

Table 6.26.4. Theoretical hydraulic loading/retention times

	Depth m	Hydraulic loading m/day	Retention time days
Whole system	0.6	0.31	2.0
Cell 1	0.4	0.79	0.5
Cell 2	1.0	2.01	0.5
Cell 3	0.5	1.99	0.3
Cell 4	1.0	1.99	0.5
Cell 5	0.5	2.03	0.2

The Environment Agency carried out trace tests using flourazine dye and found that actual retention times were less than theoretical, due to preferred pathways and incomplete mixing of water in the free-water surface, flooded compost cells. The retention time in cell 1 was greater than 8–9 hours, while in the flooded compost cells it was only about 2 hours.

Table 6.26.5. Summary data for Fe and pH for the period March 1999 to June 2001

n = 27 for Fe, 17 for pH		Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Cell 1	Fe	28.8	11.2	61.0	34.3	13.8
	pH	5.42	4.3			
Cell 2	Fe	11.2	3.5	68.8	15.1	15.6
	pH	4.3	6.5			
Cell 3	Fe	3.5	3.1	12.9	0.9	0.9
	pH	6.5	6.2			
Cell 4	Fe	3.1	0.8	74.8	4.5	4.5
	pH	6.2	6.6			
Cell 5	Fe	0.8	2.0	-158.4	-2.4	-2.5
	pH	6.6	6.5			
Whole system	Fe	28.8	2.0	93.1	52.4	8.2
	pH	5.4	6.5			

Costs

Capital costs:
Whitworth I – £214,314
Garth Tonmawr – £223,534
Phase III – £360,198

References/sources of information

Acer Environmental Ltd 1995
Collins 1995
Edwards 1995
Edwards and Maidens 1995
Edwards, Bolton, Ranson and Smith 1997
James, Elliott and Younger 1997
Price 1997
Pryce Consutant Ecologists 1994
Ranson, Reynolds and Smith 2000
Rees, Bowell, Dey and Williams 2001
Roberts 1996
Younger 1998
Younger, Large and Jarvis 1998

6.27 POLKEMMET, WEST LOTHIAN

NGR NS933640
New system

Minewater type:	Deep minewater, pumped
Type of system:	Hydrogen peroxide addition, settlement and reed beds
Year of completion:	New system commissioned November 1999
Owner/Consultant/Contractor:	Coal Authority IMC/ SRK/ SWK
Pre-treatment, if any:	Hydrogen peroxide addition (35%) Flocculant addition

Construction

Table 6.27.1. Area of component parts

	Area m^2	Depth m	Volume m^3
Primary pond 1	850	2.5	2125
Primary pond 2	850	2.5	2125
Secondary pond	1125	2.3	2588
Wetland	3200	0.4	1280

Depth, if known:	Assume depth of wetland 400 mm for calculations
Layout:	Minewater is pressure injected with hydrogen peroxide and pumped to primary settlement ponds which operate in parallel (newly constructed), flocculant is added; Flow then recombines in secondary settlement pond and then flows by gravity to pre-existing wetland area around which a bund of colliery spoil has been constructed

	Treated water then discharges to the burn Old settlement ponds and the serpentine lagoon are now used for sludge storage.
Substrate:	Existing soil
Vegetation:	Existing vegetation
Inlet/outlet structures:	Pipes

Treatment criteria

Average flow/pumped volume:	Pumping capacity 68 l/s (no data currently available)
Average hydraulic loading: (assuming 68 l/s, even split between primary ponds 1 and 2)	Primary pond 1 – 3.46 m/day Primary pond 2 – 3.46 m/day Secondary pond – 5.22 m/day Wetland – 1.84 m/day
Average retention time:	Primary pond 1 – 0.7 days Primary pond 2 – 0.7 days Secondary pond – 0.4 days Wetland – 0.2 days

Table 6.27.2. Summary water quality data for Polkemmet wetland

Nov 99 – Feb 2000 n = 29	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Wetland Fe total	2.66	0.31	88.3	13.8	4.3
Wetland Fe sol	0.23	0.00			
Wetland pH n = 4	7.0	7.4			
Burn upstream of treatment system – Fe	1.67				
Burn downstream of treatment system – Fe	0.50				

Table 6.27.3. Summary water quality data for Polkemmet treatment system

March 2000 – October 2000 n = 31	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Raw minewater Fe total	48.9				
Raw minewater Fe sol	44.3				
Wetland Fe total	1.31	0.14	89.3	6.9	2.1
Wetland Fe sol	0.04	0.00			

Costs

Capital costs – £198,000
Operating costs – £205,368
Cost per kg Fe removed – £2.15/kg
Cost per m^3 water treated – £0.11/m^3

References/sources of information

Carter 1994
Henton 1998
Marsden, Kerr, Holloway and Wilbraham 1997
Younger 2001

6.28 QUAKING HOUSES, COUNTY DURHAM

NGR NZ190508

Minewater type: Acidic spoil heap leachate

Type of system: Surface flow compost wetland (too little head for a SAPS)

Year of completion: Pilot – February 1995
Full scale –November 1997

Owner/Consultant/Contractor: Site provided by British Coal Property
Design by Newcastle University
Community involvement (Quaking Houses Environmental Trust)

Pre-treatment, if any: None

Construction

Area of component parts: 440 m^2

Layout: Embankments made from compacted pulverised fuel ash
Wetland divided into two cells by PFA central weir, 2nd cell 0.4 m lower than 1st

Substrate: Limestone at far end of wetland, cattle manure, municipal waste compost and horse manure (40:30:30 ratio) for main body of wetland – 300 to 500 mm deep

Vegetation: Mixed wetland species

Inlet/outlet structures: Inlet – 100 mm diameter siphon pipe from culvert (the original discharge point of minewater into the Stanley Burn), overflow from culvert to original stream channel
Outlet – 150 mm diameter pipe from end of wetland to an effluent channel and back to original stream channel

Adjustable 90° bend at end of effluent pipe, on the wetland side, allows control of water level

Treatment criteria

Average flow/pumped volume:

127 l/min
Range – 30 l/min to 420 l/min (rapid fluctuation during storm events)

Average hydraulic loading:

0.42 m/day

Average retention time:

(use 300 mm for calculations)

0.7 days

Table 6.28.1. Average water quality data for Quaking Houses constructed wetland

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Fe (n=65)	4.55	2.48	45.4	0.4	0.9
Al (n=61)	6.6	2.4	63.0	0.8	1.7
Acidity (n=32)	55.0	28.3	48.5	4.9	10.4
pH (n=54)	6.16				
Mn (n=65)	3.58				
Sulphate (n=56)	845				

Costs

Construction costs – £18,000
Operating casts are estimated to be approximately £500 p.a.
Cost per kg Fe removed – £15.75/kg
Cost per m^3 water treated – £0.03/m^3
Cost per m^2 reed bed – £41/m^2

References/sources of information

Jarvis and Younger 1999
Younger 1998
Younger, Curtis, Jarvis and Pennell 1997
Younger, Large and Jarvis 1998

6.29 RENISHAW PARK, SOUTH YORKSHIRE

NGR SK427377

Minewater type:	Acidic spoil heap leachate
Type of system:	Permeable reactive barrier
Year of completion:	November 1998
Owner/Consultant/Contractor:	Coal Authority IMC Consulting Engineers
Pre-treatment, if any:	None

Construction

Area of component parts:	Trench 1 – 40 m long × 2.3 m deep Trench 2 – 25 m long × 2.3 m deep Trench 3 – 40 m long
Depth, if known:	
Layout:	Downflow through trench 1, upflow through trench 2 and horizontal flow through trench 3
Substrate:	Trench 1 – ornamental bark (60%), limestone (10%) and cattle manure (30%)
Vegetation:	Mixed wetland vegetation in trench 3

Treatment criteria

No data are available

6.30 SHILBOTTLE, NORTHUMBERLAND

NGR NU 220077

Minewater type:	Acidic spoil heap leachate
Type of system:	Aerobic wetland (redundant) Reactive barrier (planned
Year of completion:	Wetland – 1995 Barrier – 2001/2
Owner/Consultant/Contractor:	Northumberland County Council Newcastle University
Pre-treatment, if any:	None
Construction	A permeable reactive barrier is planned for this site, the final details of which have yet to be finalised
Treatment criteria	Fe and Mn up to 500 mg/l, pH low
References/sources of information	Amos 1999

6.31 SKINNINGGROVE, CLEVELAND, NORTH YORKSHIRE

NGR NZ713195

Minewater type: Deep ironstone mine, ferruginous, net-alkaline, brackish

Type of system: Surface catalysed oxidation of ferrous iron (SCOOFI) within a chamber filled with trickle filter media

Year of completion: Pilot plant – autumn 1999
Full scale - autumn 2000

Owner/Consultant/Contractor: Pilot plant constructed by Cleveland Potash Ltd, monitored by local residents with guidance from Newcastle University

Pre-treatment, if any: None

Construction

Substrate: Pilot plant – gabion baskets filled with blast furnace slag
Full scale – plastic filter media

Vegetation: None

Treatment criteria

Average flow/pumped volume: Estimated flow from Carlin Howe Mmine and Loftus Mine – 24 and 22 l/s respectively

Influent Fe concentration 15–18 mg/l. Pilot plant achieved on average 50% iron removal in 9 m length of flow through 0.5 m^2 cross-sectional area

References/sources of information Younger 2000c
www.minewater.net/scoofi

6.32 ST HELEN AUCKLAND, COUNTY DURHAM

NGR NZ 199268

Minewater type:	Net alkaline deep minewater
Type of system:	Aerobic wetland
Year of completion:	August 1999
Owner/Consultant/Contractor:	English Partnerships Paul Younger, Newcastle University
Pre-treatment, if any:	None

Construction

Area of component parts:	Not known
Layout:	Area of existing wetland was contained by a clay bund, with a French drain around outside to prevent subsurface leakage
Substrate:	Existing wetland substrate
Vegetation:	Existing wetland vegetation
Inlet/outlet structures:	Pipework to distribute flow along northern edge of wetland, and to collect water exiting wetland into pipe line to take it to the River Gaunless

Treatment criteria

Average flow/pumped volume:	Flow up to 2000 m^3/day (23 l/s)

Table 6.32.1. Average water quality data for St Helen Auckland wetland

	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day
Fe (n = 4)	2.53	0.2	92.1	4.6
Alkalinity (typical)	500	480		
pH (typical)	6.3	6.7		

Costs

Not known

References/sources of information

Younger, *St Helen Auckland Mine Water Wetland*

6.33 TAFF MERTHYR, SOUTH WALES

NGR SO170020

Minewater type: Net alkaline deep mine discharge

Type of system: Settlement tanks followed by aerobic wetland

Year of completion: December 2000

Owner/Consultant/Contractor:
Coal Authority
PB Kennedy & Donkin
(part of Taff Bargoed Millemium Park scheme – a major land reclamation project by Merthyr Tydfil County Borough Council funded by the Welsh Development Agency and millennium funding)

Pre-treatment, if any: Aeration cascades

Construction

Area of component parts:
Total site area – 3 ha
Settlement tanks – 2 in parallel, 80 m long
Wetland – 16 cells, total area 1.8 ha

Layout:
Settlement tanks, cascades, reed beds
Two shaft emissions
No sludge drying facilities on site – calculated emptying interval for tanks is 5 years – excess water will be pumped from one to the other in summer and sludge left to dry for a while before off-site disposal

Substrate: Bentonite liners with 400 mm depth topsoil

Vegetation:
Phragmites and *Typha* in 70:30 mix
Nursery grown (Merton Hall)

Inlet/outlet structures: Inlet to reed bed cells comprise concrete channel across width, with openings. Openings are levelled with neoprene flanges to regulate flow. Outlets comprise openings in concrete wall across width of cell with vertical bars to allow infill of opening as sediment accumulates – 1 m accumulation allowed for. Temporarily using a geotextile membrane for filtration at final discharge from final settlement tanks – were using wool-based filter material as at Woolley but it blocked almost instantly

Treatment criteria

Average flow/pumped volume:

Approx. 60 l/s

Average hydraulic loading:

0.29 m/day

Average retention time:

1 day

There are currently no monitoring data available for Taff Merthyr as it has only recently been fully-commissioned.

