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ENGINEERING GUIDELINES FOR THE PASSIVE REMEDIATION OF ACIDIC AND/OR METALLIFEROUS MINE DRAINAGE AND SIMILAR WASTEWATERS

Written and edited by

the PIRAMID Consortium

P I R A M I D

PIRAMID: Passive In-situ Remediation of Acidic Mine / Industrial Drainage

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EXECUTIVE SUMMARY

Acidic mine drainage and similar wastewaters (such as leachates from acid-sulphate soils) are a major cause of ground and surface water pollution in the European Union. Because such pollution can persist for decades and even centuries after the cessation of industrial activity, there is a pressing need to develop cheap, sustainable remedial methods. PIRAMID has sought to harmonise research and practice efforts in Europe to create passive in situ remediation (PIR) methods for acidic drainage treatment. A key objective of this three year R&D project has been the development of these engineering guidelines for the design and installation of passive treatment systems. These guidelines are intended to provide practitioners in the field of environmental engineering with sufficient information to enable them to confidently undertake feasibility studies and develop conceptual design statements for passive mine water remediation systems.

The guidelines open with a detailed discussion of the required design information for development of a passive treatment scheme. The importance of collection of accurate flow-rate and water quality data is emphasised, since these data are the basis of treatment system design. Details of the various methods of flow measurement are outlined, together with their pros and cons. Simultaneous to measurement of flow-rate, it is necessary to collect water samples for chemical analysis, and in most cases also to undertake on-site testing of labile parameters. The requirements for water quality testing are discussed at length, including issues such as the variables to be determined, laboratory analysis, sampling frequency, and health and safety considerations.

Beyond this characterisation of the drainage itself, it is necessary to collect information on a number of other issues before detailed design and construction begins. Once a potential treatment site is selected, it is necessary to carry out a topographical survey, which is indispensable for detailed design purposes. A site appraisal, incorporating desk and walkover studies, may reveal much about the potential of a site e.g. presence of protected species at the site, location of buried services. A more rigorous appraisal is made during the ground investigation. This enables a more thorough assessment of conditions below the surface to be made. It may include excavation of trial holes, installation of boreholes, and laboratory testing of materials encountered. Details of the exact type of information to be gathered are provided. Although desk studies may reveal the likelihood of historic contamination of the potential treatment site, a dedicated assessment should be undertaken, to ensure there is no likelihood of encountering contaminated materials.

Passive treatment is most frequently applied to the treatment of waters containing elevated acidity, iron and / or aluminium, and sections 3 and 4 of the guidelines therefore consider this topic in detail. Section 3 provides details of the treatment options available for amelioration of acidity, iron and aluminium, identifying which treatment units are appropriate for particular water qualities. A central issue in unit process selection is whether the discharge is net-alkaline or net-acidic. The significance of this issue is highlighted in the text.

Installing the correct size of passive unit is essential if the treatment system is to work effectively, and section 4 of the guidelines provides sizing criteria for each of the unit processes described in section 3, as well as providing guidance on the appropriate sequencing of these units. These sizing criteria are comparatively simple to use, but a number of important assumptions are implicit in their use, of which the user needs to be aware. These assumptions are outlined, so that the apparent simplicity of the sizing criteria is supported by an accurate appreciation of their scope and limitations.

Mine drainage (and similar industrial wastewaters) occasionally contain contaminants other than acidity, iron and aluminium, and section 5 of the report provides guidance on the passive amelioration of such problematic pollutants. Specific attention is given to the removal of zinc, manganese, arsenic, cyanide and sulphate, with notes also on other contaminants such as uranium, nickel and thallium.

In some cases polluted drainage can be minimised by means of engineering interventions meant to stem pollutant release at source. These interventions principally entail the installation of a water cover or dry cover on spoil heaps and / or tailings dams. Particular attention is given to engineering aspects of applying dry covers to tailings / spoil facilities. Revegetation of mine wastes is invariably a high priority in reclamation projects, partly because it limits the potential for future pyrite oxidation, and partly because of the improved amenity resulting from revegetation. Therefore section 6.5 gives detailed advice on successful techniques for installing a vegetative cover.

Civil engineering considerations for the construction of settlement lagoons and other elements of passive remediation infrastructure are outlined in section 7. This includes discussion of a number of key issues, such as slope stability, earthworks, lining materials, construction plant and equipment, site preparation, and the construction of embankments. Materials selection is also discussed as a separate section. Key aspects of materials selection for passive treatment systems include pipework and / or channel design. The advantages and disadvantages of pipes versus channels are outlined, in relation to the particular problems arising from mine water chemistry. Other issues considered in this section are the selection of materials for drainage systems, materials for inlet and outlet structures, and selection of wetland plants.

There are always contractual matters associated with the construction of systems such as passive treatment facilities. A brief overview of the key issues is provided in section 9 of the guidelines.

In general discussions of the topic, much is made of the low maintenance requirements of passive treatment systems, but this is not to say that such systems are maintenance *free*. Indeed, regular, albeit infrequent, maintenance is essential if the system is to operate effectively. Therefore the final section of the guidelines address regular and long-term maintenance issues for passive treatment facilities.

An extensive list of references is provided.

FOREWORD

This handbook is one of the main practical outcomes of the 5th Framework RTD project PIRAMID (Passive In-situ Remediation of Acidic Mine/Industrial Drainage). As such these guidelines concern the practical application of passive *in-situ* remediation technologies to acidic and / or metalliferous mine drainage and similar wastewaters. The PIRAMID research consortium formally agreed a definition of 'passive *in-situ* remediation' as follows:

> ***Passive* in-situ *remediation* (PIR)** 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.

This in turn can be resolved into the following two subsidiary definitions:

> ***Passive treatment*** is the improvement of water quality using only naturally-available energy sources, in gravity-flow treatment systems (such as wetlands or subsurface-flow bioreactors) which are designed to require only infrequent (albeit regular) maintenance to operate successfully over their design lives[1].
>
> ***Passive prevention of pollutant release*** is achieved by the surface or subsurface installation of physical barriers (requiring little or no long-term maintenance) which inhibit pollution-generating chemical reactions (for instance, by permanently altering redox and / or moisture dynamics), and / or directly prevent the migration of existing polluted waters.

These guidelines provide the basis for developing and implementing robust engineering designs for the passive treatment and / or passive prevention of mine water pollution. The operational aspects of water management are explained in the particular context of the design and construction of passive remediation schemes. Each chapter considers a particular aspect of passive technology and describes the key principles involved. The selection of the appropriate form of passive treatment for specific types of effluent is discussed and appropriate methods of design are recommended. Without unduly lengthening the coverage, worked examples of the selection and implementation of remedial options are included where these seem likely to be helpful to the newcomer to this topic. An extensive bibliography facilitates access to more detailed research reports which further explain the scientific basis of the technologies described here.

These guidelines go far beyond previously existing guidelines[2] in specifying the detailed civil engineering techniques which need to be mastered in order to turn a generalised design concept into a robustly-constructed remediation facility. While many of these techniques are well-known amongst practising civil engineers, they are seldom described in the literature and have never been brought together under one cover for the purposes of

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[1] The PIRAMID Consortium is indebted to William Pulles of Pulles Howard and de Lange Inc., Johannesburg, Republic of South Africa, who coined this definition whilst undertaking the first external review of the project in March 2000. From this original definition the related definitions of 'passive *in-situ* remediation' and 'passive prevention of pollutant release' were subsequently developed.

[2] Most notably the guidelines of the former US Bureau of Mines (Hedin *et al.* 1994), the more recent successor document to these guidelines published by the US Department of Energy (Watzlaf *et al.* 2003) and the recent mine water text book by Younger *et al.* (2002), which devotes one of its five chapters to passive technologies.

remedial engineering design for contaminated mine sites. Given that passive remediation technologies are rapidly developing, with substantial involvement from research scientists, these guidelines seek to de-mystify practical engineering for the benefit of scientists, and also to make recent scientific advances readily accessible to experienced engineers.

The user of these guidelines is asked to bear in mind that individual field sites nearly always display some degree of uniqueness, which will require the use of scientific insight and engineering know-how to the solution of site-specific challenges. It is also stressed that these guidelines are provided in good faith, as an expression of the latest scientific consensus on the most appropriate approaches to passive remediation of acidic and / or metalliferous wastewaters. Neither the authors nor the European Commission can accept any liability for loss or injury arising from attempts to implement these guidelines in practice. Users are therefore encouraged to further prove the concepts presented here by means of site-specific pilot trials wherever possible.

This document has been produced by a team comprising:

- **Professor Paul L Younger** (the PIRAMID Coordinator, Univ. of Newcastle, UK)
- **Dr Adam Jarvis** (IMC Consulting Engineers Ltd, Nottinghamshire, UK)
- **Mr David Laine** (IMC Consulting Engineers Ltd)

The above team have drafted and edited these guidelines by means of integrating the following information sources:

- the major practical experience of the team itself, including the worldwide experience of IMC Consulting Engineers Ltd, in the design and construction of full-scale passive remediation systems
- previous literature, most notably:
  - ❖ the pioneering works on passive treatment undertaken in the USA (Cohen & Staub 1992; Hedin *et al.* 1994)
  - ❖ the more recent transatlantic synthesis of Younger *et al.* (2002)
  - ❖ various individual papers which have appeared in the peer-reviewed literature (cited in the text as appropriate)
- specific contributions derived from the experimental work undertaken by academic partners involved in the PIRAMID project, relating to:
  - ❖ the application of wetland-type passive systems to the immobilisation of arsenic (Dr Marc LeBlanc, UM2 Montpellier, France) and cyanide (Dr Jorge Loredo, ETSIMO Oviedo, Spain, and Dra Teresa Martínez Flores, Río Narcea Gold Mines Ltd, Spain)
  - ❖ the use of novel reactive substrates for neutralisation of acidic waters and removal of zinc and other metals in subsurface flow passive systems (Dr Carlos Ayora, CSIC-IJA, Barcelona, Spain, Dr Christian Wolkersdorfer, TU-Bergakademie Freiberg, Germany, and Prof Paul Younger, University of Newcastle, UK)
  - ❖ the passive treatment of uranium in wetland-type systems (Prof Miran Veselič, IRGO, Slovenia, Prof Martin Sauter, Universität Jena, Germany)
  - ❖ the use of dry covers and water covers, and the role of compaction and revegetation, in helping to passively prevent pollutant release from bodies of mine waste (Prof Björn Oehlander, Dr Jan Landin and Dr Anders Widerlund, MiMi Consortium, Sweden, and IMC Consulting Engineers Ltd, UK)

- the role of wetland vegetation in polishing metals to low residual concentrations in treatment wetland systems (Dr Lesley Batty, University of Newcastle, UK)

While every effort has been made to ensure that these guidelines are as up-to-date as possible at the time of going to press, in such a rapidly-developing field it is inevitable that these guidelines will soon be overtaken by new developments and be in need of revision. Although the formal funding of PIRAMID will no longer be available in future, the authors intend to obtain independent funding in the coming years to facilitate the issuance of updated editions of these guidelines, to take account of the latest experiences and new scientific findings. Suggestions for amendments / expansions of this document in future editions should be therefore sent to the following e-mail address: hero@ncl.ac.uk

While PIRAMID was essentially a technology development and dissemination project, a parallel programme of research has addressed many of the policy implications arising from the research of the PIRAMID team and others. This policy-oriented project, ERMITE (Environmental Regulation of Mine Waters in the EU; see www.minewater.net/ermite) is producing technical and managerial guidelines for mine water management at the catchment-scale (as distinct from the site scale which PIRAMID takes as its default focus), and policy briefs for potential use by those engaged in the development of more efficient regulatory strategies for mine waters in Europe.

Finally, it should be noted that many of the full-scale passive treatment systems in the UK used for research during the PIRAMID project have now been developed into an integrated 'outdoor laboratory' known as **CoSTaR** (Coal Mine Sites for Targeted Remediation Research). CoSTaR comprises a 'constellation' of six established mine water remediation systems located in one relatively compact fieldwork district in northern England. The CoSTaR systems have been selected to provide at least one example of each of the principal types of passive systems currently used to treat polluted mine waters at numerous sites in Europe:

Aerobic reed beds treating non-acidic waters with high iron contents: St Helen Auckland, Whittle and Acomb sites

Compost wetlands treating acidic waters: Quaking Houses site

Reducing and Alkalinity Producing System (RAPS), i.e. vertical flow compost / limestone bioreactors: Bowden Close site

Permeable reactive barrier: Shilbottle site

Hybrid active / passive system: Acomb site

Nowhere else in the world is such a complete array of passive treatment systems available within such a small geographical area. Initially designated a UK national research facility by the contaminated land remediation organisation CL:AIRE (see www.claire.co.uk). Through the good offices of the European Commission's 6th Framework Programme (FP6), it is intended to provide funding (under the FP6 'Access to Research Infrastructure' programme) so that the CoSTaR systems can be made available for wider use by the European research community, with allowances to cover travel, subsistence and research consumable costs for successful applicants. This funding is anticipated to be in place from early 2004. For the latest information on this, and all other aspects of the CoSTaR research facilities, the interested reader is directed to: www.minewater.net/CoSTaR/CoSTaR.htm

Newcastle Upon Tyne, UK
September 2003

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1. INTRODUCTION

1.1. Mine drainage and other metalliferous wastewaters

Like so many other human activities of enormous economic importance, mining does not yield its riches to society without inflicting some damage on the natural environment. In particular, aquatic pollution by mine effluents which are enriched in ecotoxic metals (and which also often exhibit a low pH, i.e. high acidity) is acknowledged by the industry to be "the most serious and pervasive environmental problem related to mining" world-wide (e.g. MMSD 2002). Mine waters include water present in and/or draining (either under gravity or by pumping) from:

- the mined voids themselves
- bodies of mine waste, including both spoil (i.e. waste rock) and tailings (i.e. the generally fine-grained non-saleable material left behind after processing of run-of-mine product to extract the economically valuable components)

In both sources, mine waters may become heavily contaminated with dissolved and/or colloidally-transported metals which can cause great damage in freshwater ecosystems or in watercourses used for public water supply purposes. Chief amongst the problematic metals are iron (Fe), manganese (Mn, which is only a problem in relation to waters used for domestic supply) and aluminium (Al). Of these three, the latter is effectively restricted to the more acidic waters, whilst Fe and Mn can occur at elevated concentrations even in mine waters with a circum-neutral pH. Next in frequency are the strongly ecotoxic metals zinc (Zn) and copper (Cu), and in some cases the rare ecotoxic / anthropotoxic metals such as Cd, Ni and Hg are also significantly mobile in mine waters. The metalloid arsenic (As) is a significant pollutant in certain mine waters, especially those associated with orebodies rich in arsenic minerals (most notably arsenopyrite, realgar and orpiment) and in some cases as a desorption product from ancient haematite orebodies. A further problem relates to sulfate (SO_4^{2-}), which is often present at high concentrations (many hundreds to several thousands of mg/l) in many mine waters, whereas the limit concentration for drinking waters is only 250 mg/l. For further detail on the origins, nature and impacts (ecological and socio-economic) of mine waters, the interested reader is referred to a recent textbook (Younger *et al.* 2002). Suffice it to say that several thousand kilometres of streams and rivers in Europe are already severely impacted by this form of pollution, and that without timely and adequate intervention this situation is likely to get considerably worse (see Younger 2002). This is simply because, in line with previous experience, once recently-abandoned mine workings have flooded up to the local base level of drainage they can be expected to give rise to large perennial flows of polluted mine drainage There is therefore a pressing need for the application of cost-effective abatement measures for mine water pollution both in Europe, and indeed throughout many current and former mining areas of the world.

Other waters which closely resemble polluted mine waters in terms of both hydrochemistry and environmental impacts also require abatement measures. Close natural analogues for acidic mine drainage are provided by the waters emanating from so-called 'acid-sulphate soils', which are commonly found in low-lying coastal areas of the world, where former marine and inter-tidal sediments naturally rich in sulphide minerals have been exposed to atmospheric weathering. This can occur naturally (due to natural geological uplift) or accidentally, as an undesired consequence of draining and tilling former coastal marshlands for agricultural purposes. Some pipe condensate waters associated with the production of coalbed methane (or with the sequestration of methane from mine gas streams) also exhibit characteristics similar to those of acidic mine waters, as do the effluents associated with a number of metal-pickling and finishing industries.

1.2. Mine water treatment: active and passive approaches

Until the 1990s, the only 'proven technologies' for abating mine water pollution were what is now termed 'active treatment' (Younger *et al.* 2002), which involves the application of industrial reagents and external power sources (for stirring, pumping, heating etc) by means of conventional unit processes common to many chemical engineering and environmental engineering plants. Active treatment has an important contribution to make to the abatement of mine water pollution in many cases, such as:

- at active mine sites where pollutant loadings can fluctuate at short notice as drainage installations are altered in response to changing patterns of production
- at both active and abandoned mine sites where the flow rate and / or pollutant concentrations are very high, such that the application of less intensive treatment methods would require too much space.

The recent textbook of Younger *et al.* (2002) devotes an entire chapter to active mine water treatment, and provides numerous references to a wide-ranging literature on the available technologies and their applicability.

As was noted in the Foreword, the PIRAMID project defined 'passive treatment' as "the improvement of water quality using only naturally-available energy sources, in gravity-flow treatment systems which are designed to require only infrequent, albeit regular, maintenance to operate successfully over their design lives". Besides the reliance on naturally-available energy sources, passive treatment primarily differs from active treatment in its economic structure: in active treatment, the overall cost of treatment (summed over the entire time period during which treatment is implemented) is distributed over time, with relatively high operating expenditure (opex) recurring throughout the period of treatment. By contrast, passive treatment systems are 'capex-intensive', that is to say by far the bulk of the entire costs associated with the implementation of a passive system are spent up-front, in the form of the capital expenditure needed to install the system in the first place.

1.3. Nature of these guidelines

This document provides detailed guidelines for the passive *in-situ* remediation of mine waters and similar wastewaters, specifically providing practical advice on the design, construction and operation of 'passive treatment' systems, and of the conceptually-similar interventions aimed at achieving 'passive prevention of pollutant release'[3]. These technologies have only emerged over the last two decades, and

[3] see the Foreword for the formal definition of this term and of the blanket term "passive *in-situ* remediation"

specific variants of them have only begun to be regarded as "proven technologies" in Europe within the last five years (see Younger 2000a). Nevertheless, passive *in-situ* remediation is still in its infancy in many spheres of potential application (hence the need for a major research project such as PIRAMID), so that many of the potential applications described in these guidelines are still to be regarded as 'provisional' in nature. Where this is so, the text makes the current level of uncertainty abundantly clear.

The document assumes the reader to be a scientist or engineer with a reasonable background in chemistry and the basics of civil engineering, albeit with no specific knowledge of mine water treatment in general, or the passive treatment of mine waters in particular. The underlying contaminant removal mechanisms that guide the recommended design criteria are discussed in sufficient detail that the reader can understand why passive systems should work, but the discussion stops short of being a formal scientific treatise on the relevant aspects of geochemical kinetics and solute transport hydrodynamics. Rather, the document guides the reader through the logical framework (represented by flowchart such as Figures 4.1 (p. 55) and 4.2 (p. 56)), enabling them to quantify the nature of the problematic drainage, and the potential for treatment that exists on the site, leading to the selection of the most appropriate form of remediation. Advice on the design of the recommended system is provided along with detailed construction advice on means of achieving completion of the scheme. All real-world schemes will have at least some element of uniqueness (be it in terms of water quality, proposed treatment area, or regulatory constraints), and therefore the guidelines cannot provide a fully comprehensive guide to all passive applications. What is intended is that sufficient information is provided to enable a user to design or complete a feasibility study on a basic system, but the guidelines cannot replace the knowledge of an experienced passive treatment engineer. There is no question that the expertise of specialist scientists and engineers will be required during the course of the design and construction of a passive treatment scheme. Indeed, to fail to enlist the services of geotechnical and civil engineers, in designing water retaining embankments for example, may result in serious health and safety risks at a site, which is clearly unacceptable.

1.4. Context of these guidelines: the PIRAMID project

1.4.1. The PIRAMID project - introduction

The principal scientific findings of the PIRAMID project have been reported in a comprehensive final report, the non-confidential sections of which are available for the cost of reproduction from the PIRAMID Coordinator (see www.piramid.org). The following summary of the PIRAMID project is adapted from Section 5 of the final report, to give the reader of these guidelines an appreciation of the scientific context within which this document was written.

The original objectives of PIRAMID (see below) have been successfully realised, providing a firm foundation for the further development and practical application of passive in-situ remediation technologies. PIRAMID has also led organically to the development of a network of scientific and engineering specialists with expertise in this form of remediation, which will have substantial positive implications for future training of technical specialists, and for integration of efforts across Europe in the remediation of contaminated mine sites.

1.4.2. Objectives of PIRAMID

The objectives of the PIRAMID project were as follows:
- To compile and distribute a database of existing passive *in-situ* remediation (PIR) systems for acidic / ferruginous mine waters in Europe (freely available for download at www.piramid.org)
- To develop process-based models of PIR systems
- To evaluate the potential applicability of PIR for countries in central and eastern Europe which were lacking the technology in 1999
- To experimentally evaluate, in the field and the lab, novel reactive substrates for PIR systems
- To produce and disseminate engineering guidelines for the future design, construction and operation of PIR systems

1.4.3. Scientific achievements of PIRAMID

The scientific research completed as part of the PIRAMID project has been extremely successful, producing an abundance of information which provide many possibilities for the application of passive *in-situ* remediation to a wide range of pollution problems. In particular:
- the critical importance of macrophytes in achieving low residual Fe concentrations in aerobic wetland systems was demonstrated, together with fundamental insights into the effects of elevated proton and metal activities on wetland plant growth
- low-cost substrates for treatment of acidic waters have been investigated (including synthetic zeolites made from PFA, caustic magnesia, and organic matter promoting dissimilatory bacterial sulphate reduction, in particular green waste composts and farmyard manures)
- the use of oxidative and reductive systems for the treatment of arsenic-rich mine drainage has been investigated, with the kinetics of oxidative systems proving the more favourable of the two; arsenic-oxidising bacteria have been identified and isolated
- the safe, passive destruction of residual cyanide leaching from gold mine tailings has been achieved using compost-based wetland-type passive systems (which also remove residual copper), thus complementing active cyanide-destruction techniques used while the mine remains operational
- detailed field investigations of: (i) conventional wetland-type passive systems in the UK, France and Slovenia, removing Fe, As and U respectively from diverse mine waters (ii) a conventional permeable reactive barrier (PRB) within an aquifer at Aznalcóllar (Spain), treating highly acidic groundwaters associated with a base metal mine, and (iii) a novel, hybrid passive system at Shilbottle (UK) which intercepts extremely acidic colliery spoil leachates using a surficial PRB (with substrates including organic matter mixed with limestone and / or blast furnace slag, promoting inorganic and bacterial neutralisation processes), which releases its effluent into a series of oxidation ponds and aerobic wetlands downstream.

Significant advances were also made in relation to natural and stimulated natural attenuation of acidity in mine pit lakes, and in the assessment of the sustainability of dry covers and water covers for tailings and waste rock piles, under both cold (sub-arctic) and warm (Mediterranean) climatic conditions.

Modelling software has been developed allowing simulation of subsurface-flow passive treatment systems (RETRASO), wetland-type systems (NOAH2D) and the natural attenuation of mine water pollutants in flooded deep mines (RUMT3D).

PIR was found to have considerable potential for application in many European countries, and offers potential solutions to pollution problems in most Newly-Associated States with significant past or present mining industries. (It is hoped that such developments, in these and other countries, will be greatly facilitated by the present document).

1.4.4. Socio-economic relevance and policy implications of PIRAMID

Working in partnership with another FP5 project, 'ERMITE'[4], PIRAMID made significant contributions to the technical background invoked in drafting the June 2003 'Proposal for a Directive of The European Parliament and of the Council on the management of waste from the extractive industries' (COM(2003) 319 final; 2003/0107 (COD)). Through continued work with ERMITE, it is anticipated that the findings of PIRAMID will also make a significant contribution to the guidelines currently being prepared by the ERMITE project in relation to implementation of the Water Framework Directive (2000/60/EC) with respect to mining operations and abandoned mines.

1.5. Relationship of these guidelines to previous work

This short section aims to outline the objectives of this book in the context of other previously published work. The volume of research conducted in the field of passive treatment of mine waters has increased dramatically over the course of the last two decades. The following paragraphs barely scratch the surface, highlighting only some of the key works which relate specifically to the design of passive mine water treatment units. More substantial reviews can be sought in the recent publications of Younger *et al.* (2002) and Watzlaf *et al.* (2003).

For many decades the mining industry has appreciated the low costs associated with the use of low-intensity methods of water treatment, such as simple settling of solids from suspension in large sedimentation ponds. Thus the coal industry world-wide has long used relatively simple settlement lagoons as a means of clarifying water from operational mines, albeit mainly for the removal of inert solids (coal fines), but also for the removal of iron hydroxides. The published work that best reflects these long-established practices is the handbook *Technical Management of Water in the Coal*

[4] ERMITE: Environmental Regulation of Mine waters In the European Union; contract no. EVK1-CT-2000-00078; see www.minewater.net/ermite.

Mining Industry (NCB, 1982). The empirical formulae and engineering guidelines provided therein, for the design of settlement lagoons, are still used today (e.g. Younger *et al.* 2002). However, despite the widespread adoption of settling pond technology by the early 1980s, no consideration was given at that time to the use of wetland systems or other unit processes now regarded as 'mainstream' passive treatment technologies.

Perhaps the earliest documented scientific investigations of the potential of wetlands for mine water treatment were those of Huntsman *et al.* (1978) and Wieder and Lang (1982). Both works focused on (independent) observations of sphagnum bogs in the USA that happened to receive mine drainage. The potential of wetlands as treatment units for mine drainage was suggested by this work, which provided the impetus for the future construction of treatment wetlands and the derivation of empirical design guidelines for them. In terms of passive treatment system design, the culmination of research and practice in the decade or so following these early works (predominantly undertaken in the USA) was arguably the US Bureau of Mines (USBoM) publication, *Passive Treatment of Polluted Coal Mine Drainage* (Hedin *et al.* 1994a). At the time, passive technologies comprised aerobic wetlands, organic substrate wetlands, and anoxic limestone drains (ALDs), and treatment was specifically targeted at the removal of Fe, Mn, Al and acidity. The key contribution of the USBoM was to review the operation of thirteen different full-scale passive mine water treatment systems (which had been constructed by a range of organizations, including the Tennessee Valley Authority, various other State agencies and local community coalitions) in order to derive empirical sizing criteria for the future design of such systems. The resulting formulae, the details and derivation of which are discussed in section 4 of the present guidelines, and they remain the formulae most commonly applied in the design of aerobic and organic substrate wetlands.

Some other notable publications, which either refined the design guidelines of Hedin *et al.* (1994a), or proposed new passive treatment technologies, are summarized in Table 1.1. It is notable that only the most recent publications in Table 1.1 (effectively since 1999) include works by researchers based in Europe. While passive treatment was introduced to Europe in the mid-1990s (e.g. Younger *et al.* 1997; Jarvis and Younger 1999), early implementations were highly derivative from the USBoM findings. However, over the last 5 years research in Europe has increased significantly, not least within the PIRAMID project, so that innovative passive unit processes are now beginning to emerge in Europe.

While all of the works cited here are important in terms of documenting the development of passive treatment technology, none of them provide sufficiently detailed information on the engineering aspects of the implementation of designs to act as practical guidance for would-be practitioners. Inherent to the philosophy of passive treatment is that hard civil engineering structures, such as concrete tanks and retaining structures, are best avoided (or at least well hidden!). Nevertheless, civil engineering issues cannot be avoided when addressing construction of a passive treatment scheme, particularly for units such as deep settlement lagoons.

Table 1.1. A non-exhaustive selection of publications which appeared after the the USBoM guidelines of Hedin *et al.* (1994a), in which new passive treatment technologies were proposed, or previous design guidelines were amended. An asterisk at the start of an entry indicates European researchers. Bold highlighting of authors' names denotes research arising from PIRAMID.

Author(s) (Year)	Subject
Hedin et al. (1994b)	Residence time of 14 hours recommended as optimum for ALDs
Kepler and McCleary (1994)	Successive Alkalinity Producing Systems (SAPS) proposed as alternative to ALDs, where Fe^{3+}, Al and dissolved oxygen concentrations are high.
Cohen (1996)	Updated previous work to demonstrate feasibility of passive treatment of aggressive metal mine effluents using bacterial sulphate reduction processes
Benner et al. (1997)	Demonstration of effective 'in aquifer' mine water treatment using a permeable reactive barrier (PRB)
*Sen and Johnson (1999)	Demonstrated the existence of acidophilic sulphate-reducing bacteria, thus disproving previous belief in pH > 4.5 as pre-requisite for reductive passive treatment
Cravotta and Trahan (1999)	Oxic Limestone Drains (OLD) proposed as a treatment option where anoxia unobtainable
Tarutis et al. (1999)	Proposed first-order kinetics as a more appropriate basis for constructed wetland design (however, see discussion in section 4)
***Younger** (2000)	Use of 'SCOOFI filters' and other novel passive unit processes outlined for the first time.
*Nuttall and Younger (2000)	Zinc removal from circum-neutral mine waters by ALD-like system demonstrated at field pilot scale
Watzlaf *et al.* (2000)	Defined area-equivalent design acidity removal rates for RAPS systems (re-named from 'SAPS' of Kepler and McCleary 1994)
*Jarvis and Younger (2001)	Use of high surface area reactors for rapid iron removal proposed as potential alternative to aerobic wetlands
***Batty and Younger** (2002)	Demonstration that plant uptake is an important process in achieving low residual iron concentrations in aerobic wetland effluents
***Amos and Younger** (2003)	Systematic approach to the design of reactive substrates for sulphate-reducing subsurface flow bioreactors
Cortina *et al. (2003)	Successful use of caustic magnesia as an appropriate passive system reactive substrate
***Casiot *et al*.** (2003)	Demonstration of feasibility of microbially-mediated oxidative immobilisation of arsenic in aerobic wetlands
Younger *et al. (2003)	Development of hybrid surficial PRB / ponds/ wetland systems for treating highly acidic spoil leachates

These engineering guidelines effectively build upon earlier North American guidelines to complement their coverage with recent insights from five leading European research institutions. Thus these guidelines include not only the removal of Fe, Mn, Al and acidity using wetlands and ALDs (as previously described by Hedin *et al.* 1994a), but also the newer passive technologies which emerged during the life of the PIRAMID project, which have greatly extended the "repertoire" of passive treatment to embrace a wider range of contaminants (including Zn, Cu, U and As). A further development beyond previous guidelines is the inclusion of much more detailed information on civil engineering aspects of passive treatment implementation.

Thus, as well as design formulae for sizing of units, consideration is also given to such issues as physical construction of passive treatment systems (e.g. retaining wall design), selection of inlet and outlet arrangements, materials selection, planting techniques (where appropriate), and contractual matters relating to civil engineering projects. These issues are usually the preserve of texts in civil engineering, landscape architecture, and specialist industry documentation. Therefore these guidelines are intended to bridge the gap between recent passive treatment technology research and other disciplines (especially civil engineering) of which knowledge is required to actually construct such systems.

2. REQUIRED DESIGN INFORMATION

2.1. Introduction

The design information required for a passive treatment system ranges from land purchase and planning issues, through contaminated land and site investigations, to water quality and flow-rate information. For any single full-scale passive treatment project it will be important to address all of the issues discussed below. Some of the data collection topics are essential irrespective of the scheme (e.g. water quality and flow-rate data), whilst others (e.g. contaminated land issues) may require only cursory investigation, if only to discount as possible constraints on site-specific designs.

2.2. Flow measurement

2.2.1. Importance and duration of flow monitoring

Determination of flow-rate is fundamental to the design of a treatment system. The volume of a discharge is crucial to calculating the lateral extent and volumetric capacity of the treatment facility. Poor measurement of flow-rate may result in a range of problems including:

- undesirable drying-up of zones containing sulphide minerals due to lack of flow in dry periods
- where mid-range flows are higher than those estimated before construction, inadequate retention times may occur within the treatment system, and / or

horizontal flow velocities may be too high to allow effective contaminant removal.

- Where high flows far exceed those estimated, erosion of bunds and other structural elements may result, possibly even leading to wholesale destruction of the passive system, with consequent contamination of downstream watercourses with sulphide minerals and silt washed from the passive system.

It is therefore vital that representative information on the flow-rate (including maxima and minima) be obtained, over as long a period of time as possible, before the design commences. If at all possible, measurement of the flow should be taken over a minimum of a 12-month period, so that seasonal variations may be determined. The presence of particularly 'dry years' will undoubtedly influence results and thus the meteorological context of the monitoring period should be taken into account when assessing findings. In some cases, it may be possible to use statistical techniques including regression of measured values against longer rainfall records etc (the so-called 'synthetic hydrology' approach) to convert a short flow monitoring record into an equivalent longer record. Nevertheless, there is no real substitute for obtaining as long a flow record as possible for the site in question.

Undue sophistication in measuring is frequently unnecessary, and simple structures such as 'V'-notch weirs (see below) can provide adequate information to give confidence in the design. These structures are simple and often easy to install and monitor. For many systems the vital design variable is the maximum flow-rate. Variations within that flow-range may not be vital, unless the aim is to effect a reasonable improvement in quality by treating a certain proportion of the discharge. Regular 'snapshot' monitoring of the flow by measurement of the depth over a notch will often suffice to provide the necessary data, without need for recourse to sophisticated monitors with continuous data logging. In any case, some allowance above the maximum flow-rate will often be included in system design, to allow for any irregular increases in flow-rate. The maximum flow-rate of a mine water over a weir can often be accurately estimated from the ochreous high water mark which most mine waters leave on such structures.

2.2.2. Methods of flow measurement

This discussion is limited to measurement of flow from point sources i.e. pipes and open channels. For diffuse sources measurement of flow-rate is more complex. At its simplest, a diffuse discharge may be intercepted by a channel e.g. along the toe of a spoil tip. Measurement of groundwater flows entails the use of boreholes and pump tests, and these operations are beyond the scope of these guidelines.

The choice of flow measuring device may well be dictated by the nature of the point of discharge, and / or the level of accuracy of flow measurement required. Unless flow-rates fluctuate over very short time-scales it is unlikely that continuous logging equipment will be required. In most case the installation of some form of weir, or even bucket and stopwatch measurement, is sufficient. The various methods are discussed below.

‘Bucket-and-stopwatch’:

Measurements of flow-rate can be made using a bucket of known volume, and repeatedly (at least 3 times) recording the time it takes to fill. This method of measurement is highly accurate as long as the time taken to fill exceeds approximately 10 seconds. For time periods less than this, accuracy is compromised. Thus, since most ordinary buckets have capacities of around 10 L, this method is most appropriate for discharges with a flow-rate of ≤ 1 L/s.

