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Engineering design aspects of passive *in situ* remediation of mining effluents

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Abstract

Passive treatment of contaminated effluents can offer a 'low cost' management opportunity to remediate drainages to the standards required by enforcement agencies. However, the initial cost of construction of passive treatment systems is significant and often in excess of that for active treatment systems. It is therefore important that the engineering design of the passive systems produces an effective and efficient scheme to enable the construction and maintenance costs to be minimised as far as possible. Possible parameters for the design of passive systems are suggested to seek to obtain uniformity in size and layout of treatment elements where this may be possible.

Passive treatment systems include aeration systems, sedimentation ponds, aerobic and anaerobic wetlands, anoxic limestone drains and reducing alkalinity producing systems. Most active treatment systems also include passive elements in the treatment stream. The basic design considerations that should be considered to ensure the construction of efficient systems are discussed.

Key words: construction, design, maintenance, mine water, passive treatment

INTRODUCTION

Mine water generation and impacts

When mining galleries are excavated, atmospheric oxygen and water may come into contact with the exposed rock. Pyrite minerals (FeS_2), ubiquitous in Coal Measures and Coal Measure shales, are readily oxidised by the atmospheric oxygen. The oxidised residue is highly soluble, and therefore easily washed from the surface of the rock by water draining through the workings. The overall reaction can be summarised as follows (from Barnes and Romberger 1968):

$$4\,FeS_2 + 15\,O_2 + 14\,H_2O \Rightarrow 4\,Fe(OH)_3 + 16\,H^+ + 8\,SO_4^{2-}$$

In reality, as many as 15 reactions are involved in the oxidative dissolution of pyrite (Nordstrom and Southam 1997), and the reaction is bacterially catalysed (Singer and Stumm 1970). However, the equation above is sufficient to illustrate that:

(i) iron is released into the water, initially perhaps as ferrous iron (Fe^{2+}), but ultimately as ferric hydroxide in the surface environment; and
(ii) the reaction generates acidity (represented here by the protons, H^+).

Other sulphide minerals, such as sphalerite (ZnS), greenockite (CdS) and arsenopyrite (FeAsS), may also release metals into solution, but these monosulphides do not have the same acid-generating potential as the disulphide, pyrite.

Discharges at the surface may be alkaline if, for example, carbonate strata overlie the Coal Measures. Discharges that are not buffered may emerge acidic. These discharges may contain other metals, such as aluminium and manganese, because of their higher solubility in low pH waters. Such is the extent of pyrite within workings, this oxidative dissolution process may continue for many decades, if not centuries, before the pollution abates without intervention.

Approximately 700 kilometres of rivers in the UK are seriously impacted by drainage from abandoned coal workings (Jarvis and Younger 2000). This includes around 60 significantly contaminated mine water discharges and around 350 less significant discharges. Any discharge with an iron concentration above approximately 2 mg/L is likely to cause visual staining due to

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the deposition of iron hydroxide (often termed 'ochre'). High-volume, poor quality, discharges may cause staining for many kilometres downstream, rendering these waters unsuitable for abstraction for many uses, and often devastating the aquatic fauna and flora of the receiving watercourses (Jarvis and Younger 1997).

PASSIVE TREATMENT SYSTEMS

Passive remediation may be defined as follows:

> Passive *in situ* remediation signifies an engineering intervention which prevents, diminishes and/or treats polluted waters at source, using only naturally available energy sources (such as topographical gradient, microbial metabolic energy, photosynthesis and chemical energy), and which requires only infrequent (albeit regular) maintenance to operate successfully over its design life.
> *Passive In situ Remediation of Acidic Mining/Industrial Discharges Research (PIRAMID). This project was funded under the EU Fifth Framework Programme, website www.piramid.org*

The development of passive systems for minewater treatment arose from the longevity of such discharges (which, in some cases, may remain polluted for centuries). Because of this, the application of 'active' treatment methods (i.e. use of chemicals and energy) is often unappealing due to the high, and long-term, on-going costs. Passive treatment systems therefore endeavour to make use of natural (micro)biological and chemical processes that remove contaminants (predominantly iron and acidity) in an equally effective manner.

There are two principal strategies in passive *in situ* remediation (PIR):

- ***Passive prevention of pollutant release*** is achieved by the installation of physical barriers (requiring little or no long-term maintenance) that inhibit pollution generating chemical reactions (for instance, by permanently altering redox and/or moisture dynamics), and/or directly preventing the migration of existing polluted waters.
- ***Passive treatment*** is achieved using constructed (or appropriated natural) gravity flow systems, in which all treatment processes used meet the criteria of the definition given above.

Passive treatment technologies embrace wetland type systems, sub-surface reactive barriers, and an increasing array of gravity flow geochemical reactors. The design of major passive treatment units is discussed further in this report.

Throughout the discussion it is important to bear in mind that passive treatment is not always a feasible option for mine water treatment. This is because pollutant removal mechanisms in passive systems are slower than the equivalent processes in active treatment. Consequently, passive systems typically have to be far greater in size, and the area of land required is therefore greater. For very high volume, poor quality discharges, designers may have no option other than to resort to chemical treatment.

In practice many systems draw on both active and passive treatment technologies. For example, the Old Meadows mine water treatment scheme, Lancashire, UK, employs sodium hydroxide dosing to raise the pH of the acidic discharge (facilitating more rapid precipitation of iron), which then flows through a sedimentation pond and tertiary treatment wetland to remove the iron hydroxide precipitate. At Vivian (Six Bells), South Wales, a net-alkaline (i.e. alkalinity > acidity) discharge contains circa 50 mg/L iron. This discharge is dosed with hydrogen peroxide to facilitate rapid oxidation of ferrous iron to ferric iron (active treatment), before discharge to sedimentation ponds and a tertiary wetland (passive treatment).

DESIGN DATA COLLECTION

It is vital that representative information on the quality and quantity of the discharge is obtained over as long a period of time as possible before the design commences. Ideally flow measurements should be made over a minimum of a twelve-month period, so that monthly and annual variations may be determined. Unfortunately, it is often the case that a 'design and construct' contract is not long enough to permit the collection of data over such a long period.