Costs

Capital costs – £1,625,000
Operating costs – £18,888
Cost per m^2 reed bed – £90/m^2 (this figure includes capital costs for the settlement tanks, so is an over-estimate of the costs of the reed beds alone. No separate figures are available for reed bed construction)

References/sources of information

Derelict coal mine site transformed *Mining Environment Management* **9(1)** 7

Environment Agency website – 29/11/200 – Millennium Park Reed Bed Scheme takes shape SM128/00LR

6.34 TAILRACE LEVEL, DURHAM (FRAZER'S GROVE)

NGR NY896440

Minewater type: Zn-rich alkaline minewater from Pb–Zn–Fe mine

Type of system: Temporary measure – holding lagoons to which lime and straw bales were added.
Closed-system limestone dissolution reactor (ALD)

Year of completion: February 2000

Owner/Consultant/Contractor:
Durham Industrial Minerals Ltd
Environment Agency
Newcastle University

Construction

Interim measure comprised lagoons to which were added lime and straw bales. Subsequent to this an anoxic limestone drain was constructed on the site of the lagoons, using limestone chippings sealed with plastic and soil to create anoxic conditions

Treatment criteria

Average flow/pumped volume: When mine was working, average pumped volume was 1895 m^3/day (22 l/s) yielded from four inter-connecting mines

Water quality: Average water quality of minewater:

Zn	up to 40 mg/l
Mn	up to 62 mg/l
Fe	up to 10 mg/l

Costs

No costs available

Comments

Long-term Zn predicted to be approx 4 mg/l

References/sources of information

Johnson and Younger 2000
Younger 2000b
Environment Agency website:
Agency’s emergency work to save stream in County Durham, 5 Jan 2000
Zinc-removing bacteria combat mine pollution, 19 Nov 1999
Agency works with site owner to control minewater discharge at Frazer’s Grove, 2 Sept 1999
Environment Agency plans urgent works to prevent minewater damaging stream, 20 Aug 1999
University study sheds new light on pollution risk from Frazer’s Grove, 11 Nov 1998

6.35a WHEAL JANE, CORNWALL
(active system)

NGR SW775428

Minewater type:	Acidic metal mine drainage
Type of system:	Active lime dosed, high density sludge process
Year of completion:	October 2000
Owner/Consultant/Contractor:	Environment Agency Hyder (now United Utilities)
Pre-treatment, if any:	Lime dosing, aeration, sludge recirculation
Construction	Active system comprising a system of tanks in which successive stages of the treatment are carried out. Minewater is pumped from the shaft to the treatment system. Also water from the toe drain and supernatant of tailings dam and pilot passive treatment plant is treated in this system.

Treatment criteria

Average flow/pumped volume:	In the shaft there are 6 pumps each with a 55 l/s capacity, to give a current maximum of 330 l/s from shaft. There is pipework to add two extra pumps – maximum capacity of the system is 440 l/s. Current discharge consent is for 350 l/s to river (treated water is also used for mixing lime slurry, carrying polymer and washing down).

Treatment process

Two identical process streams operate in parallel to allow for closure and maintenance of one stream when necessary.

The principle of the treatment process is that recirculation of sludge allows the formation of larger particles which have better settling characteristics than the rather 'fluffy' precipitate typical from lime dosed treatment alone.

Pumped minewater enters the first tank where recirculated sludge is added at a ratio of between 50:1 and 25:1 (volumetrically sludge addition accounts for about 10% of the inflow). Mixers within the tank keep minewater and sludge agitated. Addition of sludge raises pH to about 8.5.

The minewater/sludge mixture enters the second tank by flowing under the dividing wall. In the second tank aeration is carried out by an air diffuser at the base of the tank with a vertical turbine paddle above to break up bubbles. Approx. 18 tonnes/day of lime is added in this tank from a constantly recirculating loop dosing system in the form of a 5% lime slurry.. Vigorous agitation in this tank prevents the formation of a 'fluffy' flocculant. The operating pH is 9.25.

An offtake pipe in the corner of second tank takes the minewater/lime/sludge mixture, by gravity flow, through a pipe with a mesh to break up the flow before addition of a polymer flocculant (Magnafloc 10 is currently used at a dosing rate of 2.5 g/l). The mixture than flows back up the pipe where the flow is split into two and directed into two parallel clarifiers (for each stream).

The clarifiers comprise lamellar plates which give a total surface area of about 50 m^2 for each tank, thus providing a much increased settling area. Solids settle out in the clarifiers and are drawn off at the base by a rotating rake. And are then either recirculated to the first tank, or piped to the sludge holding tank. Supernatnant water decants over the overflow launders into a central channel and flows to the treated water holding tank.

Sludge in the sludge holding tank settles to over 30% solids in 5 hours and can go up to 40% solids. It is then piped to the tailings dam. Stability tests carried out on the sludge reveal that it is relatively stable in terms of mobility of metals it contains, with a low permeability and a high pH micro-environment with some residual lime. Filter press trials have shown that a 70% solids sludge could be obtained if the tailings dam were no longer available and off-site disposal had to be considered.

Water from the treated water holding tank is either used for wash down, lime mixing or to carry the polymer flocculant, or discharged direct to the river.

Retention times within the system are approximately 30 minutes in the tanks and 25 min in the clarifiers.

Table 6.35a.1. Typical water quality for the Wheal Jane active treatment system

	pH	TSS mg/l	Fe mg/l	Zn mg/l	Mn mg/l	Cu mg/l	Ni mg/l	Cd mg/l
Influent	3.8	3	159	44	5	0.4	0.4	0.056
Effluent	9	5	1.3	0.4	0.2	<0.01	<0.1	<0.001

Comments

Treatment system is currently being run on 10 year contract by Hyder (now United Utilities) on behalf of the Environment Agency. Payment is made to Hyder per litre of minewater treated plus a management fee. The contract specification for discharge is tighter than the EA discharge consent; the operators are not paid for any treated water that fails to meet these contract levels. Water that does not comply with the EA discharge consent could result in prosecution of the operators.

6.35b WHEAL JANE, CORNWALL

(passive treatment pilot plant)

NGR SW775428

Minewater type:	Acidic metal mine drainage
Type of system:	Pilot passive system comprising aerobic reed beds, anaerobic cells and rock filters, with varying pre-treatment
Year of completion:	1994 Modifications (additional pipework to allow alteration of flow paths, re-instatement of the lime-dosed and lime-free system anaerobic cells) were carried out between October 1996 and January 1997
Owner/Consultant/Contractor:	Project managers – Environment Agency Consultants – Knight Piesold Contractor – E Thomas Construction Ltd
Pre-treatment, if any:	3 systems: Lime-dosed Anoxic limestone drain No pre-treatment

Construction

Area of component parts:	Aerobic cells in lime dosed and ALD system – 5 × 260 m^2 Aerobic cells in lime-free system – 5 × 1000 m^2 Anaerobic cells – area 1000 m^2 Rock filters in lime dosed and ALD systems – 550 m^2 Rock filter in lime-free system – 800 m^2

Depth, if known:	Aerobic cell water depth < 300 mm
Layout:	3 systems have same layout of aerobic cells, anaerobic cell, rock filter – with 3 different pre-treatments as described above
Substrate:	Aerobic cells – tailings from Wheal Jane mill processing plant (from South Crofty ore) Anaerobic cells – sawdust, hay and cattle manure Rock filters – crushed granite, approx 100 mm size
Vegetation:	Aerobic cells planted with 50:50 mixture of *Phragmites australis* and *Typha latifolia* at a density of 4 plants/m^2 with exception of cells 1 and 2 of the lime-free system planted at 1 plant/m^2. Also 100 plants of *Scirpus* planted in aerobic cells of lime dosed and lime-free system Most of planting stock was commercially sourced pot-grown stock, although some locally sourced seed was propagated for use in the ALD and lime-dosed system.
Inlet/outlet structures:	Minewater transported to pre-treatment components in approx. 150 mm diameter pipe. Aerobic cell inlet and between cells comprise concrete spillways approx 450 mm wide × 3 m long, with slight drop in height (generally < 150 mm) from spillway to water level in cell. Outlet from final aerobic cell comprises orifice plate, in which adjustable orifice size can be used, to anaerobic cell, which feeds into

perforated pipe within substrate of the cell. By-pass spillway takes excess water to the outflow point of rock filter and thence to final collecting tank.

Outflow from the anaerobic cell comprises a flexible 50 mm pipe, the height of which can be adjusted to regulate flow and depth of water within the anaerobic cell. This pipe feeds a collecting tank which overflows via a v-notch into rock filter.

Outflow from rock filter is 150 mm diameter pipe set vertically at desired height, which takes effluent to final collecting tank, from where it is pumped back to the main mine site for conventional, active treatment.

Treatment criteria

Average flow/pumped volume:

Flow through system is controlled by ball valves at inflow points and measured by v-notches, so that required flow can be set and adjusted.

During operating period up to 1998, flows into aerobic cells were varied between 0.1 to 0.6 l/s, with short periods of up to 1.9 l/s flow into the lime-free aerobic cells.

Design flow rates for lime dosed and ALD system were 0.5 l/s, and for lime-free system 0.7 l/s.

Flows into anaerobic cells were generally 0.1 l/s, with short periods of increase up to 0.3 l/s.

When operation recommenced in 1999, initial flows into aerobic cells were 0.3 l/s for one week which were then reduced to 0.1 l/s. Inflow to aerobic cells was increased to 0.2 l/s in June 2000.

Average hydraulic loading:

At design flow rates, hydraulic loading to aerobic cells of lime dosed and ALD systems – 0.03 m/day, and to aerobic cells of lime-free system hydraulic loading – 0.01 m/day

Average retention time:

Tracer tests carried out in aerobic cells of lime-dosed system, when inflow rate was 0.2 l/s, gave the following results:
Determined residence time – 9.2 days
Theoretical residence time – 19.1 days
According to the design criteria, residence time within whole of each system was estimated to be about 2 weeks (EA 1998 report)
At design flow rates and depth of 0.2 to 0.3 m, the theoretical retention time in the aerobic cells is 6 to 9 days in the lime-dosed and ALD systems, and 16 to 25 days in the lime-free system.