Velocity-area method

When undertaken properly, the velocity-area method for measuring open-channel flow can provide flow-rate data to an accuracy of ± 10%. For purposes of measurement, the entire channel width channel is notionally subdivided into 10 or more equal-width sub-sections (usually 0.5m intervals will suffice), and then the depth and velocity of each subsection is measured. In practice, sub-division is often most easily achieved by suspending a graduated rope[5] across the channel at the measurement section. Using waders, the responsible technician then traverses the river, stopping to measure the velocity in each sub-section. The technician should face upstream and hold the velocity measuring device in front of them (thus avoiding the problems of flow disturbance by their own legs which would occur if flow were to be measured downstream). Velocity is typically measured using some form of impeller, and the impeller is suspended at three fifths of the total water depth (at which point the actual velocity tends to closely reflect the mean velocity throughout the full depth of water). The products of the area (m^2) and velocity (m/s) for each subsection are summed to provide a flow-rate (m^3/s). The accuracy of this method is much reduced for channels containing large boulders, which may result in inaccurate calculation of channel area. The method is also inappropriate for narrow channels (i.e. < 4 m), where errors may be ± 40% (Younger *et al*, 2002).

Control structures

If neither of the above techniques is appropriate, or there will be a need for repeated measurements of flow at the same site, hydraulic ‘control structures’ can be constructed in the path of the flow to be gauged. The objective of all control structures is to generate “critical flow” conditions, under which the relationship between flow-rate and water depth is linear.

V-notch weirs, as the name implies, are thin, vertically-oriented plates (of steel, wood, fibreglass or some other material) into the top edge of which a straight-sided v-shaped notch has been cut. The edge of the "V" should be chamfered to a sharp edge, and the apex of the 'V' (i.e. the base of the notch) is usually formed to make an angle of 90^o (though lesser angles can be used for more accurate measurement of low flows). For accurate readings to be made, care must be taken to ensure that the V-notch is correctly aligned i.e. perpendicular to the flow, and vertical. There must be a sufficient fall on the downstream face that water leaving the v-notch falls freely, with

[5] i.e. an ordinary rope around which brightly-coloured hoops of adhesive tape are fixed every 0.5m. This rope also doubles as a useful safety aid for the wading technician.

air below the water jet, to the downstream water surface. Once the v-notch is in place, flow measurement is straightforward: it simply entails measuring the depth of water above the apex of the "V", and converting this into an equivalent flow either using tables (which are widely available in textbooks, such as that of Brassington 1998), or the formula given in Table 2.1 below. For convenience a graduated scale can be fixed on the upstream face of the weir to expedite reading of the depth of water above the 'V'. Automatic data loggers can easily be installed behinds v-notch weirs to yield continuous flow records of very high quality.

An alternative to the v-notch, particularly appropriate to larger flows (> 20 L/s), is the rectangular thin-plate weir. These are similar in all respects to v-notch weirs, save for the rectangular shape of the notch. The width (L) of the rectangular notch needs to be noted in order to calculate flows from the measured depth of water flowing over the weir (Table 2.1).

One particular drawback of thin-plate weirs in mine water applications is their tendency to become clogged with ochre and other debris. Open-jawed flumes are less prone to clogging, and are far more easy to clean, than thin-plate weirs. For modest mine waters flows (≤ 4 l/s) a variety of flume known as an "H-flume" has found good application (Younger *et al.* 2002), and can be fitted with data loggers to obtain continuous flow measurements. As with thin-plate weirs, the water exiting a H-flume must do so by free fall. However, flumes need more careful construction than weirs, as the dimensions of several components need to be very precisely machined and joined for theoretical ratings to apply. Flumes also require straight length of channel upstream (usually also fabricated), the required size of which increases proportionally to the flow-rate. Comprehensive guidance on the design of such structures is available elsewhere (Ackers *et al.* 1978).

The accuracy of measurement obtained with a control structure generally increases with the provision of a turbulence-reducing "stilling basin" upstream. Calculation of the flow-rate from the vertical head of water within each of these control structures is easily calculated from the equations given in Table 2.1.

Table 2.1. Approximate formulae allowing calculation of flow (Q) from measurements of head (h) behind thin-plate weirs (after Younger *et al.*, 2002).

Type of flow gauging structure	Simplified rating equation	Comments
V-notch weir (90°)	$Q = 0.29\ h - 32.5$	Head measured above the apex of the 'V'. Yield Q in L/s if h is in mm.
Rectangular thin-plate weir	$Q = L \cdot [1.83\ (1 - h) \cdot h^{1.5}]$	Head measured above the crest of the weir. L is the width of the weir in m. The formula yields Q in m^3/s if h is in m.

Continuous Monitoring of flow in open channels

In a manner analogous to the use of data loggers behind thin-plate weirs, continuous monitoring of flow in an open channel generally requires the measurement of water depth upstream of some "control section", which may either be a large weir suited to river conditions (e.g. a Crump weir; Ackers *et al.* 1978), or else a 'natural control' such as a waterfall or a naturally straight, shallow section of channel. In most cases, theoretical ratings for these control sections will not apply so well as in the cases of thin-plate weirs, so that spot-measurement by means of the velocity-area method will be used to develop unique stage-discharge formulae for each control section. Using the appropriate formula, logged water depth measurements can be readily converted into equivalent flows.

The depth of flowing water upstream of an open-channel control section can be measured by the following methods:

- directly, from a gauging board (i.e. a giant ruler) fixed in the water.
- by a float with a mechanical linkage direct to an instrument mounted alongside the stilling chamber
- by a pressure transmitter / transducer attached to a solid-state logger
- by low pressure air bleed dip tube indicating pressure variations to an instrument located in an adjacent building
- by an ultrasonic transmitter/receiver mounted above the water surface linked to a recording instrument
- by a conductivity contact probe graduated over the range of flow and linked to a recording instrument sited locally to the probe

Most instruments capable of converting depth into flow rate feature displays giving percentage full flow indication, total quantity indication and often a graphical output. Air bleed and ultrasonic types require an electrical supply and are not suitable for use in remote locations. The mechanical type, however, can be used remotely, as can the conductivity probe type that is battery powered and can operate unattended for periods up to one year. All instruments require periodic maintenance and housing for protection from frost and vandals.

Flow Meters for Pipes

There are a great number of types of flow meter available for the measurement of clean or contaminated waters flowing through pipes. Effluents which will be considered for passive treatment will generally contain either dissolved or suspended contaminants that will create difficulty in ensuring accuracy of measurement.

Many meters are not suitable for use in mining/industrial operations. For example, electromagnetic flow meters and venturi meters are costly and more applicable to large flows than will generally be experienced in effluent treatment. Other types of insertion meters can quickly become fouled by accretion of iron deposits to the vanes or other measuring equipment within the pipe rendering the installation ineffective. Even with external meters, such as ultrasonic types, accretion of scale to the inner surface of the pipes reduces the cross-sectional area of flow resulting in overestimation of the volume. Therefore, for any permanent metering on or in pipework, means for regular maintenance and cleaning of the meter are essential to maintain accuracy.

Notwithstanding these *caveats*, the following types of meter may be considered.

- **Differential pressure meters**: A constriction placed in the fluid causes it to accelerate at the expense of its pressure energy and the measurement of this change is used to determine the flow rate. The simplest type is the orifice plate. This is inexpensive to purchase, but it creates a head loss in the pipe main and thus increases the power costs for pumping.

- **Variable area meters**: The flow of water in the main either moves a hinged obstruction, raises a float, or enlarges a variable area orifice, the movements of which are measured and converted to a flow rate. Such devices can measure over a wide range of flows with moderate accuracy. They are generally inexpensive but are not suited to effluents containing high concentrations of suspended solids or dissolved materials which will precipitate on, and accrete to, the hinged system.

- **Rotating mechanical meters**: Most meters of this type incorporate a vane, rotor or other device within the path of flow that is free to rotate as water passes. The revolution of the vane is monitored either mechanically or electronically and converted into flow, most often displayed in digital form. Mechanical types need no power supply and are therefore suited for remote locations. The weak point of all rotating flow meters is the bearing and for this reason they are generally unsuited for use in waters containing high-suspended solids concentrations or dissolved iron.

- **Ultrasonic meters**: These meters monitor flow by passing beams of ultrasonic sound into the effluent and measuring the resultant change in the transmitted signal. The measuring apparatus is mounted externally on the pipe and thus presents no obstruction to flow. There are two main types of ultrasonic meter:

  - Doppler meters, which are inexpensive, moderately accurate and are suited for use with waters that are aerated or contain dirt particles. The frequency shift of an ultrasonic wave reflected from an air bubble or dirt particle is measured and this allows determination of flow rate. The device requires mains power and is glued or clamped onto a pipeline, no cutting or special sections being necessary.

  - The diagonal beam meter which is a more expensive but accurate meter supplied in a fabricated section for connection into the pipeline. The principle of measurement is the determination of the difference in transit time of pulses of ultrasonic sound transmitted alternately upstream and downstream. Even greater accuracy can be obtained by the use of pairs of transmitting units.

Ultrasonic meters are not suitable for measurement of flow of mine waters containing dissolved iron which can decrease the internal diameter of a pipe by deposition of scale, giving rise to inaccurate readings.

- **Insertion meters**: Insertion meters provide an approximate value of flow rate at a greatly reduced cost compared with full bore meters. They measure the velocity of flow at a point one quarter of a radius distant from the pipe wall, this having been found to approximate to the mean pipe velocity. Propeller insertion meters are the least expensive and most widely used. Contamination of internal parts by the effluent is a significant problem.

Generally, it is unlikely that a sophisticated type of meter for mine water flow measurement will either be necessary or justified. Their use for mine waters is typically not appropriate anyway, for several reasons:

1. The site is likely to be remote without power supply available.
2. Vandalism of equipment on remote unmanned sites is likely to be a problem.
3. A high degree of accuracy or continual monitoring is not likely to be a requirement for most projects.

For most projects periodic flow measurement, using a weir or flume in an open channel, is adequate. Note that the agreement of the authorities responsible for flow and quality in rivers will be required for installation of a permanent, artificial control structure.

2.3 Hydrochemical sampling and analysis

2.3.1 Introduction

The importance of mine water sampling and analysis cannot be over-emphasised. No matter how well constructed, a passive treatment system that has been designed using inaccurate or unreliable hydrochemical data may well fail to treat the mine water effectively. To use most of the engineering guidelines that follow, it is essential to have reliable flow-rate and water quality data. While flow-rate data are essential in determining the size (and ultimately cost) of a scheme, it is the hydrochemical data that will determine what type of treatment units are required, how many of them are required, and what order they should be placed in.

If the paragraph above leaves the impression that flow-rate and water quality data can be collected independently, it is time to thoroughly dispel that notion. Unless it is absolutely guaranteed that the flow-rate of the water does not vary (a most unlikely situation), ***the chemical data will be virtually useless for treatment system design purposes unless a simultaneous measurement of flow-rate is made.*** It is the contaminant load (i.e. flow-rate x concentration) that ultimately determines the system type and size. Since contaminant concentrations invariably change with flow-rate*, it is not possible to reliably calculate contaminant load unless a water sample is collected and flow-rate measured at the same time.* In the UK at least, the authors of this report have repeatedly seen time and effort wasted because this has not been done.

Because hydrochemical sampling and analysis are far more costly per unit measurement than flow monitoring, there will always be a temptation to skimp on hydrochemical sampling. In the short-term, this temptation should be resisted. Only when the behaviour of a particular the discharge has been determined from several weeks or months of intense, synchronous chemical sampling and flow measurement, might it become possible to design a 'compromise' characterisation strategy for the longer term, such as a combination of continuous flow monitoring coupled with weekly or monthly chemical sampling. Even if such a strategy is implemented, it remains essential that ***flow be recorded every time a chemical sample is collected.***

2.3.2. Variables to be determined

There is nothing more frustrating for a design engineer than to sit down with an apparently thorough set of mine water chemistry data, spanning at least 6-12 months, only to discover that whoever decided the analytical suite for the samples failed to include crucial design variables in their list. What then, are the essential variables? Table 2.2 lists the 'standard suite' recommended by the authors for the characterisation of the majority of polluted mine waters. The reasons for including the listed parameters in this suite are indicated in Table 2.2 in accordance with the following list:

a) key design variable for most mine waters
b) key design variable for acidic mine waters
c) possible design variable for *some* mine waters
d) key variable where mine water will enter a watercourse used for drinking water supply
e) needed to check cation-anion balance of analysis for quality-control purposes (see 2.3.5 below) and to facilitate geochemical speciation / mineral equilibrium modelling, where necessary
f) Indicates overall salinity of mine water; this may limit applicability of wetland treatment, because plant growth is most unlikely to be successful where conductivity > 10000 μS/cm.
g) Can be present as a pollutant at mg/L levels in some mine waters.

Once the nature of a particular water has been established from repeated sampling and analysis, it may be possible to eliminate certain determinands from the standard suite. This is typically so in relation to Al in waters with pH > 5.5, and for Zn in many coal mine waters. On the other hand, it may also be necessary to add further parameters to the suite. Which parameters ought to be added will be depend to a considerable extent on the local mining geology. Metals which are often added to the standard suite include copper (Cu), cadmium (Cd), nickel (Ni), lead (Pb) and chromium (Cr), all of which are commonly found in waters draining from metalliferous ore bodies. One indication that one or more of these metals might be present in a mine water hitherto analysed using only the standard suite would be persistently negative values of the cation-anion balance (see 2..3.5 below). Other metals, such as mercury (Hg), are principally found in effluents from mines which either exploited the mineral cinnabar (HgS) or else encountered it as a gangue mineral and are very rarely found in other mine waters. Uranium (U) and other radioactive metals are commonly found in uranium mine effluents, but can also be found in coal mine effluents (e.g. in Poland and South Africa). Arsenic (As) is most often associated with minor arsenopyrite in base-metal ore bodies. Cyanide (CN) of anthropogenic origin may also be present in drainage from certain gold mining sites.

A discussion of geochemical analysis and interpretation could fill a book in its own right, and indeed it has (many times). A very accessible introduction to geochemistry in general is provided by Appelo and Postma (1993), while the second chapter of the recent mine water text book of Younger *et al.*(2002) provides comprehensive treatment of the chemistry of mine drainage.

Table 2.2. Standard analytical suite suited to characterisation of the majority of mine waters.

Variable	Measurement unit	Reasons for inclusion (see text for meaning of letters)
pH		a, b
Total Acidity	mg/L as $CaCO_3$	b
Total Alkalinity[6]	mg/L as $CaCO_3$	a, b
Conductivity	µS/cm	d, f
Suspended Solids	mg/L	c
Sulphate (SO_4)	mg/L	d, e
Chloride (Cl)	mg/L	d, e
Ammoniacal nitrogen (NH_4^+)	mg/L	c, g
Calcium (Ca)	mg/L	c, e
Magnesium (Mg)	mg/L	e
Sodium (Na)	mg/L	c, e
Potassium (K)	mg/L	e
Total Iron (Fe)	mg/L	a, b, d, e
Ferrous Iron (Fe^{2+})	mg/L	a, b, d, e
Manganese (Mn)	mg/L	a, b, d
Aluminium (Al)	mg/L	b, d, e
Zinc (Zn)	mg/L	c, g

[6] For waters with pH < 9 (i.e. virtually all mine waters) bicarbonate concentration (HCO_3^-) can be acceptably calculated from the alkalinity by multiplying by 1.22

2.3.3. *On-site analysis*

It is strongly recommended that certain key variables which are subject to change during transportation and storage, and which can thus give misleading results if analysed only in the laboratory, are instead analysed on site wherever possible. Parameters in this category are:

- Temperature
- pH
- dissolved oxygen
- oxidation-reduction potential (Eh)
- alkalinity.

In addition although conductivity rarely changes greatly during transportation and storage of mine waters, it is nevertheless commonly measured on-site for two reasons:

(i) it is very easy to measure, and

(ii) knowing the conductivity of a water whilst still in the field often helps in deciding which particular waters to sample.

With the exception of alkalinity, all of the above parameters can be measured easily using hand-held electronic meters (a single meter will often measure several of the above variables). There are numerous suppliers of such instruments, and at present prices vary from as little as €100 to in excess of €1500. Generally speaking, the more robust and accurate instruments are the more expensive. No matter what price the instrument, it will eventually give erroneous results if it is not maintained properly and calibrated regularly with laboratory-grade standard solutions. During intensive sampling trips it is recommended that probes are cleaned and calibrated daily. Dissolved oxygen probes should be calibrated before each and every measurement.

Mine waters which emerge from hydraulically confined systems are usually not in thermodynamic equilibrium with the Earth's atmosphere. Specifically they may contain elevated concentrations of dissolved carbon dioxide (CO_2), which will be liberated upon emergence to atmospheric conditions (for detailed discussion see Stumm and Morgan 1996). This may have profound effects on the chemistry of the water over time, and will certainly affect the alkalinity of the water (and in some cases the total acidity also). It is therefore advisable to measure the alkalinity (and occasionally also the acidity) on-site, using sub-samples of water collected as close to the point of emergence as reasonably possible. These two parameters can both be measured by titration using hand-held kits (the cost of which are currently around €600, plus consumables). The basic idea is that an indicator in the sample changes colour when all of the alkalinity or acidity in the water is used up (by adding an acid or alkali respectively). By knowing the volume of acid (if measuring alkalinity) or alkali (if measuring acidity) added to the sample, it is possible to calculate the concentration. The difficulty with these tests is that the indicator does not always change colour rapidly, and the analyst must therefore know the colour they are looking for. This may cause problems of analytical consistency if different analysts undertake sampling at the same site. For this reason it is sometimes advisable to have the alkalinity and acidity measured by a laboratory as well.

Because of problems with carbon dioxide degassing, it is often worthwhile calculating the acidity concentration. Acidity is calculated as follows (from Hedin *et al.*, 1994a):

$$Acidity_{calc}\ (mg/L\ as\ CaCO_3) = 50 * [2Fe^{2+}/56 + 3Fe^{3+}/56 + 2Mn/55 + 3Al/27 + (1000*10^{-pH})]$$

where the relevant metal concentrations are in mg/L. Other metals can be added into the equation if they are present in elevated concentrations, using their valency and atomic masses respectively as numerator and denominator factors in the terms inserted for that metal. If there is uncertainty as to whether a field or laboratory determination of acidity is correct, it is recommended that this calculation is used.

As long as the sample is preserved in acid (see below), metal concentrations are unlikely to fluctuate measurably in the time it takes for a sample to be analysed in the laboratory, and therefore usually metal concentrations are not measured on-site. However, measurement of total and ferrous (Fe^{2+}) iron on-site can be advisable, partly because the ratio of ferrous to ferric (Fe^{3+}) iron may change, and partly because it is sometimes good to have an instant appreciation of the concentration of this key variable (at least at coal mine sites). Portable photometers can be used for this analysis, but they typically cost in excess of €1500, and the consumables bill can also run high if regular determinations are made.

Some of these on site tests can be very useful when carrying out reconnaissance work. For example, pH and conductivity alone can often be sufficient to identify a mine water with some confidence, if visual inspection is not enough. This is because mine waters may well be acidic, and they may also contain high concentrations of sulphate, which is reflected in an elevated conductivity measurement. Where many waters have to be checked to identify sources of pollution this screening approach can often save both time and money.

2.3.4. Sample collection for laboratory analysis

For a typical mine water analysis (such as that shown in Table 2.2), two samples need to be collected. Both should be collected in thoroughly clean plastic (polyethylene) bottles of about 500 mL capacity. (Check with your laboratory if you wish to use smaller bottles, bearing in mind that bottles smaller than 60 mL are unlikely to be large enough for the purposes of any laboratory). The first sample will be sent to the laboratory for analysis of all variables except metal concentrations. This bottle needs no pre-treatment To the second sample bottle, which will be used for analysis of metal concentrations, 1 - 2 mL of concentrated acid should be added as a preservative[7]. While most hydrochemical texts advocate the use of reagent-grade nitric acid (HNO_3) for this purpose, if direct laboratory measurement of Fe^{2+} is

[7] The acid maintains the pH of the sample at less than 2. This ensures that the metals are kept in solution, and therefore that there will be no chance of metals forming a precipitate at the bottom of the bottle, or accreting to the sides of the bottle.

intended it is much wiser to use reagent-grade hydrochloric acid (HCl) instead, as this avoids the problem of the NO_3^- ion oxidising Fe^{2+} to Fe^{3+}.

On the face of it water sample collection is a simple enough task, but there are some precautionary / advisory notes to bear in mind:

- Avoid disturbing the bed of the stream or channel, which may mobilise sediment, and ultimately lead to a falsely high concentration of a determinand(s). This is a particularly significant issue at mine water discharges, where the bed may be heavily coated in metal precipitates. The best way to avoid the problem is to take a sample from a point where the water is in free fall (e.g. end of a pipe) or, if this is not possible, at least where the stream is deep. Sometimes it will only be possible to collect a sample without disturbing the bed by using a smaller vessel or a syringe to decant water into the sample bottle.
- If it is necessary to enter a watercourse when approaching a sample point, always do so from downstream, and always stand downstream of the point where the sample is collected (again, to avoid bed disturbance at the sample point).
- If the discharge is to be sampled regularly, always take the sample at the same point, and ensure that other samplers are familiar with the exact sample point location. It is also good practice to collect repeat samples at approximately the same time of day to avoid misleading impressions arising from diurnal heating / cooling and other short-term time-dependent phenomena. (If such phenomena are of specific interest, it will be necessary to specifically plan campaigns of high-frequency sampling, which will most conveniently undertaken using an auto-sampler wherever possible).
- Always endeavour to collect a sample from running water (clearly not possible for pit lakes, settlement lagoons etc), because this is most likely to represent a well-mixed part of the flow.
- The sample bottle that does not contain acid should be rinsed with the water to be sampled before collecting the sample itself.
- If the acidified bottle is submerged in the channel the acid will be washed out of it. Therefore fill the acidified bottle from the second sample bottle, topping the latter up afterwards.
- Fill the acidified bottle to the shoulder. The second bottle should be filled as completely as possible, to exclude air (which, if present, will encourage microbial reactions during transport and storage).
- Concentrated acids liberate noxious fumes. It is advisable to keep as far away from the acidified bottle as possible when filling it.
- Contact of concentrated acid with skin should clearly be avoided, and therefore rubber or latex gloves are advisable. If the sampler's skin is broken, or cut, gloves should definitely be worn, because of the risk of waterborne disease (see the final bullet-point below).
- All sample bottles should be labelled clearly with appropriate information: sample identification, sampler's name and company, date (and time) of sample collection, any preservative added to sample.

- To have confidence that a sample is representative the collection of multiple samples is sometimes advocated (3, or even 5). This may be possible for an academic institution, where laboratory analysis is usually undertaken in house, but rarely so in industry, where the client is unlikely to welcome a laboratory analysis bill 3 or 5 times higher than they envisaged. This is much less of an issue if samples are collected regularly over a period of time (as they should be – see below), because any anomalous samples will become evident when time series plots of the variables are generated.
- It may be deemed necessary to collect filtered samples. On site filtering is usually accomplished using a syringe and 0.2 μm filter. Filtering enables a distinction to be made between metal ions that may be adsorbed to particulate matter and metal ions in solution. In addition, filtering may remove bacteria that would otherwise cause changes in sample quality during transportation and storage. For the design of treatment systems for abandoned mine water discharges filtered samples are rarely required, because the suspended solids load of these waters is typically low. However, at active mine sites the suspended solids load may be significant, and therefore collection of filtered samples may be prudent.
- Local residents are often a source of valuable information about the mine water discharge on their doorstep. Local anecdotes about the variation in flow, or the appearance of the discharge, can often be a useful guide when deciding how to quantify the water in terms of flow and chemical quality.
- When planning fieldwork bear in mind that accurate and thorough on site analyses and sample collection may take up to an hour for a single discharge.
- Samplers should be aware of the **health and safety** risks associated with such work, and should comply with the risk assessment and health and safety requirements relevant to the jurisdiction within which they are working. The minimum requirement, even in the absence of legal obligations, is to assess hazards rationally and plan for their minimisation by means of safe working practices. Unusual hazards particularly associated with mine water sampling include the following:
  - Mine water discharges are often in remote locations, and / or adjacent to fast flowing rivers. Safe access must be planned carefully. As a minimum, a colleague at base should know the whereabouts of the sampler, and their estimated time of return. Radio or mobile phone contact with base ought to be maintained where appropriate / possible. For particularly hazardous sites (and for all underground sampling) the sampling party should number at least 2 people (in many large regulatory / water organisations this is in any case mandatory), and for underground sampling a 'banksman' should be in position, i.e. a person stationed at surface to await the return of the sampling party by a pre-arranged time, and equipped to raise the alarm in the event they do not return as planned.
  - People sampling mine waters are subject to the same disease risks as others working in the water / wastewater industry, and as such they need to take the relevant precautions. Rubber or latex laboratory gloves should be worn when sampling, as already mentioned. These not only help to combat risks from potentially hazardous reagents used in sampling (such as strong acids) but also

help to minimise the risk of contracting Weil's disease (*Leptospirosis*), a rare but potentially lethal disease transmitted by rats and their excreta. Less dramatically, such gloves also help minimise the minor but irritating dermatological problems which many people experience following repeated exposure to acidic, metalliferous waters. More importantly, it is essential that all personnel involved with sampling have up-to-date inoculations against the following sewage-related diseases: hepatitis A, polio, typhoid and tetanus.

- A significant proportion of mine waters exsolve potentially hazardous gases (most notably CO_2 and radon) as they encounter atmospheric pressure. Accumulation of such gases in confined spaces through which the mine water flows to surface (which in many cases will be favoured sampling locations, such as drift portals, shaft collars or access chambers) can give rise to serious hazards. Accumulation of excess CO_2 can render the air so oxygen-deficient that it can rapidly cause fainting, and even death. At the very least it will lead to laboured breathing and a severe, persistent headache. Exposure to radon gas does not tend to cause such acute problems, but repeated exposure can lead to a significantly increased risk of long-term cardio-vascular health problems. The mining industry is well-versed in the precautions necessary to minimise these 'confined spaces' risks, and the means for testing for the existence of dangerous accumulations of gases (including flame safety lamps and electronic gas detectors). Where any doubts exist as to the atmospheric safety of a mine water sampling station, the advice of qualified professionals must be sought and heeded.

Table 2.3 is a checklist of items typically required when visiting mine water discharge sites to carry out on site testing and sample collection.

2.3.5. Analytical quality checks

In industry, laboratory analysis is usually undertaken by a specialist laboratory contractor, and therefore detailed knowledge of the analytical techniques is not required (although it sometimes helps to know when checking the quality of an analysis). For organisations conducting their own analysis, there are several standard reference books that detail acceptable methods. *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998) is one such text.

Table 2.3. Checklist of items that will / may be required during on-site testing and sample collection.

Item	Comments
Personal Protective Equipment	Especially waterproof gloves and boots
On site testing equipment e.g. pH probe and digital titrator for alkalinity testing	Ensure calibration before heading outdoors
Sample bottles	Usually worth carrying more than required
Sample bottle labels and indelible marker pen	Easiest to write whilst labels are still dry
Cloth for drying sample bottles and equipment	
Field notebook and pencil	Notebooks with waterproof paper are available; biros will not work on wet paper
Camera	Photographs often help with future site identification
Mobile telephone	To contact colleagues in the event of an emergency
Cool box or similar for storing samples	
Sample collection vessel (with rope if sampling from bridges etc)	Should be plastic
Steel ruler or tape measure for measuring water depths in V-notch weirs etc.	
De-ionised water for cleaning probes etc	
Syringes and filters if required	
Keys / permits for secured sites	

If samples are sent to a reputable laboratory, and / or the laboratory work is undertaken by an experienced analyst, the results of the mine water analysis should be reliable. However, the authors have frequently received accredited analyses which the experienced eye can quickly recognise as being gravely in error (for instance, when the analytical results indicate a thermodynamic impossibility, such as 200 mg/L dissolved Al in a water with pH 7).

Identifying erroneous results can be difficult, and if the validity of results is not amenable to ready clarification, a specialist hydrochemist should be consulted. However, there are some basic checks that can be done, provided the analysis includes most of the determinands listed in Table 2.2:

- If ferrous iron and total iron concentrations are both measured, then the total iron concentration should always be equal to or higher than the ferrous iron concentration. In the same way, if dissolved metal concentration is measured it should always be equal to, or lower than, the equivalent total concentration.
- At least for long-running discharges from deep mines, check whether the analysis looks similar to previous ones. If not, why not? Sudden major changes in the water quality of deep mine discharges are not unheard of, but they are rare. It is advisable to have the analysis checked if such changes occur (laboratories usually

keep samples for some time after despatching results, exactly so that such queries can be accommodated).

- The cation-anion balance can be calculated if all major ions have been determined. This typically includes SO_4^{2-}, Cl^-, HCO_3^- (anions) and Ca^{2+}, Mg^{2+}, Na^+, K^+ (cations), and usually at least one of the main metal contaminants if the concentration is high (e.g. Fe^{2+} / Fe^{3+} and Al^{3+}). The basic idea is that, when converted to units of milliequivalents per litre (which accounts for ion charge), the sum of the cations (positively charged) and anions (negatively charged) should be equivalent. Concentration in meq/L is calculated by multiplying the ion or molecule charge (or valency) by the concentration in mg/L, and dividing the product by the atomic or molecular weight (shown in the at the bottom of the relevant box on the periodic table). The cation-anion balance (CAB) is then calculated using the following equation:

$$\text{Cation-anion balance (\%)} = \frac{\Sigma \text{ cations (meq/L)} - \Sigma \text{ anions (meq/L)}}{\Sigma \text{ cations (meq/L)} + \Sigma \text{ anions (meq/L)}} \times 100$$

  Strictly the balance should not exceed ±1-2%, although ±5% is often deemed acceptable; however, where values climb above 10% questions should be asked.
- High values of CAB can indicate analytical error. However, where this can be ruled out, it may be that the CAB is actually revealing that the analysis has not included some determinand which is actually present in the water at significant concentrations. A large negative CAB will indicate one or more unidentified metals in the water (Cd? Ni? etc), whereas a large positive CAB will indicate unidentified anions (e.g. oxyanions of chromium or arsenic?)
- As a rapid guide to the accuracy of the analysis the total of milliequivalents per litre for either cations or anions should approximately equal the conductivity (in μS/cm) multiplied by 100 (assuming the measurement of conductivity is accurate of course) (Hem, 1992).

2.3.6. Sampling frequency

It is not possible to provide an absolute answer to the question of how many samples should be collected (together with simultaneous flow-rate measurements) before a system can be designed with confidence. Formal sampling theories based on statistical distributions do exist, though in practice they almost always recommend sampling to a density beyond the most generous resources. Lesser sampling densities are therefore the norm, and these tend to be designed on the basis of heuristic knowledge. In general terms, even the most basic characterisation of a variable in statistical terms (mean plus standard deviation) requires a minimum of 6 measurements. Thus, for example, with 6 paired measurements of iron concentration and flow-rate it might be possible to statistically show that iron concentration decreases as flow-rate increases. However, to predict iron concentrations by extrapolating above and below the highest and lowest observed flow-rate values would be a risky strategy, particularly since such relationships are rarely linear.

Besides, it is still necessary to know the upper flow-rate in order to design the treatment system.

Uncontrolled discharges from large, deep, mines often have a reasonably consistent flow-rate, because the time taken for infiltration of recharge (combined with the flow balancing effect of storage in saturated mine voids) effectively dampens the peak flows that are evident in surface watercourses. Discharges from shallow workings and from spoil heaps are far less amenable to treatment system design with sparse data, because flow-rates may fluctuate significantly. Furthermore, increases in flow-rate are not necessarily reflected by decreases in contaminant concentration (Younger *et al.* 2002, chapter 3). The net result, under these circumstances, is that contaminant load may increase by an order of magnitude during storm events. Clearly, if a treatment system is designed for 'normal' weather conditions, it will be inundated during a storm event. This may result in very visible pollution of the receiving watercourse, at best resulting in embarrassment for those who designed the system and, at worst, prosecution of those responsible for it.

An added complication is that most discharges exhibit some form of cyclical fluctuation, in terms of contaminant load, flow-rate, or both. Diurnal fluctuations have been noted at some coastal deep mines, due to tidal effects, although the authors are not aware of any examples where the effect is so great as to influence treatment system design. Seasonal fluctuations are invariably associated with weather conditions. Such changes may well have an influence on treatment system design. Finally there are long term changes in quality, which are usually manifested as a trend rather than cyclical event (Younger 1997; 2000b). An example of this can be found at the Freiberg/Saxony mines in eastern Germany, from which water drains via the 50 km long Rothschönberg Dewatering Adit. During mining the total iron concentration of the water was 2 mg/L. This rapidly increased to 170 mg/L shortly after abandonment in 1969, fell to 20 mg/L in 1975, and at the time of writing is as low as 1 mg/L (Merkel *et al.*, 1997; Baacke, 1999). Whilst treatment system design can take account of these trends, it will rarely be realistic to attempt to monitor them during the periods of sampling and analysis discussed in this section.

An important point is to attempt to quantify the magnitude of seasonal fluctuations mentioned above. Therefore, samples should preferably be collected at regular intervals (at least monthly) over 12 months or, failing this, at least 6 months across the summer-winter transition. Beyond this the objective of sampling and analysis should be to characterise short-term storm events. For uncontrolled discharges from deep mines a 6-12 month sampling programme may suffice, since the effects of storms are minimal. However, for more 'flashy' discharges intensive sampling may be advisable. It is conceivable that during the very highest flow-rates it may be possible to allow some untreated mine water to discharge directly to the receiving watercourse, since the river itself is likely to be in spate. This can only be established through rigorous sampling and analysis. However, the costs of such an exercise will be far outweighed by the potential savings if a treatment system does not have to be designed to treat peak flows.

Sampling during a storm event is logistically difficult, as storm events are inherently unpredictable. If sufficient manpower is not available it may be worthwhile considering the use of automatic samplers (if they can be secured at the site). Whatever the approach, it is undoubtedly worth the effort involved, in order to properly characterise the mine water selected for treatment. It is unlikely to ever be possible to collect a data set that represents every conceivable condition of the discharge (for example due to large, low frequency, storm events). For this reason, building in flexibility to the treatment system design is often desirable, and these issues are discussed elsewhere in these guidelines.

In conclusion it may be said that it is often necessary to begin sampling and analysis before a firm decision can be made on how many samples should be collected, and when. For mature, uncontrolled discharges from deep mines as few as six samples collected over the course of a year may be sufficient. For other discharges however, especially those associated with rapid shallow infiltration and surface runoff, characterisation of the discharge may well require more intensive sampling. In practical terms the time frame for sample collection and analysis of samples may be dictated by the overall schedule of the project and financial considerations. However, to design a mine water treatment system using only sparse data is a very high risk strategy, and is not to be recommended. Clients should be persuaded of the wisdom of adequate sample collection prior to making large financial commitments.

2.4. Treatment site selection

Once the contaminated discharge has been accurately quantified in terms of water quality and flow-rate the design process begins in earnest. The design of a passive treatment is generally a somewhat iterative process, in that it may be necessary to proceed with some of the data-gathering before a design constraint becomes apparent. In some cases the constraint may be so serious that it is necessary to return to the drawing board, selecting a new site for the scheme, and repeating tasks such as the topographical survey and walkover surveys.