Sophisticated flow measurement devices (such as continuous data logging) are often unnecessary for mine waters. Simple structures such as 'V'-notch weirs, H-flumes, and even a bucket and stopwatch, can provide adequate information to give confidence in the design. Weir structures are often simple and easy to install and monitor. For treatment systems, the vital design variable is the maximum flow-rate, and variations within that flow-range are not always vital. Regular 'snapshot' monitoring of the flow, by measurement of the depth over a weir, will often provide sufficient data. The maximum flow-rate of a mine water discharge can often be estimated, since ochreous discharges invariably leave a 'tide mark' on the measuring structure.

Wherever possible, it is advisable to build a contingency into the design to allow for unusually high flow-rates. This is particularly the case where few flow-rate data are available. Particularly in the case of shallow abandoned coalfields, flow measurements should be taken in both dry and wet periods, as the influence on discharge rate can be significant.

In terms of water quality measurement, there is no substitute for field testing of the effluent at the point of discharge. This ensures that the design data are truly representative of the quality of the mine water. All too often, designs have been based on the results of laboratory analysis of water samples that have been transported and stored for many days, during which time the chemistry of the effluent may change significantly.

For discharges arising from abandoned coal mines, or their spoil heaps, the key water quality design variables are *usually* pH, acidity (including determination of permanent/temporary status), alkalinity and ferrous and ferric iron concentration. Therefore the field-testing carried out should include these variables, together with other variables such as temperature, dissolved oxygen concentration, and conductivity.

More detailed analysis should be undertaken, particularly at the outset of a project when laboratory analysis of a range of metals should also be undertaken. It is always easy to identify if iron, and often aluminium, is a problem, because of the very visible staining caused. Other metals are not visible in watercourses unless they are at very high concentrations e.g. zinc, manganese, copper. In determining which elements to analyse for, careful inspection of the local geology and land use should give clues as to what else may be present.

Analysis of calcium, magnesium, sodium and potassium will assist in establishing the source and evolution of waters (often important if the pathway of water through long-abandoned workings is initially unclear). Concentrations of the major anions (sulphate, chloride and bicarbonate) should also be determined. Sulphate is invariably present in high concentrations. Although not the case in the UK, in semi-arid countries such as South Africa, removal of sulphate is the primary objective of treatment, since these waters are subsequently required for potable supply (the processes that remove sulphate will also remove any metal contaminants).

In the case of mine waters emerging from deep workings following rebound, the quality of a mine water discharge will normally change over time. Often there is a 'first flush' of very poor water quality (Younger 1997). The quality normally improves quickly but it takes a considerable period of time to reduce to the level of contamination experienced when the mine was working. For example, it took the Morton discharge in Derbyshire, UK about 30 years to improve from the initial quality of 160 mg/L total iron to 40 mg/L total iron: a level still above that measured during operation of the mine. The following equation has been derived for the estimation of the period of the first flush (from Younger *et al.* 2002):

$$t_f = (3.95 \pm 1.2).t_r$$

where t_f = duration of first flush and t_r = rebound time i.e. time for workings to flood

In the UK there has been an increasing trend towards intercepting rising mine waters during rebound, to prevent an uncontrolled surface discharge (this is often done because the likely location of the uncontrolled discharge is unclear). This can be achieved by sinking a borehole into abandoned workings, or utilising old mineshafts and then pumping water at a rate sufficient to maintain a steady water level in the workings. The quality of water discharged from a borehole should be monitored, particularly at the outset of pumping, as it invariably deteriorates, in a similar fashion to that experienced during the 'first flush' period of uncontrolled discharges. Caution should also be taken in respect of sampling of water quality in shafts where stratification can give rise to unrepresentative results.

Mine water hydrology and quality may also change for the worse, often without warning. The Bullhouse mine water discharge, UK, turned from net-alkaline to net-acidic during construction of the treatment system, giving rise to the need to reconsider the treatment philosophy (Laine 1997). Conversely, the mine water quality at the Woolley treatment system, UK, improved significantly because of abstraction and treatment of mine water at the adjoining Caphouse mine (Laine 1997). At the Pelenna site in south Wales, the system was completed and operating successfully when the discharge dried up, only to reappear some 50 metres up the valley, presumably due to a collapse within the workings (P.L. Younger, University of Newcastle, UK, *personal communication* 2001).

These anecdotes reinforce the need for representative mine water quality. The unpredictable nature of such events makes it advisable to incorporate flexibility into designs.

PASSIVE TREATMENT PROCESS SELECTION

In the following paragraphs specific design guidance for passive treatment units is provided. This should provide potential implementers of passive treatment schemes with an overview of engineering requirements for such systems. However, as a broad guide to the reader, Figure 1 illustrates a 'decision tree' for selection of most appropriate passive treatment technology