Table 6.35b.1. Summary data for the period when the plant was operated by Knight Piesold is presented in the Environment Agency (1998) Final Appraisal Report, and is further summarised below:

	Median inflow mg/l June 1995 to Jan 1998 (whole monitoring period)	Median Outflow mg/l Jan 1997 to Jan 1998 (post-modifications)	Treatment efficiency %	Retained kg/day[1]	Area adjusted removal $g/m^2.day$[1] (aerobic cells)
Lime dosed system					
pH	3.8	3.4			
Total Fe Aerobic cells only Whole monitoring period	142	31	78.0	2.9	2.2
Total Fe Whole system	142	23	83.8		
Al	50	26	48		
Mn	23	19	17		
As	2.68	0.03	98.8		
Zn	75	50	33		
Cu	0.41	0.27	34		
Sulphate	1742				
ALD system					
pH	3.9	3.8			
Total Fe Aerobic cells only Whole monitoring period	149	17.5	88.3	3.4	2.6
Total Fe Whole system	149	5	96.6		
Al	50	5	90		
Mn	23	15	35		
As	2.57	0.007	99.7		
Zn	75	23	69		
Cu	0.42	0.05	88		
Sulphate	1720				

Table 6.35b.1. (continued)

	Median inflow mg/l June 1995 to Jan 1998 (whole monitoring period)	Median Outflow mg/l Jan 1997 to Jan 1998 (post-modifications)	Treatment efficiency %	Retained kg/day[1]	Area adjusted removal g/m^2.day[1] (aerobic cells)
Lime-free system					
pH	4.0	3.2			
Total Fe Aerobic cells only Whole monitoring period	140	34.5	75.4	6.4	1.3
Total Fe Whole system	140	28	79.3		
Al	50	39	22		
Mn	23	19	17		
As	2.49	0.019	99.2		
Zn	72	54	25		
Cu	0.40	0.29	27		
Sulphate	1694				

[1] Assuming flow rate of 0.3 l/s into lime dosed and ALD systems (which is less than design flow rate, but was the flow rate for most of the post modification period) and 0.7 l/s into lime-free system. In reality, flow rates were varied throughout the operational period

Costs

Design and construction costs – £1,300,000, maintenance costs – £100,000

Cost per kg Fe removed (assuming flows as [1] above): Lime dosed system – £67.76/kg; ALD system – £56.77/kg; Lime-free system – £30.89/kg
Cost per m^3 water treated: Lime dosed system – £8.10/m^3; ALD system – £8.10/m^3; Lime-free system – £3.47/m^3
Cost per m^2 reed bed: Lime dosed and ALD systems – £333/m^2; lime-free system – £87/m^2

NB These costs spread maintenance costs evenly between the three treatments, whereas in reality, the lime dosed system should have higher costs to allow for cost of lime and maintenance of lime dosing equipment, and the ALD system maintenance should be higher than the lime-free system to allow for cost of routine work to keep pipework unblocked, backflushing the ALD due to blockages etc.

Comments

The pilot passive treatment plant was operated from November 1994 to January 1998 to assess the potential of a passive treatment system for all the Wheal Jane mine water discharge. The final report by the consultants concluded that a passive system was not a viable option and the pilot plant became redundant until November 1999, when the current LINK project started. The current project is being carried out by a consortium of academic institutions with input from industry and funded by the DTI and the Environment Agency. The objectives of the current study are to gain a fuller understanding of the microbiological, geochemical and ecological functioning of the pilot plant, and to incorporate all the resultant data and knowledge into a computer model of the system.

References/sources of information

Dennison, Sen, Halberg and Johnson 2001
Dodds-Smith, Payne and Gusek 1995
Environment Agency 1998 and 1999
Halberg and Johnson 2001
Hamilton, Lamb, Hallett and Proctor 1999
Hamilton, Postlethwaite and Foster, 1994
N.R.A. 1992, 1996 and 1994
Williams 1994
Wright 1997

6.36 WOOLLEY, WEST YORKSHIRE

NGR SE312113

Minewater type:	Net alkaline pumped deep minewater
Type of system:	Aeration, settlement tanks and lagoons Aerobic wetland Experimental wetland for ammonium removal
Year of completion:	July 1995
Owner/Consultant/Contractor:	Coal Authority IMC Research by Imperial College
Pre-treatment, if any:	Aeration

Construction

Table 6.36.1. Area of component parts

	Length m	Width m	Area m^2	Depth m	Volume m^3
Tank 1	100	16	1600	3.75	6000
Tank 2	66	16	1056	2.77	2925
Tank 3	66	16	1056	2.77	2925
Pond 1	100	30	3000	4.00	12000
Pond 2	125	30	3750	4.53	17000
Wetland			14000		

Depth, if known:	Wetland – assume 300 mm deep
Layout:	Pumped minewater split – 60% to tank 1; 40% to tank 2; Aerated by 2 cascades – approx. 2 m fall; Flow into settlement tanks 1 and 2; Flow recombines in tank 3; Flow to 2 lagoons (pond 1 and pond 2) in series, then to

wetland area prior to discharge to river.

Substrate: *In situ* soil

Vegetation:

Scirpus lacustris (True Bulrush)
Typha latifolia (Cattail)
Typha angustifolia (Lesser Cattail)
Iris pseudocorous (Flag Iris)
Phragmites australis (Common Reed)
Phalaris arundinacea (Reed Canary Grass)
Juncus effusus (Soft Rush)

Inlet/outlet structures:

Inlet to tanks 1 and 2 – cascades
Inlet to tank 3 – weir from tank 2 and pipe from tank 1
Outlet from tank 1 – pipe to tank 3 and overflow to tank 2
Outlet from tank 2 – weir to tank 3 and overflow to outlet channel from tank 3
Outlet from tank 3 – weir to open channel and pipe under railway to ponds
Inlet to pond 1 – pipes at water level
Outlet from pond 1/inlet to pond 2 – weir and overflow channel
Outlet from pond 2 – weir and overflow channel
Inlet to wetland – four 300 mm plastic pipes with tee-fittings across top of wetland area
Outlet from wetland – concrete channel with 5 opening to wetland area, which discharges to a ditch and then to River Dearne

Treatment criteria

(figures for 1999)

Average flow/pumped volume: 162 l/s

Average hydraulic loading:

Tank 1 – 5.25 m/day
Tank 2 – 5.3 m/day
Tank 3 – 13.25 m/day
Pond 1 – 4.67 m/day
Pond 2 – 3.73 m/day
Wetland – 1.00 m/day

Average retention time:

Tank 1 – 0.71 days
Tank 2 – 0.52 days
Tank 3 – 0.21 days
Pond 1 – 0.9 days
Pond 2 – 1.2 days
Wetland – 0.3 days

Table 6.36.2. Summary water quality data for Woolley treatment system

	n = 13	Average inflow mg/l	Average outflow mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Tank 1	Fe total	12.7	8.6	32.3	34.4	21.5
	Fe soluble	11.8	1.4			
Tank 2	Fe total	17.9	7.1	60.3	60.5	57.3
	Fe soluble	16.8	1.3			
Tank 3	Fe total	8.0	7.5	6.3	7.0	6.6
	Fe soluble		2.0			
Pond 1	Fe total	7.5	4.7	37.3	39.3	13.1
	Fe soluble	2.1	1.2			
Pond 2	Fe total	4.7	2.9	38.3	25.2	6.7
	Fe soluble	1.2	1.3			
Wetland	Fe total	2.9	0.3	89.7	36.4	2.6
	Fe soluble	1.0	0.2			
Overall	pH	7.0	7.9			
	Fe total	15.3	0.3	98.0	202.8	

Costs

Construction costs – not available
Operating costs – £100,308 p.a.
Wetland costs are quoted as £5/m^2.

References/sources of information

Laine 1997, 1998 and 1999
Parker 1997

6.37 WREN'S NEST, DUDLEY

NGR SO935919

Minewater type:	Sulphide-rich under pressure in cavern of former limestone mine
Type of system:	Active system, involving pumping, lime addition, ferric chloride addition, dissolved air flotation. Sludge to landfill, water to sewer
Year of completion:	1999
Owner/Consultant/Contractor:	Dudley Metropolitan Borough Council Biffa
Pre-treatment, if any:	Lime, ferric chloride

Construction

Active system comprising pumping, treating, recirculating and discharge to sewer, with the resultant sulphide-rich sludge being disposed to landfill.

Pumps, tanks and pipes installed for period of treatment (21 weeks) and removed at contract completion. Cavern void filled with grout as water removed to prevent subsidence.

Treatment criteria

Average flow/pumped volume: Total volume pumped and treated – 41,232 m^3
Target treatment rate – 100 m^3/day: not achieved initially due to strong buffer in minewater, necessitating high lime use and slower treatment rate to achieve target for suspended solids, but achieved successfully in latter part of pumping programme

Treatment process: Lime injected into sulphide-rich minewater while still under pressure to raise pH to about pH 10 and keep

hydrogen sulphide in solution
Ferric chloride added to precipitate out sulphide as iron sulphide, with some iron hydroxide
Precipitated iron separated from water in a dissolved air flotation (DAF) unit
Sand filtration, settling tanks and H_2S scrubber included to ensure required treatment standard achieved

Table 6.37.1. Summary water quality data for Wren’s Nest treatment scheme

	Initial water quality mg/l	Discharge consent (for recirculation to cavern) mg/l
H_2S	Up to 35	< 0.2
SO_4	Up to 15	
pH	7	6–9
Suspended solids		< 15
Fe		< 1

Comments

This was a one-off pump and treat operation for discrete body of contaminated water

6.38 YNYSARWED, SOUTH WALES

NGR	Treatment plant	SN807016
	Wetland	SN804011

Minewater type:	Acidic deep mine discharge, pumped
Type of system:	Active – mechanical aeration; lime dosing; lamellar settling; centrifugation Passive – wetland
Year of completion:	Passive – June 2000
Owner/Consultant/Contractor:	Project Partners – Neath Port Talbot Borough Council, Welsh Office, Environment Agency and Coal Authority Consultancy – SRK (UK) Ltd Active system by Hyder Consulting Ltd, constructed by Birse Construction
Pre-treatment, if any:	Mechanical aeration, lime addition, polymer dosing

Construction

Area of component parts:	Wetland – 1 ha
Layout:	Active system housed in a building between the road (B4242) and the Neath Canal Wetland about 1 km away – water from active system diverted to wetland via pipe; Wetland has liner with under-drainage, is divided into 2 cells and separated from the river by constructed bunds
Substrate:	Alluvial soil from site

Vegetation:	Existing trees retained; wetland species planted
Inlet/outlet structures:	Inlet – concrete and stonework cascade Outlet – pipe through earth bund to concrete outlet structure on river bank

Treatment criteria

Average flow/pumped volume:	Design capacity of active plant – 36 l/s

References/sources of information

Birse Construction 2000
Ranson and Edwards 1997
Ranson, Reynolds and Smith 1998 and 2000
SRK 1994
Younger, Large and Jarvis 1998

7

Summary and recommendations

Thirty-eight sites where minewater treatment is being carried out have been identified and described. As mentioned in the introduction to Chapter 5, each instance of minewater discharge is unique, dependent upon numerous factors, including local geology, hydrogeology, geochemistry, meteorology, topography, biology and anthropogenic factors. Also, the type of treatment implemented will have been determined not just on the basis of the minewater chemistry, but also on the basis of factors such as land availability, sensitivity of the receiving stream and political/social issues. It is therefore difficult to provide a definitive set of design guidelines. The following section summarises the information gathered from the minewater treatment sites involved in this study in order to make suggestions and recommendations for the design of future schemes, and also ways to further increase our understanding of minewater treatment schemes in general.

SUMMARY

The minewater discharges considered in this study can be divided into net alkaline and net acidic, as shown in table 7.1. In some instances the data may be insufficient to confidently place a particular minewater in either one of these categories, but the

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allocation of types shown in table 7.1 is used in this summary for the purpose of comparing and contrasting treatment schemes.