As an example of this, the design of a treatment scheme for a discharge in South Yorkshire, UK, was recently halted due to the information gained during the ground investigation of the proposed treatment site. The topographical survey, ecological survey, and outline scheme design had all been completed at the site, but the ground investigation then revealed a very shallow water table across the site, together with poor quality ground for construction. The cost estimate for the construction of the proposed scheme increased significantly, and it therefore became questionable whether the site was the most appropriate available. It is easy to ask, why was the ground investigation not undertaken earlier? The reasons in this case (as in many similar cases) were three-fold:

1. In many cases, ground investigation cannot be undertaken until issues of land ownership / permissive access have been resolved, and in densely-populated parts of Europe these issues are the most difficult and time-consuming hurdles faced by would-be mine water remediation practitioners.

2. Ground investigation is a relatively expensive task (requiring machinery, labour, site engineer and report preparation), and therefore it is worthwhile being reasonably confident that the chosen site will be appropriate e.g. appropriate access arrangements, suitable area of land available, appropriate site relief and the absence of protected species etc.
3. Ground investigation does not normally take more than a few working days to complete. It therefore does not need to be started at the outset of a project because it is not usually likely to hold up the overall project programme.

Anecdotes such as this may leave the reader uncertain of which task to begin with when undertaking a scheme design. However, the selection of an appropriate treatment site is undoubtedly the first task after discharge characterisation. It will clearly be necessary to have at least some feel for what area of land will be required for the treatment system, and therefore some outline design work will need to have been completed (usually as part of a feasibility study). As mentioned above, acquiring a treatment site is often a lengthy process, which is one of several reasons why it should be started early. It is usually necessary to consult with planning authorities in order to secure a site, and the time-scale for the application and approval process is invariably measured in months. Early contact with the relevant bodies is therefore strongly recommended if delays are to be avoided, and the services of a qualified land agent is advisable.

The remainder of this section discusses the other information that must be gathered once a suitable treatment site has been identified.

2.5 Site topography

A topographical survey of the area should be commissioned to provide adequate information to enable earthworks quantities and site gradients to be established to a high degree of accuracy.

Modern surveying techniques, using total stations and Global Positioning Systems (GPS) can economically provide comprehensive detail on the extent of the site and the topographical gradients that exist. These data should be provided in digital form to simplify the tasks of calculation of gradients and earthworks volumes. Digital surveys should be commissioned for all schemes when this is possible and the output data specified to be compatible with normal Computer Aided Design (CAD) packages used by design engineers.

The topographical complexity of the site (e.g. slope variation) will determine the degree of detail of the topographical survey. The more comprehensive the survey, the less the risk of additional detail being required at a later date. Similarly, a survey of a larger area than initially envisaged as being necessary for the project can be valuable if it is subsequently necessary to change the treatment design or concept.

As a minimum, specific details included as part of the topographical survey should include:

- Levels of all rivers or streams to which treated effluent could be discharged (investigative work should also establish high water levels of these watercourses).
- Elevation of proposed inlet pipe to treatment system.
- Elevation of existing outlet point of discharge.
- Cover and invert levels of all manholes on the proposed site.
- Location of any above ground services on the site e.g. electricity pylons.
- Location and extent of any substantial stands of trees.
- Overall site dimensions.
- Pipe dimensions should also be recorded.

2.6 Site appraisal

2.6.1 Introduction

The topographical survey forms just one component (albeit a very important one) of an overall site appraisal. It is essential to carry out a thorough appraisal of the intended site prior to the commencement of construction works, and the data gathered will be essential in providing an accurate cost estimate to the client.

Table 2.4 summarises the items to be investigated as part of the site appraisal, details of which are provided in the following paragraphs. Table 2.5 identifies some more specific items that are particularly applicable to the design of a water treatment system. Much of this latter information will be gathered during the ground investigation.

Most EU countries have national 'best practice' guidelines for site investigation, published either by governmental agencies[8] or by professional engineering institutions. Although such guidance texts cover more complex investigation techniques than are generally required for small pond structure typical of many passive treatment systems, they do provide simple, concise, and officially-endorsed advice.

If the investigation work is to be carried out by a third party, it is important that a simple but adequately comprehensive contract is set up with the party carrying out the works. If the feasibility stage of the project suggests that specialist advice is required, it is important that this is brought in at a sufficiently early stage to allow the site investigation to be designed to suit any specific requirements of the adviser.

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[8] For instance in the UK, best practice in site investigation is specified in the British Standards Institution code of practice no. BS 5930: *Code of Practice for Site Investigations* (2000), and is also expounded in further detail in the Construction Industry Research and Information Association (CIRIA) *Site Investigation Manual* (Special Publication No. 25).

Table 2.4. Items to be investigated during a site appraisal

Item	Details
Information requirements	• collect and assess data about the ground conditions of the site and adjacent areas from existing records • seek information on buried and other services • establish land ownership if in doubt and confirm rights of way
Site survey	• walkover of the site to record the topography vegetation and general ground conditions • carry out level survey by specialist surveyor
Ground investigation	• Establish the soil profile and groundwater conditions beneath the site using exploratory holes. • Carry out laboratory testing if required
Assessment	• Determine whether the site is suitable. • Prove sufficient and suitable construction materials • Establish that there is sufficient information for the design

Table 2.5. Detailed investigative requirements for construction of a water treatment system.

Item	Details
Foundation	• Confirm the ability of the ground to support an embankment and prevent excessive seepage from occurring through the foundation
Construction materials	• The hydraulic conductivity of all potential fill materials should be assessed to establish the suitability for embankment construction
Stability requirements	• Ensure the pond system and land adjacent will remain stable after commissioning
Access requirements	• Ensure suitable access for construction plant and imported materials.

2.6.2 Desk studies

A significant amount of documented site information may be available to aid the compilation of an effective site investigation. This information may be available from:

1. A previous owner of the land.
2. The local governmental authority.

3. Local Library, Historical Society or museum
4. Specialist “one stop” companies which have already collated much of the in formation contained in the above sources in readily-usable computer-based formats.

In particular, any evidence of previous industrial use of the site will suggest the types of contamination and waste materials that may form the site, prior to a contaminated land investigation (section 2.8).

Other information sources that should be reviewed as part of the study include:

- Topographical maps and plans.
- Historical Maps.
- Geological maps and published information.
- Soil survey maps amid published information.
- Mining and mineral extraction records.
- Aerial photographs.
- Previous ground investigations in the area.
- Local utilities

Research should also be undertaken into sources of all past published maps of the area to allow previous land use to he assessed.

2.6.3 Walkover Site Survey

A walkover of the site should be made at an early stage during the course of the information study to identify and record features of geological and topographical interest. Information on groundwater levels, water features and drainage (hydrological information) for the site should also be acquired. Data collected from a site survey can then be used to supplement and clarify data collected in the information study. The details should ideally he noted on a large-scale plan at 1:200 or larger.

The proposed site should be inspected carefully and methodically for conditions that might cause construction difficulties. Topography is important. Slopes greater than 1 in 10 may be subject to soil creep as indicated, for example, by tilting walls and trees. Abrupt changes in local topography may indicate changes in ground type and a consequent variation in soil characteristics across the site. In particular, the extent of flat areas in the bottom of a valley should be identified, as these may delineate the extent of softer or weaker material infilling a valley floor.

Vegetation is an important indicator of soil types and groundwater levels. Reeds, rushes and willows indicate a shallow water table, whereas bracken and gorse usually denote a well-drained soil with a low water table. Gorse is often preferentially associated with land disturbed by human activities within the last few decades. Abrupt changes in vegetation may indicate important variations in ground or groundwater conditions. The extent and type of vegetation across the site should be recorded and linked, where possible, with topography amongst other features.

Table 2.6. Information to be gathered during a walkover survey of a proposed passive remediation site.

Item	Details
Topographical information	• access • present land use • evidence of former land use • signs of made ground or the deposition of material • gradients of slopes steeper than 1 in 10 • tilting trees, walls or other structures • abrupt changes in slope or topography • current and recently removed vegetation
Geological information	• soil type where visible or exposed by hand digging or exposures • evidence of slipped material • changes in vegetation • breaks in slope.
Hydrological information	• watercourses, whether running, damp or dry • seepage, springs and flowing water • areas of standing water • waterlogged and boggy ground • changes in vegetation • evidence of former flooding.

Removal of vegetation, particularly mature trees and large shrubs, may lead to groundwater level changes and ground movement. Evidence of past ground deformations due to soil moisture changes and movements may be revealed in existing buildings or structures on the site. Current and past land use, where known, should be noted. Previous land use may have resulted in realigned watercourses, areas of made ground or abandoned underground workings. Dumped material is likely to have been tipped without control and may contain voids, as well as combustible or hazardous material. Where household refuse or debris and residues from industrial processes are suspected the site must be treated with care. The presence of such material is likely to make most sites unsuitable and specialist advice must be sought.

The typical features that should be identified in a site survey, which should be carried out on foot, are listed in Table 2.6. Useful items to be carried by the surveyor are shown in Table 2.7.

Table 2.7. Useful items to be carried on a walk over survey.

Item	Comments
True scale plan	At least two photocopies of site plan to mark up with features and observations.
Spade	To dig small holes for initial appraisal of soil type.
Bags & Ties	For soil samples.
Tape measure	For approximate measurement of features of interest.
Camera	For a complete record of the site or a feature when used in conjunction with the site plan.

2.6.4 Buried and other services

The presence of services such as gas, electricity and telephone, which are privately owned or belong to statutory bodies and public authorities within the site, must be established. The exact position of each service must be clarified prior to commencement. Safe working practice beneath or adjacent to services must be adopted and advice concerning these matters should be sought from the relevant owners, statutory bodies and public authorities.

The statutory bodies, public authorities and companies responsible for river control and drainage, water, gas, electricity and telephone services should be informed of the proposed works. The national grid reference of the site and a large-scale plan marked up with the proposed working areas should be sent not less than two months before the construction is to start, to ensure a reply. A copy of the site plan would normally be returned, marked with the positions of any services. If adequate information is not obtained prior to commencement of the construction work, the relevant statutory body, public authority or company should be requested to inspect the site and be informed of the date of commencement.

Excavation of areas where services are thought to be present should he advanced with caution. Holes should be excavated by hand to a depth of approximately 1 m to allow inspection of the ground and to establish the presence of services prior to excavation continuing by mechanical means.

Where the proposed construction or subsequent use would affect a service, it should be diverted outside the site area. The route and method of diversion or the details of the reconstruction should be agreed prior to work commencing on site and this work is often best carried out prior to the main construction. In many instances, the diversion works will be carried out by the owners and a charge made accordingly.

2.7 Ground investigation

2.7.1 Introduction

The ground investigation comprises the work carried out to gain detailed information below the ground surface and involves the excavation of exploratory holes possibly followed by laboratory testing to assess the soil parameters. This work should involve a specialist ground investigation contractor, particularly if boreholes or laboratory testing is to be carried out. An initial ground investigation is generally carried out by excavating trial holes using a tractor backhoe, which is ideal for this purpose. A soil auger may also be used if available, whilst boreholes will normally be necessary if information is required below 4 m to 5 m depth or if groundwater conditions prevent the excavation of open trial holes.

Exploratory holes should be located across the whole of the area that will form the foundation for the treatment system. In addition, exploratory holes should also be sunk in the location of potential borrow pit areas to assess the suitability and amount of material for use as embankment fill, and elsewhere where unstable slopes might be present.

The number of exploratory holes required will depend on the size of the site and the variability of materials across the site. For even the smallest site there should be a minimum of three initial exploratory holes positioned at suitable locations across the proposed treatment site. These initial exploratory holes should indicate the general nature and variability of the ground across the site. Additional holes should then be sunk in the centre of the site.

The exploratory holes should extend through the superficial material and not less than 1.5 m into the underlying in-situ ground. The superficial materials comprise the generally weaker and softer material. It also includes alluvial clay and gravel, boulder clay and granular deposits, material moved down-slope as a result of natural processes, slipped material, highly weathered ground and any made ground.

2.7.2 Trial Holes

Trial holes allow a good visual assessment to be made of the soil profile and groundwater seepages. Soil and water samples should be taken for examination and possible laboratory testing and their depths should be noted. The sides of trial holes and other excavations may be liable to collapse if unsupported or inadequately supported and a careful approach must be adopted. Colour photographs of the excavated faces should be taken wherever possible with a prominent scale and method of identifying the photographed face. The position of the trial holes should be marked on the site plan used for the site survey. Trial holes can be excavated to a maximum depth of up to 4 metres depending on the ground conditions, the position of the water table and the size of excavator used. On completion, trial holes should be backfilled carefully with the excavated material. This is particularly important to ensure that the ground is returned to as near to its original condition as possible.

2.7.3 Soil Augers

Augers are used in cohesive soils (clays) and are unlikely to be successful in granular soils (sands and gravels). Advancing the auger may prove difficult in stony or very stiff soil, when the water table is reached, and in frozen ground. Detailed soil descriptions should be made from material removed from the auger, with soil samples taken for closer examination and laboratory testing. The augured material should be laid out in sequence on the ground and photographed to give the soil profile. The depth of water seepages may be estimated, but little information on groundwater levels is usually available with this method of investigation. Auger holes can be taken to a maximum depth of up to 6 m, depending on the ground conditions and the position of the water table.

2.7.4 Boreholes

Boreholes are more expensive than simple trial holes, but their use may be required by site conditions. Boreholes allow the extent of exploration to be continued to a substantial depth (≤ 50m) if required, and also enable exploration of very soft, loose and/or waterlogged ground that cannot be easily investigated using trial holes. The boreholes are normally sunk by specialist drilling contractors using tripod-mounted percussive rigs. Far better resolution of site geology, including accurate localisation of contaminated materials, can be achieved using the very latest 'sonic' drilling technologies (see www.sonicbore.com). Where necessary, boreholes can even be sunk through the beds of surface waterbodies, by means of mounting the drilling rig on a platform above the water level. During the process of drilling, additional information can be obtained on the *in situ* geotechnical properties of the ground. This is most commonly achieved by means of Standard Penetration Tests (the conduct of which is typically specified in national technical guideline documents[9]), which provide information on the *in situ* strength and relative density of the soil.

On completion of a borehole, it is often advisable to install support for the borehole walls in the form of 'casing' (i.e. a solid pipe), with slotted pipe ('screen') below the water table. Alternatively, piezometer tips can be installed below the water table, with narrower-diameter tubes connecting these back to surface. With such installations, groundwater levels can be monitored over time, providing valuable insights into seasonal and diurnal[10] variations in the groundwater level. It is also possible to conduct *in-situ* hydraulic conductivity tests in such boreholes / piezometers (see section 2.7.6).

On sites where the treatment scheme is to be constructed in areas of man made fills these piezometers can be combined with slotted gas monitoring standpipes. It is

[9] For instance, in the UK these are described in British Standards document BS1377 (1990)
[10] These can be especially important in coastal areas, beside tidal rivers, or in areas where intermittent groundwater pumping is taking place.

important to monitor gas levels within the soil if any enclosed structures (e.g. pump houses) are to be constructed on the site where gas may accumulate. As many of these treatment sites are located on former mine sites these are likely site for gaseous emissions and therefore require specific monitoring. Monitoring should be carried out for the levels of oxygen, carbon dioxide, methane and gas flow present within the wells. Depending on the location of the site there may be the requirement to monitor for carbon monoxide and hydrogen sulphide. On all monitoring occasions the barometric pressure and trend should be monitored and the majority of the monitoring visits should be made on a falling barometer. The groundwater and gas levels within the piezometers should be monitored over as long a period as possible in order that the variation in site conditions can be fully ascertained.

For sites that may have be undermined by shallow mining activity it is recommended that additional boreholes are drilled. These boreholes will probably have to be drilled using rotary methods. If a simple stratigraphic succession is required and the absence of shallow mineworkings proven then these boreholes can be drilled by rotary percussive methods and the chippings logged at surface by the supervising geotechnical engineer. However it may be a requirement to core the rock to provide continuous samples for description. This is especially important where excavations for any structure associated with the site may extend into solid geological strata and therefore an assessment of the suitability of the excavation of the rock is required.

Piezometers can also be installed in these rotary boreholes to monitor long term ground water levels that may be causing gas expulsion from mining systems or rises in groundwater levels.

2.7.5 Information required from borehole logging

Information recorded from the exploratory boreholes should include:
- Thickness of each layer encountered (including topsoil and subsoil).
- Type, classification and variability of material present in each layer.
- Compactness / strength and hydraulic conductivity of the material present in each layer.
- Position of the water table and any seepage, giving the relative rate of inflow.

A description of the material in each layer should be made on the basis of visual observations and the response to simple field tests. Soil description systems are noted in many national standards and it is recommended that any soils examined during the site investigation are described by an experienced geotechnical engineer to an appropriate national standard (e.g. the UK standard BS 5930, 2000).

In clayey soils the presence of any planar or undulating polished surfaces, identified from trial holes, should be clearly noted. These may be of a differing colour and a lower strength than the adjacent ground. If imported fill is to be used, samples should be taken from the source, with appropriate and sufficient testing and visual assessment carried out. This should be sufficient to assess that the material is free from any contaminants and that the type, classification and variability of the material and the compactness/strength and hydraulic conductivity when used as fill.

2.7.6 Hydraulic conductivity assessment

'Hydraulic conductivity' is the parameter which describes the permeability[11] of earth materials with respect to fresh water. Assessment of the hydraulic conductivity characteristics of a site or a particular soil with any degree of accuracy and confidence is not straightforward and should only be entrusted to a suitably experienced hydrogeological / geotechnical expert.

A number of field tests can be carried out to give an indication of the hydraulic conductivity of materials in the field. Where the water table is encountered, borehole tests are most appropriate. In reasonably permeable ground, it should be possible to test-pump a borehole for a number of hours (4 hours being a realistic minimum) and monitor the drawdown in water level within the well over time. Monitoring the recovery of water levels after the cessation of pumping yields a second set of comparable data. Analysis of drawdown and recovery data can directly yield estimates of 'transmissivity', which can be considered to be the product of the hydraulic conductivity and the saturated thickness of the ground. Provided the latter is known from drilling records, hydraulic conductivity can be readily calculated. In cases where pumping is not an option, either for lack of suitable equipment or because the well quickly runs dry, two further options for testing hydraulic conductivity. In the first of these, the so-called "slug test", a known volume of water is suddenly and rapidly tipped into the borehole, and the subsequent decline in water levels back to the initial level is monitored. Alternatively, a bailer (i.e. a long, narrow, bucket-like tube which fills from its base via a simple non-return 'clack' valve) can be inserted into the borehole, in which it will fill with water as it descends. It is then rapidly withdrawn and the recovery in water levels is monitored. This is known as a "bail test". A useful variant on slug and bail tests is to use as the 'slug' a solid rod (usually of stainless steel) which is only slightly narrower than the borehole. When the rod is inserted, water levels rise and their decline can be monitored. When the rod is withdrawn, the recovery of water levels can be monitored. The interpretation of pumping, slug and bail tests is discussed in detail in many hydrogeology text-books, with the most comprehensive coverage being offered by Kruseman and de Ridder (1990).

Testing for hydraulic conductivity in unsaturated soils (i.e. those above the water table) is far more problematic than testing below the water table. In many cases the least expensive option will be to collect samples and submit them for lab-based testing. Rigorous *in-situ* testing of unsaturated zone hydraulic conductivity is possible, using instruments such as Guelph permeameters, but their use demands high levels of scientific training and skill, and is also very time-consuming; hence such instruments are seldom used outside of research investigations. Simpler tests of unsaturated zone

[11] 'permeability' is sometimes used, inaccurately, as a synonym for hydraulic conductivity. However, permeability is an intrinsic property of the material in question, and is independent of fluid properties. 'Hydraulic conductivity' is derived mathematically from permeability by including the properties of fresh water in the overall expression for the ease of water movement through a porous medium.

hydraulic conductivity are often implemented, for instance filling trial holes with water and recording the time taken for this water to soak away. These tests can give a vivid impression of relative permeability of different zones, but because of the complex hydraulics of wetting and infiltration they seldom yield defensible quantitative values for hydraulic conductivity. In respect of material to be used in a construction, it is unlikely that the condition of materials found on site will replicate the properties they will have in future after excavation, emplacement and compaction; consequently, results from *in-situ* testing of possible fill materials must be treated with caution.

Whenever possible, the final assessment of the hydraulic conductivity should be based on both *in-situ* tests (especially on foundation materials) and laboratory tests (especially on fill materials). Soil from at least three locations should be tested from exploratory holes spaced evenly about the potential site area, with two samples at varying depths selected at each location.

Acceptance criteria for use of the soils as fill for an impermeable layer or barrier are:
1. A hydraulic conductivity not greater than 10^{-9} m/s
2. A minimum clay content of 10%, although a value of 20% to 30% is desirable.
3. Adequate compaction characteristics.

Table 2.8 shows the hydraulic conductivity ranges for various soil types and gives an indication of the meaning of what attaining a hydraulic conductivity of 10^{-9} m/s implies in practice.

Table 2.8. Hydraulic conductivity of various soil types, and description of characteristics.

Soil type	Typical hydraulic conductivity (m/s)	Hydraulic conductivity / drainage characteristics
Clay	Less than 10^{-8}	Practically impermeable
Silt	10^{-8} to 10^{-6}	Low permeability Poor drainage
Sand	10^{-6} to 10^{-3}	Medium/high permeability Moderately free draining
Gravel and cobbles	Greater than 10^{-3}	High permeability Free draining
Peats / organic-rich soils	Highly variable, with values $\leq 10^{-4}$ in shallow layer rich in plant-debris (the 'acrotelm'), but with values $< 10^{-8}$ in deeper layers of humified material (the 'catotelm')	Except where artificially drained, generate rapid surface runoff above saturated catotelm

Where clay is to be imported to the site to form the impermeable layer, sufficient tests must be carried out on the material to assess the clay content and the hydraulic conductivity when re-compacted as fill. The amount of testing will be dependent on the quality control arrangement for the source, but a minimum of five sampling locations should be selected.

2.7.7. Summary of ground conditions

The data collected by the information study and site survey should be used in conjunction with the results of the ground investigation to assess the ground conditions at the site. The results of the ground investigation should be provided in a factual report. If an interpretation of the site investigation data has been made by a suitably qualified engineer then this separate report should be included in the design documentation.

Subsurface profiles of the ground should he prepared from the information and thickness of the various layers in the exploratory holes. Additional subsurface profiles will be necessary for more complex sites. Sections across any proposed borrow area will also be required to assess the availability of materials and constraints on the area. It will allow the variation of the ground across the site to be assessed and, together with the properties of each soil layer, will provide the basic information to assess the suitability of the site for construction.

2.8. Contaminated land assessment

2.8.1. Introduction

The possibility that ground proposed for use for ponds, wetlands and other water treatment features might be contaminated must be considered. Contamination might have resulted from activities associated with mining, for example spoil disposal or spillage of oils, or may be related to other current or historical land uses. Historically, a very wide range of industrial processes were deliberately located close to mines, to minimise transport costs for raw materials. A great variety of potentially contaminating historical activities are thus characteristic of many present and former mining districts.

Contamination of the ground may pose risks to various receptors, including:

- health hazards to personnel carrying out excavations and other works as a result of skin contact, ingestion or inhalation of harmful materials. Disturbance of materials might also create off-site risk to occupiers of nearby land, for example by the escape of airborne dust.
- potential for damage to building materials, for example due to the presence of soluble sulphates which may attack cement-based building materials, or organic contaminants with the potential to damage plastics or permeate potable water supply pipework.
- ground gases that may be explosive, flammable, toxic or asphyxiating, and may accumulate in any buildings constructed.
- soluble contaminants which may leach into the water within the treatment system, causing further pollution of the water and resulting in additional discharge

constraints. Excavation may also increase the risk of leaching causing pollution of other surface and ground waters.

- adverse effects on plant growth, including landscape planting and plants used as a part of the treatment system.

For these reasons, assessment of ground contamination should form a part of the ground investigation programme. The desk study (section 2.6.2) should provide background information on present and former land uses both at the proposed excavation site and in the vicinity. In this section the further investigatory steps needed to establish the contamination status of land ear-marked for passive remediation uses.

2.8.2 Sampling

Soil and groundwater samples should be obtained from trial pit or borehole excavations (see sections 2.7.4 - 2.7.6). Consideration must be given to the likely nature of contamination, based on the desk study, and appropriate sampling techniques used. For example, volatile contaminants such as organic solvents or light petroleum hydrocarbons will require the use of well-sealed sample containers (usually of glass) and cooled storage, to prevent loss of the contaminant in transit to the laboratory. Guidance should be sought from the testing laboratory on the selection of appropriate containers, or requirements for chemical preservatives in the case of unstable contaminants. Sufficient samples should be obtained for testing to adequately represent the materials which may be exposed during subsequent excavations, or be available as contamination sources by leaching. Groundwater samples should also be obtained and suitably preserved for transport.

2.8.3 Chemical testing requirements

The analytical suite required must be suitable for the site-specific circumstances, and should reflect the contaminants possibly associated with the former uses of the site. Commonly required analytes are shown in Table 2.9.

Table 2.9. Common substances to be analysed for as part of a contaminated land assessment.

Contaminant	**Typical sources**
Metals (including arsenic, cadmium, copper, chromium, lead, mercury, nickel, zinc)	Mine spoil, ash, metallurgical processing wastes, other industrial waste disposal, natural mineralogy.
Sulphates (particularly water-soluble)	Mine spoil, ash, demolition waste, peat, natural mineralogy
Hydrocarbons and other organic compounds (analysis for total hydrocarbons, specific hydrocarbon groups such as diesel range, and various classes of organics as appropriate)	Fuel and lubricating oil spillage, coal carbonisation, other chemical storage / manufacture / disposal.
Asbestos	Building waste

The method of analysis should be chosen to reflect the potential exposure pathways of concern. Assessment of human health risks by ingestion / inhalation is typically carried out based on the determination of 'total' concentrations. Potential for sulphate attack of concrete is typically based on a 2:1 water-soil extract. Where there is potential for harm due to leaching of contaminants into waters being treated, or into groundwater or surface water, leachability testing may be required in addition. If potentially contaminated material is expected to be removed from site to be landfilled elsewhere, some landfill operators will also require leachability data to support risk assessment of the potential for contaminant migration.

2.8.4 Assessment

Contaminant concentration data should be appraised on a risk assessment basis, consistent with the approach of the relevant regulatory authority. In most EU countries, statutory regimes for the assessment of contaminated land are now in force which are based on the occurrence or likelihood of 'significant harm' or pollution of controlled waters. In most regimes of this type, land may only be determined as statutory contaminated land if harm to specified receptors is demonstrated. Other receptors may be relevant to assessment of ground proposed for development as a water treatment facility, as follows:

Health: Assessment of human health risk may be based on a simplistic comparison with national generic standards[12]. However, this level of assessment may be misleading due to the presence or absence of relevant exposure pathways in site specific circumstances. As an example, the UK 'threshold trigger concentration' for arsenic in 'open spaces' was set at 40mg/kg. This concentration is frequently exceeded, sometimes by a substantial margin, in colliery spoil. However, unless the material in question is exposed at the surface and a pathway exists by which regular ingestion of soil might take place by a receptor group, then no harm is likely to occur. Remedial action would then be unnecessary even where the buried spoil exhibits concentrations considerably in excess of the generic guideline level.

A recent development in assessing human health risks associated with contaminated land has been made with the release of the CLEA (Contaminated Land Exposure Assessment) model by the UK regulatory authorities (DEFRA, 2002). The CLEA model may prove to have some application in the assessment of health risk at potential mine water treatment sites, though at present the range of contaminants available for assessment using the CLEA software is limited, as are the available land use scenarios. The US Risk-Based Corrective Action (RBCA) approach is similar, and software is commercially available which is much more flexible than the CLEA model. The assessment at any specific site should consider the most appropriate means of health risk assessment, taking into account site-specific factors including realistic appraisal of available pathways and receptor exposure.

[12] for example the ICRCL guidelines in the UK (ICRCL, 1987).

Building Materials: Risks to building materials are excluded from the CLEA and RBCA models. However, guidance is available from various published sources (e.g. BRE, 1994; 2001). Given the very high concentrations of sulphate associated with mine waters, it is most important when planning mine water remediation systems to assess the potential for sulphate attack on concrete (BRE 2001).

Gases: At sites where the construction of buildings is planned, particularly where the building or any services trench or other void may be poorly ventilated, monitoring of gas emissions from the ground should be undertaken. The gases of greatest significance are methane (flammable / explosive), carbon dioxide and carbon monoxide (toxic / asphyxiant) and radon (carcinogenic). All of these gases are at their most dangerous where they can accumulate in confined spaces with poor ventilation, such as basements or storage sheds which are only rarely entered. Methane emissions may result from the biodegradation of organic materials (e.g. as landfill gas) or from shallow coal mine workings (often passing from the workings via drained, permeable sandstones within the Coal Measures sequence). Radon is principally (though not exclusively) associated with granitic and metamorphic rocks, certain mudstones and metalliferous mine workings accessing various types of mineral vein. CO_2 (the accumulation of which can lead to the development of oxygen-deficient air) can emanate from natural coal- or limestone-bearing strata, and is especially prevalent in unventilated, abandoned coal workings. Carbon monoxide, and a range of other potentially hazardous gases such as sulphur dioxide, ammonia and various smoke-related organic pollutants (including polycyclic aromatic hydrocarbons (PAHs)) are associated with underground fires in old mine workings or in bodies of coal-rich spoil. Other gases or vapours may be relevant at certain sites. Monitoring should be carried out over an extended period, preferably under variable atmospheric pressure conditions, and should include gas flow rates in addition to gas concentrations. Assessment of the need for gas protection measures should then be undertaken.

Water Environment: The assessment of risk to the water environment (surface and groundwaters), and of pollution of water in the treatment system may be based on soil leachability testing and / or groundwater analysis. A tiered approach is commonly adopted in the assessment of results, the upper tiers taking into account progressively more factors regarding pollutant dilution / dispersion / degradation prior to impact on a water body of concern. The lowest tier may compare the soil leachability data with standards relevant to a specific receptor, such as Environmental Quality Standards (EQS) for surface watercourses, or drinking water supply regulations in cases where the potential target is a river or aquifer used for potable supply. If the leachability result (expressed as mg/L in a water extract, commonly 10 parts water to 1 part soil) or groundwater concentration exceed these criteria, then consideration might be given to more detailed site-specific factors. A cautious, pragmatic, approach to the assessment of potential for pollution of waters being treated would be to exclude any soils whose leachability results exceed the Water Supply or EQS criteria from the lining material within ponds etc. Any soils contaminated by oils or similar should also be excluded.

Plants: It is important to bear in mind that the absence of plants, or the presence of only stunted unhealthy plants, are not in themselves conclusive evidence that the soil is toxic with respect to plants (i.e. 'phytotoxic'): they may equally indicate a lack of plant nutrients in otherwise clean soil, and / or effective 'drought' conditions where the soil is so coarse-grained that all rainfall landing on the soil drains quickly below the root zone. Where neither of these conditions obtains, then phytotoxicity may well be the explanation for a lack of healthy plants. It is important to bear in mind that phytotoxic effects are not solely associated with certain 'toxic metals' but also with other factors, such as low soil pH and high salinity. Of the contaminants commonly encountered on mine sites, copper, nickel, zinc and boron most frequently give rise to phytotoxic effects. In terms of total concentrations in soils, the following 'threshold' values are widely used to identify situations in which site-specific investigations of phytotoxicity are warranted:

Copper	130mg/kg }	
Nickel	70mg/kg }	ICRCL threshold trigger concentrations
Zinc	300mg/kg }	
Boron	3mg/kg (water soluble)	
pH	< 5.0	
Salinity	> 3000 µS/cm	(Rule of thumb criterion for electrical conductivity of saturated calcium sulphate extract, UK MAFF method).

In reality, the phytotoxicity of most contaminants is much more likely to be related to the concentration of water-soluble fractions than to total concentrations. Specific plant-availability test methods have been devised, such as analysing for EDTA-extractable materials, which aim to more accurately reflect potential phytotoxicity risks than do simple analyses of the total concentrations of contaminants in bulk samples of soil.

Further discussion on the conditioning of mine site soils to favour plant growth follows in Section 6.5.

3. PASSIVE TREATMENT OPTIONS FOR ACIDITY, IRON AND ALUMINIUM REMOVAL: PRINCIPLES OF UNIT PROCESSES

3.1 Introduction

The discussion in this section is limited to technologies applicable to ameliorating acidity in mine water discharges, and the removal of two of the most common metal contaminants found in mine drainage, i.e. iron and aluminium. The removal of manganese (typically as common as iron and aluminium in many mine waters), zinc, arsenic, cyanide and sulphate is dealt with elsewhere in this text (section 5), since removal of these contaminants from mine waters is more difficult, and many of the technologies are still at the development stage. (This is not to say that some of the technologies below will not prove appropriate for their remediation in many cases).

These engineering guidelines are devoted to the design and construction of passive treatment systems. However, when considering the options for mine water treatment at the outset of a project, the use of chemicals and energy (active treatment) should not be disregarded out of hand. Some mine waters are of such poor quality and high volume that passive treatment alone will not attain the required improvement in qater quality. There are also some other cases where consideration should be given to a partially active scheme. For example:

- It may be worth considering the use of alkali chemicals on a drip-feed basis for marginally net-acidic waters. It is shown below that passive treatment units for acidic discharges are not without their problems. On restricted sites there may not even be sufficient land available to construct the passive units. Where compsot wetlands or RAPS would prove difficult to maintain (e.g. in cold or arid climate zones), the cost of supplying (weekly or perhaps even monthly) and maintaining a simple alkali dosing facility may be more cost-effective.
- In many situations it is simply not possible to design a mine water treatment system that is entirely fed by gravity. In many cases, therefore, pumping may be employed to lift water to the head of the treatment system, from where it can flow through the treatment units under gravity.
- Aeration units require significant hydraulic head. In some cases drip feeding the mine water with an oxidant, such as hydrogen peroxide, may be more cost-effective than pumping water to a sufficient height to facilitate cascade aeration, especially where there would be no other reason to pump the mine water.

In other words, laudable as passive treatment may be on ‘environmental’ grounds, it will not always be the correct treatment solution, at least not in isolation.

3.2. Passive technology options

3.2.1 Introduction

Before it is possible to select appropriate passive technologies it is necessary to first understand what types of technologies are available, and what their respective functions are in terms of remediation of mine waters. This section provides an overview of passive technologies, while details of sizing criteria etc are reserved for section 4.

3.2.2 Aeration units

For mine waters that initially have low dissolved oxygen concentrations, aeration is vital where the intention is to remove iron as a ferric (Fe^{3+}) precipitate. The reason for this is that upon emergence to the surface environment iron is often present in the reduced, ferrous (Fe^{2+}) form. Direct precipitation of ferrous iron as a hydroxide requires a higher pH (approximately 8.5) than that of ferric iron (approximately 7.0). Attaining pH values of 8.5 in a passive system is difficult, whereas reaching pH 7 is a realistic objective. Therefore it is necessary to oxidise the ferrous iron to ferric iron. Dissolved oxygen must be present in the water column to facilitate this oxidation, and aeration is the simplest means of accomplishing this.