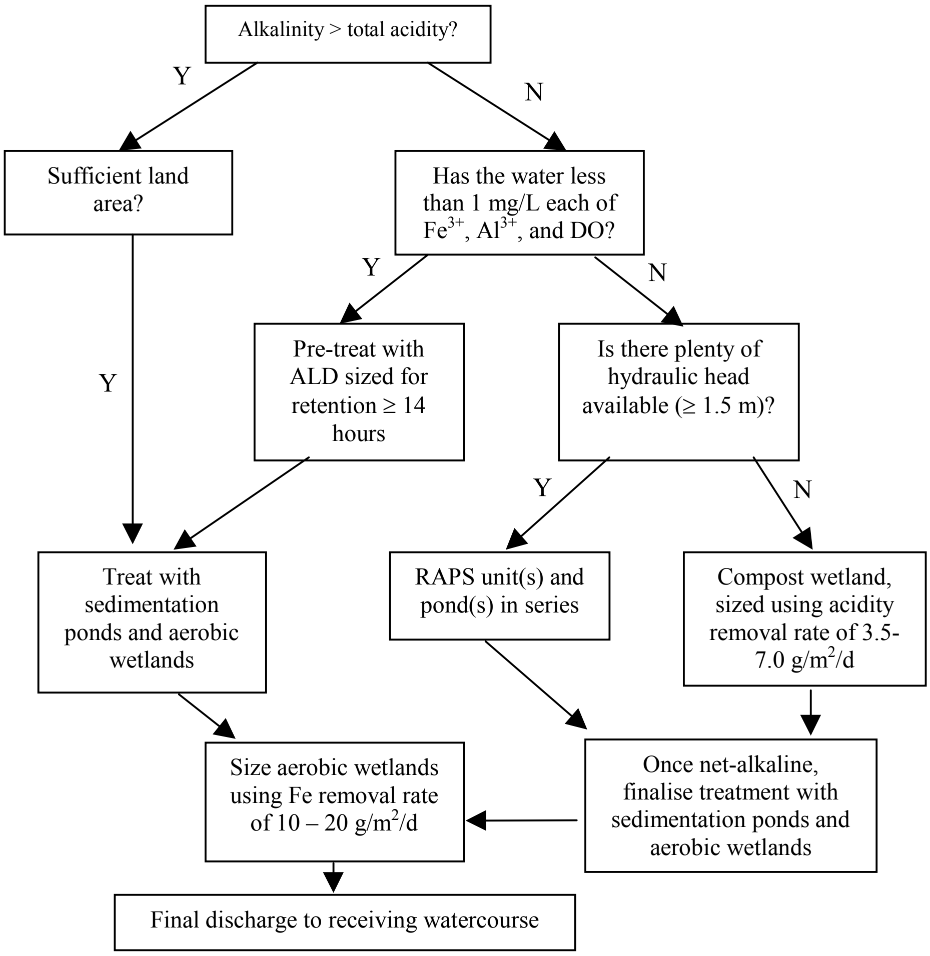

Figure 1. Simplified decision flow chart showing passive treatment unit selection (adapted from Younger et al. *2002) (see text for further details)*

for any given discharge. The flow chart includes only treatment technologies discussed in this article. Other technologies do exist, though they are less widely applied, at least in the UK. For example, if the answer to the 'sufficient land area?' question is 'No', passive treatment may still be possible, by using inorganic media systems (Younger *et al.* 2002). Details can be found in Jarvis and Younger (2001) and Younger *et al.* (2002). Subsurface flows may be intercepted using permeable reactive barriers (PRBs). Benner *et al.* (1997) and Younger *et al.* (2002), give details.

For some discharges (especially those of very poor quality and high volume) passive treatment will be inappropriate. It is not possible to give a specific iron or acidity loading at which passive treatment becomes unfeasible, because these decisions are site specific. Such a decision is typically based on an assessment of the costs and benefits (over the short and long term) of the options available (be they passive or active), in view of the land area available and its topography.

Examples of a variety of passive treatment systems currently operational in the UK are shown in Table 1. The reasons why particular treatment unit(s) were selected at each site should become clear in the following paragraphs.

GUIDELINES FOR THE DESIGN OF SEDIMENTATION PONDS

Introduction

There are significant advantages in having standard guidance for water treatment systems in any industry, based upon experience of design and operation of such structures in compliance with enforcement agency standards. In the past, water treatment facilities for drainages have been considered the 'poor relations' in the development of a site, and were relegated to whatever small area of land remained when the development proposals were completed. With the increasing

Table 1. Examples of different passive treatment systems in the UK

Site name	Water quality/flow			Treatment type	Comments
	Mean flow-rate (L/s)	Net-alkaline/ net-acidic	Mean iron concentration (mg/L)		
Quaking Houses, Co Durham[1]	2	Net-acidic	6	Compost wetland and aerobic wetland	Insufficient hydraulic head for RAPS
Deerplay, Lancashire[2]	2	Net-acidic	22	RAPS and aerobic wetland	Steeply sloping site, limited land availability
Edmondsley, Co Durham[3]	10	Net-alkaline	30	Aerobic wetland	Flat site, Fe load not excessive
Woolley, West Yorkshire[4]	150	Net-alkaline	15	Sedimentation ponds and aerobic wetland	Large flat site; high iron load

1. Jarvis and Younger (1999)
2. Jarvis and England (2002)
3. Younger *et al*. (2002)
4. Laine (1997)

aspirations of the enforcement authorities for higher quality effluent discharges, often driven by European Union legislation, it is increasingly unlikely that facilities allocated in such a manner can comply with the discharge quality constraints imposed. The area required for effective facilities needs to be related to the size of the catchment and/or the quantity and quality of discharge, if effective remediation is to be achieved. Whilst it is accepted that the area allocated must not be excessive, as it may impair the development, it is nevertheless required to be a realistic area that can achieve the water quality improvements necessary.

In the UK coal industry, basic water treatment design guidance for engineers was provided by NCB (1982). To some extent this text was instrumental in achieving a uniformity of effective design throughout the industry. Although not in print, copies remain available in some libraries. However, other than sedimentation ponds, NCB (1982) did not consider the design of passive systems for mine water remediation (i.e. wetlands, biogeochemical reactors). The paragraphs following provide guidance on the design of sedimentation ponds, wetlands and other passive treatment technologies, and provide references to some of the key texts in the field.

Sedimentation pond size

Sedimentation ponds invariably form the main repository for metal precipitates at mine water treatment sites. However, for them to be effective in removing iron solids, the water must be net-alkaline at the influent point, and there must be sufficient dissolved oxygen present to enable the oxidation and precipitation reactions to occur. Therefore sedimentation ponds are invariably preceded by an aeration step, and sometimes passive or active means of alkali generation. Aeration may be chemically induced, or it may achieved by a simple cascade system. Where iron concentrations are high, i.e >30 mg/L, multiple cascades may be required (NCB 1982).

In the UK coal mining industry, sedimentation pond areas were designed on the basis of providing 40 square metres of surface area of pond for every hectare of surface drained. This is effectively equivalent to an overflow rate of 1×10^{-5} metres/second in conventional water industry design parlance, or 100 m^2 of pond surface area for every litre/sec of drainage (NCB 1982). This concept was based upon experience in the USA, and is primarily relevant to inert suspended solids.

Whilst this ‘rule of thumb’ approach is useful for quick estimates of area requirements, it is based on settlement of particle sizes of >4 μm. However, particles of iron hydroxide are typically smaller than this (often less than 2.5 μm, at least initially) (Younger *et al.* 2002). The combination of the time for the formation of $Fe(OH)_3$ precipitates, and the smaller particle size, means that residence time is a crucial design consideration. A retention time of 24 hours is likely to be adequate, but because actual retention time will be less than theoretical retention time (i.e. pond volume divided by flow-rate) as sludge depth increases, total pond volume is often designed on the basis of at least 48 hours retention time.