Table 7.1. Summary of minewater types and treatment system implemented

Case study	Site	Treatment type
Net alkaline		
5.1	Acomb	Settlement ponds; Wetland
5.5	Blaenavon	Wetland
5.7	Bullhouse	Settlement pond
5.12	Edmondsley	Wetland
5.13	Fender	Wetland
5.16	Gwynfi	Wetland
5.17	Kames	Settlement tanks; Wetland
5.18	Kimblesworth	Pilot scale reactor
5.19	Mains of Blairingone	Wetland
5.20	Minto	Wetland
5.21	Monktonhall	Alkali addition; Settlement ponds; Wetland
5.23	Nenthead	Pilot scale reactor
5.27	Polkemmet	Hydrogen peroxide addition; Settlement ponds; Wetland
5.31	Skinningrove	Reactor
5.32	St Helen Auckland	Wetland
5.33	Taff Merthyr	Settlement tanks; Wetland
5.34	Tailrace Level	Reactor
5.36	Woolley	Settlement tanks and ponds; Wetland
Net acidic		
5.2	Allanton	Settlement ponds
5.3	Aspatria	Wetland
5.4	Benhar Bing	Settlement pond; Wetland
5.6	Bowden Close	Pilot scale RAPS
5.8	Caphouse	Alkali addition; Settlement tanks
5.9	Craigenbay	RAPS; Wetland
5.10	Dulquharan	Dilute and disperse; Wetland
5.11	Dodworth	Settlement pond; Wetland
5.14	Foss	Pilot scale RAPS, wetland and ALD
5.15	Gilmerton Bing	Wetland
5.22	Nailstone	Alkali addition; RAPS; Settlement ponds
5.24	Oatlands	Wetland
5.25	Old Meadows	Alkali addition, Settlement ponds; Wetland
5.26	Pelenna	Wetland; RAPS
5.28	Quaking Houses	Wetland
5.29	Renishaw Park	Permeable Reactive Barrier
5.30	Shilbottle	Wetland; Permeable Reactive Barrier
5.35	Wheal Jane	Alkali addition; Wetlands; ALD
5.37	Wren's Nest	Alkali addition
5.38	Ynysarwed	Alkali addition; Wetland

Net alkaline minewater treatment schemes

Net alkaline minewaters are most often treated by means of aeration, settlement and wetlands. The excess alkalinity in the minewater means that there is sufficient bicarbonate buffer to maintain a near-neutral pH as oxidation and hydrolysis of iron occurs. Thus, the main objective of treatment is to introduce sufficient oxygen into the minewater to oxidise ferrous iron to ferric iron, at the same time degassing carbon dioxide, and then to provide conditions in which the precipitated ferric iron can settle out of suspension. Examination of table 7.1 reveals that most of the net alkaline minewaters are indeed treated in this way, i.e. by aeration, settlement and wetlands. There are, however, some exceptions, as outlined below.

Various forms of passive reactors, either at full or pilot scale, are used at Kimblesworth, Nenthead, Skinningrove and Tailrace Level. In the case of Kimblesworth and Skinningrove, reactors have been proposed because there is insufficient land area suitable for aerobic wetland establishment: the purpose of the reactor is to provide surface area on which iron oxidation can occur, in a more contained setting that in an aerobic wetland. At Nenthead and Tailrace Level, experimental work has demonstrated that anoxic limestone drain-type systems can be installed to treat high-Zn minewater.

Two of the treatment systems for net alkaline minewaters use chemical addition prior to settlement and wetland treatment. At Monktonhall, lime and flocculant are added to the pumped minewater. Although theoretically a simple aeration and settlement system would be suitable for the Monktonhall discharge, chemical dosing has been included in the treatment process, presumably to provide consistently high treatment efficiency of the large volume of minewater. An ongoing review of the volume and quality of the minewater should be carried out at this site to identify the point at which adequate treatment could be achieved without the addition of chemicals.

At Polkemmet, treatment initially comprised aeration via a cascade and settlement in tanks, but this was modified in 1999 to include chemical addition to the pumped minewater. The chemicals used are hydrogen peroxide, to act as an oxidising agent, and flocculant, to improve the settling characteristics of the precipitated material. A wetland area was also added at this time. Presumably, as at Monktonhall, the addition of chemicals was incorporated into the treatment process to ensure consistently high quality of treatment of large volumes of minewater.

The remaining schemes for net alkaline minewaters comprise ‘conventional’ settlement and/or wetland treatment and are considered below in terms of efficiency, operating parameters, cost effectiveness and design features (the settlement tanks and wetland areas of Montonhall and Polkemmet are also included in this assessment).

Settlement tanks

Settlement tanks are included in the systems at Acomb, Bullhouse, Kames, Monktonhall, Polkemmet, Taff Merthyr and Woolley. Since Acomb is not yet operational there are no data for the performance of these tanks, and there are no detailed data currently available for Taff Merthyr. Data for the other settlement tanks are given in tables 7.2 and 7.3. All values presented are average values, or, in the case of Polkemmet where no flow data are available, the flow used in the calculations is the value for the pumping capacity of the system.

Table 7.2. Operating parameters of settling tanks

		Flow l/s	Volume m^3	Hydraulic load m/day	Retention time days
Bullhouse		40	50,000	0.49	16.2
Kames		16.7	360	4.01	0.2
Monktonhall		98	21,000	2.0	2.5
Polkemmet	Old system				
	Main	68	2590	5.22	0.4
	‘1m’	34		4.2	
	‘2m’	34		4.2	
	New system				
	Primary 1	34	2125	3.46	0.7
	Primary 2	34	2125	3.36	0.7
	Secondary	68	1280	5.22	0.4
Woolley	Tank 1	97.2	6000	5.25	0.7
	Tank 2	64.8	2925	5.30	0.5
	Tank 3	162	2925	13.25	0.2
	Pond 1	162	12,000	4.67	0.9
	Pond 2	162	17,000	3.73	1.2

Hydraulic loading rates vary from 2.0 to 13.25 m/day and are generally in the region of 3 to 5 m/day. Retention time varies from 0.2 to 2.5 days.

Examination of table 7.2 reveals that there are a wide range of operating parameters for settlement tanks, as would be expected from the diversity of minewater discharge chemistry and volume, and the variable constraints on land availability. The treatment scheme at Bullhouse used an existing quarry void; as such it is atypical in terms of the design and sizing criteria used for settlement tanks. The minewater at Bullhouse is also not a typical net alkaline minewater but can be successfully treated in this manner due to the long retention time that can be achieved in the large lagoon.

Table 7.3. Treatment performance for Fe removal in settling tanks

		Fe in mg/l	Fe out mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day	Volume adjusted removal g/m^3.day
Bullhouse		51.8	2.5	95.2	170.4	24.3	3.4
Kames		12.7	12.4	2.4	0.4	1.2	1.2
Monktonhall		50.9	12.4	75.6	326.0	77.1	15.5
Polkemmet	Old system						
	Main	33.1	20.9	36.9	71.7	63.7	27.6
	'1m'	11.7	2.5	78.6	27.0	38.6	
	'2m'	11.7	2.6	77.8	26.9	44.6	
	New system*	48.9	1.3	97.3	279.6	164.5	
Woolley	Tank 1	12.7	8.6	32.3	34.4	21.5	5.7
	Tank 2	17.9	7.1	60.3	60.5	57.3	20.7
	Tank 3	8.0	7.5	6.3	7.0	6.6	2.4
	Pond 1	7.5	4.7	37.3	39.2	13.1	3.3
	Pond 2	4.7	2.9	38.3	25.2	6.7	1.5

* assumes most settling occurs in the first two, primary, tanks

Treatment performance is also highly variable. The most notable examples of poor Fe removal are Kames and Tank 3 at Woolley. Both of these have a low retention time (0.2 days). Woolley Tank 3 also has a high hydraulic load (13.25 m/day) although the hydraulic load at Kames is not excessive when compared to values for the other treatment schemes (4.01 m/day). These data therefore indicate that a retention time of 0.2 days (about 5 hours) is not sufficient to allow all precipitated Fe to settle out of suspension. Retention times in excess of 0.5 days would seem to be desirable to achieve reasonable treatment efficiency (for example, Woolley Tank 2 has a retention time of 0.5 days and a treatment efficiency of 60.3%).

The use of chemicals, including flocculants, at Monktonhall and Polkemmet are seen to be beneficial, giving good treatment efficiency – these two systems also have the highest daily retention of Fe, in the region of 300 kg/day.

Although it is difficult to make design and sizing proposals on the basis of such a varied data set, it would seem that, in general, settlement tanks should be designed to allow a retention time in excess of 0.5 days (12 hours), and a hydraulic loading rate not exceeding 5.5 m/day. Volume adjusted removal rates of up to 20 g/m^3.day would seem to be achievable with a reasonably high treatment efficiency, although two of the three systems that achieved volume adjusted removal rates in excess of 15 g/m^3.day used flocculants, so a more conservative value should probably be used for systems which are intended to operate without chemical addition.

As far as design features are concerned, the settlement tanks vary in their mode of construction from the use of an existing void, as at Bullhouse, through earth-bunded lagoons, as at Acomb, Monktonhall, Polkemmet and the latter-stage settlement lagoons at Woolley, to concrete tanks, as at Kames, the early-stage tanks at Woolley and Taff Merthyr. The main operational functions of settlement tanks that will influence their design are the requirement to achieve optimum hydraulic flow conditions and ease of sludge removal. It is generally considered that optimum hydraulic flow conditions are achieved in tanks with a length to width ratio of between 1:2 and 1:4 and a depth determined using the following formula:

$$\frac{1}{35} < \frac{H}{L} < \frac{1}{20} \qquad (7.1)$$

where H is the wetted depth of a rectangular tank and L is the length of the tank (Water Treatment Handbook 1991).

Thus, tanks of an essentially rectangular nature would seem to be necessary in order to achieve optimum settlement of precipitated particles. The design of the settlement tanks at Acomb, while conforming to the overall rectangular shape, has attempted to soften the lines by introducing slight irregularities into the basic shape.

Cascades, if incorporated into treatment schemes, should be such as to achieve maximum aeration and degassing of carbon dioxide. This means that the design needs to be such as to break the fall of water into small droplet sizes. Generally a stepped cascade structure with a fall in the region of 2 m, as at Woolley, seems to achieve adequate aeration. The cascading water at Fender and Edmondsley issues from the inflow pipes as a single stream of water, without intermediate steps to reduce the droplet size; the latter feature would probably improve aeration. The aeration cascade at Bullhouse achieves the desired effect by passing the cascading water over a series of steps within a manhole.

As a general consideration, the settlement tank stage of a treatment scheme cannot be considered to be particularly aesthetically attractive or natural-looking, since the accumulation of ochre will necessarily give the characteristic orange colouration. Also, from a safety point of view, public access to the settlement tanks would be fraught with potential hazard, so even in a scheme where public access is desirable this will probably need to be confined to the reed bed stage. Thus it may be preferable to accept that the settlement tanks are a necessary functional unit of the treatment scheme and not to compromise their performance by trying to make them aesthetically attractive. Even on sites where public access is encouraged, it would seem to be advisable to keep the settlement tanks within a separate fenced-off area.

Having accepted that the settlement tanks are not particularly attractive or aesthetically pleasing, the matter of construction materials, whether earth bunds or concrete, becomes more an issue of cost and functionality. As long as the tanks can be readily accessed and emptied of ochre, the final choice of construction materials would be determined on the basis of cost benefit analysis.

It is not possible in this review to compare the settlement tanks of the various treatment schemes in terms of cost effectiveness since the costs associated with the settlement tanks are not available as separate figures.

Wetlands

The use of constructed wetlands or reed beds is common practice for minewater treatment in general and for alkaline minewater treatment in particular. Most of the alkaline minewater treatment schemes covered in this review include a wetland stage, either as the main treatment component or as a polishing mechanism. Exceptions to this are the four sites where there are passive reactors (see above) and at Bullhouse, although this site does have some wetland-type polishing ponds, for which there are no separate monitoring data. Summary data for the other sites, which include a wetland component, are presented in tables 7.4 and 7.5. (There are currently no data available for Acomb and Blaenavon which are still in construction, and no detailed data are available for Taff Merthyr or St Helen Auckland.)

Examination of table 7.4 reveals a wide range of operating parameters for constructed wetlands, with hydraulic loading rates varying from 0.03 to 1.84 m/day, and retention times between 0.2 and 9.2 days. These values compare with average retention times of between 0.3 and 30 days reported by Hedin *et al.* (1994) in a review of constructed wetlands receiving minewater, and values for hydraulic loading of 0.014 to 0.22 m/day for domestic wastewater treatment wetlands, with a retention time of 5 to 14 days, reported by Mitsch & Gosselink (1993). The latter authors quote work published in 1989 that recommends a hydraulic loading rate of 0.05 m/day and a retention time of greater than 1 day for wetlands receiving coal mine drainage (Fennessy & Mitsch 1989).