It should be noted that the oxidation of ferrous iron is not only dependent on the presence of oxygen. It depends also on the concentration of ferrous iron present and on the pH, as shown by the following expression (valid within the pH range 4 to 8; Singer and Stumm 1970):

$$\frac{d\,Fe^{2+}}{dt} = K\,\frac{[Fe^{2+}]\,[O_2]}{[H^+]^2}$$

pH is, of course, represented in this concentration by the term for protons, H^+. The raising of $[H^+]$ to the power of 2 indicates that the reaction is second order with respect to H^+. In other words, the reaction rate is very sensitive to changes in pH. For this reason, raising pH and generating alkalinity is crucially important when dealing with acidic discharges. Methods of generating alkalinity in passive systems are discussed below.

For some strongly alkaline mine waters aeration may also serve to release CO_2, which leads to an increase in pH, further increasing the rate of ferrous iron oxidation.

In passive systems, the most common means of aeration is some form of cascade. Typically these are arranged as a series of steps, down which mine water is allowed to cascade. Despite their apparent simplicity, the design of aeration cascades is still a matter of debate (Younger *et al.* 2002). One widespread school of thought that the cascades are best designed to break the flow into thin films and droplets so that a greater water surface area is available, and therefore the potential for oxygen transfer from air to water is maximised. To attain this aim, a simple flight of steps will be adequate. An alternative formulation with a firmer theoretical basis, backed up by a large body of experimental and computational work (Novak 1994), is that efficient mass-transfer of oxygen from the gaseous to aqueous form is best achieved by ensuring that jets of falling water can dissipate thoroughly within a standing water column below.

It is recommended that aeration cascades for mine water treatment be constructed from sulphate resistant concrete. While such concrete is more expensive than some potential alternatives, it has the advantages of (a) a considerably longer life-time than most alternative materials and (b) being relatively easy to clean.

An alternative to simple aeration cascades is in-line venturi aeration. A small pipe, open to the atmosphere at one end, is submerged in a pipe flowing at full bore. The submerged end of the pipe faces downstream. As mine water flows down the pipe air is entrained through the end of the pipe open to the atmosphere, as illustrated in Figure 3.1. Up to 900 mg/L ferrous iron may be oxidised in a single aeration step via in-line venturi aeration, making this a particularly attractive proposition for mine waters with high iron concentrations. One potential drawback of such systems is that to work most effectively the mine water pipe requires a minimum operating head of about 15 m. Further information about the principles and application of such systems can be found in Ackman and Place (1987), Ackman and Kleinmann (1993) and Ackman (2000).

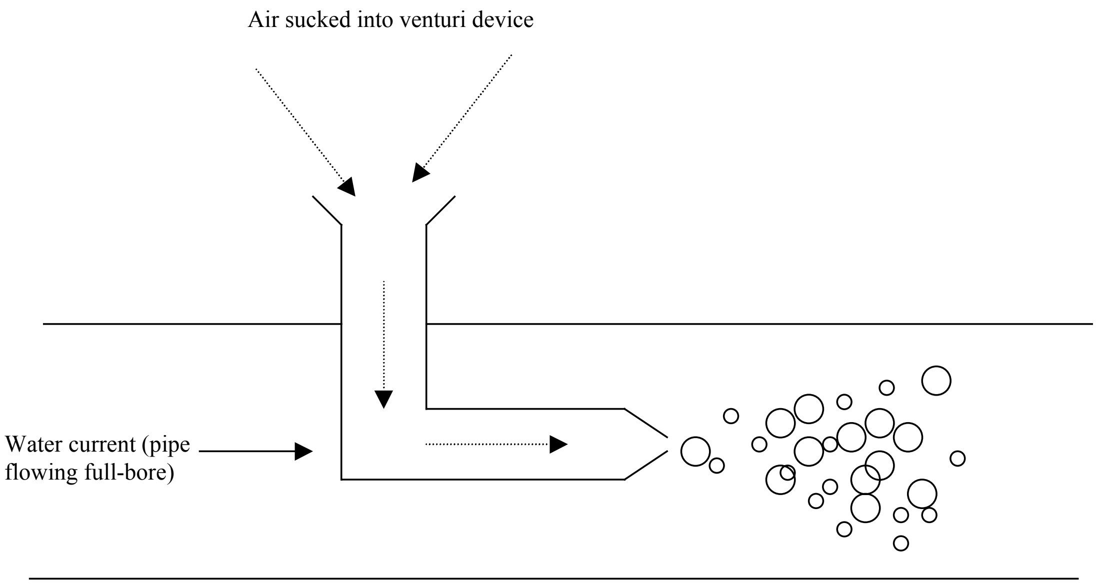


Figure 3.1. Sketch of in-line venturi aeration device (after Younger *et al.*, 2002)

For a truly passive system (i.e. no energy or chemicals), the topography of a site must be such that mechanical aeration (by cascade or venturi) can be driven by gravity. Where this is not the case a number of alternatives ought to be considered, including:

- violating the strict 'passive' rule by pumping the water high enough to facilitate cascade aeration / venturi deployment
- relying on the considerable aeration afforded by radial oxygen leakage from plant roots in aerobic wetlands (see Batty and Younger 2002), which has been found to be sufficient to raise mine waters oxidation status from wholly to oxygen-saturated within a few hours' retention time
- use of mechanical agitators (which could conceivably be driven by wind power) to aid aeration
- use of chemical dosing, using hydrogen peroxide or some other oxidant (see Younger *et al.* 20002 for further discussion)

3.2.3. Settlement lagoons

Although less attractive as landscape features than wetlands, it is often desirable to include settlement lagoons ponds as the first step in the sequence of unit processes which comprise a complete passive treatment system. This is simply because it is far easier to routinely remove sediments from a settlement lagoon than from a vegetated wetland. Hence trapping virtually all inert sediments and a large proportion of the iron hydroxide precipitates in settlement lagoons makes the long-term maintenance of a passive system much easier than would be the case if only wetlands were used. In general, if the total iron content of a mine water exceeds 5 mg/l, consideration should be given to using a settlement lagoon upstream of the first wetland in the treatment system. In Europe's largest programme of passive treatment (that of the UK's Coal Authority) settlement lagoons are routinely deployed with the intention of removing at least 50% (and, where possible, >70%) as the first step in passive treatment.

If the lagoons are to be effective, the water which they receive must be net-alkaline. Also, if the initial iron concentration in the mine water exceeds 50 mg/L, and aeration cascades are used for oxygenation, a series of cascades and settlement lagoons will be required (one cascade-lagoon pair for every 50 mg/L Fe in raw water). If this level of provision is not made, not all of the ferrous iron will be converted to the ferric form, and it is likely that ferrous iron will be carried over in the effluent from the settlement lagoons.

High concentrations of suspended inert solids are not often a problem in discharges from long-flooded underground workings, but they can be a major problem at active mine sites (due to the inevitable disturbance of ground inherent in mining operations). Lagoons for the removal of inert solids are typically smaller than those required for the removal of iron precipitates, because the inert sediment particles are typically larger and denser (and therefore settle more rapidly) than ferric hydroxide flocs.

3.2.4. Aerobic wetlands

Aerobic wetlands are amongst the most popular passive unit processes, for they are relatively simple to design and build, and often develop a very attractive appearance (e.g. Figure 3.2). Indeed aerobic wetlands usually provide substantial amenity value and wildlife habitat, which can be particularly advantageous in sensitive locations.

Figure 3.2. Ferruginous mine water entering a recently-commissioned aerobic wetland system at St Helen Auckland[13], County Durham, UK.

As with settlement lagoons, aerobic wetlands are primarily suited to the treatment of net-alkaline, ferruginous waters. While there *are* examples of successful removal of iron from acidic mine waters using aerobic wetlands, removal rates under these circumstances are very low (thus requiring very large wetland areas), pH usually drops across the wetland, and the plants are seldom very vigorous.

It is best if the water entering such a wetland has been pre-aerated, although this is not absolutely essential, since oxygenation will occur during flow through the wetland (by direct exchange across the air / water interface, and by means of radial oxygen loss from the roots of wetland plants; Batty and Younger 2002).

Aerobic wetlands remove problematic metals from mine waters by the allied action of a number of processes, including sedimentation of suspended flocs, filtration of flocs by stems of plants, adsorption of aqueous metal species, precipitation of hydroxides on plant stems and the wetland sediment surface, and direct plant uptake of iron and other metals (which are retained primarily in the plant roots). While the latter process had previously been discounted as a quantitatively important sink for metals (e.g. Hedin *et al.* 1994a), at least in terms of the overall metals budgets of aerobic wetland systems, recent work undertaken during the PIRAMID project has demonstrated that direct plant uptake is crucial in 'polishing' the last few mg/L of iron in a mine water in order to achieve low residual iron concentrations (Batty and Younger 2002).

[13] This is one of the component systems of the CoSTaR research facility mentioned in the Foreword

Although the principal focus in most wetlands is the removal of iron, other metals are subject to attenuation in aerobic wetlands, either by direct precipitation as (hydr)oxide or carbonate phases or by co-precipitation with iron in ferric hydroxides.

While most European applications of aerobic wetlands have been for mine waters pre-treated (using settlement lagoons) so that they enter the wetlands with modest iron concentrations (typically around 5 mg/l), it should be borne in mind that there are numerous examples of aerobic wetlands (particularly in the USA) in which good plant growth and efficient water treatment is occurring despite receiving waters with iron concentrations in excess of 50 mg/L (e.g. Watzlaf *et al.* 2003). Common reed species, such as *Typha latifolia* and *Phragmites australis*, adapt well to such conditions. However, there are other reasons why subjecting wetlands to such high iron loads may not be advisable. Specifically, rapid accumulation of ochre, particularly towards the influent end of the wetland, may cause channelling of water through the system (reducing efficiency) and will certainly reduce the life of the wetland. Clearly removal of ochre from wetlands is difficult because of the presence of the reeds.

In some cases it has proved possible to combine a settlement lagoon and aerobic wetland in a single basin. At the influent end of such systems water depth may be > 2 m, and there are no reeds. However, where the water shallows to less than 0.5 m, reed colonisation will commence. Thus, ochre accumulates predominantly in the proximal, deep part of the basin, with 'polishing' treatment afforded by the reed-dominated section at the distal end of the pond.

3.2.5 SCOOFI reactors

Surface-Catalysed Oxidation Of Ferrous Iron (SCOOFI) reactors are described in detail by Jarvis and Younger (2001), Younger (2000), and Younger *et al.* (2002). The basic idea is that ochre will accrete to high surface area media when oxygenated mine water is passed over it. Ferrous iron is adsorbed to this ochre layer, and oxidises in situ. This in situ oxidation of ferrous iron is more rapid than open water oxidation (such as occurs in an aerobic wetland), and therefore iron removal can proceed very rapidly.

As with settlement lagoons and aerobic wetlands, SCOOFI reactors are appropriate for net-alkaline waters where the key objective is the removal of iron. Two types of reactor have been investigated:

- saturated flow SCOOFI reactors (Younger, 2000) and
- unsaturated flow SCOOFI reactors (Jarvis and Younger, 2001)

Saturated flow SCOOFI reactors provide more intimate contact between water and media, but are only suitable where the mine water is already well oxygenated, and for waters with iron concentration less than 50 mg/L. This type of reactor achieves higher rates of removal of iron (because of the intimate contact between water and media) than unsaturated reactors. Also, they may be aligned horizontally, which means that the hydraulic head requirements are less onerous than for unsaturated-flow SCOOFI reactors.

Unsaturated flow SCOOFI reactors do not need to be preceded by an aeration facility, since by their nature they will oxygenate water. However, such reactors need to be vertically aligned, and the removal rate of iron is not so great as in saturated flow reactors. Nevertheless, unsaturated flow SCOOFI reactors have proved successful in lowering iron concentrations from 5 mg/L to less than 0.5 mg/l (Jarvis and Younger 2001), and have also been shown (at pilot-scale) to remove approximately 50% of manganese at some sites.

3.2.6. Anoxic Limestone Drains (ALDs) and Oxic Limestone drains (OLDs)

Both ALDs and OLDs utilise the dissolution of calcite ($CaCO_3$) to raise pH, neutralise acidity, and generate bicarbonate alkalinity. Limestone is found wide application in passive treatment systems for these purposes. It has the advantages of being low cost, non-hazardous, and generally widely available in mining areas. On the negative side, it's rate of dissolution is slow when compared to the alkalis used in chemical treatment, and it has a tendency to 'armour' with ochre if it's used in the wrong environmental conditions.

ALDs are simply buried trenches containing single size (typically 50-75 mm) limestone. The limestone must have a high calcium carbonate content (> 80%), and therefore carboniferous limestone is more appropriate than dolomite. The limestone is buried because conditions within the trench (as the name suggests) must be anoxic, so that all dissolved iron remains in the ferrous form, rather than converting to the ferric form, which would quickly lead to hydrolysis and armouring of the limestone with ochre precipitates, leading to a reduction in limestone dissolution rate and, ultimately, clogging of the pore space of the ALD.

The single objective with ALDs is to raise pH / generate alkaline conditions, thus pre-conditioning the mine water so that iron will readily form a precipitate in subsequent aerobic passive units (e.g. settlement lagoons, aerobic wetlands). The great drawback with ALDs is that (to avoid the problems of ochre armouring previously mentioned) concentrations of ferric iron and aluminium must be below approximately 2 mg/L, and dissolved oxygen concentration must be less than 1 mg/L (to prevent oxidation of ferrous iron to ferric iron). Because these conditions are very rarely met with in acidic waters found in Europe (which commonly contain concentrations of ferric iron and aluminium well into double figures as mg/L), there are no known full-scale, long-term applications of ALDs in Europe to date[14]. Practitioners should be very cautious about recommending the installation of ALDs due to the problems of armouring – there are several cases known where ALDs have become blocked within 6 months of

[14] Several pilot-scale applications have been made (most notably at Wheal Jane, UK), all of which ended within 6 months due to clogging of the ALD with ochre and / or aluminium hydroxide precipitates.

installation due to elevated concentrations of ferric iron and aluminium causing clogging.

Oxic Limestone Drains (OLDs) are identical in engineering terms to ALDs but, as the name suggests, are designed to accept oxygenated waters. The principle, devised by Ziemkiewicz *et al.* (1997) and Cravotta and Trahan (1999), is that the precipitation of iron and aluminium hydroxides is encouraged within the limestone bed (through the dissolution of limestone), but velocities through the system are kept high enough (> 0.1 m/min) that the solids are kept in suspension, for subsequent settlement in a lagoon or wetland. Cravotta and Trahan (1999) show good rates of alkalinity generation (albeit not as high as ALDs) within the space of 2-3 hours residence time.

There seems little doubt that the rate of limestone dissolution is hindered by the armouring process, but that it does not cease completely is encouraging. Given this, there may be some instances where the installation of open or closed limestone drains may be the most appropriate passive treatment technology e.g. in very remote locations.

3.2.7. Compost wetlands

Compost wetlands superficially resemble aerobic wetlands for they receive surficial inflows of mine water, have shallow water depths (typically around 100 mm) and usually support dense stands of wetland plants. However, they differ from aerobic wetlands in having thick (≥ 300 mm) anoxic substrates comprising various forms of organic matter. These substrates are the loci of powerful bacterial processes (reductive precipitation of iron and sulphur etc) which serve to consume acidity / generate alkalinity and remove metal contaminants (predominantly iron) from solution. In simplistic terms, the following reactions may summarise the processes responsible for water quality improvement in compost wetlands (Hedin *et al.*, 1994a; Walton-Day, 1999):

$$2\,CH_2O + SO_4^{2-} + 2\,H^+ \rightarrow 2\,CO_2 + H_2S + 2\,H_2O$$

(reduction of sulphate to hydrogen sulphide, consuming protons i.e. acidity)

or

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

(reduction of sulphate to form H_2S, with generation of bicarbonate alkalinity)

and then

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

(reaction of metal, M^{2+}, with H_2S to form insoluble metal monosulphide)

Limestone may be mixed with the compost to further encourage generation of alkalinity. Because the limestone is mixed with the compost, it too is under anoxic conditions, and therefore armouring with metal precipitates should not be a problem.

Aluminium, which is poorly soluble above a pH of approximately 4.5, forms deposits of $Al(OH)_3$ (aluminium hydroxide) within and on tope of the compost substrates of such systems, due to the elevation of pH via sulphate reduction and limestone dissolution.

In an effective compost wetland the effluent water will have a low dissolved oxygen concentration, and therefore it is normal practice to follow a compost wetland with an aerobic wetland. This serves to oxygenate the water, and remove residual iron and aluminium (both as hydroxides).

The range of (bio)chemical reactions occurring in compost wetlands is far greater than that implied by the very brief overview given above. Indeed, from a research point of view compost wetlands are probably the most challenging in terms of establishing the exact processes occurring. Some of these processes may actually release metals from as well as attenuate them. For this reason it is very difficult to predict the performance of such systems, and Younger *et al.* (2002) wisely advise that they should be employed only when there is insufficient hydraulic head to allow the installation of a RAPS (see below).

3.2.8 Reducing and Alkalinity Producing Systems (RAPS)

RAPS were developed by Kepler and McCleary (1994) in response to the restrictive applicability of ALDs for neutralising acidic mine waters with elevated concentrations of dissolved oxygen, ferric iron and / or aluminium[15]. A RAPS is essentially an ALD overlain by a compost bed. The principle is that dissolved oxygen is stripped from the mine water in the compost bed, and Fe^{3+} is reduced to Fe^{2+}. The water then flows down through a limestone bed, where alkalinity generation occurs (Figure 3.3).

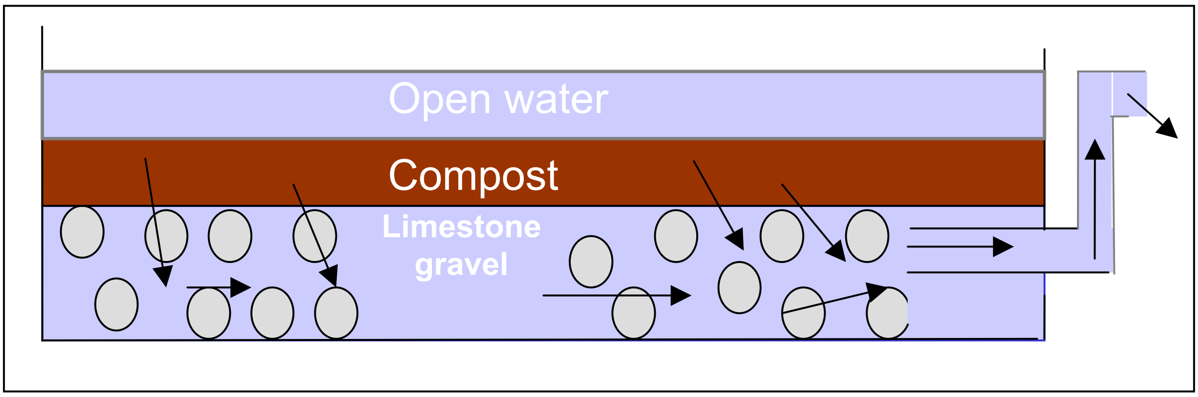


Figure 3.3. Conceptual diagram of a Reducing and Alkalinity Producing System (RAPS) (after Younger 2000).

[15] It should be noted that Kepler and McCleary (1994) originally termed these systems 'SAPS' (successive alkalinity producing systems), on account of the potential use of several successive units in series to treat very acidic waters; however, at most sites, a single RAPS is installed followed by aerobic passive unit processes. The more descriptive term 'RAPS' was therefore introduced by Watzlaf *et al.* (2000).

Thus, a RAPS can be used where acidic mine waters contain elevated concentrations of ferric iron, aluminium and dissolved oxygen. Although the redox reactions in the compost bed are equally complex to those of compost wetlands the risks are not as great, since the objective is only to generate alkaline conditions in the water. It is therefore necessary to follow a RAPS with settlement lagoon(s) and / or aerobic wetland, but RAPS are nevertheless the favoured system for treatment of acidic mine waters in the UK, at the present time.

It is conservatively assumed in designing such systems that iron will not be removed in the compost layer via the process of sulphate reduction described above, but observations of operational systems suggest that such attenuation does in fact occur, both within and upon the surface of the compost layer. Iron hydroxide accumulation above the compost layer of a large RAPS unit in South Wales, UK, is so prolific that it has stimulated research into the deliberate adaptation of the hydraulic characteristics of such systems as a possible high-intensity iron removal process (Dey *et al.* 2003).

Because a RAPS is constrained such that all of the water must contact the compost and limestone, these systems occupy as little as 20% of the land area which would be needed to achieve the same degree of treatment using a compost wetland. The principal drawback of RAPS is that there must be sufficient relief on site below the point of mine water emergence to accommodate the substantial hydraulic head losses ($\leq$ 1.5 m) associated with subsurface flow through the compost and limestone. Additionally at least 1 m freeboard is advisable above the surface of the compost. Thus a minimum site relief on the order of 2.5m below the point of mine water emergence is a pre-requisite for deployment of a RAPS unit, at least without rendering the system semi-passive by the use of pumping.

In response to the challenges of accommodating RAPS systems in low-relief settings, a recently-commissioned RAPS system in the UK (the Coal Authority's Deerplay mine water treatment scheme in Lancashire) has been designed such that compost and limestone are arranged side by side. Water flows down through the compost layer and then up through the limestone bed (Jarvis and England, 2002). While this necessitates a slightly larger land area, it reduces the hydraulic head requirement, and ensures even better efficiency through the limestone bed since water is driven upwards through it.

Even with the latter arrangement, a further problem with conventional RAPS designs in line with Figure 3.3 still remains. This relates to the compost-before-limestone flow pattern, which has the following drawbacks:

(i) the entire flow through the system is throttled by the low hydraulic conductivity of the compost layer, which is typically some orders of magnitude lower than that of the limestone gravel layer, and

(ii) from a public safety perspective, the presence of more than 0.5m of saturated organic matter as the surface layer of the RAPS substrate is a significant hazard, for it will not bear the weight of even a small child. At the very least, this makes the compost layer a potential source of distress and discomfort; for an unaccompanied child, it could prove extremely dangerous.

Drawing upon recent experiences with permeable reactive barrier substrates (Amos and Younger 2003), and on the observation in section 3.2.7 to the effect that limestone fragments mixed with organic materials in compost wetland substrates are under sufficiently anoxic conditions that they are not subject to armouring with metal precipitates, a possible solution to these two limitations of RAPS is to thoroughly mix the limestone gravel and compost components. The result is as shown in Figure 3.4.

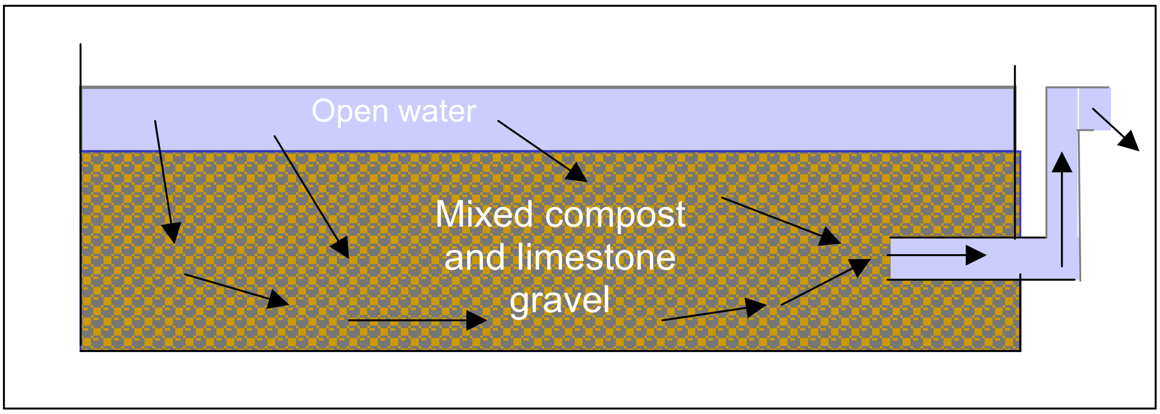


Figure 3.4. Modification of the RAPS concept to dispense with a discrete compost layer of low hydraulic conductivity.

At the time of writing this concept is being introduced to Europe for the first time at the Bowden Close CoSTaR system in northern England (see Younger *et al.* 2003).

3.2.9 Permeable Reactive Barriers (PRBs)

All of the unit processes introduced in sections 3.2.1 through 3.2.8 are used for treatment of discrete discharges at the Earth's surface (so called point sources). However, under some circumstances, polluted mine waters migrate into aquifers, i.e. bodies of soil / rock containing ground water. Where these aquifers are used for supply purposes, the impacts of this pollution may be very grave. Even where the aquifer is minor and not used directly for supply purposes, it is likely to discharge ultimately into a surface water body, so that any polluted water it might contain will emerge in a rather diffuse manner and give rise to ecological damage. In such cases, the only way to address the pollution using the methods thus far described is to pump groundwater to the surface via a well or borehole. Such 'pump and treat' schemes are expensive to operate (e.g. Parker 2003), and do not strictly fall within the definition of passive treatment.

Permeable reactive barriers (PRBs) offer a passive alternative to pump-and-treat systems. In essence, a PRB is a zone of reactive materials emplaced within the flowpath of contaminated ground water, such that polluted ground water is improved in quality as it flows through the PRB (Figure 3.5).

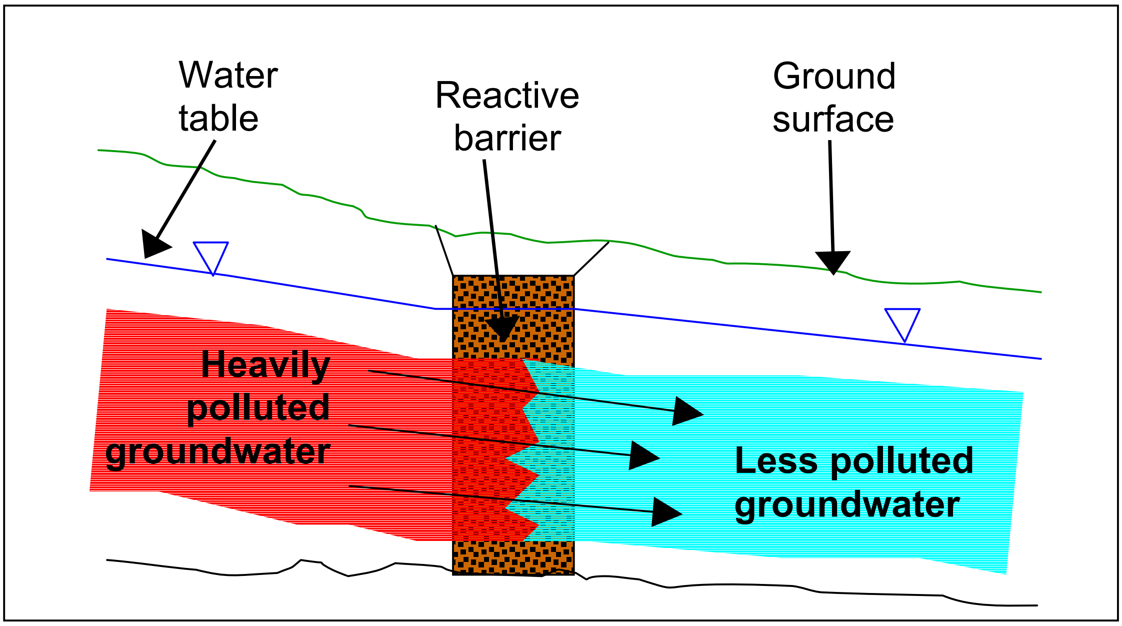


Figure 3.5. Conceptual diagram of a permeable reactive barrier (after Younger *et al.*, 2002).

The majority of full-scale PRBs constructed to date world-wide have been designed to reductively degrade organic micro-pollutants, principally by reaction with a substrate containing zero-valent iron (ZVI). ZVI is also attractive for the reductive precipitation of uranium and chromium, and as such PRBs containing ZVI certainly do have niche applications in those few mine waters which are heavily contaminated with these metals. However, for the vast majority of mine waters ZVI is not an especially attractive reagent, as it tends to *liberate* rather than immobilise dissolved iron. While proposals have been repeatedly made for the use of sorptive materials (such as zeolites; section 5.7) in PRBs receiving acidic mine drainage, and progress has been made within the PIRAMID project in assessing the suitability of caustic magnesia as a soluble reactive component (see section 5.6 and Cortina *et al.* 2003), the majority of successful PRB applications to date for polluted mine waters have been based on the same compost-based, bacterial sulphate reduction processes already described (section 3.2.7) in relation to compost wetlands and RAPS. The only difference between the former two technologies and compost-based PRBs lies in the details of hydraulic design[16]. As with RAPS and compost wetlands, where PRBs discharge their effluent at the ground surface (as can occur, for instance, where they

[16] This minor difference has not prevented the granting of a European process patent (EP-0-502-460-B1), textually derived from an earlier Canadian patent, which covers the application of sulphate-reduction based PRBs for the treatment of acidic, mine-derived waters moving through aquifers. It is important to appreciate that the claims of this process patent relate solely to "within aquifer" application of such technology, and do not cover compost wetlands, RAPS or other surficial permeable reactive systems such as those described by Cohen (1996) (see also Section 5.10).

are installed at the toes of spoil heaps (Younger *et al.* 2002), they PRBs should be followed by settlement lagoon(s) and / or aerobic wetlands to facilitate the further removal of metals and oxygenation of the effluent.

4. PASSIVE TREATMENT OPTIONS FOR ACIDITY, IRON AND ALUMINIUM REMOVAL: DESIGN CRITERIA

4.1. Treatment of net-alkaline waters

4.1.1. Selection and sequencing of appropriate units

Units and processes applicable to the treatment of net-alkaline mine waters (from sections 3.2.1 through 3.2.8) are as follows:

- aeration units
- settlement lagoons
- aerobic wetlands
- SCOOFI reactors

When addressing ferruginous, net-alkaline, mine waters the first step of treatment will always be aeration. The prime objective with net-alkaline mine waters is to convert ferrous iron to ferric iron (section 3.2.1), to facilitate the removal of iron as ferric hydroxide ($Fe(OH)_3$). The rate equation for the oxidation of ferrous iron, given in section 3.2.1, shows that the process is influenced predominantly by the concentration of ferrous iron, the concentration of protons (H^+), and the concentration of dissolved oxygen. Ferrous iron concentration is clearly predetermined, for a net-alkaline water the proton concentration is acceptably low by definition, and therefore the only variable that can be influenced is the dissolved oxygen concentration. Therefore for the conversion of ferrous iron to happen at an acceptable rate, every effort must be made to ensure that the mine water is well oxygenated.

Once a net-alkaline mine water is well oxygenated the next step is to remove iron as a precipitate. This can either be done using settlement lagoon(s) and / or aerobic wetlands. Settlement lagoons alone may be adequate to meet regulatory conditions for the final discharge in some instances. However, usually an aerobic wetland is included, either as the sole means of removing iron, or as a polishing facility following removal of a large proportion of the iron in a settlement lagoon. In summary, therefore, the options are as follows:

- settlement lagoon(s) only
- aerobic wetland(s) only
- settlement lagoon(s) followed by aerobic wetland(s)

Note that a settlement lagoon would never logically follow an aerobic wetland.

It is not possible to provide specific quantitative guidance (in terms of iron load) about which option will be most appropriate for a given mine water. Such decisions may be driven by factors such as available land area, site topography, planning regulations, target effluent quality and desired lifetime of the system, all of which are site specific. Nevertheless, in general terms:

- aerobic wetlands alone will be used where iron loadings are comparatively low.

- "settlement lagoons only" may be used where iron load is high, but target effluent quality is no stricter than about 5 mg/L total Fe.
- both settlement lagoons and wetlands will be used where iron load is high and strict regulatory standards for effluent quality are imposed (e.g. total Fe $\leq$ 0.5 mg/L).

An indication of which is the most appropriate option can be gained by calculating the predicted rate of accumulation of iron. An example is provided below for illustrative purposes:

A net-alkaline mine water has an iron concentration of 50 mg/L and a flow-rate of 50 L/s. An efficient aeration cascade has facilitated complete oxidation of ferrous iron. Regulatory conditions are imposed such that the effluent must not contain an iron concentration of greater than 2 mg/L (a typical numerical consent in the UK). Therefore 48 mg/L iron must be removed.

The first step is to calculate the mass of iron removed per day. The calculation (to give the figure in units of kg/d) is as follows:

$$(48 \text{ mg/L} \times 50 \text{ L/s} \times 60 \times 60 \times 24) / 10^6 = 207 \text{ kg/d}$$

But the iron will be removed (for argument's sake) as iron hydroxide – $Fe(OH)_3$, which has a greater mass than iron alone. The molecular weight of iron hydroxide is 107 (the summation of the atomic weights of Fe (56), O (16), and H (1)), and therefore the 207 kg/d is multiplied by 107 / 56 to give the mass of iron hydroxide: 396 kg/d.

396 kg/d is the mass of dry solids, and assumes that the precipitate will only comprise iron solids. As an estimate, settled ochre has a dry weight content of only around 5%. Therefore the predicted volume of ochre accumulating is 20 times the mass: 7.9 m^3/day.

Using the widely applied empirical formula of Hedin *et al.* (1994a) (see section 4.3.2.3), a wetland for treatment of a discharge of 50 L/s, with iron concentration of 50 mg/L, would need to have an area of 21,600 m^2. This large size alone may be sufficient to make the design of a wetland-only system infeasible, and necessitate the inclusion of settlement lagoons ahead of the wetland. However, another consideration is the rate of accumulation within the wetland. Assuming a freeboard of 1.0 m in the wetland, the total capacity would be 21,600 m^3. At a rate of accumulation of 7.9 m^3/day the wetland would, theoretically, be completely choked with ochre after 7.5 years. In reality, ochre would accumulate most rapidly at the influent end of the system, and problems with clogging of inlet channels and so on would likely be evidenced much sooner. For these reasons, a settlement lagoon would definitely be recommended for such a discharge. However, because of the strict regulatory limit of 2 mg/L, a polishing wetland would still be advisable.

The example above serves to illustrate the type of calculations that can be performed during the design process. Although based on some fairly broad assumptions, such exercises can still serve to assist in the decision-making process. However, other factors, particularly land area and topography, will always constrain the designer.

SCOOFI reactors (section 3.2.4) are a newer technology, and this is reflected in the fact that there is only one full-scale example in the UK (see Younger, 2000). Because quantitative design guidance is not currently available caution should be taken in recommending the installation of such reactors at the current time. However, if land area and topographical restrictions are such that the options are limited to a SCOOFI reactor or no treatment at all, then such technology may be worth considering. Preliminary pilot-scale experiments are definitely advisable.

If sufficient hydraulic head is available to facilitate effective aeration a saturated flow SCOOFI reactor is the recommended type, because the efficiency of iron removal is reportedly much better than the alternative unsaturated variety of reactor (Younger *et al.*, 2002). Unsaturated reactors facilitate aeration and iron removal simultaneously, and therefore may find a niche where conditions preclude the installation of an aeration cascade. However, the saving in hydraulic head made by omitting an aeration facility may be offset by the need to design unsaturated reactors such that flow-rate is downwards.