A further consideration in pond design is the frequency of sludge removal required. As a guide level, sludge should be removed before it is within 1 m of the water surface, but the best guide to when sludge removal is required is the effluent iron concentration. Systems designed by IMC are usually designed in such a way that sludge removal will not have to be undertaken more than once per year.

Chemical dosing in sedimentation ponds

Where inadequate space is available to allow the construction of sedimentation ponds of the desired size, resort may have to be made to chemical dosing. The intention of the dosing is to achieve rapid generation of suspended solids so that settlement takes place immediately over the full area of the pond. There are notable cases of schemes designed without appreciation of the time of reaction required to convert ferrous iron to ferric iron, where mine water had flowed about 50% through the pond system before precipitation of the iron solid allowed sedimentation of generated particles to commence.

IMC Consulting Engineers have recent experience of the construction of active chemical treatment schemes for mine water using hydrogen peroxide at the Vivian (Six Bells) and Acomb mine water schemes. A 'rule of thumb' guideline suggests a provision of 50 m^2 of pond surface area for every 1 L/s of influent flow – half the guideline for inert solids, and significantly less than for fine ferrous particles requiring the addition of reaction time.

Rapid precipitation of solids can also be achieved by the use of alkaline materials such as sodium hydroxide, whether the mine water be acidic or alkaline. A provision of pond surface area also of 50 m^2 area for every 1 litres/sec of influent flow is also thought to be appropriate.

The use of chemical dosing involves cost, and the health and safety implications of using aggressive chemicals should be avoided unless land availability constraints dictate active treatment.

Sedimentation pond geometry and layout

Pond geometry and layout are important considerations, which influence both the operational efficiency of treatment, and the amenity value of the completed system. Exact dimensions and layout will clearly be governed to an extent by local site conditions, but there are certain general rules that should be applied in all circumstances.

Where a number of sedimentation ponds are proposed, often to allow for continuation of treatment during desilting, the ponds should be arranged to operate in parallel and not in series. Splitting the flow between two ponds reduces the velocity of flow to 50%, which promotes greater efficiency for sedimentation. Where ponds of equal size are arranged in a series, there is no reason why an inert particle that has not been detained in the first pond should subsequently settle in the second pond, as the sedimentation potential is identical in each.

Practical experience suggests that the best sedimentation performance is obtained when the ratio of the pond length to width is between 3:1 and 5:1. Higher length to width ratios may result in 'streaming' across ponds, whilst lower length to width ratios may promote short-circuiting.

Multiple sedimentation ponds have other advantages:

(i) by arranging ponds so that they may be operated in parallel, sludge may be removed from one pond (or maintenance undertaken) whilst the second continues to operate;

(ii) carefully designed, a site may look more attractive with multiple smaller ponds as opposed to a single large pond with an equivalent volume.

Aeration stages

The oxidation of ferrous iron to ferric iron uses up dissolved oxygen in the water. Since there is a maximum concentration of oxygen in water (9.1 mg/L at 1 atmosphere and 20°C, in freshwater) it stands to reason that high iron concentrations will exhaust the supply. For this reason it is sometimes necessary to have multiple aeration cascades, and therefore multiple ponds. Based on the theoretical rate of oxygen consumption, there should be one aeration cascade per 50 mg/L ferrous iron (Younger *et al*. 2002). Practical experience suggests that inefficiencies in oxygen transfer effectively make the requirement one aeration cascade per 30 mg/L ferrous iron (NCB 1982). Thus, for a discharge with 90 mg/L ferrous iron, there should be three aeration cascades, each followed by a sedimentation pond.

Flow inlet and outlet structures

Good flow distribution increases operational efficiency by keeping inlet velocities low, and minimises the potential for short-circuiting. The objective of inlet and outlet structures is to spread the flow evenly across the whole of the pond width. These guidelines apply equally to sedimentation ponds and wetlands.

Conventional means of distribution on sedimentation ponds generally comprise both full-length weir inlet and outlet systems or a multiple pipe inlet arrangement. Full-length concrete weir systems are expensive to install, but operate by minimising velocity over the length of the weir. In reducing the velocity, significant settlement of solids often occurs in the inlet channel. Whilst simple to rectify, safety of operatives working close to deep water must be ensured.

Multiple pipe inlet systems are much cheaper to install and generally comprise a number of pipe outlets over the full width of the pond to achieve even distribution. Whilst this may not be achieved as effectively as a full-length weir, it is nevertheless adequate for most sedimentation pond systems. It is probably slightly more inconvenient to maintain such pipes, as rodding equipment will be necessary, but the locations at which the work will be carried out are generally reasonably convenient and safe. Pipe flow velocity can be designed to minimise accretion of ochre within the pipe network.

Irregular shaped pond systems will provide more natural looking features in the landscape, although it must be recognised that such shapes are not necessarily complementary to effective sedimentation performance. The even passage of water through the system is more difficult to achieve, and features such as shallow ledge areas around the pond perimeter significantly reduce sedimentation capacity. It is vital that the minimum calculated pond requirement is provided with the full depth required for sedimentation purposes and that the aesthetic curved areas to form the landscaping element are *in addition* to the basic requirement.

GUIDELINES FOR WETLAND AND OTHER PASSIVE SYSTEM DESIGN

Introduction

There are two types of wetland that are generally employed for mine water treatment:

(i) aerobic wetlands;
(ii) anaerobic wetlands

Aerobic wetlands are used where treatment requirement is limited to oxidation of dissolved contaminants and the detention of the generated suspended solids. Most commonly they are used for the retention of ferric iron precipitates. Such systems typically have a soil media in which metal-resistant wetland plants are grown.

Anaerobic (compost) wetlands may be used for net-acidic discharges (i.e. acidity > alkalinity). Such systems have a compost medium, the exact nature of which is discussed below. Plants are less of an essential component in compost wetlands, but may still be encouraged for aesthetic reasons. In compost wetlands biogeochemical reactions are encouraged to generate alkalinity through sulphate reduction processes and precipitate metals (especially iron) as insoluble monosulphides.