Table 7.4. Operating parameters of constructed wetlands for alkaline minewater treatment

	Flow l/s	Area m^2	Hydraulic load m/day	Retention time days
Edmondsley	4.4	3235	0.12	3.4
Fender	25	6000	0.36	0.8
Gwynfi	10	1360	0.64	0.5
Kames	16.7	4000	0.36	1.4
Mains	7.8	20667	0.03	9.2
Minto	37.8	10000	0.33	0.6
Monktonhall	98	5070	1.67	0.2
Polkemmet	68	3200	1.84	0.2
Woolley	162	14000	1.00	0.3

Table 7.5. Treatment performance for Fe removal in wetlands

	Fe in mg/l	Fe out mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day	First order constant* k_1
Edmondsley	13.4	0.6	95.5	4.9	1.5	0.038
Fender	9.53	1.85	80.6	16.6	2.8	0.220
Gwynfi	6.2	0.4	93.7	5.0	3.7	0.230
Kames	12.4	0.2	98.4	17.6	4.4	0.087
Mains	12.5	0.5	96.0	8.1	0.4	0.010
Minto	11.0	4.3	60.9	34.7	3.5	0.348
Monktonhall	12.4	0.7	94.7	99.4	19.6	0.569
Polkemmet	2.66	0.31	88.3	13.8	4.3	0.854
Woolley	2.9	0.3	89.7	36.4	2.6	0.441

* First order constant k_1 is shown, see discussion later in this chapter.

There can thus be seen to be a disparity in the values, particularly for hydraulic loading, recommended in the literature and those found at the sites studied in this review. If the two highest values of hydraulic loading are omitted (Montonhall and Polkemmet), the other wetlands have loading of 1 m/day or less. As mentioned above, Monktonhall and Polkemmet are both atypical in that they have chemical pre-treatment, and it is likely that the use of flocculants improves the settling characteristics of the precipitated material so that a higher loading can be applied to a wetland area without compromising performance in terms of removal of iron from the water.

Treatment performance of the wetlands is generally good, with the majority of the sites giving treatment efficiency in the region of 90% or greater. Exceptions to this are at Minto, where average treatment efficiency during 1999-2000 was only 60.9%. However, two sets of data from 2001 indicate that a treatment efficiency of 96.1% is now being achieved at Minto, perhaps due to denser plant growth. Treatment efficiency at Fender is 80.6%, which is slightly lower than that achieved in the other wetlands – this could be due to the fact that the wetland at Fender is newly planted, so the data reflect the fact that the plant material has not yet achieved sufficient density to provide an efficient filtration mechanism. There is a general trend observed at most reed bed sites of increasing treatment efficiency with maturation of the reed bed.

The values for area adjusted removal are all low, with the exception of Polkemmet where the value is 19.6 g/m^2.day. It is thus not possible to re-assess the design and sizing criteria proposed by Hedin *et al.* (1994) (see table 2.7) in the light of this review. These authors note that accurate assessments of removal capabilities of wetland systems can only be made if there are excessive concentrations of contaminants entering the systems. If effluent contaminant concentrations are low (lower than 2 mg/l), the rate of contaminant removal achieved in the system may be limited by the loading of the contaminant and not be a true reflection of what the system is capable of, given greater loading. Thus the very success of these constructed wetlands can be considered to be counter-productive to the formulation of revised sizing criteria!

As with the settlement tanks, it is difficult to provide general rules for design of constructed wetlands in terms of their operating parameters, although it would seem that, for a wetland receiving raw minewater that has not had any chemical additions, the hydraulic loading rate should not exceed 1 m/day, and retention time should be in excess of 0.3 days (about 8 hours).

It is also interesting to note that none of these wetlands is receiving minewater with a concentration of Fe greater than 13.4 mg/l. This is perhaps an indication of the increasing trend in recent years, particularly in UK constructed wetlands, to use wetlands as a polishing mechanism, rather than relying on them to carry out the whole of the treatment process. This is particularly pertinent where Fe loading is very high, since rapid accumulation of ochre in a wetland is likely to be more problematic in terms of maintenance and disposal than using settlement tanks to retain the bulk of the precipitate. In addition, if a use can be found for the ochre, it is important that it is in as pure a form as possible and not ‘contaminated’ with organic matter from a wetland.

Consideration of the above points gives rise to the proposal that it is more appropriate to provide the correct hydraulic operating parameters, i.e. hydraulic loading and retention time, than to design treatment wetlands on the basis of area adjusted removal. The question of whether first-order or zero-order kinetics are the most appropriate to use in consideration of treatment wetlands (see Chapter 2) may also have relevance in this context and is discussed later in this chapter.

The design features of the reed bed stage of treatment schemes include such considerations as functional, ecological and aesthetic factors.

The main functional considerations in reed bed design are the achievement of even flow distribution and optimum flow regime through the reed bed, provision of sufficient ‘free board’ to allow build up of organic matter and retained material in the reed bed as it matures, and structures to collect flow out of the reed bed and allow manipulation of water depth in the reed bed cells. The requirement to control and regulate the inflow and outflow of the wetland means that a certain amount of ‘hard’ engineering is necessary for the inlet and outlet structures. As a general rule, simplicity of design of these structures should be a major aim. For example, the use of open channels, as at Kames and Taff Merthyr, rather than enclosed pipes makes blockages less likely and facilitates maintenance, and the use of simple flow control devices such as removable weir boards is preferable to complicated valve arrangements, as at Minto, which once again are more prone to becoming blocked with ochre. Ochre blockage of pipes is a major operational problem, particularly, for example, at Bulhouse and Old Meadows, where around 40% of the pumped volume is lost in only a few months due to ochre accretion in the pipes (Parker pers. comm.). If pipes are unavoidable in the layout of the treatment scheme, it is considered to be necessary to incorporate features into the design to routinely clear ochre from the pipes. For example ‘pigging’ arrangements can be included at the design stage to facilitate regular maintenance.

Optimum flow within the reed bed cells is important to ensure consistent and reliable treatment efficiency. To a certain extent the design consideration pertaining to settlement tanks in terms of the optimum length to width ratio also applies to reed bed cells, but there is less need to have a strictly rectangular shape. Irregular shapes, such as at Gwynfi and Woolley, are aesthetically more pleasing and more likely to integrate into the surrounding environment, giving a more natural appearance. If flow is evenly distributed across the width of the reed bed at the inlet, and there is a well-established, dense stand of wetland vegetation, short-circuiting of flow through the cell should be discouraged. For example, at Wheal Jane, the inlet is via a single channel in one corner of the cells, and plant establishment is poor, which leads to a substantial amount of short-circuiting across the cell and greatly decreased retention time. Inclusion of areas of deeper water, such as at Mains and Polkemmet, performs a functional role by increasing retention time, as well as providing ecological and aesthetic benefits by providing habitat diversity.

Ecological and aesthetic considerations generally involve trying to make the reed beds appear as natural as possible, which can be achieved by using irregular, curved shapes, diverse emergent macrophyte species and the use of marginal species, such as willows and alders, to make the wetland area blend in with the surroundings. Good examples of this latter feature are seen at Mains of Blairingone and Polkemmet.

Net acidic minewater treatment schemes

As a general rule, acidic minewaters are more problematic and harder to treat since the treatment requirement is to add alkalinity and precipitate out metals, rather than just precipitate out metals as is the requirement for net alkaline minewaters. Consequently, the range of those strategies that have been applied to treatment of acidic minewaters is more varied than that applied to treatment of net alkaline minewaters. Treatment techniques include active treatment, passive reactors, permeable reactive barriers, settlement ponds, Reducing and Alkalinity Producing Systems (RAPS) and wetlands.

Active treatment

Active treatment is often used for acidic minewaters, either as the whole treatment process or as a pre-treatment, followed by passive treatment elements. Examples of active treatment can be seen at Wheal Jane, where a high density sludge-type process is in operation; at Caphouse, where lime dosing and flocculant addition are followed by settlement tanks; at Old Meadows, where addition of caustic soda precedes settlement ponds and a wetland; at Wren's Nest, where a one-off pump and treat active scheme was implemented to remediate a confined body of water; and at Ynysarwed, where active treatment has been installed, followed by a wetland which it is hoped will in time be the sole treatment method. Each of these active treatment schemes is highly site specific, having been designed and modified to suit the specific minewater chemistry requiring treatment. The obvious drawbacks of active treatment are the ongoing costs of labour, power and reagents. Labour costs are difficult to reduce, since even with an automated dosing system, such as at Old Meadows and Wheal Jane, a skilled technician needs to be present or on call to ensure consistent and reliable operation and to deal with equipment or process failures. Another consideration at Old Meadows is the use of caustic, which is a more hazardous chemical to handle than lime, thus requiring a more skilled and highly trained labour force to operate the process.

Power costs may be reduced by maximising the use of 'off peak' supply, for example, by pumping at night and using a balancing pond to regulate the flow through the treatment system, as is the proposal for the modified treatment scheme at Caphouse.

Reagent costs can be reduced by 'fine-tuning' the treatment process to ensure it is operating at optimum efficiency. For example, at Wheal Jane, substantial savings in reagent use have been achieved in the first year of operation of the new active treatment scheme.

Passive reactors

Passive reactors are an attempt to achieve satisfactory treatment while reducing active involvement in the treatment process. Thus, anoxic limestone drains (ALDs) can be used to introduce alkalinity without the need for an active lime dosing plant. At Wheal Jane, an ALD was installed in the pilot passive treatment plant as a pre-treatment before the flow entered a series of wetlands. However, this was not successful due to the aluminium content of the minewater causing blockages within the ALD. This emphasises the importance of ensuring that a particular treatment process is suitable for the specific minewater chemistry. The other example of a passive reactor included in this review is the pilot-scale ALD installed at Foss, Aberfeldy, as part of a multi-component passive treatment scheme. The main aim of this ALD was the removal of Zn, an application also used at sites in the Pennines (Nenthead and Tailrace Level, see above).

Permeable reactive barriers

Permeable reactive barriers (PRBs) are a relatively new concept for treatment of migrating plumes of contaminated groundwater. They are thus ideally suited for use in situations where there is a diffuse source of minewater, for example, leachate and percolation through spoil heaps. The only PRB currently operational for minewater treatment in the UK is at Renishaw Park, where the indications are that it is functioning well, although there are no data to substantiate this. A PRB is also planned for Shilbottle; and an increase in the use of this technology in the future may be anticipated as more treatment schemes for spoil heap-generated minewater are implemented.

Settlement tanks

Settlement tanks are used at a number of acidic minewater treatment sites (Allanton, Benhar Bing, Caphouse, Dodworth, Nailstone, Old Meadows, and Pelenna). However, data on operational parameters and performance efficiency are not generally available since the settlement tanks are not individually monitored. The available data are presented in tables 7.6 and 7.7.

Table 7.6. Operating parameters of settling tanks

	Flow l/s	Volume m^3	Hydraulic load m/day	Retention time days
Old Meadows Pond 1	49	11640	1.45	2.7
Old Meadows Pond 2	49	3030	2.79	0.7
Pelenna - Gwenffwd	8.9	850	0.9	1.1
Caphouse	38	1040	3.16	0.3*

*assumes depth of settling tanks is 1 m.

Table 7.7. Treatment performance for Fe removal in settling tanks

	Fe in mg/l	Fe out mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day	Volume adjusted removal g/m^3.day
Pelenna – Gwenffwd	5.4	1.5	72.9	3.0	3.5	3.5
Caphouse	50	3	94.0	154.3	148.4	148.4*

*assumes depth of settling tanks is 1 m.