Figure 4.1 summarises the selection procedure for net-alkaline mine water treatment units in the form of a flow chart. For net-acidic waters use this flow chart following selection of acid water treatment units shown in Figure 4.2.

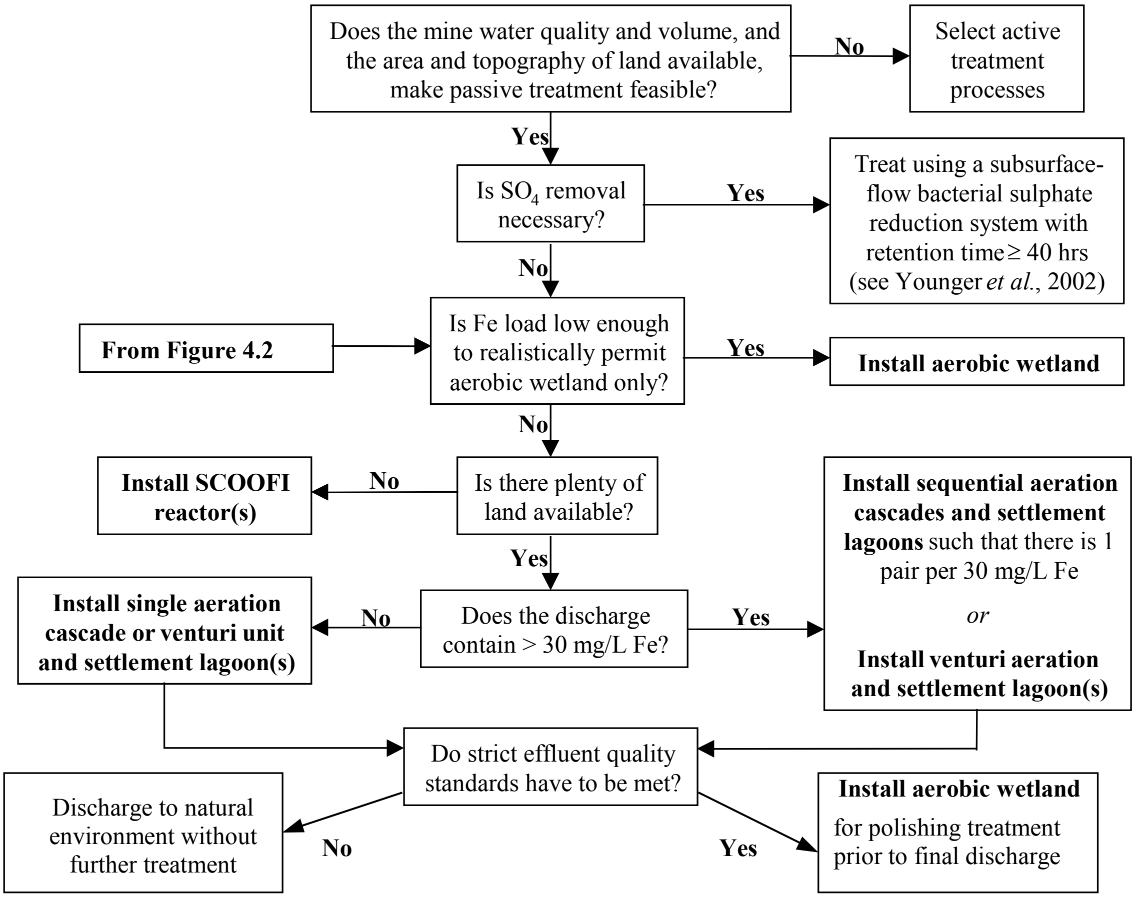


Figure 4.1. Flow chart for unit passive treatment process selection for net-alkaline mine water discharges (NB: use Figure 4.2 first if discharge is net-acidic)

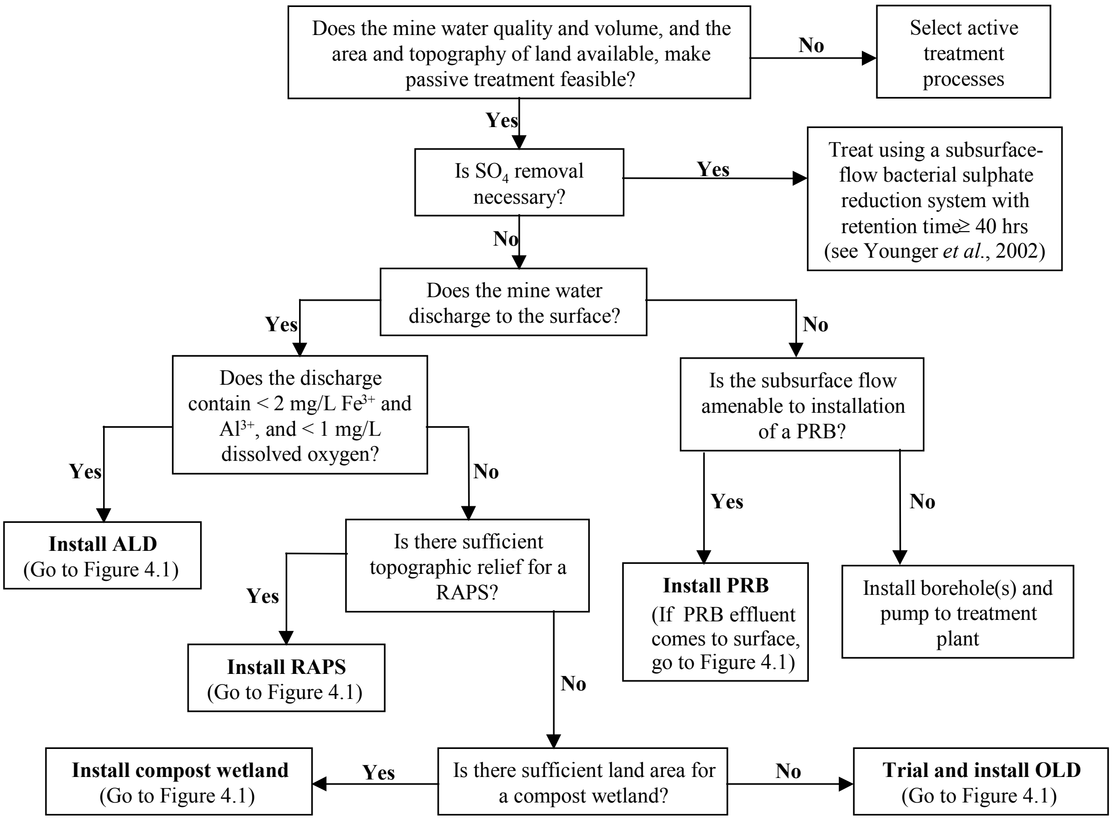


Figure 4.2. Flow chart for unit passive treatment process selection for acidic mine waters (NB: "Go to Figure 4.1" indicates that after the water has been exposed to the unit process specified in the box in question, net-alkaline water should result, so that further unit process selection can be carried out using Figure 4.1).

4.1.2 Sizing criteria and engineering design considerations for passive treatment unit processes applicable to net-alkaline mine waters

4.1.1.1 Aeration cascades

Most aeration cascades installed to date have been designed as simple flights of wide steps, with the aim of maximising the surface area of water available for oxygen transfer from the atmosphere to the water. To ensure a thin film of water across the cascade Younger *et al.* (2002) recommend 100 mm of step width for every 1 L/s of water requiring aeration. Four to six steps are typical, each with a height of between 500 and 800 mm. A single aeration cascade should be capable of oxidising approximately 50 mg/L ferrous iron, although 30 mg/L is sometimes used as a more realistic target. Thus, for mine waters with concentrations in excess of this, it may be necessary to install successive cascades, with settlement lagoons between them.

More recently, alternative approaches to cascade design have been mooted, based upon rigorous research into the hydraulics of a wide range of cascade designs (Novak 1994). In line with these findings, the cascade should be constructed with the aim of ensuring that the water falling over each step enters a plunge pool, where entrained bubbles can fully dissipate in the water column before the water continues over the next step. Experimental observations suggest that the plunge pool should be at least as deep and as wide as the preceding fall of water is high (Figure 4.3a).

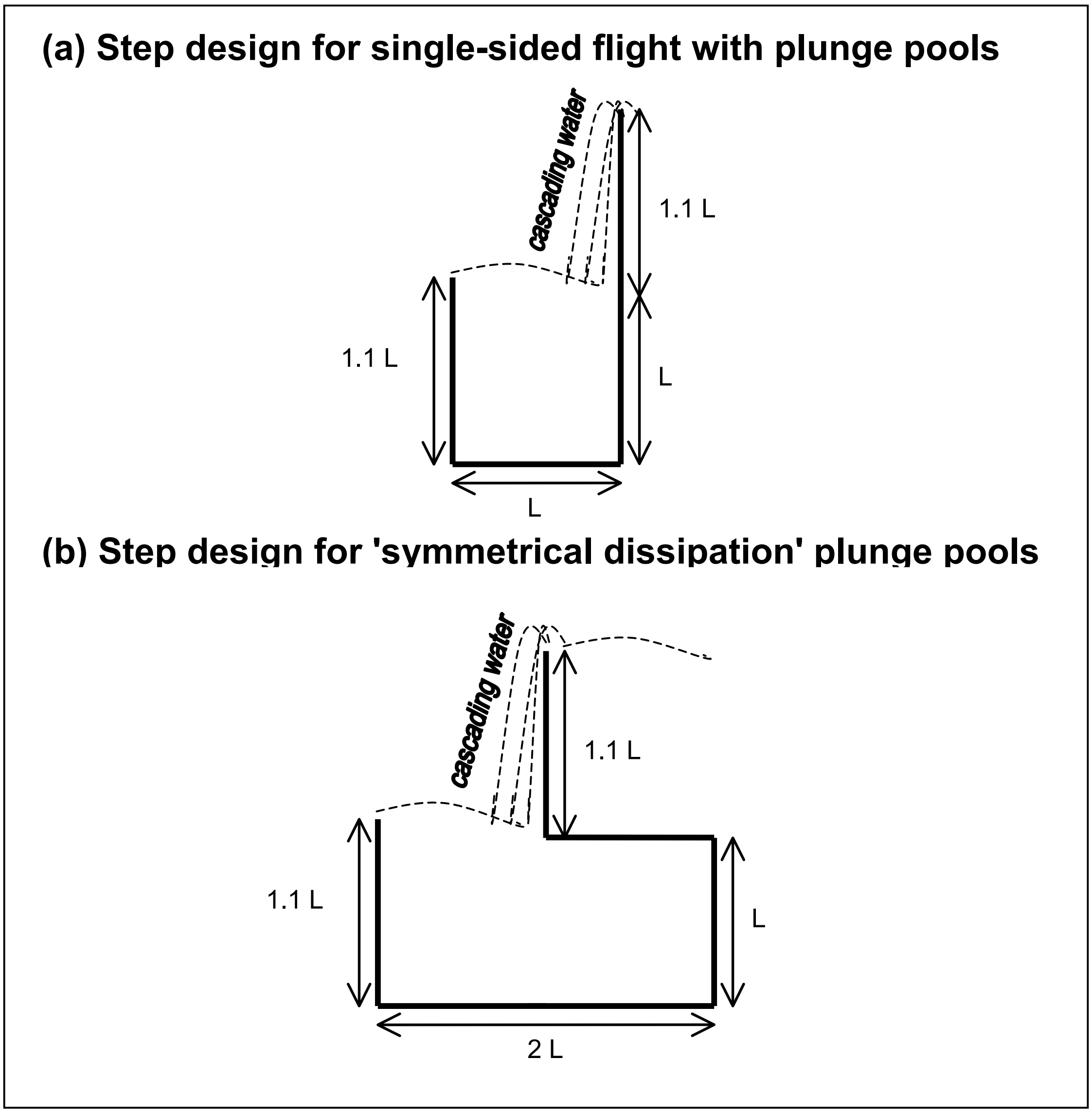


Figure 4.3. Improved aeration cascade design based on the research findings of Novak (1994) and supplementary theoretical and experimental development.

Where space permits, theory suggests (J Aumônier, *personal communication* 2003) that a further enhancement of performance can be achieved by having the falling water enter a basin which affords symmetrical dissipation (see Figure 4.3b). It should be noted that a slight "lip" is added to the crest of each plunge pool to improve cascade development.

4.1.1.2 Settlement lagoons

The key to the design of settlement lagoons is to ensure that the retention time within the lagoon(s) is sufficiently long that iron precipitates will settle out effectively. In the past retention time recommendations for such purposes have ranged from as little as 8 hours to more than 72 hours. This range of estimates reflects the difficulties inherent in designing such systems from first principles, such as application of the physics represented by the Navier-Stokes equation: ferric hydroxide particles are initially very small (< 2.5 μm) and the rates at which they flocculate and increase in density in an open water column vary greatly depending on factors such as water turbulence, temperature and the concentrations of various anions in solution.

Sizing formulae previously used include:

- Stipulating a standard, nominal hydraulic retention time[17] of 48 hours
- Stipulating 100m^2 of lagoon area per L/s of mine water to be treated
- Application of the aerobic wetland sizing criterion of (i.e. assuming an iron removal rate of 10 g/m^2/d)

Table 4.1 demonstrates the wide variations in lagoon sizing which these three formulae yield.

Table 4.1. Calculated settlement lagoon dimensions and capacities using three different design formulae, and three different combinations of flow-rate and iron concentration.

Design criterion	Area (m^2)	Illustrative dimensions (L x W x D) (m)	Volume (m^3)	Retention time (days)	Sludge removal frequency (years)
50 L/s; 50 mg/L Fe					
48 hours retention time	3600	90 x 40 x 3	8680	2	1.8
100 m^2 per 1 L/s flow	5000	100 x 50 x 3	12520	2.9	2.6
Fe removal at 10 g/m^2/d	21600	216 x 100 x 1	21000	4.9	4.6
10 L/s; 50 mg/L Fe					
48 hours retention time	880	42 x 21 x 3	1730	2	1.5
100 m^2 per 1 L/s flow	1000	50 x 20 x 3	1960	2.3	1.7
Fe removal at 10 g/m^2/d	4320	100 x 43 x 1	4020	4.7	4.3
50 L/s; 10 mg/L Fe					
48 hours retention time	3600	90 x 40 x 3	8640	2	8.8
100 m^2 per 1 L/s flow	5000	100 x 50 x 3	12520	2.9	12.9
Fe removal at 10 g/m^2/d	4320	100 x 43 x1	4020	0.9	4.3

Note: Dimensions, volumes and areas are calculated on the basis that a trapezoidal basin is used (internal slopes 2:1). Water depth is 3 m for first two formulae, but only 1 m for third option (see text for explanation). Sludge removal assumed necessary when lagoon two thirds full.

[17] calculated simply as the ratio of the volume of water stored in the lagoon to the design flow rate

For discharges with high iron concentrations the area-adjusted removal rate method of calculation generates the largest lagoon area; indeed the sizes suggested will be impracticably large in most cases, since operational realities are such that effective de-sludging of such a vast lagoon using established means would be virtually impossible. Conversely, for high flow, low concentration discharges (50 L/s; 10 mg/L Fe), the retention time in lagoons designed on this basis is low (0.9 days), and may not be sufficient to allow effective iron removal.

Using the 100 m^2 per 1 L/s flow formula has the advantage that it ensures that retention time is consistently high. However, for discharges with high flow and low iron concentration the size of lagoons calculated is greater than alternative design formulae.

Sizing based on nominal hydraulic retention time (such as the 48 hours used in Table 4.1) may in fact be the most robust approach to use. Recent research by the UK's Coal Authority has revealed the existence of a relatively robust linear relationship between the percentage reduction in the influent iron concentration and nominal hydraulic retention time (Parker 2003). Interestingly, this relationship appears to be largely independent of both flow rate (it has been found to obtain for flows between 16 and 97 l/s) and of the absolute value of initial iron concentration (values between 4.7 and 44.2 mg/L being included in the data-set from which the relationship was derived). The relationship may be summarised by the following simple expressions.

Required hydraulic retention time (hours) = 0.5 x (% lowering in Fe concentration desired)

and conversely:

% lowering in iron concentration achievable = 2 x (hydraulic retention time in hours)

For both of the above expressions, the 95% confidence interval seems to be approximated by a range of ± 10% of the values obtained from the formulae. It is easy to see how these two expressions might be used for the following purposes:

- to size a proposed settlement lagoon
- to determine the iron concentration leaving a settlement lagoon of a specified size

Other important considerations when designing settlement ponds include:

- The length to width ratio should be within the range 2:1 to 5:1 (NCB, 1982), to help minimise possible streaming and short-circuiting.
- The depth of the pond should be sufficient to prevent resuspension of settled particles due to the horizontal velocity of water and / or wind. 3 m is a typical water depth.
- Bear in mind that as solids accumulate in the pond the effective volume (and therefore retention time) will decrease. If possible, it is therefore worth building in safety factors to accommodate this during design.

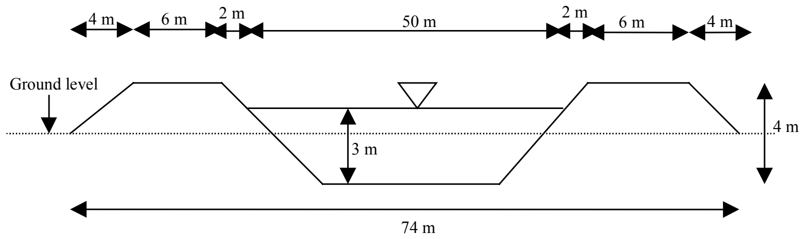
- The most effective shape of ponds, from a hydraulic point of view, is rectilinear. However, there is an increasing desire to minimise the visual impact of treatment schemes, and there is therefore a move away from hard engineered structures. Typically this means that settlement lagoons have a somewhat larger land area requirement than initial calculations would suggest.

The foregoing discussion (and Table 4.1 in particular) is open to the misinterpretation that a single settlement pond is the likely outcome of design calculations. In practice, this is rarely, if ever, the case. It was stated earlier that the maximum concentration of ferrous iron that can be oxidised in a single aeration cascade is 50 mg/L. This is the theoretical maximum, but practical experience suggests that 30 mg/L ferrous iron is a more realistic figure (NCB, 1982). For discharges with in excess of 30 mg/L it will be necessary to have a series of aeration cascades, with settlement lagoons in between. For example, for a mine water containing 100 mg/L ferrous iron, there will need to be 100 / 30 = 3.3 aeration cascades (i.e. 4), and 4 settlement lagoons in series. Even where the initial iron concentration is less than 30 mg/L it is good practice to have at least two settlement lagoons available, so that one may remain in use while the other is taken off-line for de-sludging[18]. Under ordinary operation the two lagoons would typically be operated in parallel, each receiving 50% of the total flow. (Performance of either one of the lagoons on its own during a period when it receives the total flow can be estimated using the expressions given above). It should be noted that the use of a minimum of two lagoons in parallel, rather than a single treatment line, has implications for the overall land area requirement for lagoon construction. The worked example in the box below illustrates this point.

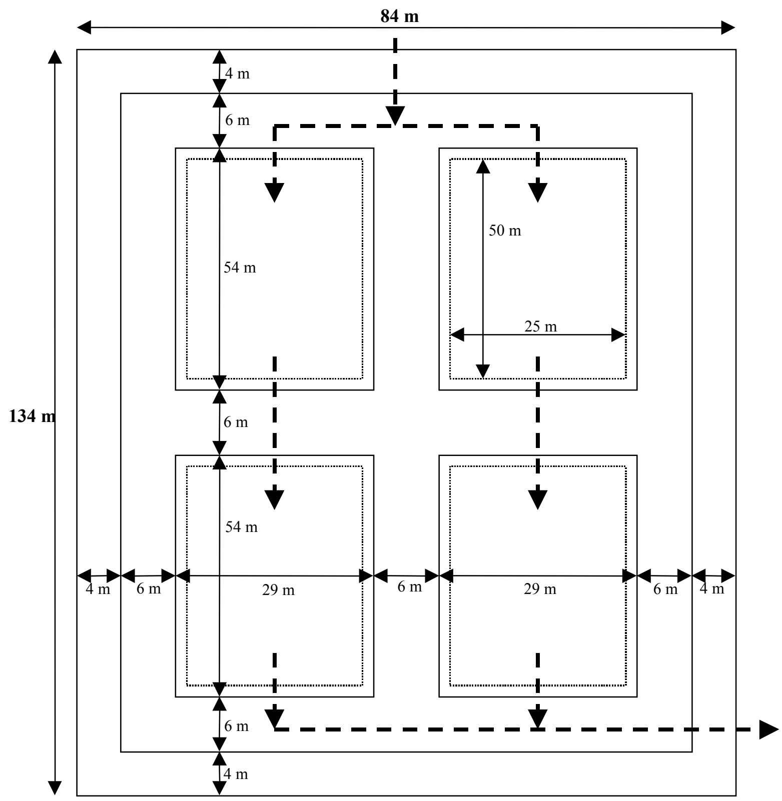
Disposal of sludge from settlement ponds is a significant cost element in the long-term maintenance of passive systems. A thorough discussion on this topic, including consideration of possible uses for recovered ochre, is beyond the scope of this document. However, it is worth noting that sludge management can be greatly facilitated by the provision of sludge drying beds in close proximity to the settlement lagoons, into which sludge may either be pumped, tipped or else allowed to flow by gravity via a pipeline installed for the purpose at the time of lagoon construction. A suitable sludge drying bed will typically be a shallow basin with a gravel base, which will be under-drained by a pipe network. After sludge from the settlement lagoons has been emplaced within such a drying bed, it will gradually dry out over a period of several months, with an increase in its solids-by-weight content from as little as 2% at the time of emplacement to 30% or more by the time it is sufficiently dry to exhibit desiccation cracks. This natural dehydration of sludge makes it far more amenable to possible recycling, or at the very least reduces the volume of material destined for waste disposal.

[18] One of the few exceptions to this rule relates to instances in which mine water is pumped to the treatment system from a shaft or borehole. If the recovery of water levels within the borehole or shaft is slow enough, it may be that pumping can cease for the period of sludge removal from a single-line settlement lagoon without resulting in a polluted discharge to surface.

For the hypothetical mine water in Table 4.1, with a flow-rate of 50 L/s and ferrous iron concentration of 50 mg/L, the area of settlement pond calculated using the 100 m^2 per 1 L/s flow is 5000 m^2. The nominal dimensions of such a pond are 100 m x 50 m, giving a volume of 12,520 m^3 (assuming 3 m water depth).



The cross-section above is typical of the design of such a lagoon. The width of water surface is 50 m, as calculated. For structural stability all slopes have to have an angle of at least 2:1. With a 1.0m freeboard, this means that the width at bank top level is 54 m. Vehicular access is likely to be required around the ponds to facilitate maintenance and sludge removal. A safe (albeit generous) width for tracks is 6 m, increasing the width to 66 m. Finally, the external slopes must be considered. In this case the retaining walls are 2 m high, and therefore an additional 4 m width is required on each side of the pond. The total width of land required is therefore 74 m (24 m greater than the width of the water surface). The same additions will apply to the long access of the settlement lagoon, and therefore the overall dimensions will be 124 m x 74 m, giving a total land area requirement of 9,200 m^2 (cf. water surface area of 5,000 m^2).



For a water containing 50 mg/L ferrous iron two aeration cascades, and therefore two settlement lagoons, will be required, in series. Additionally, unless there is another set of lagoons in parallel it will not be possible to continue operation during sludge removal. Therefore in fact 4 ponds are required, each with an area of 1,250 m^2. Nominal dimensions for each lagoon might be 50 m x 25 m. The most efficient layout, with vehicular access around each lagoon, is illustrated in the plan view above. The overall dimensions now are 134 m x 84 m, giving a total land area requirement of 11,260 m^2 (cf. 9,200 m^2 for a single pond). The effect of splitting the system into four ponds is to slightly reduce the retention time and increase frequency of sludge removal. The total water volume in the 4 lagoon system is 10,460 m^3, nominal retention time is 2.4 days, and sludge removal frequency is 1.9 years (cf. 12,520 m^3, 2.9 days, and 2.6 years respectively for a single 5,000 m^2 lagoon).

4.1.1.3. Aerobic wetlands

The most robust design approach for aerobic wetlands remains that of Hedin *et al.* (1994a). In this approach, a simple formula is used to calculate the area of wetland required for a given flow-rate and influent and target (i.e. effluent) iron concentration (or other metal). In its most general form, the sizing formula may be written as follows (Younger *et al.* 2002):

$$A = \frac{Q_d \,(C_i - C_t)}{R_A}$$

where,	A	=	required wetland area (m^2)
	Q_d	=	mean daily flow-rate (m^3/d)
	C_i	=	mean daily influent contaminant concentration (mg/L)
	C_t	=	concentration of contaminant in final discharge (mg/L)
	R_A	=	area-adjusted contaminant removal rate ($g/m^2/d$)

If the guidance provided in section 2.2 (Hydrochemical sampling and analysis) has been followed, reliable values for Q_d and C_i should be known. A value for C_t should have been agreed with the relevant regulatory authority and / or client.

The area-adjusted contaminant removal rate was derived by Hedin *et al.* (1994a) by monitoring a range of wetland systems in the USA. For systems that must meet regulatory standards a value of 10 $g/m^2/d$ is recommended. For wetlands where a 'reasonable improvement' in water quality is deemed sufficient, a less stringent figure of 20 $g/m^2/d$ may be admissible.

Increasingly the design of wetlands is guided by the fact that they should provide amenity value as well as effective water treatment. Consequently, there has been a move away from rectilinear shapes and 'hard' engineered structures. It is with amenity in mind that Younger *et al.* (2002) provide general guidelines for the design of aerobic wetlands. These guide notes are paraphrased as follows:

- Ensure the wetland surface has an uneven surface (such as spits and islands) to maximise habitat potential and help baffle flow across the wetland
- Try to ensure that internal slopes have a gradient of 3:1 or less, again to maximise habitat value, but also to reduce health and safety risk.
- Try to avoid tall trees in close proximity to the wetland, as these may cause shading and limit development of emergent vegetation in the wetland. Smaller shrubs, in contrast, may be a positive advantage for riparian mammals and amphibians.
- Avoid the use of exposed concrete structures, which detract from amenity value.
- Avoid artificial liners if possible. These add significantly to capital costs, are easily damaged once in place, and may be unattractive and hazardous if exposed.
- Avoid rectilinear corners when laying out a wetland. Such features tend to be ugly, and square corners often result in stagnant zones, which do not aid treatment.
- Distribute influent water evenly across the width of the wetland

It will be apparent that the preceding guidelines are more focused towards the amenity and habitat aspects of wetland construction. There are equally important engineering guidelines that should be followed. Inevitably the physical constraints of a site may determine the particular layout of a wetland, but the following engineering guidelines should be adopted as closely as possible:

- Inlet structures can be one of two types, both of which should extend the full width of the inlet end of the wetland, to minimise short-circuiting / streaming:
  - crenellated concrete channels
  - a long length of pipe with T-pieces for discharge of water at intervals along it

  The pros and cons of each are discussed in section 8.4. Because of the amenity value of wetlands, inlet structures comprising pipes are often favoured. For outlet structures care must be taken if installing pipes, since there is the potential for blockage by vegetation. This can be avoided to some extent by not planting reeds in the immediate vicinity of the outlet pipe.
- Both types of inlet (and outlet) structure should be such that water level can be varied. Although typical water depth in a wetland should be 100-200 mm, recent experience in the UK has suggested that for newly planted wetlands an initial water depth of approximately 50 mm may ensure better reed establishment.
- Soil depth should typically be around 300 mm. Important aspects of 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 a reliable analysis is made of it.
  - It contains sufficient nutrients, in appropriate proportions, to support reed growth.

- The height of freeboard above the soil surface will be one factor governing the lifetime of the wetland, and it should therefore be maximised within practical reason - 1 m is typical.
- Retaining bunds can be constructed along similar lines to those for settlement lagoons (see section 7), if the wetland needs to be constructed above original ground level. There may not be a need to have wide track access (as illustrated in the box above), since there should not be the need to use heavy machinery around the wetland, at least until such time as soil and reeds require replacement.
- Aquatic plants in wetlands serve a number of useful functions (Laine and Jarvis, 2002):
  - 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.
  - Stems and leaves may provide additional surface area for adsorption and surface-catalysed, oxidative precipitation of metals.
  - Direct uptake of metals by plants (which occurs principally in roots and rhizomes in common wetland plants, with very little translocation to the subaerial stems and leaves) can remove metals down to lower residual concentrations than would be achieved in open-water settling ponds, a process which is particularly important in 'polishing' applications of wetlands at the end of a treatment system (Batty and Younger, 2002).
  - Emergent macrophytes significantly improve the aesthetic appearance of a site, and contribute to wildlife habitat.
  - In both nominally aerobic wetlands and compost wetlands, the continuous carbon source provided by decaying plant debris may encourage bacterial sulphate reduction reactions in the subsurface, further encouraging the immobilisation of metals (Mitsch and Wise, 1998).

It is good practice to make efforts to avoid establishing monocultures in wetlands. Although *Typha latifolia*[19] has been most commonly used in mine water treatment wetlands constructed to date in Europe, other commonly-used aquatic plants *include Phragmites australis* (Common reed), *Juncus effusus* (Soft rush), *Scirpus lacustris* (Bullrush) and *Iris pseudacorus* (Yellow flag iris). The

[19] Common name: reedmace in the UK, cattails in the USA

ecological value of a wetland will be enhanced where attention is paid to the natural wetland plants of a given zone when selecting species for planting. For instance, while *Phragmites australis* is native in most of central and southern Europe, its northern limit lies around 53^{o}N. While it is possible to successfully introduce *Phragmites* north of this line as pot-grown seedlings, the adult plants produce only infertile seed, and thus reproduce only asexually, by means of rhizome migration. In more northerly latitudes therefore, it is difficult to make an ecological case for inclusion of *Phragmites* in an introduced suite of wetland plants.

Typically reeds are planted at a density of 3-4/m^2, and they are most likely to thrive where they are introduced to the new wetland between mid-April and late June in the form of pot-grown seedlings around 200 mm. These tall seedlings are favoured where resources permit, as they are sufficiently advanced in terms of growth to survive unanticipated adverse conditions in the new wetland. Nevertheless, considerable success has been achieved with cheaper 90 mm plugs (Laine and Jarvis, 2002), and even by growing wetland plants from seed (Younger *et al.* 2002). Indeed, where there is no urgency in achieving peak wetland performance, it is possible to simply create the right hydraulic conditions for wetland plant establishment (i.e. with water depths between 15 and 40 cm depth) and await natural colonisation by wetland plants. Further details of recommended reed species and planting strategies for constructed wetlands are provided in section 8.2.

<u>4.1.1.4. SCOOFI reactors</u>

Sizing criteria for SCOOFI reactors are based on far fewer examples than aerobic wetlands, and therefore confidence in them is significantly lower. Only 4 systems have been built to date (all in the UK; 2 pilot-scale and 2 full-scale). Saturated flow reactors are intrinsically more efficient than the unsaturated, but the latter may find application where simultaneous oxidation and accretion is required on a cramped site.

When normalised for the surface area of the media, the following area-adjusted removal rates are derived:

- 25 g/m^2/d for saturated flow reactors
- 0.05 g/m^2/d for unsaturated flow reactors

For saturated flow reactors the value is greater than that for aerobic wetlands (10 – 20 g/m^2/d), but the equivalent figure for unsaturated flow reactors is considerably less. However, the residence time of waters within SCOOFI reactors is significantly less than in wetlands (as little as 90 – 120 *seconds* for unsaturated reactors; Jarvis and Younger, 2001). Dividing by the residence time of the various reactors (a mean of 3.5 days for aerobic wetlands; Hedin *et al*, 1994a), gives the following figures (after Younger *et al.*, 2002):

- Aerobic wetlands: 2.9 – 5.7 g/m^2/d
- Unsaturated flow SCOOFI reactors: 36 g/m^2/d
- Saturated flow SCOOFI reactors: 4000 g/m^2/d

In this context, the significantly more rapid removal of iron in SCOOFI reactors is illustrated. Younger *et al.* (2002) point out that these figures will be substantially amended as experience of SCOOFI reactors increases, but they are nevertheless useful starting points in the design of such passive treatment units.

Perhaps the most important design and engineering consideration for SCOOFI reactors is how to ensure that the media can be removed to be cleaned, or cleaned *in situ*. Jetting or brushing should be sufficient to clean the media, but in a system of any size it may difficult to gain access to do so. Placing media in metal gabions or similar, that can be sequentially lifted out during operation, for cleaning, is one solution that has been adopted.

4.2. Treatment of net-acidic waters

4.2.1 Selection and sequencing of appropriate units

Units and processes applicable to the treatment of net-acidic mine waters (from sections 3.2.1 through 3.2.8) are as follows:

- Anoxic Limestone Drains (ALDs) and Oxic Limestone Drains (OLDs)
- Compost wetlands
- Reducing and Alkalinity Producing Systems (RAPS)
- Permeable Reactive Barriers (PRBs)

For acidic waters, a unit capable of generating alkalinity / neutralising acidity / raising pH will always precede units designed for metal removal (i.e. those discussed in sections 3.2.1 to 3.2.4 and 4.3.2.1 to 4.3.2.4). There are only two possible exceptions to this rule (Younger *et al.*, 2002):

1. For highly carbonated waters, a pre-aeration step may be employed to de-gas CO_2, and hence raise pH.
2. A sedimentation lagoon may be installed as the first treatment unit where the content of inert solids is high.

In all other cases generating alkaline conditions is the first priority, since the subsequent removal of iron is so much more rapid (and effective) at higher pH.

It is normal that only one of the four types of unit listed above will be used in any particular treatment scheme. The selection of which unit is most appropriate for alkalinity generation is based upon water chemistry, land area, topography, and the nature of the discharge (i.e. surface or subsurface). Figure 4.2 is a flow chart that can be used as a decision-making tool for selection of the most appropriate passive treatment unit for acidic discharges. This should be used in combination with the engineering design guidance that follows.

4.2.2. Sizing criteria and engineering design for net-acidic mine water treatment units

4.2.2.1 Anoxic Limestone Drains (ALDs) and Oxic Limestone Drains (OLDs)

Because of the lack of full-scale ALDs in Europe, the engineering design guidance for these systems is still wholly dependent on lessons drawn from experiences in the USA (e.g. Hedin *et al.,* 1994a).

The sole purpose of an ALD is to generate alkalinity, and studies have demonstrated that alkalinity generation levels off after approximately 8-14 hours retention time (Hedin *et al.*, 1994a). Early systems resembled buried trenches, but more recently rectangular shaped systems have been favoured (Younger *et al.*, 2002).