Aerobic wetlands

The design guidelines of Hedin *et al*. (1994*a*) are still those that are most commonly applied in the design of aerobic wetlands. For a net-alkaline discharge, where removal of iron is the objective, the following formula is used:

$$\text{Wetland area (m}^2\text{) = iron load (g/d) / removal rate (g/m}^2\text{/d)}$$

It will be evident from this that the knowledge of the flow-rate and iron concentration are required to calculate the numerator of the equation. Hedin *et al*. (1994*a*) derived the removal rate from monitoring wetlands in the USA. For a wetland from which the effluent must meet regulatory standards, a figure of 10 g/m^2/d was derived. A second figure of 20 g/m^2/d was derived for situations where regulatory standards were not so stringent, and a ‘reasonable improvement’ in water quality was sufficient. It should be noted that this equation assumes removal of *all* of the iron. In practice this is both unlikely and unnecessary. The iron load can therefore be calculated by subtracting the target effluent iron concentration (e.g. 2 mg/L) from the influent iron concentration, and then multiplying by the flow-rate.

With the increasing number of wetlands now operational in the UK, it has become possible to design wetlands on a *pro rata* basis – i.e. take the removal rate from an operational wetland and apply it to one under design. However, such an approach must be applied with caution, since mine waters are unique, and therefore removal rates may vary. It is always worth calculating the area according to the removal rates as a check.

Two examples of actual removal rates in aerobic wetlands in the UK are as follows:

(i) the Old Meadows wetland, which has a flow-rate of 40 L/s, and influent total iron concentration of up to 10 mg/L. The surface area of the wetland is 1800 m^2, and effluent iron concentration is (less than) 2 mg/L. Therefore the removal rate is 15.4 g/m^2/d;
(ii) the Woolley wetland, which had a flow-rate of 200 L/s, and influent total iron concentration as high as 10 mg/L. The surface area of the wetland is 14 000 m^2, and effluent iron concentration is (less than) 1 mg/L. Therefore the removal rate is 11.1 g/m^2/d.

The general dimensions of aerobic wetlands should be as for sedimentation ponds. However, as wetlands may form effective amenity areas, there are significant advantages in avoiding rectilinear shapes. Nevertheless, steps should still be taken to minimise streaming or short-circuiting. Planting will help significantly in spreading water over the full width of the wetland without any engineering intervention.

Vegetation and growing media for aerobic wetlands

Typically the growing medium in aerobic wetlands is a good quality soil, usually placed to a depth of 300–400 mm. Important aspects of the soil quality are as follows:

- it does not contain excessive large stones or other sharp objects, which may puncture pond liners and impede plant growth;
- the soil is not contaminated. The best way to ensure this is to check the source of the soil, and ensure that a reliable analysis is made;
- it contains sufficient nutrients, in appropriate proportions, to support reed growth.

Research is presently taking place into the potential for use of suitable 'waste' materials as a growing medium, as often these can be obtained at minimal costs, thus avoiding the costly import of soil.

Aquatic plants in wetlands serve a number of useful functions:

(i) plants are excellent at ensuring flow is distributed evenly across the wetland, as long as they are planted across the direction of flow rather than parallel to it;
(ii) stems and leaves may provide additional surface area for adsorption of metals;
(iii) they significantly improve the aesthetic appearance of site, and may form a wildlife habitat;
(iv) even in aerobic wetlands, the continuous carbon source provided by plants may encourage sulphate reduction reactions in the subsurface, in turn encouraging the immobilisation of metals (Mitsch and Wise 1998).

Bioaccumulation is not generally recognised as an important removal process in wetlands for mine water treatment, and certainly not in respect of iron. However, recent research at a wetland in the north-east of England suggests that approximately 30% of the iron (influent concentration 3 mg/L) is removed by bioaccumulation (P.L. Younger, University of Newcastle, *personal communication* 2002). This may well be related to the comparatively low influent concentration.

Recent laboratory research (Batty and Younger 2002) has suggested that subjecting wetland plants to water with iron concentrations in excess of 10–20 mg/L will result in limited root development, and potentially poor overall growth and death. Reed die-back at some sites in England has recently been ascribed to the influent iron concentration being too high. However, it is more likely that the depth of water in these wetlands was too great, and the plants were therefore inundated. It is important to limit water depth above the surface to around 100–200 mm to prevent this, although plants such as *Scirpus lacrustus* can tolerate greater depths.

Evidence that reeds are capable of tolerating high iron concentrations comes from the USA. In deriving the design formula discussed above Hedin *et al.* (1994*a*) studied 16 wetlands, nine of which received water with >100 mg/L iron. All of the sites had been operational for several years, but none had suffered significant loss of reeds. At the current time the *laboratory-based* evidence suggesting that reeds will not tolerate iron concentrations greater than 20 mg/L is far outweighed by the *field-based* experiences to the contrary.

Nevertheless, most aerobic wetlands in the UK are used as tertiary treatment systems, following sedimentation lagoons. However, this has nothing to do with reed tolerance to iron but is because:

(i) using the formulae above, a wetland receiving a high iron concentration will require a larger land area than an equivalent sedimentation pond; and
(ii) removing large volumes of ochre from a wetland is far more difficult than removing it from a sedimentation pond (indeed, the plants must be removed as well).

A range of emergent aquatic plants have been used in wetlands for mine water treatment. Most commonly *Typha latifolia* (common name greater reedmace in the UK, or cattails in the USA) have been used. Increasingly monocultures are being avoided, and other types are planted in addition to *Typha*. Other commonly used aquatic plants include:

Phragmites australis (common reed)	Widely used, but may not thrive on exposed sites.
Juncus effusus (soft rush)	A naturally common species in wet upland areas
Scirpus (bulrush)	Is tolerant of deeper water depths where other types may not survive
Iris pseudacorus	An attractive species, improving appearance of wetland.

Typically, reeds are planted at a density of 3–4/m^2, between the months of May and June in the UK. 200 mm pot grown varieties are often favoured, as they are sufficiently advanced in terms of growth to survive conditions in the wetland, although considerable success has been achieved with cheaper 9 cm plugs.