The scarcity of data for the settling tanks in acidic minewater treatment schemes limits the conclusions that can be drawn regarding design and sizing criteria. However, the low estimated retention time for the Caphouse tanks could be a contributory factor in the fact that the concentration of Fe in the discharge often exceeds the target concentration of 3 mg/l.

The proposals made on the basis of data from settlement tanks in alkaline minewater treatment schemes, i.e. a minimum 12 hour retention time and maximum hydraulic load of 5.5 m/day, are not contradicted by the limited data presented.

Reducing and Alkalinity Producing Systems (RAPS)

RAPS, or SAPS as they are alternatively known, are a means of passively introducing alkalinity into acidic minewater and, hopefully, overcoming the problems of blocking that can occur with anoxic limestone drains.

Pilot-scale RAPS have been trialled at Bowden Close and Foss, and full-scale RAPS are operational at Craigenbay, Nailstone and Pelenna.

The pilot-scale RAPS at Bowden Close achieved good treatment efficiency for Fe (93.7%) and in the region of 60% efficiency for Mn and Al. The retention time within the system was about 1.3 days and the hydraulic load was 0.79 m/day. The data for monitoring of the Foss pilot-scale RAPS are not available, but it is reported that a significant proportion of Fe was removed in the RAPS, with a lesser proportion of Al and Mn being removed. The retention time is calculated as 19 hours and the hydraulic load was 1.3 m/day.

Of the full-scale RAPS that are operational in the UK, the only ones for which there are any data are those at Pelenna, which are presented in tables 7.8 and 7.9.

Table 7.8. Operating parameters of RAPS for acidic minewater treatment

	Flow l/s	Area m^2	Hydraulic load m/day	Retention time days
Pelenna Whitworth A	9.2	1825	0.44	2.3*
Gwenffwd	8.9	2425	0.32	3.2*
Garth Tonmawr Whole system	22.6	6370	0.31	2.0
Garth Tonmawr Cell 2	22.6	970	2.01	0.5**
Garth Tonmawr Cell 4	22.6	980	1.99	0.5**

* assuming depth of 1 m.

** actual retention time is less than this theoretical value due to surface flow across flooded cells.

Table 7.9. Treatment performance for Fe removal in RAPS

	Fe in mg/l	Fe out mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day
Whitworth A	59.7	8.5	85.8	40.7	22.3
Garth Tonmawr Whole system	28.8	2.0	93.1	52.4	8.2
Garth Tonmawr Cell 2	11.2	3.5	68.8	15.1	15.6
Garth Tonmawr Cell 4	3.1	0.8	74.8	4.5	4.5

Examination of tables 7.8. and 7.9 reveals that the hydraulic loading rate for RAPS varies between 0.32 and 2 m/day and the retention time varies from 0.5 to 3.2 days. The figures of 2 m/day hydraulic loading and 0.5 days retention time are for individual cells RAPS-type cells at Garth Tonmawr. If the figures for the whole Garth Tonmawr system are considered, the respective values are 0.31 m/day and 2 days, which more closely accord with the values for other RAPS at Pelenna. Reports of RAPS in the USA have retention times of between 3.6 hours and 26 days (Jage *et al.*2000, from a review of eight RAPS), of about 3 days (Barton & Karathanasis 1999) and between 2.3 and 10 days (Rose *et al.* 2001). The latter workers report on a multi-stage RAPS system in which the hydraulic loading rates vary between 0.1 and 0.3 m/day.

Treatment efficiency of the systems for which adequate data are available to make the necessary calculations is generally good, in excess of about 70%, and area adjusted removal rates of up to 22 g/m^2.day have been achieved. The latter figure compares with values of 17 g/m^2.day (Nairn *et al.* 1999) and 40 g/m^2.day (Barton & Karathanasis 1999) reported in the literature for RAPS in the USA. The figure most commonly reported in the literature is that of alkalinity generation within RAPS, which cannot be calculated for the UK systems due to lack of data.

It would thus seem that as a general rule, RAPS systems should be designed to receive a hydraulic loading rate in the region of 0.3 m/day, with a target retention time of 2 to 3 days and a target area adjusted removal of Fe in the region of 20 g/m^2.day. However, it should be stressed that these are only very approximate guidelines; a considerable amount of further work and data collection needs to be carried out to gain more knowledge and refine the design and sizing criteria for RAPS.

Wetlands

Constructed wetlands are used as part or all of the treatment at Aspatria, Benhar Bing, Craigenbay, Dulquharan, Dodworth, Foss, Gilmerton Bing, Oatlands, Old Meadows, Pelenna, Quaking Houses, Shilbottle, Wheal Jane and Ynysarwed. Available data are presented in tables 7.10 and 7.11.

Hydraulic loading rates vary from 0.01 to 2.3 m/day, although if the values for Old Meadows (2.3 m/day) and Wheal Jane (0.01 and 0.03 m/day) are omitted, the hydraulic loading rate to the other wetlands is generally less than 0.4 m/day. Retention time varies from 0.2 to 16.5 days.

The performance of the wetland systems generally gives average iron removal of greater than 75%, with the exception of the two Bings (Benhar and Gilmerton), Old Meadows and Quaking Houses. The two Bings both have very high Fe concentrations in the minewater to be treated (226 and 403 mg/l, respectively), so it is hardly surprising that the percentage Fe removal in the wetlands is low. In fact, in each instance the concentration of Fe is reduced by about 100 mg/l. Old Meadows has the opposite situation, where the influent Fe concentration is low (average value 1.7 mg/l) and thus reduction in concentration to 1 mg/l represents a treatment efficiency of only 41.2%. This, however, can be considered to be successful treatment, despite the low removal efficiency, since effluent Fe conforms to the usual consent concentration.

The highest area adjusted removal rate achieved is 6.9 g/m^2.day, at Craigenbay, but since there are no reliable data available for this site, this figure cannot be taken as representational for the removal achieved. Other values for area adjusted removal are all less than 6 g/m^2.day. This value may indicate that greater area adjusted removal efficiencies are unlikely to be achieved with net acidic minewaters. It is not possible to compare these values with those proposed by Hedin *et al.* (1994), since these authors reported rates of acidity removal rather than Fe removal. The only site for which acidity data are available is Quaking Houses, where an area adjusted removal of 11.1 g/m^2.day is reported. This compares favourably with the figure of 7 g/m^2.day reported by Hedin *et al.* (1994) as being a suitable value to use for sizing wetlands to treat acidic discharges to abandoned mined land (AML) criteria (see table 2.7).

As was the case for wetlands receiving net alkaline minewaters, it is not possible, on the basis of this review, to propose alternative sizing criteria to those already put forward. However, as a general guideline, it would seem advisable to restrict the hydraulic loading rate for wetlands receiving acidic minewater to less than 0.4 m/day and to aim for a retention time in excess of 1 day.

Table 7.10. Operating parameters of constructed wetlands for acidic minewater treatment

	Flow l/s	Area m^2	Hydraulic load m/day	Retention time days
Craigenbay	3.8	1371	0.24	0.6
Dodworth	1.9	1500	0.12	2.8
Foss	0.03	12	0.22	1.4
Old Meadows	49	1825	2.3	0.2
Whitworth 1	3	900	0.29	1.0
Whitworth A	9.2	4500	0.18	3.4
Gwenffwd	8.9	2000	0.38	1.6
Quaking Houses	2.1	440	0.42	0.7
Wheal Jane Lime dosed and ALD	0.5	1300	0.03	6.0
Wheal Jane Lime-free	0.7	5000	0.01	16.5

Table 7.11. Treatment performance for Fe removal in wetlands

	Fe in mg/l	Fe out mg/l	Treatment efficiency %	Retained kg/day	Area adjusted removal g/m^2.day	First-order constant** k_1
Benhar Bing*	226	119	21		0.8	
Craigenbay	30	1	96.7	9.5	6.9	0.070
Dodworth	7.1	0.5	93.4	1.1	0.7	0.041
Gilmerton Bing	403	310	23.1			
Old Meadows	1.7	1.0	41.2	3.0	1.6	4.372
Whitworth 1	24.2	4.3	82.2	5.2	5.7	0.167
Whitworth A	8.5	1.7	80.0	5.4	1.2	0.110
Gwenffwd	13.1	0.9	92.8	9.4	4.7	0.144
Quaking Houses	4.6	2.5	45.5	0.4	0.9	0.676
Wheal Jane Lime dosed	142	31	78.2	4.8	3.7	0.022
Wheal Jane ALD system	149	17.5	88.3	5.7	4.4	0.016
Wheal Jane Lime free	140	34.5	75.4	6.4	1.3	0.009

* Values for Benhar Bing are median values.

** First-order constant k_1 is shown, see discussion later in this chapter.

Removal kinetics

Chapter 2 introduced the concept of removal kinetics operative in constructed wetlands, and the ongoing debate about whether it is more appropriate to use zero order kinetics (i.e. area adjusted removal rates) or first order kinetics, in which the relationship between wetland size, area and contaminant concentrations is expressed by equation 7.2 (Tarutis *et al.* 1999):

$$\text{Area} = \frac{\text{flow}}{k_1 \ln(C_{in} / C_{out})} \tag{7.2}$$

It is proposed by Tarutis *et al.* (1999) that the use of first-order kinetics is more appropriate, especially in instances where the volume of flow is high and the concentration of contaminants is low, which is often the case in wetlands used as a final polishing step in a treatment system.

Table 7.12 summarises the values for removal efficiency, area adjusted removal and first order removal of Fe obtained from sites in this review. For comparison purposes, table 2.8 is also reproduced (as table 7.13) to show the values presented by Tarutis *et al.* (1999).

Table 7.12. Summary statistics for treatment efficiency, area-adjusted removal and first-order removal of iron in wetlands covered in this review*

Performance measure	Mean ± std. deviation	Median	Range
Fe efficiency (%)	83 ± 17	88	41 to 98
Fe area adjusted removal, k_0, (g/m^{-2} day^{-1})	3.0 ± 1.9	3.2	0.4 to 6.9
Fe first-order removal, k_1, (m/day^{-1})	0.23 ± 0.25	0.13	0.009 to 0.854

* extreme outlying values (Monktonhall area adjusted removal (19.6) and Old Meadows first order removal (4.37)) are omitted.

Table 7.13. Descriptive statistics for treatment efficiency, area-adjusted removal and first-order removal of iron and manganese for 35 natural wetlands (Tarutis et al. 1999)

Performance measure	Mean ± std. deviation	Median	Range
Fe efficiency (%)	64 ± 28	72	-32 to 99
Fe area adjusted removal, k_0 (g/m^{-2}.day^{-1})	6.5 ± 8.8	3.5	-7.7 to 45
Fe first-order removal, k_1 (m/day^{-1})	0.29 ± 0.34	0.18	-0.061 to 1.8

Examination of tables 7.12 and 7.13 shows that in both studies there was a very wide range of values for the parameters considered, especially for the area adjusted removal and first-order removal. However, the median values are very similar from both studies. Tarutis *et al.* (1999) suggested that it was perhaps most appropriate to use the median value for the first-order removal constant as a more conservative estimate for calculating approximate wetland sizes, until more accurate values for this constant have been obtained. Collation of data from this review seems to corroborate this statement and it could be extended to include a recommendation to use median values for area adjusted removal as well.