In terms of ALD size, the calculation is straightforward. It entails the calculation of the volume of drain required (accounting for the porosity of the limestone) to provide a retention time of 14 hours. Younger *et al.* (2002) present the relevant equations as follows:

$$V_v = Q_d \cdot \varphi$$

where V_v is the volume required to store 14 hours worth of water, Q_d is the design flow-rate (in m^3/d), and the factor φ is in this case equal to 14. The minimum total volume (V_t) is then calculated by accounting for the porosity of the limestone (typically in the range 38%-50% (Younger *et al.*, 2002)):

$$V_t = V_v / n_e$$

where n_e is the effective porosity expressed as a decimal (e.g. 0.5 as opposed to 50%).

The porosity of the limestone should ideally be checked experimentally before finalising the design. The choice of limestone may depend on the particular topography of the site. For sites where significant hydraulic gradient is available, small size (e.g. 10-20 mm) limestone may be appropriate (because it is effectively more reactive), but at flatter sites 50-75 mm aggregate may be a better choice. In either case, single size limestone must be specified, and the limestone selected must have a $CaCO_3$ content of > 80% to be effective. In terms of the layout of the ALD at a particular site, Younger *et al.* (2002) recommend maximising the following factors:

- the hydraulic conductivity (K) of the ALD
- the cross-sectional area (A) through which flow occurs and
- the hydraulic gradient (i) across the ALD.

The hydraulic conductivity (K) can be derived experimentally by conducting constant-head permeameter tests (see, for example, Freeze and Cherry (1979)), and the hydraulic gradient (i) can be determined once the site is surveyed, and the inlet and outlet points of the ALD are known. Knowing the design flow-rate (Q_d), it is then possible to calculate the required cross-sectional area (A) by rearranging Darcy's Law:

$$Q = K \cdot A \cdot i$$

Wherever feasible the cross-sectional area of flow across the ALD should be maximised, so as to reduce the build up of back-pressure, and minimise the likelihood of total clogging of the system.
As limestone dissolution proceeds in an ALD the retention time, and therefore the effectiveness of the ALD, will begin to decrease. For this reason it is recommended that 14 hours is used as a minimum retention time. Wherever possible, retention time should be maximised (say to 20-40 hours).

The construction of an ALD necessitates the excavation of a suitable trench or rectangular-section basin, typically of around 2-3 m depth. Unless the side-walls are stabilised these may need to be sloping, which may increase the overall surface footprint of the ALD. In the USA only 5-10% of ALDs are lined with heavy duty PVC or similar (Younger *et al.*, 2002), but it may be necessary in particularly permeable ground. Once in place the limestone needs to be covered, to ensure that oxygen ingress is minimised, CO_2 retention is maximised, and that the system is hydraulically confined.

Inlet pipes are typically arranged so as to feed into the base of the drain, with effluent pipes exiting at in the upper portion of the far end of the system.

In engineering terms OLDs are designed in a similar manner to ALDs, but with some subtle differences:

- Water velocities through the system must be > 0.1 m/min, and therefore limestone size must be larger than an ALD (ca. 100 mm).
- Total retention times are consequently in the region of 3 hours (Cravotta and Trahan, 1999).
- Cravotta and Trahan (1999) recommend the inclusion of a scour pipe, a large diameter perforated pipe along the base of the system that can be periodically opened to allow sudden flushing of any metal hydroxide solids retained in the OLD.
- Although the influent water may have high dissolved oxygen concentration, Watzlaf (1997) and Cravotta and Trahan (1999) suggest that OLDs are most effective when applied to mine waters with Fe^{3+} and Al^{3+} concentrations in the region of 10-20 mg/L, and acidity concentration of ≤ 90 mg/L as $CaCO_3$.

<u>4.2.2.2 Compost wetlands</u>

The design of compost wetlands is best performed using the same equation as that for aerobic wetlands (section 4.1.1.3). The difference is in the removal rates used. The processes that attenuate acidity and metals in a compost wetland are slower than those that remove metals in an aerobic wetland. Recommended removal rates for design of compost wetlands fall in the range 3.5 – 7.0 $g/m^2/d$, with the lower figure being used where the receiving watercourse is particularly sensitive and / or strict regulatory standards need to be met in terms of effluent quality.

The processes that ameliorate contaminants in a compost wetland are highly complex, and the caveats discussed in section 4.5 regarding the predetermination of effluent quality apply even more significantly in the case of compost wetlands. It is for this reason that the layout of Figure 4.2 is such that RAPS will be selected ahead of compost wetlands when selecting passive treatment units for acidic discharges.

The design of compost wetlands is essentially the same as that for aerobic wetland systems. Beyond making the wetland an appropriate size, the selection of a suitable compost media is perhaps the most important decision to be made. Important properties of the media are as follows (Younger *et al.*, 2002):

- The media must be sufficiently fibrous that it will retain its hydraulic conductivity when self-loaded to a depth of approximately 0.5 m, with additional loading due to water of 0.1-0.2 m.
- It should contain sulphate reducing bacteria (SRB). This is usually assumed to be the case for any compost containing mammalian faecal material, and it is not normal to conduct microbiological tests to confirm SRB content prior to installation.
- The media should be alkaline if possible, and should certainly not be likely to release acids into solution.
- It should not contain potentially harmful viruses (such as BSE or foot-and-mouth disease).

- Preferably it should be locally available at low cost.

The depth of compost used is typically in the region of 0.5 m. Compost media that have been successfully used in wetlands include (Younger *et al.*, 2002):

- Spent mushroom compost (e.g. Hedin *et al.*, 1994a).
- Horse manure and straw (e.g. Younger *et al.*, 1997).
- Cow manure and straw (e.g. Cohen and Staub, 1992).
- Composted municipal waste (e.g. Jarvis and Younger, 1999).
- Composted conifer bark mulch (e.g. Younger, 1998).
- Sewage sludge cake (e.g. Laine and Jarvis, 2002).
- Paper waste pulp (e.g. Laine and Jarvis, 2002).

The planting of reeds is not considered essential in compost wetlands. However, it is usually undertaken nevertheless, for two main reasons:

- to improve the appearance and habitat value of the wetlands, and
- so that annual senescence (i.e. die-back) of the reeds will provide a regular input of fresh ceullose-rich material to the wetland substrate, where it will be subject to bacterially-mediated decay, releasing short-chain organic compounds such as acetate, which are the favoured metabolites of sulphate-reducing bacteria. Root exudates from many wetland macrophytes also include these compounds.

4.2.2.3 Reducing and Alkalinity Producing Systems (RAPS)

RAPS are increasingly viewed as the unit process of first choice for treatment of net-acidic waters, largely because of their lower land area requirements in comparison with compost wetlands, and also because of their ability to cope with wider ranges of raw water quality than ALDs. In terms of system dimensions, the following factors need to be borne in mind:

- A much greater freeboard is required above the compost layer in a RAPS than in a compost wetland system. Wherever possible, freeboard should be ≥ 1.5 m, to ensure that any decrease in hydraulic conductivity of the compost over time can be automatically compensated by a rise in head of water above the compost without running the risk of over-topping the RAPS bunds (Younger *et al.*, 2002).
- Typical depths for compost in a RAPS are in the range 0.15 - 0.60 m (Watzlaf et al, 2003).
- The limestone gravel layer beneath the compost is typically made 0.5 – 1.0 m thick, using single-size gravel of 25 - 50mm diameter (Watzlaf *et al.*, 2003; Younger *et al.*, 2002). It is usual practice to size the RAPS such that this limestone gravel layer has a nominal retention time (= total pore volume / design flow rate) in excess of 14 hours, and to then fit adjust the area and thickness of the layer to achieve this total pore volume. It is important to remember that end-tipped, single-size limestone gravel almost invariably displays a porosity of 50%, so that the excavation required to provide a 14 hour retention time will be double the calculated volume of limestone. Thus, for a mine water with a flow-rate of 5 L/s, the volume of limestone required (assuming 50% porosity) would be 504 m^3. (Given that limestone has a relative density of around 2.7, this equates to about 1360 tonnes).
- Alternatively, the area of a RAPS can be calculated by assuming that the thicknesses of the compost and limestone gravel layers are each ≥ 0.5 m, and then using an areal removal rate, similar to that used for sizing aerobic and compost wetlands. From monitoring of well-established systems in the eastern USA, Watzlaf *et al.* (2003) suggest that acidity removal rates of 25-30

g/m^2/d are appropriate for sizing RAPS. Note that these values are an order of magnitude higher than the equivalent figures for compost wetlands.

- Allowance should be made for the water to be able to lose at least 1.5m of head during its flow through the compost layer. Early in the life of the RAPS, this will be a gross over-estimate, but as clogging of the compost and limestone layers gradually increases, the ability to increase the head across the reactive media will become more and more important. The effluent pipe from the system should thus be configured such that its height may be varied vertically. A number of possible ways of achieving this exist, including the use of penstocks, hinged valves and weirs with stop-logs.
- One of the most important civil engineering considerations for RAPS is the sealing of the effluent pipe-work where this passes through the downstream retaining bund. If not properly sealed against the bund material, leakage along the boundaries of the pipe can occur, which may eventually lead to erosion and failure of the retaining bank.
- It is good practice to include a "scour pipe" in the effluent pipe-work of the RAPS (see Figure 4.4), the purpose of which is to facilitate periodic flushing of the system, by opening a valve on the scour pipe and subjecting the interior of the RAPS to very high hydraulic gradients, thus transporting loose solids (be they metal hydroxides or inert sediments) out of the RAPS pore space. This is especially important where the RAPS receives aluminium-rich waters, which are prone to forming fluffy flocs of aluminium hydroxide within the pores of the RAPS media. Periodic operation of the scour pipe in this manner can help to extend the operating life of the RAPS unit considerably (Kepler and McCleary 1997).

Figure 4.4 illustrates the application of these guidelines for a case in which the mine water has a flow of 5 L/s, with a net-acidity concentration of 100 mg/L as $CaCO_3$. Having calculated a total volume of limestone required (see above) as 504 m^3, if the limestone gravel layer is set to be 0.3 m thick, then the surface area of the limestone layer will be 1680 m^2. Bearing in mind that the horizontal dimensions of the excavation will necessarily decrease with depth (since few engineering soils have a vertical angle of repose), so that the internal slopes will lie at an angle of at least 2:1, the surface area of the overlying compost layer will be greater still, at around 1,730 m^2.

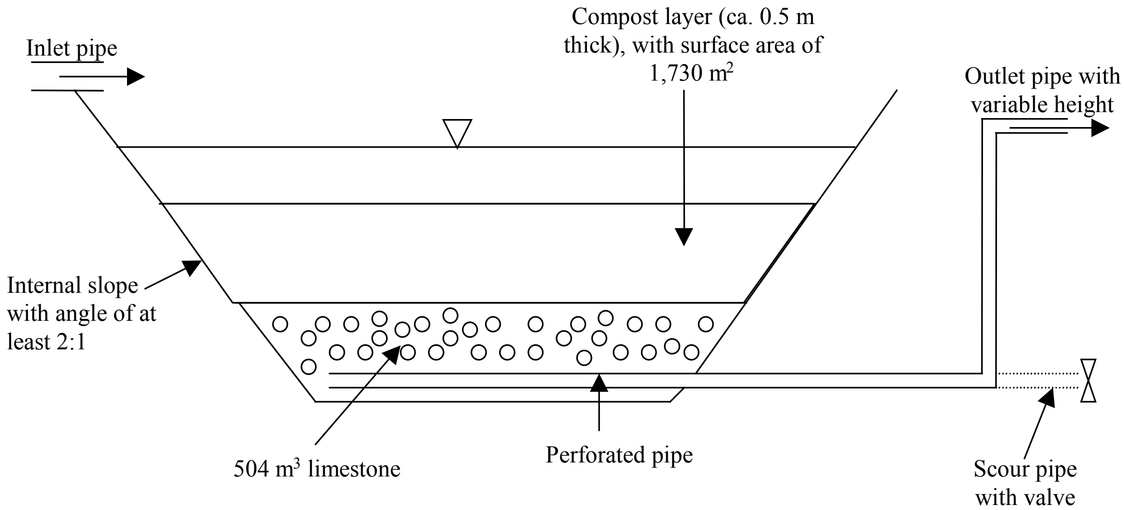


Figure 4.4. Conceptual drawing of a RAPS to treat net-acidic mine water with a flow of 5 L/s and acidity concentration of 100 mg/L as $CaCO_3$.

As previously mentioned, alternative RAPS configurations are beginning to appear, such as the design of Jarvis and England (2002), who arranged the compost and limestone layers side-by-side, such that water flows downwards through the compost, and then upwards through the limestone gravel. This further reduces the possibility of short-circuiting through the limestone gravel layer, and also reduces the depth of excavation required (typically at least 2.5 m), albeit at the expense of some land area. For the latest RAPS designs, in which a thoroughly mixed substrate of organic matter and limestone gravel is prescribed (with 50% limestone in the mixture, and favouring larger limestone clast sizes (around 50mm), definitive design rules currently remain a topic for further research (e.g. Amos and Younger 2003; Younger *et al.* 2003), but preliminary indications suggest that volumetric removal rates on the order of 25 $g/m^3/d$ may reasonably be assumed.

4.2.2.4 Permeable Reactive Barriers (PRBs)

A prerequisite of good PRB design is an accurate quantification of the groundwater flow, and specifically (Younger *et al.*, 2002):

- the hydraulic conductivity of the aquifer
- the hydraulic head distribution within the aquifer, from which flow directions and velocities can be derived and
- the quality of the contaminated groundwater, which is likely to be worst near the source of the pollution.

It will be apparent from these requirements that the installation of a borehole network is a necessary forerunner to treatment system design. Identifying suitable positions for boreholes, selecting the appropriate method of drilling, and interpreting the subsequently collected data, is beyond the scope of these guidelines, and specialist advice should be taken on these matters.

To date PRB systems applied to mine water treatment have mainly been of the ‘continuous wall’ type (Younger *et al.*, 2002), in which the reactive media are installed across the full width of the polluted groundwater plume. The simplest approach to installation is ‘simultaneous cut and fill’, in which reactive media is placed in the trench as it is excavated. In reasonably cohesive soil it is possible to excavate a trench of up to 6 m depth without needing to install temporary supports. Beyond 6 m depth, construction of PRBs becomes both more difficult and more costly (e.g. Carrera *et al.* 2001).

The criteria for selection of a suitable reactive media are similar to those for compost wetlands and RAPS (sections 4.4.2.2 and 4.4.2.3 respectively). Even more so than for either of these types of system, selecting a suitably permeable media is crucial. Laboratory-based permeameter tests are typically undertaken to establish the hydraulic properties of the media (e.g. Amos and Younger, 2003), bearing in mind that up-scaling to field applications rarely results in identical values. There are no fixed rules on the width of PRBs, or on the most appropriate residence times. To date systems have had widths of 1.4 – 4.0 m, and residence times within successful PRBs range from 3 – 90 days. However, it appears that 3 days is adequate for effective treatment (Cohen and Staub, 1992). Younger *et al.* (2002) (pp. 380-381) tentatively suggest a means for calculation of PRB width (W_{PRB}), which involves calculation of the average linear velocity of flow through the PRB (V_{PRB}). This is then used to calculate the barrier width, using the equation:

$$W_{PRB} = V_{PRB} / \varpi$$

where the factor ϖ (the residence time, in days) should be greater than or equal to 2 and, for practical purposes, within the range 2-6. Media mixtures used in operational PRBs are shown in Table 4.2, below.

Table 4.2. Media mixtures used in operational PRBs around the world (from Younger *et al.*, 2002). References in bold typeface originated from research in the PIRAMID project.

Site	*Mixture chosen*	**Mean K ($m.d^{-1}$)**	***References***
Sudbury, Canada	20% municipal compost, 20% leaf mulch, 9% wood chips, 50% pea gravel, 1% limestone	345	Benner *et al.* (1997); Waybrant *et al.* (1998)
Vancouver, Canada	70% pea gravel, 30% compost	130	McGregor *et al.* (1999)
Shilbottle, UK	50% limestone aggregate, 25% cattle slurry screenings, 25% green waste compost	6	**Amos and Younger (2003)**
Aznalcóllar, Spain	50% limestone aggregate, 40% compost, 10% river sediment	12	**Carrera *et al.* (2001)**

4.2.3 Aluminium removal in passive systems

The preceding discussion has focused on the removal of iron and acidity in passive treatment systems, but it will be noted that the title of the section also includes removal of aluminium. The reason the discussion has not addressed aluminium thus far is because there are no design criteria for passive systems based on aluminium concentration or load.

Dissolved aluminium only occurs in elevated concentrations at pH < 4. In the range pH 5-8 dissolved aluminium is rarely present in concentrations > 1 mg/L. It is invariably the case, for mine waters at least, that at pH < 4 iron is also present, and typically in higher concentrations than aluminium. Since aluminium is only sparingly soluble at pH > 4.5, and forms a precipitate above this value, it is more difficult to remove iron from mine waters than aluminium. Therefore the design of treatment systems is based on the removal of iron for net-alkaline waters (in which aluminium is rarely a problem metal anyway), and acidity for net-acidic waters (since the main objective is to raise pH through alkalinity generation).

Aluminium is nevertheless an ecotoxic metal, and may cause unsightly coatings and froths (because of its low density) in receiving watercourses. Thus, its presence in acidic drainage should not be overlooked.

The most common reaction proposed for the removal of aluminium in passive treatment systems is (e.g. Hedin *et al.*, 1994a):

$$Al^{3+} + 3\ H_2O \rightarrow Al(OH)_3 + 3\ H^+$$

Aluminium is invariably present in solution in its trivalent form (Al^{3+}) and therefore, unlike the hydrolysis of ferric hydroxide, an oxidation step is not required to facilitate its removal. Aluminium removal as a hydroxide precipitate may therefore occur under either aerobic or anaerobic conditions.

Consequently, aluminium is likely to be removed in passive systems for treatment of net-alkaline or net-acidic waters, as long as the systems are efficient in the removal of the primary target contaminant i.e. acidity or iron.

The aluminium hydroxide precipitate formed in passive treatment systems is initially a low-density, amorphous material, which is easily re-suspended with agitation. Such turbulence may lead to the formation of the unsightly froths mentioned above, and therefore care must be taken if this is to be avoided. Over time, the hydroxide will crystallise to form one of several $Al(OH)_3$ minerals, such as gibbsite, which are all relatively stable and non-toxic (Younger *et al.*, 2002).

4.3. Implicit assumptions in apparently simple sizing criteria

The preceding discussion has provided simple sizing formulae for passive treatment units. However, engineers should not be left with the impression that this is a reflection of the simplicity of operation of systems in terms of the physical, chemical and (micro)biological reactions occurring. Even in systems where treatment is founded on purely physico-chemical reactions (such as ALDs and SCOOFI reactors), predetermining system performance with accuracy is virtually impossible. There are numerous reasons for this, of which the following are just some of the major ones (in no particular order of importance):

- The rates of many of the reactions involved in passive treatment units are influenced by site-specific environmental conditions, particularly temperature.
- Few investigations of the flow-dynamics through passive treatment systems have been undertaken. It is therefore difficult to define rates of removal with accuracy since the time element is difficult to quantify.
- In compost-based systems in particular, the net effects of the numerous (and sometimes competing) redox reactions occurring, which vary both temporally and spatially, have eluded quantification to date. It is unlikely that a generic model that can predetermine these effects with any accuracy will ever be developed.
- All mine waters are chemically unique to some degree, and this is bound to have an influence on reaction rates and overall treatment system performance.
- Wetland and compost-based systems are ‘living’ entities, and thus evolve over time. How this evolution influences water quality, within a single system and between systems, defies quantification.

It is exactly because of this complexity that it is necessary to use simple sizing criteria for design purposes. Although we have an ever increasing understanding of the specific (bio)geochemical processes occurring within passive treatment systems, for design purposes it is still necessary to treat these units as 'black box' reactors. Thus, for example, the simple design formulae of Hedin *et al.* (1994a) remain the most appropriate for design of aerobic and compost wetlands.

To emphasise the level of assumption made in designing a system, it is instructive to look at the derivation of the sizing criteria recommended by Hedin *et al.* (1994a) for aerobic wetlands. The area-adjusted removal rate of 20 $g/m^2/d$, recommended by Hedin *et al.* (1994a), was derived from data for 6 wetlands. These data are illustrated in Table 4.3.

Table 4.3. Area-adjusted removal rates for 6 wetlands in the USA, used to derive aerobic wetland design formula (after Hedin *et al.*, 1994a).

Name of wetland	Area-adjusted Fe removal rate ($g/m^2/d$)
Cedar Grove	6.3
Keystone	20.7
Morrison Ditch	19.2
REM-L	28.3
Howe Bridge Upper	42.7
REM-R	20.1

It can be seen from Table 4.3 that the rates of removal vary significantly at the different systems (from 6.3 – 42.7 $g/m^2/d$). In general terms the authors were largely able to explain these differences. Nevertheless, it illustrates that the performance of aerobic wetlands is not consistent. Rather, wetland performance depends not only upon the size of wetland constructed, but also upon the pre-treatment provided, site-specific environmental conditions, and the quality of water that is being discharged onto the wetland. For example, the poorly performing Cedar Grove wetland comprised square cells, potentially resulting in short-circuiting, and limited topographic relief meant that the provision of aeration structures was restricted.

The more conservative area-adjusted removal rate of 10 $g/m^2/d$ was recommended by Hedin *et al.* (1994a) because the authors recognised that as influent iron concentration decreases the rate of iron removal may also decrease. This concept, that iron removal in wetlands is a first-order reaction, was pursued by Tarutis *et al.* (1999), who proposed a first-order reaction kinetics model for the design of aerobic wetlands. However, although laboratory experiments show that the oxidation of iron is a first-order reaction, the complexity of reactions occurring within a wetland are such that the area-adjusted removal model (a zero-order reaction) more closely reflects overall iron removal (Younger *et al.* 2002).

Therefore the area-adjusted removal model remains the best design tool for aerobic wetlands, but the complexity of wetland systems is such that a removal rate of 10 $g/m^2/d$ cannot be guaranteed. Consequently effluent iron concentrations cannot be predetermined.

The good news is that because experience of constructing aerobic wetlands is quite wide, good engineering practices have been developed to minimise the risk of not meeting the objectives of aerobic wetland treatment. For other passive treatment units (e.g. SCOOFI reactors) our experiences are more limited, and therefore the risks are greater.

How is it possible to minimise these risks when passive treatment is the desirable water treatment solution? There are a number of ways that the risk of a system not meeting expectations can be minimised:

- Particularly for newer passive treatment units, such as SCOOFI reactors, construct and monitor pilot-scale systems prior to installing a full-scale system.
- If possible, examine successful passive treatment systems in the same region or country. Useful information regarding good engineering techniques can be gleaned from such visits. Equally, potential pitfalls or weaknesses may also be observed.
- As far as possible, design the passive treatment system so that it is flexible. For example, if 4 settlement lagoons are included in a design, try and ensure that they can operate in series as well as in parallel.
- If land and finance permit, endeavour to build safety factors into the design e.g. increase wetland size by 10-15% above the calculated area requirement.
- Consult colleagues and other experts who have constructed passive treatment systems in the past.

4.4 Special considerations for cold climates

Temperature is an influential factor in almost all chemical and microbial reactions. It therefore follows that it is an important consideration in terms of designing passive treatment systems. However, the difficulty is that many of the experiments illustrating the temperature dependence of reactions are demonstrated in a laboratory, under controlled conditions. Subsequently making use of this information in the actual design of a treatment system is very difficult, since the kinetics of a given reaction in the complex natural environment rarely reflects the kinetics determined in the laboratory.

Because of the difficulty in translating laboratory results into field applications, the following notes, on the design of passive systems in cold climates, are cautionary rather than specific.

The rate of sulphide oxidation decreases with temperature, though the relationship is not linear. For those who may wish to investigate this issue further, the influence of temperature on chemical reactions (including pyrite oxidation) be approximated by the Arrhenius equation:

$$ln(k_1/k_2) = E_a/2.3(T_1-T_2)/RT_1T_2$$

where, k_1 and k_2	=	reaction rates
T_1 and T_2	=	Temperature for reaction rates k_1 and k_2
E_a	=	Activation energy of the reaction
R	=	Gas constant

Field studies suggest that oxidation of sulphide minerals such as pyrite does not cease completely under cold climate conditions (e.g. Dawson and Morin, 1996; Elberling *et al.*, 2000). The bacterial reactions that catalyse the oxidation of pyrite are also influenced by temperature. At low to moderate temperatures the Arrhenius equation is reported to be valid for microbiological reactions. The oxidation reactions mediated by *Thiobacillus ferrooxidans* apparently reach their fastest rates at a temperature of 30ºC (Otwinowski, 1994). However, at temperatures above 30ºC the rate decreases, and therefore the Arrhenius equation does not predict reaction rates. The rate of reactions mediated by *Thiobacillus ferrooxidans* may decrease by several orders of magnitude as the temperature approaches 0ºC. However, it has been suggested that *Thiobacillus ferrooxidans* may adapt to cold temperatures given a sufficient supply of oxygen and unfrozen water (Dawson and Morin, 1996).

The major principle here is that sulphide mineral (including pyrite) weathering rates are reduced under cold climate conditions. This is clearly to be welcomed, although it is clearly not practically possible to engineer such a conditions. Perhaps of more significance is the influence that low temperatures may have on the performance of passive treatment systems.

In aerobic wetlands, the function of which (in the UK at least) is typically to remove iron, may be affected by low temperatures. The oxidation of ferrous iron (Fe^{2+}) is crucial if iron is to precipitate as a ferric hydroxide ($Fe(OH_3)$) within the confines of a wetland. Laboratory experiments show that the relevant oxidation rate is decreased by a factor of about 10 for a 15ºC decrease in temperature (Stumm and Morgan, 1996). Such decreases in reaction rates have also been demonstrated in field applications of passive treatment (e.g. Sjöblom and Håkansson, 2003).

In anaerobic systems, where the objective is often to encourage sulphate reduction processes (to precipitate metals as sulphides), treatment performance may also be reduced by low temperatures. For example, Gammons *et al.* (2000) found that the rate of sulphate reduction was reduced by 5 – 10 times during the months of winter, when mean temperatures were 2 –3ºC lower. Whilst this had little effect on the retention of cadmium and copper, the attenuation of zinc was significantly reduced.

In addition to the influence of temperature on chemical and biological reactions, cold temperatures may also have important physical impacts on passive treatment systems. These are largely unavoidable, but efforts should be made to limit their effect during the system design stage. Some of the main physical effects of cold climates are as follows:

- Ice covers across lagoons and wetlands will reduce the potential for oxygen transfer between atmosphere and water (despite the increased solubility of oxygen in water at low temperatures), which may influence oxidation of metals.
- Freeze-thaw cycles may affect the integrity of engineered structures, and large blocks of ice may also damage structures during the thaw.
- Extensive formation of ice may significantly reduce the effective volume of lagoons, which in turn will reduce retention time of water within them. Gammons *et al.* (2000) noted a significant reduction in residence time in a wetland, which was reflected in poor removal of zinc.
- During snow melt, volumes of groundwater and surface runoff may increase dramatically. This may cause flushing of oxidised metal residues on exposed surfaces and, if the water discharges to the treatment system, may significantly reduce effective retention times within the system.

Physical conditions in cold climates, however, may have some beneficial effects:

- Ice covers on settlement / tailings lagoons will restrict turbulent conditions due to wind action, and more amenable conditions for particulate settlement may result.
- Evaporation rates are lower in cold climates, and therefore the risks of lagoons and wetlands drying out is much less, and the risk of soil covers suffering desiccation is also reduced.
- Snow cover acts as insulation, and the retained heat may help to keep the rates of chemical and microbiological reactions discussed above at reasonable levels (depending upon the type of treatment system).

In terms of the design of passive treatment systems, it is not possible to make any quantitative recommendations regarding modifications that should be made to accommodate the effects of cold climates. The best recommendation that can be made is that when designing systems for discharges in cold climate locations, it may be advisable to install systems somewhat larger than the sizes calculated using the design formulae quoted through this text. If uncertainty remains, if strict regulatory limits have to be met, or if the treatment scheme will be costly, the best approach is to begin by installing a pilot-scale system. The results of water quality monitoring in a pilot-scale system will give much greater confidence in the performance of a full-scale system for a specific geographical location.

5. PASSIVE TREATMENT FOR REMOVAL OF OTHER METALS (Zn, Cu AND Mn), METALLOIDS (As), CYANIDE AND SULPHATE

5.1 Introduction

Beyond iron and aluminium there are a host of other substances (many of them metals) that may arise in drainage from mining and industrial activities, which may also cause severe pollution of surface water and ground water. Manganese and sulphate concentrations are often elevated in coal mine drainage, adding to the pollution problems associated with iron, aluminium and acidity (see sections 3 and 4). However, the removal of manganese and sulphate is much more difficult than the amelioration of acidity, iron and aluminium. The removal of zinc (which arises due to the oxidative dissolution of sphalerite, ZnS, principally in metal mines but also in some coal mine settings) is also difficult to achieve, for reasons outlined below. Although cadmium (Cd) is rarely found at very high concentrations in mine waters, its geochemical behaviour tends to mimic that of zinc, so that despite the fact that its passive treatment has not as yet been studied in any great detail, the comments relating to zinc in the following paragraphs tend to apply also to cadmium. In contrast to the relative mobility of Zn and Cd, copper (Cu) is quite readily immobilised in the form of sorbed phases and / or precipitates, and is thus farily amenable to treatment using a number of passive unit processes.

The passive treatment technologies discussed in sections 3 and 4 may remove the above contaminants to, but with the exception of Cu the rates of removal are never as great as those for iron and aluminium. To remove all of these contaminants down to low residual concentrations compliant with rigid regulatory standards using only the common types of passive systems discussed in the preceding sections would require the use of prohibitively large systems. As a consequence specific passive technologies are being developed for the amelioration of waters contaminated with Mn, SO_4^{2}, Cu and As. These nascent technologies are the focus of this section. Although cyanide does not occur naturally in mine waters, it is a common component of leachates from certain spent heap-leach pads and abandoned tailings dams, especially in gold mining areas. The possibilities for passive treatment of cyanide are therefore discussed in outline also.

Table 5.1 summarises possible sinks for a number of 'exotic' mine water contaminants. In addition to the processes listed in Table 5.1, it is important to bear in mind that nearly all of the contaminants listed are liable to be strongly sorbed to clays, ferric hydroxide particles and various types of organic matter, especially when pH is near neutral (see chapter 2 of Younger *et al.* 2002). Two important exceptions to this generalisation relate to arsenic and chromium, which are transported predominantly as oxyanions which tend to be more mobile at near-neutral pH than at very low pH.

Table 5.1. Possible removal processes for less common contaminants, which might be applicable in future passive system design (from Younger *et al.*, 2002).

Contaminant	Possible removal process for passive systems	Examples / sources of further information
Arsenic	Oxidation in the presence of iron As^{5+}, forming AsO_4^{3-}, which sorbs to Fe oxides; can also precipitate as ferric arsenate (scorodite).	McRae *et al.* (1999)
	Reduction of As^{5+} to As^{3+} in compost-based systems, forming sulphides such as AsS and As_2S_3	Cohen (1996)
Cadmium	Precipitation as a sulphide (CdS) in compost-based anaerobic systems	Cohen (1996)
Chromium	Reduction of Cr^{6+} to Cr^{3+} in compost-based systems, with hydrolysis to form $Cr(OH)_3$	Cohen (1996)
Copper	Oxidation in alkaline solution to form carbonate minerals (azurite / malachite etc)	Brown (1997)
	Reduction in compost-based systems to form sulfides	Cohen (1996); Thompson (1996)
Cyanide	Photolysis (in tropical regions) in open ponds	Young and Jordan (1996)
	Reduction to form CO_2 and NH_4^+ in compost-based systems	Thompson (1996)
	Bacterially mediated oxidation to ammonia and nitrogen gas	Thompson (1996)
Lead	Oxidation in alkaline solution to form carbonate minerals	Thompson (1996)
	Precipitation as a sulphide in compost-based anaerobic systems	Cohen (1996)
Nickel	Precipitation as sulphides in a compost-based system	Ettner (1999)
Thallium	Reduction in compost-based systems to form sulphides	Mueller (2001)
Zinc	Precipitation as sulphides in a compost-based system	Cohen (1996) Lamb *et al.* (1998)
	Precipitation as a carbonate in aerobic ponds or limestone drains	Kalin (1998) Nuttall & Younger (2000)

Table 5.2. Summary of tentative areally-adjusted removal rates for less common mine water contaminants in conventional, wetland-type passive systems, according to PIRAMID findings and re-interpretation of data reported in the literature.

Pollutant	Type of system	Areally-adjusted removal rate ($g/d/m^2$)	Comments
As	Aerobic reed-bed	18	Bacterially-catalysed under acidic conditions; value derived from PIRAMID Carnoulès data-set
Cd	Compost wetland	0.02	Cadmium can be immobilised as a sulphide (greenockite; CdS) within the anoxic substrate (see Ettner 1999)
CN^- (WAD)	Compost wetland	4	Preliminary result from PIRAMID studies in Asturias (Spain)
Cu	Aerobic reed-bed dominated by *Phragmites*	0.05	Value from a non-engineered wetland in which flow was not well-constrained; areas used to calculate removal rates likely over-estimated so that this is a minimum value (cf Brown *et al.* 1994; Brown 1997)
Cu	Compost wetland	10	Cu removal likely to be as a carbonate phase, formed by reaction with CO_2 released by microbial respiration.
Mn	Aerobic reed-bed[b]	0.5	Higher rates achievable in warm climates with algal growth[b]
Ni	Aerobic reed-bed	0.04	Preliminary findings of Eger *et al.* (1994)
Ni	Compost wetland	2	Nickel can be immobilised as a sulphide (millerite; NiS) within the anoxic substrate (see Ettner 1999)
U	Aerobic reed-bed	0.1	Single value from PIRAMID study of volunteer wetland at Boršt (Slovenia)
Zn	Aerobic reed-bed dominated by *Phragmites*	0.04	Subject to substantial seasonal fluctuations; remobilised from wetland on occasions during winter (cf Brown *et al.* 1994; Brown 1997)
Zn	Aerobic reed-bed with floating algal mats[b]	7	Removal varies seasonally and will only be as high as indicated during the growing season; it may be negative in the winter (see Kalin 1998)

[a] as $CaCO_3$ equivalent. [b] emerging subsurface flow reactors provide much higher rates of removal; see section 5.2 (Mn) and 5.3 (Zn).

5.2. Removal of less common contaminant metals and metalloids using conventional treatment wetlands.

Aerobic wetlands and compost wetlands are frequently advocated by the uninitiated as suitable unit processes for the removal of Cu, Zn, Cd, As and other 'less common' contaminants from mine waters. For a number of reasons, wetland systems are generally not the best option for removal of these contaminants. In the cases of cadmium and arsenic, for instance, even though removal rates in wetlands can be fairly high (especially for As), the toxic nature of these metals means that leaving them readily-accessible to wildlife in surficial wetland sediments is hardly recommendable. Even those metals which are less toxic to birds and mammals, such as Mn and Zn, are not ideally suited to wetland treatment because of the relatively poor removal rates they exhibit in such systems. Nevertheless, where wetlands are to be used in any case to remove iron and / or acidity from polluted mine waters, some appreciation of the likely removal rates for the less common contaminants can be helpful when making preliminary design calculations (e.g. for additional pre- or post-wetland treatment processes to remove these other contaminants by alternative means). In other words, if the rate of removal of these contaminants in wetland systems can be estimated, then it is possible to estimate design concentrations for either the water flowing into the wetland from a pre-treatment unit, or else for the water flowing from the wetland into a post-treatment unit.