Anaerobic (compost) wetlands

The reactions occurring in compost wetlands are more complex than those of aerobic wetlands. There are two key aspects to the removal of contaminants in compost wetlands. The first is the generation of alkalinity (and therefore neutralisation of acidity). This is accomplished in one or both of the following routes:

(i) generation of alkalinity via microbially mediated sulphate reduction. The media used (see below) must be suitable for the colonisation of sulphate reducing bacteria (SRB);
(ii) dissolution of high calcium carbonate limestone, mixed into the compost media during placement.

For SRB to be active, conditions must be anoxic, the bacteria must have a source of low-carbon number compounds to metabolise, and there must be high sulphate concentration (>100 mg/L). In very simple terms the reaction that occurs (where CH_2O represents the carbon source) is as follows (Younger *et al.* 2002):

$$SO_4^{2-} + 2\,CH_2O \Rightarrow H_2S + 2\,HCO_3^-$$

Metals may subsequently form insoluble precipitates in the following manner (where M^{2+} represents a divalent metal ion) (Hedin *et al.* 1994*a*):

$$M^{2+} + H_2S + 2\,HCO_3^- \Rightarrow MS + 2\,H_2O + 2\,CO_2$$

Since the primary objective of compost wetlands is to generate alkalinity under anoxic conditions, it is usually necessary to follow a compost wetland with another treatment unit, to aerate the water and remove metal contaminants as hydroxide precipitates (a reaction that will happen effectively once alkalinity is raised). In terms of passive treatment, this is ideally achieved using an aerobic wetland.

The design of anaerobic wetlands is usually based on the acidity load of the discharge and the anticipated removal rate of that acidity. As with aerobic wetlands, the design formula of Hedin *et al.* (1994*a*) is most commonly applied:

$$\text{Wetland area (m}^2\text{)} = \text{acidity load (g/d)} / \text{removal rate (g/m}^2\text{/d)}$$

For a wetland from which the effluent must meet regulatory standards, a removal rate of 3.5 g/m^2/d was derived. A second figure of 7 g/m^2/d was derived for situations where regulatory standards were not so stringent, and a 'reasonable improvement' in water quality was sufficient.

Anoxic limestone drains (ALDs)

An anoxic limestone drain (ALD) is a buried bed of high calcium carbonate limestone, the objective of which is solely to raise pH. Dissolution of the calcite neutralises proton acidity and generates bicarbonate alkalinity, as shown by the following reactions (Younger *et al.* 2002):

$$CaCO_3 + 2\,H^+ \Leftrightarrow Ca^{2+} + H_2O + CO_2$$

$$CaCO_3 + H_2CO_3 \Leftrightarrow Ca^{2+} + 2\,HCO_3^-$$

The ALD is buried in order to promote anoxic conditions. This prevents oxidation and precipitation of ferrous iron and manganese, which would otherwise coat the limestone surfaces, preventing further dissolution (and therefore alkalinity generation). This coating process is referred to as 'armouring'. In ambient conditions aluminium only occurs as Al^{3+}, and will readily hydrolyse to Al $(OH)_3$ under anoxic conditions.

For these reasons ALDs are only appropriate where Fe^{3+} and Al^{3+} concentrations are less than approximately 2 mg/L, and for other ferruginous waters where dissolved oxygen concentration is ≤1 mg/L (Younger *et al.* 2002).

The limestone should have a calcium carbonate content in the order of 90% to work effectively. To prevent blocking problems this should be single size limestone of size in the range 50–75 mm. Sizing of ALDs is based on residence time required to generate maximum alkalinity (around 300 mg/L as $CaCO_3$). Currently this time is taken to be 14 hours (Hedin *et al.* 1994*b*). For the purposes of design calculations, limestone of the size quoted is assumed to have a porosity of 50%.

Because of the restrictions on the quality of mine water suitable for treatment in an ALD (i.e. the Fe, Al and dissolved oxygen concentration), they are not particularly suited for use in the UK. Acidic discharges in the UK typically contain elevated iron and aluminium concentrations, and therefore for these discharges either a compost wetland or a Reducing and Alkalinity Producing System (RAPS) is adopted.

Reducing and alkalinity producing systems (RAPS)

RAPS were developed in response to the shortcomings of ALDs by Kepler and McCleary (1994). Conceptually a RAPS is an ALD overlain with a layer of compost. Water is driven downwards through the compost substrate, which removes dissolved oxygen, converts Fe^{3+} to Fe^{2+}, and is a sink for Al^{3+} (as aluminium hydroxide). In its reduced state water can pass through the underlying limestone layer without 'armouring' problems. Although not specifically designed to do so, sulphate-reducing bacteria may colonise the compost layer, adding the potential for alkalinity generation and immobilisation of Fe^{2+} as FeS. However, it is unwise to assume this in the design of such systems.

The design of RAPS is along similar lines to ALDs, i.e. the limestone element of the system should be designed to ensure 14 hours residence time (Kepler and

McCleary 1994). Once the volume of limestone is calculated, it is a simple matter to calculate the area required (usually assuming a limestone depth of 0.5–1.0 m. The depth of compost above the limestone should be at least 0.5 m. In addition the compost layer should be submerged to a depth of at least 200 mm.

From the preceding discussion it will be obvious that there must be sufficient hydraulic head available to drive the water through the compost and limestone. Flat sites will not therefore be suitable for RAPS if this is to be accomplished by gravity. However, one of the great advantages of RAPS is that because they are downward flow systems it is much more likely that the entire volume of compost and limestone is utilised, and therefore the land area requirements are far less than equivalent (horizontal flow) compost wetlands.

To maintain the water level above the compost, the effluent discharge channel must be at a similar level to the water surface, which can present some engineering problems. Most simply this is achieved by attaching a 90° elbow joint and suitable length of pipe, to the pipe draining the limestone. However, the permeability of the compost will decrease over time, due to accumulation of solids, and therefore the height of the effluent pipe should be variable. This will enable the height of the effluent pipe to be reduced, such that the effective hydraulic head across the system can be increased, thus compensating for the reduction in compost permeability. The design of a system to allow backwashing of the stone and compost layer would be a distinct advantage.