Cost effectiveness

Where costs are available for treatment schemes it has been possible to assess them in terms of cost effectiveness, i.e the cost per kilogram of Fe removed from the minewater, the cost per cubic metre of minewater treated and the cost per square metre of reed bed constructed. Although this forms a useful means of assessment, it should be borne in mind that every case of minewater discharge and treatment is unique, with its own set of problems and constraining circumstances, with cost being only one of the issues to consider. The treatment costs (i.e. for Fe removal and volume of water treated) are calculated on the basis of a 10 year period. Thus, capital costs are spread over the 10 years, and annual operating costs incorporated to obtain an annual cost figure. Table 7.14 summarises the available cost data. Table 7.15 presents cost data for the treatment schemes where minewater is pumped to the treatment system, with the cost of electricity used for pumping excluded. This allows a comparison of treatment costs on the grounds of 10 year operational costs assuming that the site topography was such as to allow gravity discharge.

As mentioned in the introduction to Chapter 5, there are numerous conditions and exclusions applicable to these cost data; in particular the following points should be noted about the quoted figures for Coal Authority schemes:

- Capital costs are for contractors costs only and do not include consultancy fees, site investigations and project management costs.
- Operating costs similarly do not include consultancy/management fees to oversee operations.
- Operating costs do not include pump repairs/replacements. There is a separate budget for this of approximately £200,000 p.a.

Other ad hoc items are also excluded from the operating costs, e.g. dayworks for improvements and repairs, dealing with vandalism, additional reed planting.

Table 7.14. Summary of costs associated with minewater treatment schemes

	Capital cost (£)	Operating cost (£)	Cost per kg Fe removed (£)	Cost per m^3 water treated (£)	Cost per m^2 wetland constructed (£)
Blaenavon	301,000	1,560			75
Bowden Close	20,000				
Bullhouse	1,200,000	43,404 (P)	2.63	0.13	
Dalquharran		4,560			
Dodworth		20,000 (C)			23
Edmondsley	277,000	24,036 (P)	28.93	0.37	86
Fender	385,000	21,204 (P)	9.85	0.08	64
Gwynfi	235,000	1,560	13.73	0.08	173
Kames	622,000	4,080	19.95	0.13	156
Minto	260,000	2,460	3.56	0.02	26
Monktonhall	297,000	97,260 (P, C)	0.82	0.04	59
Old Meadows	949,000	132,696 (P, C)	5.01	0.15	520
Polkemmet	198,000	205,368 (P, C)	2.15	0.11	62
Quaking Houses	18,000	500	15.75	0.03	41
Taff Merthyr	1,625,000	18,888 (P)			90
Wheal Jane – lime dosed	433,333	33,333 (P, C)	67.76	8.10	333
Wheal Jane – ALD	433,333	33,333 (P)	56.77	8.10	333
Wheal Jane – lime-free	433,333	33,333 (P)	30.89	3.47	87
Woolley		100,308 (P)			5

P = pumping costs included; C = cost of chemicals included. Otherwise costs are for labour and general operation and maintenance.

Table 7.15. Summary of costs with electricity (for pumping) excluded

	Capital cost (£)	Operating cost (£)	Cost per kg Fe removed (£)	Cost per m^3 water treated (£)
Bullhouse	1,200,000	41,554	2.60	0.13
Edmondsley	277,000	23,036	28.37	0.37
Fender	385,000	20,204	9.69	0.07
Monktonhall	297,000	93,260	0.79	0.04
Old Meadows	949,000	128,696	4.92	0.15
Polkemmet	198,000	201,468	2.12	0.10

It should thus be stressed that tables 7.14 and 7.15 have to be considered in the light of the limitations, exceptions and specific circumstances pertaining to each site. For example, the figure for cost per area of wetland constructed is calculated as the capital cost for the whole treatment scheme divided by the area of reed bed (separate costs for wetland establishment are not available for most sites), which in some cases, such as Old Meadows, gives a vastly inflated figure due to the scope and scale of pipework, infrastructure, equipment etc. involved in the overall treatment process. The figures shown in table 7.14 show the large range of costs that can be incurred in minewater treatment schemes and serve to demonstrate that cost effectiveness and cost benefit analysis can only really be applied on a site specific basis, or extrapolated to other sites only if very similar conditions pertain.

RECOMMENDATIONS

The principal objective of this review was to draw together all available information on minewater treatment schemes currently operational in the UK and to use this information as a basis for designing and implementing future treatment schemes. The case studies and site summaries presented in Chapters 5 and 6 illustrate the diversity both of minewater types and the techniques employed for treatment. It also became apparent while compiling the information and data for this review that the quantity and quality of data are highly variable from site to site, which in turn has implications for the usefulness of these data for informing future schemes. Therefore, outlined below are some recommendations for future work and data collection/handling protocols that would form the basis of an important resource for ongoing minewater remediation programmes throughout the UK.

Standardised sampling strategy

It would be highly advantageous to have a standardised sampling and monitoring strategy for all minewater treatment sites, so that a consistent and directly comparable record of performance of treatment schemes could be compiled and maintained. At the present time the scope and detail of monitoring procedures varies greatly; this imposes limitations on the usefulness of the resultant collective data set. A more consistent and complete data set would allow the crucial issue of sizing criteria, as discussed above in relation to removal kinetics, to be further explored and elucidated.

Outline of proposed sampling strategy

The following could form the basis of a standardised sampling strategy, the details of which would need to be confirmed by a consultative procedure between interested stakeholders such as the Coal Authority, the Environment Agency, academic institutions, local authorities and industry.

Sampling period	Minimum of monthly
Sampling points	Influent (untreated minewater); effluent (treated discharge); intermediate points (e.g. between individual components such as settlement tanks, wetland cells); Receiving watercourse (upstream and downstream of discharge point)
Determinands	On site – pH, alkalinity, dissolved oxygen, conductivity, temperature and possibly redox potential Laboratory – total and dissolved iron, aluminium, manganese, plus other metals that might be anticipated – an initial suite of metal analyses could be carried out and later reduced to those found at significant concentrations Major ions (calcium, magnesium, sodium, potassium, sulphate, chloride, bicarbonate) – should be included initially to characterise a minewater, and periodically thereafter to allow an ionic balance calculation to be carried out as part of the quality control procedure Other possible contaminants, e.g. nitrate, ammonium;
Flow/pumping rate	At least on every sampling occasion
Quality Control	Use of blanks, replicates, standards etc. Ionic balance calculation
Personnel	Personnel carrying out sampling should be fully trained in the standardised protocol
Equipment	All equipment should be suitable for the task and maintained appropriately e.g. meters regularly calibrated and sampling containers thoroughly cleaned

The above is just a very brief outline of a strategy that could form the basis for a standardised monitoring and sampling protocol.

Database

All field and laboratory data should be compiled in a standardised database. A high-quality database would provide a powerful resource that could be interrogated as required and would facilitate the manipulation and dissemination of data. The standardisation of the sampling and monitoring regime, and compiling these data in an easily accessible format, would mean that more confidence could be placed in assessments of such features as performance criteria and treatment efficiency. This in turn would allow refinement of the design and sizing criteria for application in future treatment schemes.

The database could also include a record of capital and operating costs of individual schemes which would allow an ongoing assessment of cost benefit to be carried out.

Appendix A

International minewater initiatives

There are numerous networks and internet-based minewater groups worldwide. The following is a compilation of some of these, access to which will provide links to further sites of interest and relevance.

ACMER

The Australian Centre for Mining Environmental Research (ACMER) is an industry initiative with the stated vision of being an internationally recognised centre of excellence supporting continual improvement in environmental performance in the minerals industry for the benefit of all stakeholders. The Centre's mission is to provide the scientific rigour and support to enable the minerals industry to enhance its environmental performance from exploration to active mine management and closure. Research interests include water systems, water treatment and disposal and ecosystem restoration.
Website address: www.acmer.com.au

Minewater Treatment: Technology, Application and Policy.
Melanie Brown, Bob Barley and Harvey Wood. ISBN: 1 84339 004 3

ADTI

The Acid Drainage Technology Initiative (ADTI) was initiated in 1995 by federal agencies, the National Mining Association and the Interstate Mining Compact Commission to identify, evaluate and develop cost-effective and practical acid drainage technologies (Hornberger *et al.* 2000). In 1999 the ADTI was expanded through the addition of the metal mining sector group. The ADTI is a technology development and transfer programme that focuses on mine drainage sampling/monitoring, prediction, avoidance/remediation and modelling Website addresses: www.neetc.org/adti, www.mt.blm.gov/bdo/adti

AMIRA

There is active research into acid mine drainage prediction and prevention strategies being carried out by the Australian Mineral Industries Research Organisation (AMIRA, Greenhill 2000).
Website address: www.amira.com.au

BIOWISE

Biowise is a major UK Government Programme funded by the DTI. It aims to improve the competitiveness of UK industry through the use of biotechnology and support the development of the UK biotechnology supplier industry. Biowise does not currently have any demonstration projects directly relevant to minewater treatment, but passive minewater treatment technology would fall within the Biowise remit.
Website address: www.dti.gov.uk/biowise

CL:AIRE

CL:AIRE (Contaminated Land: Applications in Real Environments) is a public/private partnership involving the following stakeholders: government policy makers; regulators; industry; research organisations and technology developers. CL:AIRE provides a link between the main players in contaminated land remediation in the UK, to catalyse the development of cost-effective methods of investigating and remediating contaminated land in a sustainable way. CL:AIRE currently has projects concerned with minewater hydrogeochemistry and passive minewater treatment.
Website address: www.claire.co.uk

CLARINET

The Contaminated Land Rehabilitation Network for Environmental Technologies in Europe (CLARINET) includes a working group on the Protection and Rehabilitation of Water Resources that focuses on groundwater contamination affecting groundwater and drinking water resources, aquatic ecosystems etc.
Website address: www.clarinet.at/clarinet/groundwater.htm

CLARRC

The Contaminated Land Assessment and Remediation Research Centre (CLARRC) was founded in June 1998 by the University of Edinburgh, Napier University and the Scottish Universities' Research & Reactor Centre with seed funding from the Scottish Higher Education Funding Council. CLARRC's overall goal is to develop strategies and techniques for safely assessing and treating contaminated sites to a standard consistent with its beneficial use. CLARRC operates as a "virtual centre" with a core staff based at the King Buildings campus of Edinburgh University. Other academic participants are drawn from the three founding institutions and also from the British Geological Survey. It was formed to bring together representatives of the contaminated land industry, regulatory bodies, consultants and researchers with considerable expertise in the spectrum of disciplines relevant to contaminated land. CLARRC is primarily interested in contaminated land assessment and remediation. However, their activities and expertise covers a wide range of related areas. These include environments impacted as a result of land or groundwater pollution such as wetlands, natural watercourses and marine sediments.
Website address: www.clarrc.ed.ac.uk

ERMITE

The Environmental Regulation of Minewaters in the European Union (ERMITE) initiative is a European Commission Fifth Framework Programme under the key action Sustainable Management and Quality of Water. ERMITE is a 3 year research and development project which commenced on 1 February 2001. The goal of ERMITE is to provide integrated policy guidelines for developing European legislation and practice in relation to water management in the mining sector. These guidelines need to be coherent with the catchment management approach defined by the Water Framework Directive and the sustainability principles enshrined in the Amsterdam Treaty. It is co-ordinated in

Spain and includes partners based in the UK, Netherlands, Sweden, Germany, Slovenia and Bosnia-Hercegovina.
Website address: www. minewater.net/ermite

ICME

The International Council on Metals and the Environment (ICME) promotes sustainable development policies and practices related to the mining and production of primary metals, which will ensure the safe production, use, recycling and disposal of metals. ICME was founded as an environmental non-governmental organization in 1991. It brings together companies from different regions of the world that represent a major portion of the production of non-ferrous and precious metals. ICME provides its members with an opportunity to pool their expertise, exchange information and shape future industry practice while contributing to international discussions on important environmental and health issues. Their areas of interest include assessment of the ecotoxicological effects of metals in the environment and mining and environmental policy issues.
Website address: www.icme.com

INAP

The International Network for Acid Prevention (INAP) was formerly based in Australia and administered by Rio Tinto, but it has recently come under the auspices of MIRO (see below) and has the mission of formulating a global strategy for acid minewater, with case studies and research projects forming an integral part of the network.
Website address: www.inap.com.au

MEND 2000

The initial Mine Environment Neutral Drainage (MEND) programme started in 1989 and was a 9 year programme to develop scientifically based technologies to combat acidic drainages and it established Canada as a leader in research and development of remediation strategies for acidic discharges from metal mines. The current MEND programme, MEND 2000, is a 3 year programme that started in 1998, and has the key aim of technology-transfer, to provide state-of-the-art information and technology developments to users (Tremblay 2000).
Website address: http://mend2000.nrcan.gc.co/

MIMI

Sweden has a multi-disciplinary research programme (Mitigation of the Environmental Impact from Mining Waste, MiMi) that has wide-ranging research interests into acid mine drainage within five key areas: cost-effective prevention and control methods; prediction methods; microbial processes and their importance for AMD; management of process waters and leachate waters from mines and deposits; establishment of vegetation (Hoglund 2000).
Website address: www.mimi.kiruna.se

MINEWATER INFORMATION AND LINKS WEBSITE

This website is intended as a service to the disparate community of people around the world who are concerned with any and all aspects of mine water. The site is edited and maintained by scientists and engineers of the Minewater Research Team at the University of Newcastle, UK. The website address is www.minewater.net .