With these considerations in mind, Table 5.2 summarises tentative removal rates for a number of the less common contaminants in conventional wetland-type passive treatment systems. Since these values are in many cases derived from single sites (albeit these were studied intensively) they must be regarded as preliminary only, and most of them are likely to be revised significantly in future. Nevertheless, comparison of these values with the area-adjusted removal rates recommended for iron in aerobic wetlands (i.e. 10 $g/d/m^2$; see section 4.1.1.3) and acidity in compost wetlands (7 $g/d/m^2$; see section 4.2.2.2) affords a ready appreciation of the relative mobility of these contaminants compared with the better-known contaminants.

From the evidence currently available, it seems reasonable to assume that (with the exception of manganese in the presence of iron; see section 5.2.1) removal of all contaminants listed in Table 5.2 can be expected to occur simultaneously, so that the total area of wetland needed will be that required by the most demanding contaminant.

It should be noted that a number of studies corroborate the observation that the removal of Zn in wetland systems may be only seasonal, related to biosorptive and microbial (algal) processes which are most active in summer, with little removal of Zn in the winter, or even Zn export under some conditions (e.g. Brown 1997; Kalin 1998). This behaviour contrasts markedly with that of Cu, Cd, Ni and As, which appear to be strongly fixed within the wetland substrates once they are removed from solution.

5.3. Passive treatment for manganese removal

5.3.1 Introduction

Manganese is a common contaminant in many mine waters and, though not as ecotoxic as other common contaminant metals found in such waters (such as Fe, Al and Zn), it nevertheless has various undesirable properties, including a propensity for precipitating in water distribution pipe networks (eventually causing blockage of supply pipes), imparting an unpleasant 'metallic' taste to drinking water, and staining laundry. Manganese removal is notoriously difficult using either active

or passive treatment systems. The reason why manganese is generally more difficult to remove from water than iron is because a higher pH is generally required, and the kinetics of manganese oxidation are much slower than for iron (Stumm and Morgan 1996).

The slower oxidation kinetics of manganese are reflected in the area-adjusted removal rates recommended for aerobic wetland design for manganese amelioration (Hedin *et al.*, 1994a): 0.5 – 1.0 $g/m^2/d$ compared to 10 – 20 $g/m^2/d$ for iron. Furthermore, Nairn and Hedin (1993) report that no manganese is removed in aerobic wetlands as long as dissolved ferrous iron is present at concentrations >1 mg/L. This apparently major constraint on the design of passive treatment systems for the removal of manganese, summed up in the aphorism that "you can't remove your manganese until all of your iron is gone", needs to be tempered by consideration of what can happen to dissolved Mn when ferrous iron is oxidised to the ferric form. Recent field evidence from a passive treatment system which receives raw mine water with around 40 mg/L Fe^{2+} and between 1 and 4 mg/L Mn^{2+} shows that suspended flocs of ferric hydroxide (as opposed to dissolved Fe^{2+}) *are* capable of effectively 'scavenging' dissolved Mn from the water column by sorption. On the site in question, this occurred in the settlement lagoons which immediately follow the primary aeration cascade (Nuttall 2003). With an average retention time in the lagoons of around 24 hours, nearly 20% of the dissolved Mn present in the raw mine water was removed in this manner, at a rate approximating 0.4 $g/m^2/d$. Further Mn removal in the succeeding wetlands occurred at a very slow rate (0.03 $g/m^2/d$), but given the size of these wetlands and the high removal rates achieved in the lagoons, the overall effect was to remove a further 15% of the Mn, which was sufficient to ensure a final discharge with < 0.5 mg/L Mn. This example illustrates that the antagonism between *dissolved* Fe^{2+} and dissolved Mn^{2+} does not extend to the interactions between manganese ions and highly-sorptive, fresh $Fe(OH)_3$ precipitates.

5.3.2. Current specialist passive technologies for manganese removal

Because of the prohibitively large size of aerobic wetlands required for the removal of Mn^{2+} down to low residual concentrations, more intensive processes are of considerable interest. One approach is to pass wetland effluents through oxic 'rock filters' (i.e. shallow ponds with a substrate of large cobbles which frequently rise above the water surface) which host algal and / or bacterial consortia (Phillips *et al.* 1995).. These microbial consortia form 'mats' in which micro-environmental conditions, (most notably high pH) promote manganese removal by means of oxidation and precipitation of characteristically jet-black deposits of MnO_2 (usually X-ray amorphous homologues of the mineral pyrolusite). However, the microbial mats require light to support photosynthesis, and are thus prone to under-perform in turbid waters. Winter temperatures often kill off the mats altogether (Phillips *et al.* 1995), with the consequence that this approach is most likely to be successful in tropical regions where air temperatures are warm year-round.

An increasingly popular method used in the USA for passive removal of manganese is the 'Pyrolusite Process®', in which a bed of limestone is inoculated with Mn-oxidising bacteria. Heavy deposits of MnO_2 form within the biofilm which, over time, transforms into its crystalline equivalent, pyrolusite. Although a naturally occurring reaction, in the patented Pyrolusite Process® aerobic bacteria are cultured in the laboratory (on a site-specific basis), for subsequent inoculation in the reactors. Such reactors are finding increasing application in the coal fields of the eastern USA (Younger *et al.*, 2002).

Effective manganese removal in the 'rock filters' of Phillips et al. (1995) requires well-aerated water, prior removal of essentially all dissolved Fe and Al, and pH above about 6.5. Since oxidising conditions are required there is still the requirement for the treatment system to be shallow, necessitating large land area requirements. Removal rates for these systems range from approximately 1.5 – 5 $g/m^2/day$, with residence times of at least 8 hours.

5.3.3. Novel passive treatment method for manganese removal

Recent work by Johnson (2002, 2003) appears to represent a breakthrough in passive manganese treatment, since it holds forth the possibility of manganese removal using very little land area. While the process does depend on Mn-oxidising bacteria, it is effectively kick-started without elaborate micorbial inoculations simply by means of installing two active ingredients: dolomite and manganese dioxide. Dolomite has been shown to be a much more suitable substrate for manganese removal systems than limestone. Although it does not generate as high a pH as does calcite, dolomite has a greater catalytic effect on manganese oxidation (Johnson 2002, 2003). Oxidising conditions are maintained at depth within the system using passive aeration technology (patent pending). This aeration ensures that the treatment system is robust enough to operate at low temperatures, in complete darkness and also in the presence of ferrous iron. The combined effects of both the catalysts and the aeration provide conditions under which the normally slow kinetics of manganese oxidation are significantly accelerated. However, although simultaneous iron and manganese deposition is feasible in this system, one problem is that iron oxyhydroxides are much more voluminous (i.e. less dense) than the equivalent manganese oxide / oxyhydroxide deposits. Consequently hydraulic conductivity decreases more rapidly in a system in which iron is present.

Mn removal rates achieved using this system are generally in the range 0.5 – 3.0 $g/m^2/d$, with peak removal rates reaching as high as 60 $g/m^2/day$. These values are an order of magnitude greater than removal rates quoted by Nairn and Hedin (1993) and demonstrate the ability of this type of treatment system to overcome the slow oxidation kinetics usually associated with manganese oxidation. Residence times in these reactors do not need to be very long: experimental reductions in retention time to less than 8 hours were achieved without noticeable reductions in manganese removal efficiencies. Furthermore, as operation of this passive treatment process continually generates fresh manganese dioxide (which is a very powerful sorbent for most pollutant metals) it has major ancillary benefits as a removal process for other mobile metals such as zinc (Johnson 2002, 2003).

At the time of writing manganese removal is not a major priority at most sites in the UK, at least not for discharges from abandoned coal mines (volumetrically the main source of mine water pollution in the UK). However, increasing legislative pressure from the EU will undoubtedly result in a need to address manganese removal in the future. The work of Johnson (2002, 2003) certainly represents a promising addition to the more established Pyrolusite Process®, and the short residence times reported in Johnson's process are a distinct advantage. However, since the process is still at the development stage full-scale applications should be approached with caution at this time, at least without preliminary field pilot-scale trials. The quoted Mn removal rates of 0.5 – 3.0 $g/m^2/d$ will doubtless be refined once further field-scale trails have been undertaken.

5.4. Carbonate-based reactors for Zn, Cu, Cd and As removal

The use of carbonate minerals has found application to the removal of some of the less common contaminants occasionally present at high concentrations in mine waters. Again, the technologies discussed below are still at early stages of development, and thus cannot be considered to be 'proven technologies'. However, it seems likely that in due course full-scale applications of such technologies may be forthcoming (albeit perhaps in modified forms).

Nuttall and Younger (2000) have proposed the use of calcite dissolution in a closed-system as a means of precipitating zinc as $ZnCO_3$ (smithsonite). The process, pioneered on hard, circum-neutral waters, relies on attaining a pH of approximately 8.2, at which point zinc becomes insoluble as its carbonate. A residence time of 8 hours was concluded to be the optimum for removal of 20-40% of the zinc from this water. Further reactors, in series with an aeration step between, would be required to facilitate greater treatment efficiencies. Thus, these closed-system reactors can be designed in the same way as ALDs (section 4.4.2.1), with the sole exception that φ be set equal to 8.

While it is possible to use oxidation ponds and wetlands for the removal of arsenic and cadmium from mine waters (see section 5.2), the open-air accumulation of As- and / or Cd-rich solids is not recommended on public health and wildlife conservation grounds. One possible alternative process, which results in sub-surface entrapment of arsenic where it cannot contact macrofauna, has been proposed by Wang and Reardon (2001), who propose the use of a packed-bed reactor filled with siderite ($FeCO_3$) and calcite, which reacts to remove both arsenic and cadmium from waters which are already low in dissolved iron concentration (either naturally or following some other treatment process). Siderite dissolution is achieved in a saturated flow reactor, which results in an effluent with Fe concentrations $\leq$ 15.2 mg/L. The water is subsequently aerated over calcite clasts, such that cadmium is precipitated as its carbonate, $CdCO_3$ (otavite), due to degassing of CO_2 and dissolution of O_2. Simultaneously ferric hydroxide is precipitated, and this acts as a strong sorbent for AsO_4^{3-}. Wang and Reardon (2001) report removal of As and Cd to concentrations below detection limits when the residence time of water within the reactor is 2 hours. Thus, as above, the ALD sizing criteria can be used, but this time with φ equal to 2.

5.5. Attenuation of problematic metals using caustic magnesia

Laboratory-scale experiments conducted during PIRAMID have revealed the potential utility of caustic magnesia as an effective media for the attenuation of problematic metals such as Zn, Mn and Cu (Cortina *et al.* 2003). Given the dearth of viable, rapid passive unit processes for these contaminants, these experiments merit reporting in some detail.

Caustic magnesia is a by-product of the production of magnesium oxide when the process used is calcination of magnesium carbonate. The main constituents of caustic magnesia are MgO (approximately 75% by weight) and CaO (approximately 10% by weight). Saturated column experiments, of duration 3 – 10 months, were conducted using laboratory-prepared mine waters which were designed to be analogous in quality to the acid mine waters arising in the Rio Tinto area of SW Spain. The first batch of experiments, conducted over a period of 3 months, investigated columns comprising caustic magnesia only. Over 80% of the particles of caustic magnesia fall within the grain size 0.5 – 4.0 mm, and the effective porosity of the media is approximately 0.3. Residence time of water within these columns was calculated (with the aid of tracer tests) as 1.3 – 1.6

hours. Influent water quality for the first experiment was pH 3, 100 mg/L Zn, 50 mg/L Cu, 10 mg/L Al, 20 mg/L Fe, 360 mg/L Ca, and 960 mg/L SO_4. There were significant reductions in concentrations of metals during this experiment, associated with the high pH conditions of the treated water. The high pH was caused by dissolution of lime initially (giving pH ca. 12), followed by dissolution of brucite ($Mg(OH)_2$) (giving pH ca. 8.5), once the lime was exhausted.

Mineralogical analyses indicated that significant metal removal during this experiment may have been linked to co-precipitation reactions of metals with amorphous Fe and Al oxy-hydroxides. Therefore a second group of experiments was conducted using waters with only one major contaminant (either Zn, Cu, Mn or Pb). In these experiments it was found that, for waters containing Zn and Cu, pH dropped to 6 before the conclusion of the experiments. This was attributed to armouring / clogging of the caustic magnesia by Cu and Zn precipitates. Under the lower pH conditions the reactors became a net *source* of Zn and Cu, due to re-dissolution of the metal precipitates initially formed. In contrast, Mn continued to be removed effectively throughout the 10-month period of the experiment. Use of caustic magnesia may therefore offer a promising addition to the Mn removal processes discussed in section 5.4, above, for Mn-only polluted waters.

In an effort to increase the effective porosity of the media (and therefore reduce problems of clogging), a final set of experiments were conducted using a mix of caustic magnesia and quartz. The effective porosity of these reactors was in the range 0.45 – 0.58. The quality of water used for these experiments was pH 5.5, 75 mg/L Zn and 1000 mg/L SO_4. This water quality was chosen to reflect the fact that the researchers considered that the use of a compost / calcite treatment system (e.g. RAPS, compost wetland) would need to precede a caustic magnesia reactor, partly in order to remove Fe and Al, and partly because the use of caustic magnesia may be more economic if pH is raised initially with these lower cost media.

Using at least 50% caustic magnesia, the pH of the effluent water from the mixed-media systems was consistently around 9, there was no indication of a drop in pH, and removal of Zn was highly efficient. It was concluded that such reactors may be a useful addition to the (limited) range of systems currently available for removal of problematic metals. Further, larger-scale, experiments may help to derive metal-removal rates (and hence design guidance) for such systems (Cortina *et al.* 2003).

5.6. Use of zeolites for sorption of metals from mine waters

Zeolites, minerals consisting of hydrated aluminium silicates of Ca, Na, K and Ba, are known for their strong ion exchange capacities. To harness this property of zeolites, PIRAMID investigations have probed the attenuation of metals from acid mine waters in their presence. The zeolites used were synthesised from fly ashes produced in Spain (from Teruel and Narcea). The water used for the experiments was from two boreholes contaminated by the Aznalcóllar tailings spill, and also from the Rio Tinto.

Preliminary experiments demonstrated that the zeolites were capable of retaining a significant proportion of the zinc and copper in reactor influent waters. This is despite relatively high concentrations of calcium in the water, which competes for exchange sites on the zeolite. The optimal dose of zeolite was concluded to be in the range 10 – 30 g/L, depending upon the concentrations of major cations in the water, and the specific zeolite used. Using a zeolite dose of 40 g/L metal removal from the borehole waters, in the laboratory-scale reactors, was as follows:

Zn 174 mg/L to 0.2 mg/L
Mn 74 mg/L to 6.0 mg/L
Cd 400 µg/L to 0.1 µg/L

Other metals were also shown to be removed by the zeolite reactors. Qualitative interpretation suggested that the zeolite had the following affinity for metals (from strongest to weakest):

$$Fe^{3+} = Al^{3+} > Cu^{2+} > Pb^{2+} > Cd^{2+} > Zn^{2+} > Mn^{2+} > Ca^{2+} = Sr^{2+} > Mg^{2+}$$

The only drawback observed with this cation exchange reaction was the release of high concentrations of sodium (from 54 mg/L to 490 - 1100 mg/L), and minor releases of chromium, vanadium, molybdenum and arsenic (all of which are present in the raw fly ash).

Treatment of the Rio Tinto water using these reactors produced somewhat different results. This was due to the presence of elevated concentrations of Fe^{3+} and Al^{3+}, which were preferentially exchanged, and therefore restricted the number of exchange sites available for other metal contaminants. Thus, an alternative treatment unit may be required to remove Fe and Al, upstream of the zeolite reactor, if this technology were to be applied at full-scale.

In all of the experiments there was a rise in pH between influent and effluent. It is therefore difficult to assess the relative contributions of ion exchange and metal precipitation (as hydroxides) in the reactors. Certainly it appeared that the predominant removal mechanism for iron and aluminium was the precipitation of amorphous oxy-hydroxides.

From the range of experiments carried out it appears that cation exchange on zeolite media may be an effective remediation measure for some metal-polluted waters. However, the volume of waste produced is approximately 4 times greater than that produced by conventional lime treatment, and therefore the greatest utility of zeolites may be as a polishing treatment, particularly for the removal of metals such as copper, zinc and cadmium.

Further experiments are required before design criteria are forthcoming, and long-term experiments may be required to quantify the time-scale for exhaustion of exchange sites (effectively determining the lifetime of such systems).

5.7. Passive treatment of cyanide residuals from tailings dam leachates

Cyanide destruction is commonly practised at working gold mines, and is such an efficient and effective process that there is little motivation to develop alternative passive unit processes for most applications. However, where tailings dams at former gold mining sites are decommissioned, it is possible that leachates from these tailings may contain modest concentrations of cyanide. Tailings generally leave gold processing plants with variable quantities of cyanide (usually about 10 mg/L, but occasionally higher) and elevated pH (around 9.5). Natural degradation of cyanide in tailing ponds is a well-known phenomenon. It appears that natural degradation of cyanides in tailing ponds occurs as a result of the interaction of several processes, including volatilisation, hydrolysis, photodegradation, dissociation, chemical and bacteriological oxidation and precipitation. While these processes can be effective 'passive' cyanide destruction methods in themselves, they are mainly active in open waters receiving solar radiation (and are therefore most vigorous in warm climates). By contrast, deep within the tailings pore waters, modest concentrations of cyanide may well persist for many years or even decades. To assist in the safe and sustainable long-term decommissioning of gold mine tailings dams, therefore, preliminary investigations have been made of low-intensity, passive options for cyanide destruction.

The chemistry of cyanide is complex, and many forms of cyanide exist in the effluent from a gold mining plant: at 25 °C cyanide can exist in a solid, liquid or gaseous state! From a toxicity point of view there are four major categories of cyanide compounds, which are, in approximate order of increasing stability:

- Free cyanide
- Iron cyanide
- WADs (weak acid dissociable cyanides)
- Cyanide related compounds

In aqueous solution cyanide may be present in its free form or as some cyanide complex, with the form dependent upon the pH and redox potential of the solution. Free cyanide (HCN and CN^-) is acutely toxic to humans and animals if ingested or inhaled. Cyanide in the form a complex is not as toxic as free cyanide (the toxicity of complexed cyanide relates to the potential breakdown and release of free cyanide, rather than the inherent properties of the compound itself).

One of the most appropriate options for passive cyanide destruction, in terms of minimising exposure risks etc, is the reductive hydrolysis of cyanide to formic acid and ammonium formate. This can be achieved in compost-based bioreactors / wetlands like those used for treating acidic drainage. Experiments to date suggest that WAD removal in such systems can be expected to average around 10.6 $mg/m^3/d$, which equates to an area-adjusted removal rate of 4.2 $g/m^2/d$ for a substrate 0.5m in depth. This removal rate may be a conservative starting point for the design of full-scale systems. Site-specific pilot testing is nevertheless advisable until a sufficiently large number of such systems exists to allow confident adoption of design cyanide removal rate values.

5.8. Sulphate removal in passive treatment systems

The removal of sulphate in passive treatment systems is arguably more difficult than for any other contaminant thus far discussed in these guidelines. Although it is not seen as a priority in the temperate climates of northern Europe, in more arid zones, such as the Mediterranean (and certainly in countries such as South Africa), the scarcity of water necessitates that sulphate removal from mine waters and industrial drainage be considered.

The aim when addressing sulphate attenuation in passive systems is to remove it by reduction to HS^-, followed by (limited) oxidation to remove elemental sulphur (S^0) as a solid. Thus, the main part of any system for sulphate removal is a compost-based reactor, which must maintain strongly reducing conditions. In compost wetlands (designed primarily for acidity removal), decreases in sulphate concentration are usually < 20% (e.g. Jarvis, 2000), in part reflecting the great molar excess of sulphate over metals (the removal of sulphur is attributed to precipitation with iron and other metals as monosulphides). The degree of bacterial sulphate reduction in compost reactors for sulphate removal must therefore be significantly greater. Conversely, oxidation of the HS^- produced must be controlled if oxidation back to SO_4^{2-} (with concomitant re-acidification of water) is to be avoided (Younger *et al.* 2002).

Younger *et al.* (2002) provide a brief review of research to date into compost-based reactors for sulphate removal, and distil the key points from these studies as follows:

1. A suitable source of carbon, that will support bacteria in the long-term, must be identified.
2. The hydraulics of the system must be such that highly reducing conditions can be maintained.
3. Tentative design values for SO_4^{2-} removal in such systems range from 300 millimoles per cubic metre per day (mM/m^3/d) (Gusek, 1998; Lamb *et al.*, 1998) to approximately 800 mM/m^3/d (Willow and Cohen, 1998).
4. The hydraulics of the system must also be arranged so that high residence times can be achieved (at least 40 hours), but without resulting in either clogging of the substrate or short-circuiting of it.
5. For long-term planning purposes, it is necessary to be able to quantify the rate of consumption of organic matter.
6. The oxidation of HS^- leaving the system must be controlled such that direct oxidation back to SO_4^{2-} does not occur.

Research into these various fields is on-going. At the current time design of full-scale passive systems for sulphate removal is not advisable without considerable research, and wide consultation with experts in the field. The attenuation of sulphate is one area in particular where, if strict regulatory requirements need to be met, entailing the construction of a high capital cost scheme, active treatment may be the preferable option.

6. DESIGNING COVER SYSTEMS FOR PASSIVE PREVENTION OF POLLUTANT RELEASE

6.1. Introduction

Passive prevention of pollutant release from mining waste is achieved by the surface or subsurface installation of physical barriers (requiring little or no long-term maintenance) which inhibit pollution-generating chemical reactions (for instance, by permanently altering redox and / or moisture dynamics), and / or directly prevent the migration of polluted waters. For pyrite-rich mine waste, the basic principle is to minimise the oxygen supply in order to reduce the weathering of pyrite. In this context, passive prevention relates to the use of dry and water covers. A dry cover is a layer of material that overlies sulphide mineral-rich waste, such as that often found in spoil tips and abandoned tailings facilities. A water cover is essentially a lagoon, that may contain tailings material or other sulphidic wastes.

In both cases the objective of the cover is to prevent (or at least reduce to negligible levels) the weathering of the sulphidic mineral (which is the root cause of mine water pollution). Thus, the philosophy is somewhat different to the (curative) mine water treatment techniques discussed elsewhere in these guidelines.

The oxidation of sulphide minerals is either directly or indirectly driven by the accessibility of pyrite (or other sulphide mineral) surfaces to oxygen. Conventional techniques for the prevention of mine drainage are therefore designed to minimise the supply of oxygen to pyrite and other sulphide minerals. As the oxygen transport into mine tailings generally proceeds by diffusion (advection and convection may be more important in waste rock piles), covering the tailings with water or water-saturated soil reduces the diffusion of oxygen to the waste significantly, thereby limiting the oxidation of sulphides. The oxidation of sulphides is significantly reduced since the diffusion coefficient for oxygen in water is a factor of 10^4 lower than the diffusion coefficient for oxygen in air (Evangelou *et al.*, 1998).

In addition to limiting the oxygen diffusion in the pore spaces of mine waste, a water-saturated barrier also sets an upper limit on the oxygen concentration available for sulphide oxidation. The solubility of oxygen in water will determine its maximum concentration in the waste pore water. For a given temperature, the solubility of oxygen in water is controlled by the partial pressure of oxygen according to Henry's law.

Although a well-constructed cover system may prevent sulphide mineral oxidation, the potential for future pollution remains. Therefore one of the most important considerations when designing a cover system is its long-term integrity. These issues are discussed further in section 6.5 of these guidelines.

Water covers and dry covers are primarily appropriate for abandoned tailings facilities, waste rock piles and spoil heaps. The objective is to provide a long-term barrier that will prevent the diffusion of oxygen into areas rich in sulphide minerals, thus preventing the generation of polluted waters. A secondary objective may be the effective reclamation of such waste repositories, particularly in the case of spoil tips and waste rock piles, and therefore dry covers may be designed in such a way that they will support a variety of vegetation.

6.2. Mine wastes and the development of spoil heaps and tailings ponds

Spoil heaps and tailings ponds are the principal repositories for mine wastes. Given the present-day prevalence of surface mining, which results in the disturbance of far more barren rock than deep mining, the volumes of mine waste generated at this point in history are impressively large. Hartman and Mutmansky (2002) estimate that some 70% of all the material excavated during mining operations world-wide is waste. Even though active mining is now a relatively modest component of total economic production in the EU, the European Commission (2003) has recently estimated that the mining industry still produces about 29% of total waste generated in the EU each year, with an annual volume in excess of 400 million tonnes.

These mine wastes fall into two main categories:

- Waste rock or spoil ('overburden') and
- Tailings (sometimes referred to as 'finings' in the coal industry)

Tailings are only generated during mineral processing, whereas spoil is generated throughout a mining operation i.e. before and during mining, and during mineral processing. However, the main difference is in grain size: tailings are generally fine particles (say < 1 mm), whereas spoil can be comprised of particles in the range 1 mm – 50 mm (and sometimes greater).

Spoil heaps[20] are usually created by loose tipping of waste material. It is rare that any intentional compaction of the spoil is undertaken. Sorting of the various grain sizes is generally by gravity alone, with large particles typically falling to the toe (lower edge) of the heap, and finer particles remaining in the upper layers. Although in recent times active efforts have been made to ensure revegetation of spoil heaps, this has not always been the case. Consequently there are many exposed spoil heaps across Europe. These are a significant source of (often acidic) mine waters. Indeed, the quality of waters arising from spoil heaps (either as surface runoff or as seepage from the toe of the tip) is often worse than that from deep mines, because of the greater potential for oxidative dissolution of sulphide minerals in the comparatively shallow, open structures of these deposits.

Tailings arise from mineral processing operations. Following crushing and grinding, processing of ores is typically by techniques such as gravity separation, froth flotation or leaching. All of these operations are carried out in water, with one or more organic or inorganic compounds being added depending upon the process being employed. Some of the water from these processes will be recovered for re-use, but there is always a remainder, heavily charged with fine particles, that must be disposed of. The disposal of this water is invariably to a tailings lagoon. The water content of the tailings reduces over time, through natural processes of gravity drainage and evaporation. Ultimately a desiccated crust will form on the surface of the lagoon (typically of around 0.5 m thickness), but the lower layers may remain high in water indefinitely.

Rather than treating polluted drainage arising from spoil heaps and tailings lagoons, a more holistic solution is to prevent the oxidative dissolution of the sulphide minerals in the first place ('prevention is better than cure'!). For this reason 'capping' of spoil heaps and abandoned tailings facilities is

[20] also known as 'waste rock piles' (Canada, Australia), 'gob piles' (USA), 'bings' (Scotland; NZ) and 'culm banks' (India), amongst other terms.

now widespread practice. This section of the guidelines outlines the technical and engineering aspects of such capping operations.

6.3. Technical aspects of dry and water covers for spoil heaps and tailings

6.3.1 Dry covers

The main objective for dry covers is to prevent the ingress of oxygen to the spoil dump / tailings, so as to limit the potential for oxidation of the sulphide minerals therein. A second objective may be to limit the infiltration of surface waters. This will both help to reduce oxygen transfer to the spoil material, and will reduce the volume of leachate discharging from the toe of the tip.

With regard to leachate, improvements in quality and quantity should not be expected until the cover has become established, since the necessary oxygen depletion (to limit sulphide mineral oxidation) will take time to develop. Also, it is very difficult (as with many passive treatment systems) to predict the long-term leachate quality. This is primarily because there are so few systems that have been monitored over sufficiently long timescales.

Dry covers can be classified according to their function, as shown in Table 6.1.

Table 6.1. Classification of dry covers according to their function

Cover type	Primary function
Oxygen diffusion barriers	To limit the supply of oxygen by acting as a barrier against diffusion of oxygen into the waste
Oxygen consuming barriers	To limit the supply of oxygen by consumption of oxygen which penetrates into the cover
Low hydraulic conductivity barriers	To limit the supply of oxygen and the formation of leachate by acting as a barrier against the diffusion of oxygen as well as the infiltration of precipitation
Reaction inhibiting barriers	To provide a favourable environment to limit reaction rates and metal release

Oxygen diffusion barriers must contain a layer with a low effective diffusivity for oxygen. The best way of accomplishing this is to ensure that the cover can remain saturated with water. The barrier material must, therefore, be able to withstand drying and retain water. Since the water retention capacity of soils is enhanced by fine pore size, such materials are recommended. Ideally covers should be as close to the groundwater level as possible, as this limits capillary suction (which may also lead to drying of the barrier). A single layer cover is typically adequate, as long as it comprises fine-grained materials such as clays or clayey silts. Such a cover can form an effective oxygen barrier, particularly where shallow groundwater tables favour water retention within the cover. Where groundwater tables are deeper, caution needs to be taken because of the risk of the cover draining. As a successful example, the work of Lindvall *et al.* (1999) describes the application of single layer covers on tailings ponds at the Kristenberg Zn-Pb mine in northern Sweden.

The calculated values of Naturvårdsverket (1993) suggest that a 1.0 m thick single layer cover of till should result in a reduction in pyrite weathering of approximately 80%, and a 1.5 m thick layer should result in a 90% reduction. It is recommended that the hydraulic conductivity of layers should be lower than 5 x 10^{-9} m/s, and preferably ≤ 1 x 10^{-9}.

Frost action may have a deleterious effect on covers, and in these conditions they should be protected. The simplest way to do this is to increase the depth of the cover layer. Some calculations for northern Europe, where the possibility of snow cover is low (and therefore the ground is not insulated), suggest that a layer may need to be in the order of 1.5 – 2.4 m thick, depending on the exact location. In arctic and sub-arctic regions, the possibility of not having snow cover is very low. As a consequence, in Sweden the standard depth for a single layer cover is 1.5 m (Furugård, 1985). Conversely, the thickness of a clay layer required to resist frost action in the USA is considered to be in the range 0.3 – 1.8 m (Chamberlain *et al.*, 1997). The best way to assess the necessary depth of a cover layer is to statistically analyse historical observations of the frost index and snow depth at the particular site of interest. More general recommendations for estimating frost depths can be found in geotechnical design guidelines such as Knutsson (1984).

Multi-layer covers are an alternative to the single layer cover, and have the advantage that, in theory, water held in the cover layer will not be lost due to evaporation or drainage. The principle of the capillary barrier effect is that a fine grained material, with a high capacity to hold water, is placed between two coarse-grained layers ('capillary breaks') that do not have the ability to transfer capillary suction (Rasmuson and Eriksson, 1987).

The capillary barrier concept has been investigated at field scale. Bell *et al.*, (1994) constructed a multi-layer cover as part of the reclamation of the 2,500 m^2 waste rock pile at Heath Steele Mine, New Brunswick, Canada. A compacted fine-grained till was used for the 0.6 m thick capillary layer, and was placed between two 0.3 m thick layers of sand. The barrier was covered by a 0.1 m superficial layer as protection against erosion. The cover was established in September 1991. Hydraulic conductivity tests by single-ring infiltrometers indicated a hydraulic conductivity in the capillary layer of 1 x 10^{-8} m/s. A subsequent study showed no significant change in the water content of the till during the reported period (until May 1993). The oxygen concentration in the pile decreased successively from 18 – 21 % in May 1991, to 0.8 – 1.1 % in May 1992, and then to 0.1 – 0.2 % in May 1993, demonstrating the effectiveness of the cover layer.

Depending on the particular characteristics of the site, capillary barrier covers can be an economically feasible passive prevention system. The technology has been proven at full-scale at several mine sites. However, the importance of adequate characterisation of the construction materials, and modelling of the actual site and water budget prior to design and construction of a multi-layer barrier, are vital if implementation is to be successful (Aubertin *et al.*, 1997; Ricard *et al.*, 1997).

Oxygen consuming barriers are an alternative to oxygen diffusion barriers, which rely on microbial consumption of oxygen within the cover layer. Dumping of wood wastes on the tailings at East Sullivan Mine in Quebec, Canada, began in 1984 (Tremblay, 1994). In 1990 active management of wood waste, with a view to reclamation, started. The cover consists of 2 m of organic waste (85 % bark, 10 % pulpwood and 5 % sawdust). A follow-up study performed in 1991 showed that the oxygen decreased with depth in the cover. At a depth of 0.7 m the oxygen concentration in the pore gas was 1.5 %. Monitoring also showed increasing pH and decreasing metal release in leachate from the covered areas. On a cautionary note, some organic contamination with low concentrations of phenol and tannin were found. A vegetative cover may be advisable, since this will provide additonal biomass as the organic media is used up.

Low hydraulic conductivity barriers are commonly used to limit the infiltration of precipitation into landfills. In addition, low-hydraulic conductivity soil generally shows favourable water retention properties, such that a high degree of saturation may be mantained in the barrier. Limiting the percolation of water through the barrier also leads to water saturation in a portion of the layer above the barrier for long periods, unless this layer is drained. Consequently, low-hydraulic conductivity barriers also act as water saturated layers, preventing oxygen transport, provided they are covered by a protective layer.

Acting as a barrier against both the diffusion of oxygen and percolating water, covers including low-hydraulic conductivity barriers often show high potential for limiting the generation of metal-polluted acid mine drainage from mine waste dumps. Reductions in the oxidation rate of pyrite from 95 % up to more than 99 %, and a probable reduction of the percolation rate from 80 % up to more than 95 %, depending on the soil type in the barrier, have been reported (Naturvårdsverket, 1993).

Low-hydraulic conductivity barriers can be constructed of fine-grained soils, mainly clay and clayey till, geosynthetic clay liners (geotextile/bentonite liners), geomembranes (plastic liners), cement-stabilised products and some fine-grained residues from industrial processes (mainly sludge) (MEND 1994; Lundgren, 1995).

In arid and semi-arid climates, where rainfall is infrequent but usually of very high intensity on the rare occasions it falls, low-hydraulic conductivity covers may be subject to severe erosion by surface runoff. To counteract this problem, a particular variety of cover technology, termed 'storage-and-release covers' (Durham *et al.* 2000; Younger *et al.* 2002), has been developed. These covers combine a high-hydraulic conductivity surface layer (gravels) with a lower-hydraulic conductivity horizon (clays) at depth. Typically, the coarse-grained surficial layer will be deliberately shaped into hummocks, resulting in a micro-topography which inhibits the development of direct surface runoff (and therefore surface erosion) and favours infiltration. The infiltrating water ponds above the compacted clay layer, temporarily functioning as an oxygen diffusion barrier in the same way as a conventional water cover. This ponded ground water can later be released by slow, lateral flow to a carefully-engineered seepage zone, or else retained *in situ* until such time as evapotranspiration by trees rooted into the gravel can remove the moisture. Although most examples of storage-and-release covers to date are in Australia, they clearly have potential applicability in Mediterranean Europe and other dry regions.