RAPS systems have been designed and constructed at La Coruña in Spain (Laine 1998) and the Pelenna Valley in South Wales.

As an alternative to a configuration whereby compost overlies limestone, the compost and limestone units may be separated. In this case water flows *downwards* through the compost, and *upwards* through the limestone. Such a system has been installed by IMC at Deerplay, Lancashire, UK (Jarvis and England 2002). The advantages of this system are:

(i) because the limestone bed is an upward flow unit, efficiency of operation is at a maximum;
(ii) the required hydraulic head is less than that of a system with compost overlying limestone.

The only disadvantage of this configuration is that the area of land required is greater (although the limestone is not actually visible).

Because a RAPS is not designed to remove metals, it must be followed by another treatment unit. A sedimentation pond and/or aerobic wetland is the usual option.

Media selection for compost based systems

Selection of media for compost based systems is not (as yet) a precise science. For compost wetlands the primary objective is to select media that will support colonisation by sulphate reducing bacteria. For RAPS an additional requirement is that the media must be suitably permeable/porous. Standardised tests for assessing the potential of a medium to support SRB do not exist, and neither are there recommended values for the permeability and porosity of a compost for a RAPS. Rudimentary laboratory experiments can be used to assess whether compost will support SRB (see for example Younger *et al*. 1997). Tests can be conducted to ascertain permeability and porosity values, but it is not clear what values are acceptable. As a general rule, designers should aim to use compost with the maximum permeability and porosity, whilst still promoting reducing conditions.

Laboratory experiments are clearly of benefit, but because of the uncertainties outlined above, pilot-scale trials are strongly advisable when considering installation of compost based systems. Potential candidates for compost systems include the following:

(i) pressed digested sewage sludge;
(ii) horse manure;
(iii) composted domestic waste;
(iv) waste from paper production;
(v) shredded timber or bark;
(vi) mushroom compost.

Of these, (ii), (iii), (v) and (vi) have been used in successful systems. It will be noted that, with the exception of shredded timber, these media are all waste products. This is preferable because:

(i) using waste products is environmentally sustainable, since otherwise they would present a disposal problem;
(ii) because they are waste products they are usually available at very low cost, if not for free.

CASE STUDIES

Pilot-scale compost wetland to treat acidic colliery leachate at Aspatria, Cumbria

The design of an anaerobic compost system for a site in Cumbria for the UK Environment Agency is being progressed by IMC Consulting Engineers. Surface water run-off from an abandoned colliery spoil heap contains 400 mg/L Fe, 50 mg/L Al, 10 mg/L Mn, and is pH 3. Laboratory testing of effluent from the site mixed with pressed digested domestic sewage showed vigorous reduction of sulphate and generation of alkalinity. Further tests will be completed with paper waste and shredded timber to select the most appropriate compost mix for the site. A pilot testing scheme comprising four cells, each ten metres wide and 20 metres long, will be constructed on the site, and the performance of each in treating the effluent will be monitored, with the assistance of the Hydrogeochemistry Engineering Research and Outreach (HERO) group at Newcastle University. Following successful completion of the field testing, a full-scale treatment system, comprising a 20 000 m^2 surface flow anaerobic wetland and a 20 000 m^2 'volunteer' aerobic wetland is likely to be constructed at the site, to treat the entire volume of run-off from the site.

Design of a RAPS for the Sheephouse Wood mine water discharge, Yorkshire, UK

IMC Consulting Engineers have been commissioned by the UK Coal Authority to design a treatment system for the Sheephouse Wood mine water discharge, located north-west of the city of Sheffield. The discharge is from an abandoned mine adit, at a flow-rate of approximately 12 L/second. Limited field testing results available to date show a pH of 5.6, 50 mg/L total iron, 40 mg/L ferrous iron and acidity >200 mg/L as $CaCO_3$.

Traditional wastewater engineering for such a system would involve chemical injection followed by sedimentation ponds and, possibly, a tertiary treatment wetland. This is a proven approach, and about 20 such installations were constructed in the Yorkshire coalfield in the 1970s and 1980s. Latterly, similar systems have been constructed by IMC at Old Meadows, Lancashire, and Vivian (Six Bells), South Wales (because passive-only treatment was not feasible). Such systems have the advantage of being adaptable to variations in mine water quality. However, this security comes at the cost of provision of power and chemicals for the life of the treatment system, together with the enhanced maintenance costs associated with active treatment systems.

Sufficient land is available at Sheephouse Wood to make a passive-only treatment system a possible option for this acidic drainage. IMC is therefore considering the construction of a large RAPS at the site, in order to reduce the operating costs to the Coal Authority over the life of the site. Construction and long-term cost estimates for the passive RAPS are compared to cost estimates for an active scheme as shown in Table 1. It can be seen that the capital construction costs are in fact slightly higher for the RAPS. The RAPS cost estimate assumes that a man-made liner will be necessary, and includes the cost of high-calcite (>90%) limestone (significant elements of the capital cost). It is always hoped that the site may contain clay below the surface, which can be used as a liner for no cost, and that a cheap source of limestone may be available. However, this is never known until the detailed design of such a system begins. It is something of a misconception that the capital costs of passive treatment schemes will be significantly less than those for active chemical options.

The figures in Table 2 assume that the compost and limestone in the RAPS will require replacement after a period of 15 years (because of the uncertainties associated with the permeability/lifetime of compost – see above).

Table 2. Comparative cost estimates for a passive and active treatment system for the Sheephouse Wood mine water discharge (see text for details)

Item	Passive treatment (£)	Active treatment (£)
Capital cost of construction	600,000	500,000
Total cost over 40 years	1,100,000	1,500,000

40 year life; 6% interest.

Nevertheless, it can be seen that over the long-term, passive treatment systems are significantly cheaper than their active treatment equivalents, and this is the great advantage of passive schemes.

The concerns regarding the construction of such a large RAPS are as follows:

(i) how to ensure that the permeability of the compost remains adequate to pass the design flow. It is possible to raise the hydraulic head to increase flow by a greater depth of supernatant water, should problems occur;

(ii) can a backflushing system to maintain permeability operate successfully?