MIRO

MIRO (Mineral Industry Research Organisation) is an industry funded research and information organisation for the international minerals industry, that specialises in assisting member organisations to locate, develop and transfer innovative technology. MIRO was established in the early 1970's and is a non-profit making organisation limited by guarantee. It is run by its members for the benefit of those members. It is currently involved with the LINK project at Wheal Jane in Cornwall.
Website address: www.miro.co.uk

M-WINE

M-WINE (Minewater pollution: European Interdisciplinary Network) is a newly implemented network to facilitate exchange of information and knowledge between European scientists on the generation of polluted minewaters, mine waste management and mine accident prevention, and to promote international research collaboration in these fields. The final goal will be to improve practical measures for prevention and/or remediation of the minewater problem.
Contact/coordinator of the group: Bernhard.Dold@cam.unil.ch

PIRAMID

The PIRAMID project (Passive In-situ Remediation of Acidic Mine/Industrial Drainage) is a research project of the European Commission Fifth Framework Programme (Key Action 1: Sustainable Management and Quality of Water). It is concerned with the development of low-cost ecologically friendly solutions to long-term problems of water pollution from active and abandoned mine workings. There are British, Spanish, French, German, Swedish and Slovenian partners in the project, with the co-ordination of the project being carried out at the University of Newcastle.
Website address: www.piramid.org

PRB-NET

PRB-NET is an internet-based communications framework to bring together an inter-disciplinary group of people with interests in groundwater contamination and remediation technologies, with particular emphasis on permeable reactive barrier (PRB) technology. The aims include the advancement of technology transfer between academia and industry, and the dissemination of information throughout the scientific community.
Website address: www.prb-net.org

Appendix B

Active treatment suppliers

pH MODIFICATION TECHNOLOGY

Tetra Technologies Inc. 25025 I-45 North The Woodlands TX 77380-3034 USA	HDS Process www.tetratec.com
Unipure Europe Ltd Dr Richard Coulton 5 Auden Close Osbaston Monmouth NP25 3NW UK	Unipure Process www.unipure.co.uk

Melanie Brown, Bob Barley and Harvey Wood. ISBN: 1 84339 004 3

KEECO UK Ltd Wheal Jane Mill Baldhu Truro, Cornwall TR3 6ED, UK	Keeco Process – KB-1 www.mywebpage.net/keeco-uk/applications.htm
Robertson Technologies Corporation Suite 640 580 Hornby Street Vancouver BC V6C 3B6 Canada	Aquafix www.aquafix.com
Hazleton Environmental 125 Butler Drive Hazleton PA 18201 USA	Hazleton Iron Removal System www.hazletonenvr.com
Modular Environmental Technologies, Inc P.O. Box 8128 The Woodlands TX 77387 USA	Hardtac Process www.modenvtech.com
Virotec P.O. Box 188 Sanctuary Cove 4212 Queensland Australia	Bauxsol Process www.virotec.com
Mintek Hydrometallurgy Division Private Bag X3015 Randburg 2125 South Africa	Savmin Process www.mintek.co.za
Hydrometrics, Inc 2727 Airport Road Helena MT 59601 USA	CESR Process http://techlink.msu.montana.edu/nebc/members/hydrometrics.html

ION EXCHANGE

Chemical Effluent Treatment Process (Pty) Ltd
Consolidated Building
P.O. Box 590
Johannesburg 2000
South Africa

GYP-CIX Process

Mine Remediation Services Pty Ltd
GPO Box 975
Brisbane
Queensland 4001
Australia

KAD Technology
www.mine-remediation.com.au

Dynaphore, Inc
2709 Willard Road
Richmond
VA 23294
USA

Dynaphore
Forager Sponge

Metre-General, Inc
3771 Monarch Street
Unit A
P.O. Box 1149
Frederick
CO 80530
USA

Octolig MRP
www.octolig.com

BIOLOGY-BASED TREATMENTS

Paques Environmental Technology
T. de Boerstaad 24
P.O. Box 52
8560 AB Balk
The Netherlands

Thiopaq
www.paques.nl

NTBC Research Corporation
5320 McHardy Street
Vancouver, BC
V5R 4C5
Canada

Biosulphide Process
www.direct.ca/ntbc

Lawrence Consulting Ltd
1363 Chestnut Street
Vancouver, BC
V6J 3K1
Canada

www.members.home.net/lawrence.rick

Hydrometrics, Inc
2727 Airport Road
Helena
MT 59601
USA

ABC Process
http://techlink.msu.montana.edu/nebc/members/hydrometrics.html

Resource Management and Recovery
4980 Baylor Canyon Road
Las Cruces
NM 88011
USA

Algasorb

Microbial Technologies, Inc
1057 Roberts Creek Road – Unit 21
P.O.Box 193
Roberts Creek, BC
V0N 2W0
Canada

Innovative Bioremediation
www.microbialtech.com

Oak Ridge National Laboratory
Oak Ridge
TN 37831
USA

Ligand-grafted biomaterials

OTHER ADSORPTION TREATMENTS

Nalco

Nalmet
www.nalco.com

IBC Advanced Technologies
856 E Utah Valley Drive
American Fork
Utah 84003
USA

Molecular Recognition Technology
www.ibcmrt.com

Lewis Environmental Services, Inc 550 Butler Street Etna PA 15223 USA	Enviro-Clean Process www.lesvc.com/ EnviroClean
Selective Environmental Technologies, Inc 8601 Dunwoody Place Suite 302 Atlanta, Georgia 30350 USA	Mag*Sep
Electrochemical Design Associates, Inc 829 Heinz Street Berkeley CA 94710 USA	Selective Ion Recovery System www.e-d-a.com

ELECTROCHEMICAL TREATMENT TECHNOLOGIES

Electrochemical Design Associates, Inc Orinda CA 94563 USA	Rotating Cylinder Electrode Cells and Electrochemical Ion Exchange www.geokinetics.com
Aprotek Corbin Way Sacramento CA 95827 USA	Ion Conduction 3316 Agglomeration System

PHYSICAL PROCESS TECHNOLOGY

Ionics 65 Grove Street P.O. Box 9131 Watertown MA 02272 USA	RCC Brine Concentrator www.ionics.com

USF Memcor
Derby Road
Wirksworth
Derbyshire
DE4 4BG
UK
Continuous microfiltration system
www.usfilter.com

Enviro-Main Filter, Inc
Membrane technology
www.filter-this.com

Stewart Environmental Consultants, Inc
3801 Automation Way
Suite 200
Fort Collins
CO 80525
USA
Membrane technology
www.stewartenv.com

EPOC Water, Inc
3065 North Sunnyside
Fresno
CA 93727
USA
Membrane technology

Appendix C

Minewater in the UK: academic work

Academic Institution	Current Researchers	Department	Current/recent Projects
University of Newcastle	Paul Younger	Water Resources Systems Research Laboratory	PIRAMID
	Leslie Batty		
	Charlotte Nuttall		Manganese removal
	Karen Johnson		Modelling
	Russell Adams	Civil Engineering	Laboratory and numerical modelling studies of iron release from a spoil heap in County Durham
	Catherine Gandy		
University of Sheffield	Steve Banwort	Groundwater Protection & Restoration Group, Dept Civil & Structural Engineering	Minewater risk assessment – mineral weathering rates
	Sean Burke		Modelling of Fe kinetics at Blaenant-Ynysarwed (with Newcastle)
	Stephanie Croxford		Origin of minewater contaminants
	Chris Lloyd		Assessing the long-term contaminant load of minewater discharges (with Newcastle)
	Katy Evans		

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Academic Institution	Current Researchers	Department	Current/recent Projects
University of Edinburgh	Kate Heal	Ecology & Resource Management	Constructed wetlands for acidic spoil heap runoff (Benhar Bing)
	Colin Cunningham	CLARRC (Contaminated Land Assessment And Remediation Research Centre)	Uses for ochre from minewater treatment
University of Nottingham	David Reddish	Chemical, Environmental & Mining Engineering	Ammonia removal at Woolley
University of Leeds	Kath Morris Rob Raiswell John Hine	Environment Earth Sciences	Research at Dodworth
	Tobias Puhlman	MSc Environmental Geochemistry	Fe kinetics at Wheal Jane
University of Cardiff	Matthew Dey Simon Dominy	Engineering Earth Sciences	Ochre build-up at Pelenna
	Chris Bullen		Research based on Wheal Jane active treatment
Imperial College	John Monhemius Peter Swash	RSM	Wheal Jane LINK project
	Bill Dudeney O Demin		Re-use of iron rich sludges Ammonia at Woolley Bulhouse project (completed)
University of Wales, Bangor	Barrie Johnson Kevin Halberg Chris Freeman	Biological Sciences Ecology Group Wetland biogeochemistry	Wheal Jane LINK project Microbiology Constructed wetland biogeochemistry
University of Reading	Paul Whitehead Hannah Prior	Geography	Wheal Jane LINK project
	Geoff Warren	Soil Science	Adsorption capacity of ochre for arsenic
Camborne School of Mines	Keith Atkinson Melanie Brown Bob Barley		Wheal Jane LINK EMWaTCh
	Terri Stoddern		Longevity of minewater treatment wetlands
	David Watkins		Hydrogeological modelling

Academic Institution	Current Researchers	Department	Current/recent Projects
Queens University, Belfast	Graham Saunders		Removal of heavy metals from wastewaters using polyelectrolyte chelates
	Matt Dring Adam Mellor		Use of seaweed to remove organic nutrients and heavy metals from wastewaters
	Bob Kalin Kayleen Walsh		Computer modelling of reactive barrier performance PRB-NET
University of Stirling	Carol Salt	Environmental Science	Work with Edinburgh on wetland systems for spoil runoff (Benhar Bing)
Natural History Museum	Eva Valsami-Jones	Mineralogy	Adsorption studies (using apatite); effect of acid mine drainage on river water chemistry (REE) and soils
University of Bristol	Vala Ragnarsdottir Tony Kemp	Earth Sciences	Undergraduate research projects at Pelenna
University of Exeter	Phil Smith Jonathan Rogers		Use of glauconite clay to remove copper from wastewaters
	Garrett Betlam Kate Getliffe	Law	Legal aspects of minewater - ERMITE

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Much work has been carried out in recent decades concerning minewater treatment, both in the UK and worldwide. Many different bodies and organizations are involved in developing minewater treatment processes and schemes. *Minewater Treatment* addresses the need for a single source of state-of-the-art information that draws all the latest research material together.

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