(iii) ensuring adequate retention time for water in the compost layer;

(iv) for acidity levels >200 mg/L, a conservative design would allow for two RAPS stages, in that Kepler and McCleary (1994) considered that between 150 and 300 mg/L for a single unit could be achieved. A RAPS would not only have to neutralise acidity, but it would also have to impart excess alkalinity to allow sedimentation of dissolved solids from the mine water. A twin RAPS system, with intervening sedimentation ponds and re-aeration systems, would significantly increase construction costs and leave little, if any, financial advantage in a net present value calculation of the whole life of a system.

RESEARCH AND DEVELOPMENT INTO PASSIVE TREATMENT SYSTEMS: THE PIRAMID PROJECT

The 'Passive In situ Remediation of Acid Mine and Industrial Drainage' (PIRAMID) research project (Ref: EVK1-CT-1999-00021) is an EU Framework 5 sponsored project led by the University of Newcastle upon Tyne, UK. Its objective is to focus and bring together all available information on passive treatment systems from across Europe. The ultimate goal of the project will be a handbook giving guidelines on the basic principles of the design and construction of passive treatment systems. Guidance on the engineering aspects of compost reactors, aerobic and anaerobic wetland systems, sedimentation ponds and permeable reactive barriers will all be included in the document. The PIRAMID research project, co-ordinated by Professor Paul Younger, brings together work groups throughout Europe researching passive treatment technologies. The handbook will be compiled and edited by IMC Consulting Engineers and Newcastle University.

REFERENCES

Barnes, H.L. and Romberger, S.B. (1968) Chemical aspects of acid mine drainage. *Journal of the Water Pollution Control Federation*, **40** (3), 371-384.

Batty, L.C. and Younger, P.L. (2002) Critical role of macrophytes in achieving low iron concentrations in mine water treatment wetlands. *Environmental Science and Technology*, **36**, 3997-4002.

Benner, S.G., Blowes, D.W. and Ptacek, C.J. (1997) A full-scale porous reactive wall for prevention of acid mine drainage. *Ground Water Monitoring and Remediation*, **17**(4), 99-107.

Hedin, R.S., Nairn, R.W. and Kleinmann, R.L.P. (1994*a*) *Passive Treatment of Polluted Coal Mine Drainage.* Bureau of Mines Information Circular 9389. United States Department of the Interior, Washington DC. 35 pp.

Hedin, R.S., Watzlaf, G.R. and Nairn, R.W. (1994*b*) Passive treatment of acid mine drainage using limestone. *Journal of Environmental Quality*, **23**, 1338-1345.

Jarvis, A.P. and England, A. (2002) Operational and treatment performance of a unique reducing and alkalinity producing system (RAPS) for acidic leachate remediation in Lancashire, UK. *Proceedings of the 9th International Mine Water Association Congress*. Freiberg, Germany, 15–21 September 2002. In press.

Jarvis, A.P. and Younger, P.L. (1997) Dominating factors in mine water induced impoverishment of the invertebrate fauna of two streams in the Durham Coalfield, UK. *Chemistry and Ecology*, **13**, 249-270.

Jarvis, A.P. and Younger, P.L. (1999) Design, construction and performance of a full-scale wetland for mine spoil drainage treatment, Quaking Houses, UK. *Journal of the Chartered Institution of Water and Environmental Management*, **13**, 313-318.

Jarvis, A.P. and Younger, P.L. (2000) Broadening the scope of mine water environmental impact assessment: a UK perspective. *Environmental Impact Assessment Review*, **20**, 85-96.

Jarvis, A.P. and Younger, P.L. (2001) Passive treatment of ferruginous mine waters using high surface area media. *Water Research*, **35** (15), 3643-3648.

Kepler, D.A. and McCleary, E.C. (1994) Successive Alkalinity-Producing Systems for the treatment of acidic mine drainage. *Proceedings of the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage*, pp. 195-204. Pittsburgh, PA, April 24-29, 1994.

Laine, D.M. (1997) The treatment of the pumped mine water discharge at Woolley Colliery, West Yorkshire. In: Younger, P.L. (ed.) *Mine Water Treatment Using Wetlands*. Proceedings of a National Conference. University of Newcastle, UK, 5th September 1997, pp. 83-103. Chartered Institution of Water and Environmental Management, London.

Laine, D.M. (1998) Treatment of pumped and gravity minewater discharges in the UK and an acidic tip seepage in Spain. *Proceedings of the Symposium on Minewater & Environmental Impacts, International Minewater Association*, p. 471. Johannesburg, South Africa, September 1998.

Laine, D.M. (1999) Remediation of the Old Meadows Gravity Minewater discharge. *Proceedings of the International Congress on Mine, Water & Environment.* International Mine Water Association Congress, Seville, Spain, 1999, p. 581.

Laine, D.M. (2000) Passive water treatment. *World Coal*, **9** (8), 43-46.

Mitsch, W.J. and Wise, K.M. (1998) Water quality, fate of metals, and predictive model validation of a constructed wetland treating acid mine drainage. *Water Research*, **32**(6), 1888-1900.

NCB (1982) *Technical Management of Water in the Coal Mining Industry.* 129 pp. Mining Department, National Coal Board, London.

Nordstrom, D.K. and Southam, G. (1997) Geomicrobiology of sulfide mineral oxidation. In *Reviews in Mineralogy (Geomicrobiology: Interactions Between Microbes and Minerals)*, eds J.F. Banfield and K.H. Nealson, pp. 361-390. Mineralogical Society of America, Washington DC.

Singer, P.C. and Stumm, W. (1970) Acidic mine drainage: the rate-determining step. *Science*, **167**, 1121-1123.

Younger, P.L. (1997) The longevity of minewater pollution: a basis for decision-making. *The Science of the Total Environment*, **194/195**, 457-466.

Younger, P.L., Curtis, T.P., Jarvis, A.P. and Pennell, R. (1997) Effective passive treatment of aluminium-rich, acidic colliery spoil drainage using a compost wetland at Quaking Houses, County Durham. *Journal of the Chartered Institution of Water and Environmental Management*, **11**, 200-208.

Younger, P.L., Banwart, S.A. and Hedin, R.S. (2002) *Mine Water: Hydrology, Pollution, Remediation*. 442 pp. Kluwer Academic Publishers, Dordrect, The Netherlands.