Reaction inhibiting barriers are a novel passive prevention technology, which make use of the ability of moderately soluble ferric phosphate precipitates to immobilise heavy metals, such as lead and cadmium, as phosphates (Evangelou, 1994; Kalin *et al.*, 1997). The technology has only been tested at bench-scale, but there may be potential for using phosphate-containing covers to both inhibit pyrite oxidation and attenuate release of heavy metals such as lead and cadmium.

6.3.2 Compaction

In some circumstances it may be possible to generate a capillary rise barrier through compaction of the waste material. By reducing the porosity of the waste its water retention capacity is increased. This may be particularly applicable at new waste dumps, where self-consolidation of the material (i.e. due to its own weight) has not begun. The water content of the material must be at an optimum. If it is too dry then friction will prevent effective compaction, but if it is too wet, locally saturated zones will develop, making further compaction impossible.

The weight of a soil cover itself will facilitate compaction, thus helping to increase the water retention capacity of the waste. To be able to assess and predict the performance and sustainability of remediation by compaction and / or increased capillary rise due to soil covering, experimental determination of material characteristics and site specific hydrogeological and meteorological assessments must be undertaken. It is this information that forms the basis for sizing cover systems. For the collection and interpretation of such data specialists should be consulted.

6.3.3 Water covers

Water covers can be created in three ways:

1. Sub-aqueous emplacement of mine tailings in a natural lake or
2. Submergence of an existing body of spoil or tailings
3. Maintenance of standing water cover in a formerly-active tailings impoundment

The first of these options, though implemented with some success in the past in Norway (Arnesen *et al.* 1997), is no longer encouraged by current EU legislation, and as such is not discussed further here.

The second option can be implemented where site conditions permit spreading of the spoil or tailings over a larger area than they previously occupied, and then impounding water such that all of the mine wastes are permanently submerged. For long-term submergence to be achieved, site conditions have to be very favourable, and few sites are likely to be amenable to this approach. Nevertheless, good examples exist in northern Sweden and in Canada. On the other hand, this approach is unlikely to be feasible in warm parts of Europe where evaporation rates are very high.

Long-term retention of standing water in a previously-active tailings impoundment is a significant engineering challenge, the success of which depends on the geotechnical stability of the tailings dam, the permeability of the dam and the impounded tailings, and the availability of inflowing water to compensate for that lost to evaporation and seepage. Details on the construction of tailings dams and analysis of seepage through them are beyond the scope of these guidelines, and the interested reader is referred to specialist texts such as that of Vick (1983).

6.3.4 Influences on the long-term performance of cover layers

Processes that may influence the performance of a cover layer over time can be divided into physical, chemical and biological factors. Important physical processes are:

- Erosion, e.g. at the surface due to flooding, ponding and surface runoff; inner erosion or slides, due to the development of excessive pore pressures or high gradients.
- Repeated drying and re-wetting leading to fracture formation, settling etc.
- Freeze and thaw damage. For cold climates it is recommended that the cover be designed so as to guarantee a freeze-protected depth for the sealing layer.
- Consolidation should be achieved in a controlled way, e.g. by compaction. This may lead to differential settling and fracture formation.

- Slides, particularly in wall sides. Significant risk if high pore pressures occur.

Important chemical processes are:

- Dissolution of specific mineral grains in the cover material, e.g. calcite, clay minerals, sulphide grains etc.
- Osmotic effects caused by differences in salinity between the tailings and the cover material. This could cause increased hydraulic conductivity in certain clay materials.
- Precipitation/cementation: dissolved ferric (oxy)hydroxides, for example, may be transported towards the surface by capillary action and re-precipitated as a result of evaporation of the water or uptake of water in plants in the root zone. Formation of hardpans may be a problem.

Important biological processes are:

- Bioturbation, i.e. mixing effects caused by earth-living organisms, e.g. worms.
- Root penetration. Roots of different plants may, under certain conditions, penetrate deep into the ground. When the plant dies, a hole is left where the root has grown. In a sealing layer, this could lead to reduced barrier efficiency. However, it has been questioned if roots can grow in a continuously water saturated sealing material. The oxygen content is very low, and typically so is the nutrient availability. Certain plants, however, have the ability to actively transport oxygen to the roots.
- Up-rooted trees. Caused by trees that are expected to establish on a tailings impoundment with time. Uprooted trees may cause severe localised damage to soil covers.
- Digging animals, in particular animals living in burrows. This type of damage typically occurs on slopes, although the moist conditions may not be attractive.
- Human intrusion may occur during different types of construction work (houses, roads etc) but also due to quarrying. It is virtually impossible to maintain a long-term protection against deliberate intrusion by humans. However, different administrative precautions need to be taken to reduce the risk of unintentional intrusion, e.g. by notification in public planning documents.
- Bacterial growth must be expected to occur in mining waste deposits. Different types of biofilms are commonly observed. At the current time it is not clear whether such microbial populations influence the performance of cover systems in any way.

6.4. Engineering aspects of dry covers for tailings ponds

6.4.1. Tailings disposal

Restoration of tailings ponds involves some unique engineering challenges, which is why this topic warrants a dedicated section of text. In section 6.2 the origin of tailings was briefly outlined. Following dewatering to recover water for re-use, the resultant thickened slurry, in which the weight of solids per unit weight of slurry may be between 15 and 55 percent, is usually pumped to an impoundment for disposal. Depending upon the nature of the mineral, the mining methods, and processing operations, there may be sufficient overburden material or processed coarse discard to permit the construction of adequate retaining banks for retention of these tailings. This is typical of the situation in the coal industry of northwest Europe, but in many branches of metalliferous mining it is not uncommon that nearly all of the waste is produced as fine-grained tailings in suspension and various expedients need to be adopted in order to create safe impoundments.

During discharge of tailings into the impoundment most of the coarse particles will tend to be deposited relatively near the point of discharge. The remaining coarse particles and the finer particles are carried further to the ponded area of the impoundment where they settle more slowly. It is usually desirable that process water is recovered from the impoundment for re-use and this may be done by employing decant towers, which must be built before the impoundment is brought into use, by pumps located on floating barges or by gravity drainage or syphons located at the perimeter of the impoundment at the edge of the ponded area. The method used will affect the location of the deposits of the finest sediments within the impoundment upon completion. Where coarse waste or borrow materials for construction of the containing embankments are in short supply the slurry suspension may be cycloned on the crest of the impoundment bank to remove the coarser fraction of the tailings which may then be placed by earthmoving plant to raise the perimeter banks.

Where the pumped tailings contain a large proportion of very fine silt or clay particles both the rate of sedimentation in the ponded area and the rate of consolidation after deposition may be very slow. The tailings deposit will have a very low density and extremely low strength for an extended period. As deposited, the tailings are usually stratified with a random variation in particle sizes of the layers or lenses due to variation in sedimentation conditions during filling. The coarser, more permeable layers provide lateral drainage paths such that the average horizontal hydraulic conductivity is much higher than the hydraulic conductivity in the vertical azimuth.

6.4.2 Completion of filling operations

When pumping of tailings ceases substantial areas of the surface of the tailings where the tailings are sloping and relatively coarse, such as the 'beach' areas near the inlets, and these may become dry under the influence of a combination of evaporation and gravity drainage. However, because of the fine-grained nature of most of the deposits[21], drainage of the bulk of the tailings rarely proceeds to completion, even under arid climate conditions. Very fine tailings therefore tend to remain very wet and very weak almost indefinitely. If the surface water is removed or evaporates in periods of dry weather a surface crust forms due to desiccation. However the thickness of the crust will be limited because as drying occurs shrinkage of the surface layer leads to significant fissuring which breaks the capillary column which initially pulls water to the top of the deposits. Subsequently the presence of the overlying dry crust largely prevents further drying by evaporation, with the tailings below remaining very wet.

Although the thickness of the desiccated crust is small (maximum thickness of about 0.5m) its presence has been found from practical experience to be of significant benefit in facilitating the over-tipping of a cover layer onto otherwise very soft tailings, as it helps to spread the imposed load at the advancing edge of the initial capping layer. Therefore, the first step in the implementation of any over-tipping scheme must be the removal of ponded water from the surface of the impoundment.

6.4.3 Creating the initial cover layer by over-tipping of tailings deposits

An appropriate specification of works for initial capping of a tailings dam calls for an operation which is contrary to all the instincts of uninitiated plant operators: the creation of single layer of around 1m in thickness, spread in a very gentle and progressive manner. In contrast most plant

[21] Gravity drainage of pores tends to cease where pore necks are smaller than about 10μm in diameter, which is often the case in mud- or silt-grade tailings.

operators will be used to working a layer thickness of two or three metres, bulldozing uphill towards a ridge at the advancing face, dumping incoming spoil as close to the advancing face as possible, and commencing the laying-out of the second and even third layers close behind the initial advancing face. Over tailings, this normal operational approach can quickly lead to large out-of-balance loadings in the underlying deposits, with potentially serious consequences for construction personnel and the overall stability of the tailings deposit. The reasons for the alternative approach of installing an initial layer of one metre thickness using a low ground pressure bulldozer should become clear in the following paragraphs. To ensure its satisfactory implementation will demand close management by a well-trained and highly experienced resident engineer. They will need to ensure that no earthmoving equipment be allowed onto the surface of the tailings without prior authorisation, and that no dumping of material be allowed on the surface of the tailings unless specifically approved by the Engineer.

In areas of the deposit surface close to the inlets and anywhere the materials are coarse and therefore well drained, the placing of a capping layer is likely to be relatively straight forward. In these areas, provided the deposits are coarse to some depth, any excess porewater pressures generated by the loading imposed by the capping material should rapidly dissipate so that adequate strength is maintained in the granular tailings. Where fine silts and clays accumulated in the area which was ponded during deposition the tailings are likely to be generally soft and wet at depth, even if a desiccated surface crust has formed, as discussed above.

The extent of the hazardous zones in which soft tailings are present will depend on the type of tailings, the type of impoundment and the size and location of the ponding that existed during disposal operations. At some impoundments the surface of the lagoon will have been deliberately used as a reservoir of process water and at these sites the size of the pond area may have been a large proportion of the impoundment surface. Also at many sites the location of the tailings inlet will have been moved during the filling of the impoundment to promote more even filling, to maintain the ponded area as far as possible from the perimeter bank and to control the location of the pond to facilitate recovery of process water. Unless there are reliable records of the former locations of ponding, it is prudent to assume that soft material may occur beneath any location on the impoundment surface and design the over-tipping operations accordingly.

The minimum shear strength of the impounded tailings required to permit successful over-tipping is theoretically very small, provided operations are strictly controlled to require the placing of the initial capping material in a thin layer using the smallest bulldozer. Once an initial layer has been carefully placed subsequent layers can be placed much more easily though it is still necessary to maintain tight control of layer thickness, plant movements and stocking of imported material to avoid overloading of the tailings. Once the total thickness of material placed is more than 2 - 3m the required tailings strength to retain stability at the advancing face and to carry earthmoving plant on the cap surface is less than that required for the placement of the initial layer.

It is crucial to achieve completion of the initial covering layer without significant heave. Heave may be induced in this layer if the initial advancing layer is too thick, the spreading plant is too heavy and / or heavily-loaded earthmoving plant is allowed to run on the capping layer to deliver material to the spreading bulldozer. In order to overtip the weakest materials the use of a thin initial capping layer of about 1m thickness will usually be necessary. This will need to be spread by the lightest bulldozer[22]. The aim is to install the capping layer in such a manner that overloaded bearing capacity failure is

[22] Low Ground Pressure D5 bulldozer or equivalent

avoided, since this would result in settlement of the capped area and heave of the material which has not yet been capped. The magnitude of the heave may be local affecting only the area just ahead of the advancing edge of the layer or general with a deep-seated displacement of the tailings from beneath the capping and a heave of the uncapped tailings. The occurrence of a local heave is some indication that the risk of a general heave is high. Where general heave of the tailings occurs this has severe adverse implications for the successful completion of the over-tipping to the design profile to programme and within cost budgets. The particular difficulties which arise are

1) that capping material is 'lost' in attempting to maintain levels in the areas which are subsiding,
2) the material which heaves loses the small strength is possessed due to the disturbance which makes further advancement of the capping layer over it extremely difficult until a further period of desiccation and consolidation has been allowed
3) the stratified structure of the deposit is disrupted eliminating the horizontal drainage and
4) the heave makes it impossible to cap the deposits to the designed levels. In some cases after the occurrence of heave, efforts to retrieve the situation have led to the formation of a 'volcano' of tailings through the capping materials which have only been finally capped at a level many metres above design level and after extensive delays.

The process of over-tipping will be greatly facilitated if the surface of the deposits have been allowed to form a desiccated crust. Therefore this should be encouraged by keeping the surface free of surface water and draining the deposits as far as possible by installing drainage outlets as deep as possible through the lagoon banks.

It is essential to maintain very tight control of the over-tipping operations to avoid inducing significant heave in the lagoon deposits. This must be done for two reasons; firstly, to avoid 'wasting' over-tipping material and secondly, to avoid inordinate delays of months or even years arising because access on to the heaved material may be impossible. Unlike conventional earthworks contracts where contractors are seeking to optimise their plant to achieve maximum earthmoving in the minimum time, in lagoon over-tipping it is usually a case of 'more haste, less speed'. The rate of placing must be determined by consideration of the condition of the lagoon deposits NOT plant availability and efficiency. The normal Lump Sum earthworks form of contract is therefore not conducive to satisfactory completion of this type of work. There could be advantage in considering carrying out the initial capping on a plant hire basis and then, as a separate contract, do the main earthworks to final profile as a lump sum.

6.5. Revegetation of mine wastes

6.5.1. Introduction

Once a dry cover has been installed on an abandoned spoil heap or tailings lagoon a natural next step is the development of a vegetative cover across the site. A dense cover of vegetation helps to prevent soil erosion, thus reducing the risk of pollution of surface watercourses with suspended matter. A green sward on a former mine waste repository will significantly increase the amenity value of a site, and in some cases it can even lead to the site being taken into agricultural production. In terms of pollution prevention, a vegetative cover will help to consume water which would otherwise infiltrate to the sulphide minerals below, and hence contributes to limiting the generation of polluted subsurface water. In addition, organic matter associated with the vegetative cover may well consume oxygen which would otherwise diffuse further into the subsurface, thus further limiting the possibility of oxidation of these sulphide minerals. This section of the guidelines provides details of the technical and engineering aspects of successfully developing such a vegetative cover on reclaimed spoil heaps and tailings lagoons. It should be noted that by far the greatest body of worj on this topic to date has been implemented in relation to colliery spoil heaps, so that the account which follows, though mentioning metalliferous mine sites where the available information permits, inevitably draws heavily upon experiences in the coal sector.

6.5.2. Constraints arising from the character of spoil and tailings

Colliery spoil consists of relatively inert minerals such as silica, clay minerals and some coal, as well as small amounts of other primary minerals such as iron pyrite, and the carbonate minerals ankerite and siderite. The inadequacies of such material as a soil for plant establishment are discussed below. However, it should be borne in mind throughout the following discussion that:

a) Wide variations occur in the character of spoil, not only between tips but also within tips and often over short distances, and
b) Weathering and leaching processes tend to exacerbate variations in spoil characteristics over depth, such that any sampling needs to be representative of the anticipated rooting depth of the plants it is proposed to establish.

Colliery spoil usually has reasonable physical properties in terms of particle size distribution and moisture holding capacity when in a loose state, but lacks both structure and texture. However, because it is well-graded and deficient in organic matter it is prone to compaction by wheeled machinery, which reduces air and water hydraulic conductivity and inhibits root growth. Because colliery material tends to be dark in colour it has a low albedo, and therefore suffers from temperature stress in some climates.

The extent of potential problems resulting from the effects of pyrite oxidation within spoil heaps depends primarily on the percentage of pyritic sulphur occurring in the spoil (expressed as S), which can be determined by laboratory tests. Table 6.2 illustrates in broad terms the likely significance of various percentages of pyritic sulphur in terms of pollution potential (and, by association, difficulty of vegetation establishment).

Invariably as the sulphur percentage in the spoil increases, the likely pH of the soil and associated water drops. This has serious implications for the range of plants that may become established on the heap, as illustrated in Table 6.3. Low pH is itself a serious hindrance to healthy plant growth, and it may also mobilise potentially phytotoxic metals such as Al, Cd, Ni etc. On the other hand, at moderately low pH (3 - 5), preferential release of micro-nutrients such as Mn, and of phosphate bound to ferric oxides, may occur, thus potentially counter-balancing negative effects. The balance of these positive and negative effects will determine the outcome in terms of plant health in any one locality.

Table 6.2 Pollution potential from spoil as a function of total sulphur content

% sulphur in spoil	Potential pollution
< 0.5	None
0.5 – 2.0	Moderate
2.0 – 5.0	Severe
> 5.0	Very severe

Table 6.3 Survival of plants at different soil pH values

pH value	Plant growth potential
< 4	Very few species can survive
4.0 – 5.5	Certain grasses and clovers grow successfully
5.5 – 8.0	Most plants grow, but many thrive best at a specific pH e.g. pH of 6 for grassland; pH of 6.5 for arable crops
8.0 +	Few species likely to survive

Some spoil heaps contain rocks which contained brines in their native state underground. Leaching of these brines into the pore water of the spoil heap can lead to significant salinity stress for sensitive plants. Electrical conductivity is the most convenient method of measuring the salinity of a spoil. The influence of conductivity on plant growth is indicated, in general terms, in Table 6.4.

Table 6.4 Effect of spoil heap salinity (as measured by Conductivity of pore water) on plant growth

Conductivity (μS/cm)	Effect on plant growth
2000 - 2100	Normal values for agricultural soils
2100 - 2500	Suitable for all plants
2500 - 3000	Sensitive plants injured or retarded
3000 - 3500	Many plants injured or retarded
3500+	Few species likely to survive

Colliery spoil is deficient in two of the three main plant nutrients: nitrogen and phosphorus. As a consequence it is typically necessary to apply both of these nutrients in an appropriate form during plant establishment. The finer particle of tailings, and the nature of their generation, means that they are often water saturated and have a low oxygen concentration. These factors define specific requirements for plants to establish in such areas. Certain plants have the ability to transport the oxygen produced in the photosynthesis to the root zone. Some oxygen may be released from the plant root to the surrounding soil material. The tailings are usually an environment with a deficiency of nutrient for plants. For plants to establish in tailings, some kind of fertiliser must usually be applied. Various waste products rich in nutrients have been studied as potential amendments (e.g. Stoltz and Greger 2001). Results show that sewage sludge is the most efficient amendment for plant establishment. When plants have been well established it is reasonable to expect that a self-sustaining ecosystem will eventually develop, and further human intervention may cease to be necessary. For example, at the tailings lagoon at Höbäcksdalen, Boliden, northern Sweden, municipal sewage water has been discharged into the impoundment. At this location a spontaneous establishment of dense and diverse vegetation has occurred. The most frequent species observed are *Carex nigra, Carex rostrata, Eriophorum angustifolium, Equisetum fluviatile, Equisetum palustre, Salix phylicifolia, Salix borealis and Triglochin palustre*.

6.5.3 Amelioration of adverse spoil and tailings conditions

6.5.3.1 Introduction.

The objective of revegetation of a spoil heap or tailings lagoon is to provide a suitable soil quality, structure and texture, that will support plant growth in the long-term. To achieve this the general aim is to provide a depth of approximately 300 mm of topsoil overlying a 600 mm depth of subsoil. In some instances suitable materials for soil and subsoil may already be available on a site, though perhaps not in the location or layered structure that is required. In this case the first operation is to undertake soil stripping, to provide a stockpile of soil and subsoil than can subsequently be re-graded across the full extent of the spoil heap. To assess the availability of suitable materials it is necessary to conduct a comprehensive soil survey, which will identify both the volume and quality of materials. If suitable materials are available the next operation will be soil stripping and storage. If the stockpiles of soil and subsoil are insufficient to satisfactorily cover the full extent of the tip it may be necessary to import soil from another location. If possible this should be avoided, however, since the capital cost of soil and its transport may be high.

The final task is the replacement of subsoil and soil across the spoil heap. Depending upon the quality of the soil, and in particular the salinity, acidity potential, and nutrient content of the material, it may be necessary to augment the soil with lime and nutrients before and during replacement.

In summary, therefore, the key steps in the development of a soil cover suitable for plant growth are as follows:

- Comprehensive soil survey, including chemical testing
- Stripping of soil materials
- Storage of soil materials
- Replacement of soils (including chemical additions if required)

Each of these steps will now be considered in greater detail.

6.5.3.2 Comprehensive soil survey

The soil survey entails augering, excavation of trial pits, and soil sampling and chemical analysis. The soil survey should investigate the potential of sub-layers in the spoil as potential soil-making material. It is for instance possible that if topsoil is in short supply then certain subsoils might be used to supplement the topsoil. Similarly, the quantity of subsoil may be supplemented by stripping suitable weathered parent material (e.g. weathered sandstone). The soil survey report must identify the soil types and agricultural land classification and make recommendations on stripping depths, lime requirements, suitable earthmoving plant, soil moisture contents for minimising soil damage during soil stripping, soil storage requirements, soil-making materials, soil reinstatement and future management. The soil survey report must also make recommendations for chemical additions, such as the need for lime treatment and nutrient (especially N and P) additions.

6.5.3.3 Soil stripping

The method of stripping of soils will normally be by motorised scraper. The routing of scrapers during this operation must be planned to minimise the travel of the machines on soils designated as topsoil or subsoil.

It is essential that soil stripping (as indeed the other operations involved in restoration) is carried out when the soils are as dry as possible. This reduces the risk of compaction and damage to the soil structure by smearing and remoulding. The condition when the soil is in a suitable condition can either be assessed by reference to the soil moisture deficit for the site on the day when operations are to be carried out, or alternatively by reference to the soil moisture content. Data pertaining to suitable conditions for earth-moving operations (e.g. moisture content) at the specific site may be included in the soil survey report.

In most of Europe, suitable conditions for soil stripping are likely to occur between May and October in an average year, since this is the season when evapotranspiration tends to exceed precipitation. Prolonged summer rainfall can make conditions unsuitable, and therefore the arrangements for stripping operations must be sufficiently flexible to allow work to cease or be delayed. Where the stripped soil is to be placed immediately, operations should normally be completed by early September to allow an autumn sowing to become established before the onset of winter.

Careful control of operations is required to ensure that the planned stripping depths of topsoil, subsoil and soil-making materials are adhered to. It will also ensure that the various materials are stored separately or replaced on a completed surface in the correct sequence and do not become mixed (unless mixing is part of the specification).

6.5.3.4 Soil Storage

As far as is practicable long-term soil storage in heaps should be avoided since stocking increases the risk of compaction and is expensive in terms of double handling and occupation of land. In addition, the condition of biologically active soils deteriorates. Wherever possible progressive restoration should be practised, with the soil from each new area stripped being immediately placed on a completed pond or lagoon profile.

In order to maintain maximum levels of biological activity in topsoil heaps they should ideally be constructed to provide the maximum surface area, and have slopes capable of being traversed by plant with implements for cultivating, seeding and maintenance and weed control.

The subsoil and soil-making material may, where necessary, be stock-piled for storage. Although the same considerations do not apply from a biological viewpoint, these heaps should be seeded and maintained in a similar manner to topsoil heaps. Where soil-making material is being stocked it may benefit from weathering and a limited heap height will encourage this. Certainly, deep (> 1.5m) soil stores ought to be avoided wherever feasible, as deep storage can give rise to anoxic conditions, which will result in greater rehabilitation of the stored soil being necessary before it can be successfully re-used as topsoil.

6.5.3.5 Completion of tipping operations and soil replacement

The final tipping operation is to complete the surface of the pond or lagoon accurately to the required profile. Earthmoving plant operations and weathering often create a compacted impermeable surface that is undesirable from a restoration viewpoint as it inhibits penetration of air, water and roots. In some cases testing of the spoil and consideration of the proposed cropping will indicate that lime and/or fertiliser are required and these should be spread at this stage (see section 6.5.3.6, below). They can then be thoroughly incorporated into the surface of the spoil by ripping which is carried out primarily to break up the surface pan and relieve compaction. As an alternative to ripping, the placing of a loose layer of spoil on the tip surface to bring it to final contour may serve the same purpose of achieving continuity of drainage and looseness of structure. However, such spoil should be of low salinity and acid producing potential.

Large stones and extraneous debris should be collected and disposed. Discing and some light re-grading may be required to restore a smooth surface.

A motorised scraper is usually the most efficient machine for soil replacement. However, scrapers inevitably cause soil compaction since they must travel over it to spread their load. Careful control of plant movements is therefore required. As far as possible scrapers should travel in common tracks across the lowest exposed soil horizon and only pull on to the subsoil or topsoil surface for the dumping operation itself. To this end it is advantageous for subsoil and topsoil replacement to be carried out almost concurrently, such that traffic movements over subsoil are minimised during topsoil placement. For the highest quality restoration a bulldozer equipped with ripping tines should be in attendance to rip each track made by the machines prior to the placing of further soil.

An alternative to soil handling by scraper operation, shovel-loaded dump trucks can be used for spot dumping of soil. Final placement is achieved using tracked shovels or back-actors[23]. Such an operation largely eliminates compaction, but is clearly very expensive if large areas are involved.

Soil replacement should only be carried out in dry weather when the moisture content is low enough to prevent damage to the soil fabric. Close supervision is necessary to ensure that the soils do not become intermixed, and that they are spread evenly to the specific thickness, with the minimum of compaction. If ripping has not been carried out as described above, general ripping upon completion of an appropriate area or layer should be carried out. Ripping should be of such a depth that the tines extend into the previous layer. After each ripping operation large stones and any other debris, which could interfere with subsequent cultivation, should be collected and disposed of.

Where laboratory tests on the subsoil show that it would benefit from the addition of lime and/or fertiliser, such additions are best made immediately after subsoil spreading. The lime or fertiliser can then be mixed into the soil by the ripping operation. After ripping the surface may be uneven with large clods in clayey soils and will benefit from discing to break it down further and restore a reasonably smooth surface.

6.5.3.6 Chemical additions

If testing shows the pH of the spoil to be less than 6 then lime, in the form of limestone or lime powder, should be added to neutralise the acidity. The lime demand can not be directly assessed from the measured pH but needs to be assessed by chemical analysis. The quantity of lime added must be calculated to ensure that the existing acidity is neutralised, but also an additional quantity may be added to counteract the effects of potential acidity generation from pyrite oxidation. The exact amount depends upon the sulphur percentage of the spoil. Typically the addition of lime, for potential acidity remediation, is based upon provision of 20% of that which would theoretically be required to neutralise the acidity resulting from complete oxidation of all the pyrite present in the relevant layer thickness. This somewhat arbitrary provision should ensure that acidity is not a problem in the short-term, and may provide a permanent solution, particularly where the depth of soil

[23] In American English these are termed "backhoes"

is substantial. As a guide, an additional 20 tonnes/ha of lime should be added to spoil heaps with sulphur content in the range 0.5 – 5.0%.

Where conductivity of the spoil is < 3000 μS/cm no action is required to combat salinity. However, above this level, salinity concentrations may have a deleterious effect on vegetation. Because of the highly soluble nature of sodium chloride (the principle compound contributing to salinity) it is naturally removed from the surface layers of exposed spoil heaps by leaching. However, prolonged exposure may be unacceptable because of the potential for pollution of watercourses due to pyrite oxidation. If leaching is deemed necessary, the preferable time to do this is over winter, when evaporation and capillary action may draw salinity towards the surface, and high rainfall will allow the salinity to be leached from the heap. Nevertheless, such a process may take on the order of 6 - 12 months to be effective. If this is not an acceptable option, the only means to counteract the salinity is to ensure a thickness of at least 300 mm of clean soil.

The rate of application of phosphorus to the soil is dependent upon the thickness of the soil cover. As a general rule no application is required if nutrient-rich soil is to a depth of > 300 mm; apply 150 kg/ha P_2O_5 to soil depths of 150 – 300 mm; and 250 kg/ha P_2O_5 to soil depths of < 150 mm. In some cases the incorporation of organic material, such as farmyard manure and certain sewage sludge, may be a low cost means of improving nutrient content, as well as assisting with the improvement of the structure, texture and moisture holding capacity of the soil cover.

Application of nitrogenous fertilisers needs to be undertaken with particular care, for nitrate is a powerful oxidant with respect to pyrite, and thus excessive nitrate fertiliser addition can actually exacerbate acidity problems in sulphur-rich mine spoils.

7. DESIGN & CONSTRUCTION DETAILS FOR LAGOONS AND OTHER COMMON ELEMENTS OF PASSIVE REMEDIATION SYSTEMS

7.1. Introduction

Probably the single most important civil / geotechnical engineering consideration in the installation of a passive treatment system is the construction of water retaining structures, and in particular the construction of settlement lagoons, which may contain water to depths of around 3 m. The consequences of inadequate design of such structures could be catastrophic, at best resulting in a sudden release of large volumes of contaminated water, and at worst resulting in loss of life. The excavation of trenches also requires careful design. Particular care needs to be taken with regard to trench wall stability, and possible safety risks associated with this issue. Section 7.8 discusses the particular techniques available for the excavation of trenches for installation of permeable reactive barriers.

The findings of the site survey and ground investigation must be considered together to assess fully the effects of constructing the treatment system at the chosen location. The effects on the surrounding area must also be considered during the construction and future operation of the system. The design should also ideally provide flexibility for future modification where these may be required. Desilting, access points, and long-term planting arrangements must also be considered as part of the design process.

Much of the text in this section, and in section 9 of these guidelines, has been adapted from the Construction Industry Research and Information Association (CIRIA) publication R161, *Small Embankment Reservoirs* by Kennard *et al.* (1996), which provided excellent, thorough guidance on this and related topics.

7.2. Slope stability and factors of safety

Slope stability of retaining embankments is a crucial design consideration. The assistance of a geotechnical engineer should be sought to establish maximum heights and side-slopes of embankments. In general, for a well-graded compact colliery spoil or quarry waste material, inner slopes of ponds not exceeding 1 in 2 and external slopes not exceeding 1 in 3 should result in stable structure where the ground-slope of the site does not exceed 1 in 10 on a strong foundation. If there is any concern about the nature of the foundation materials, the materials in which the embankments are constructed, or if the gradient of the proposed slopes exceed those indicated above, then geotechnical advice must be taken to assist the design.

The embankments constructed to form the treatment system must have acceptable factors of safety against failure. The factor of safety used depends upon the possible consequences of failure. A higher factor of safety is usually used where there is a risk of danger to persons or property. Thus, it is usually the case that a higher factor of safety is used in the design of the outer slopes of a settlement lagoon than the inner slopes, since failure of the outer slopes carries the greater risk to lives and property.

Assessment of factors of safety against failure is a complex analysis that is normally carried out using computer software packages. It requires a significant understanding of soil mechanics and is best completed by an experienced geotechnical engineer. It should be noted that the geotechnical parameters required should be obtained during the course of the original site investigation and therefore if it is believed that this information is to be needed then it should be procured at the initial site investigation stage rather than subsequent return visits.

Ordinarily it would not be engineering practice to construct treatment lagoons on spoil heaps arising from mining, quarrying or other industrial activity. This is because escape of water from a treatment system, unless comprehensively controlled by drainage arrangements, could imperil the stability of the flanks of the tip or infiltrate because of the variable nature of spoil placed without engineering controls.

However, for mine water treatment systems it is not unusual that there may be no alternative location for a treatment system other than on a spoil heap. The assistance of civil and geotechnical engineers will be required to ensure a stable and secure development. The vital design elements will ensure that required factors of safety against failure for side-slopes are achieved and that percolation of water into the tip is minimised by the use of liners or compacted impermeable materials. The consequences of failure of a lagoon bank constructed in an elevated position on a spoil heap will also have to be considered. A comprehensive site investigation and facilities for monitoring internal water levels in the tip may be required for such proposals.

7.3. Design requirements

The principal requirement for the design of a small water-retaining embankment is to ensure that it is safe and stable during all phases of its construction and operation. The following criteria must be met:

- the embankment must not impose excessive stresses upon the foundation.
- the embankment slopes must be stable under all conditions, including rapid drawdown of the treatment system e.g. during sludge removal operations or major maintenance.
- seepage through and beneath embankments must be controlled so as to not affect the stability (this can often be achieved by including a liner, such as HDPE in the design).
- the system must be safe against overtopping by provision of adequate overflow capacity.
- slopes must be protected against wave action and erosion.

The stability of a small embankment depends upon the type of fill, the type of soil underlying the embankment and the past geological history and land use of the area. It also depends on the slope of the upstream and downstream faces, the overall foundation constructed and the extent to which the design provisions are put into effect. The stability of excavated slopes also needs careful consideration.

It is not possible to construct an earth embankment that is completely watertight and seepage, however insignificant, will always occur. A soil should be considered to be an adequately impermeable barrier if it has a hydraulic conductivity of less than 10^{-9}m/s and a minimum thickness of 500 mm. To ensure stability, the embankment and foundation interface should be designed and constructed so that the rate of seepage is reduced to a minimum and seepage paths are adequately controlled by suitable drainage and filtering.

The fill material must be adequately compacted to maximise the density and reduce the amount of air and / or water-filled voids left in the soil. Compaction will also increase the strength, decrease the hydraulic conductivity and minimise post construction settlement.

Stable pond embankments will *generally* be achieved on a suitable site if:

- an acceptably level, sound foundation is present (<1 in 10 ground-slope)
- the soil types are known and are suitable.
- the fill is compacted properly
- the required drainage, cut-off and seepage control measures are provided
- the height is not in excess of 5 m, measured from the crest to the excavation level
- the outside slope should not be steeper than 1 in 3
- the inside slope should not be steeper than 1 in 2
- the crest width of the embankment should be a minimum of 5 metres.

These slopes may be adopted where a site investigation has found no adverse features. Specialist advice must be obtained if unfavourable features are found or where the conditions listed above are not achieved.

Small embankments constructed of high hydraulic conductivity soils, e.g. sands and gravels, with an impermeable membrane on the upstream slope without soil cover, may be built with slightly steeper slopes due to a general increase in fill and foundation strength associated with the dry conditions. If adequate information is available from a site investigation, slope angles may possibly be amended following specialist advice on embankment stability.

A record of the design process should be kept for future reference. This should be drafted in simple terms with sketches where required, but should be sufficient to record the development of the scheme from initial conception, through the feasibility stage to the final design proposals. The reasons for the chosen approach and solutions to problems should be recorded together with any particular difficulties or unusual aspects.

This information will prove invaluable during the construction stage and in future years for maintenance purposes. In the event of difficulties or need for remedial works, the records may prove a useful source of information. Any future owner taking over the site at a later date will also require this information.

Subsequent records of construction, maintenance, remedial or alteration should also be kept to allow a full record of the history and the development the site to be available. A large number of titled, dated photographs are often useful.

7.4. Earthworks

7.4.1. General comments

Selection of the appropriate design techniques and soil parameters will require specialist advice from an experienced geotechnical engineer. In the UK, some useful general guidance on earthworks design is contained in UK BS 603 1: Code of practice for earthworks. It is normally more economical to use the locally available materials in preference to importing fill materials from off site. Suitable designs can often be prepared using apparently unsuitable materials but these require specialist advice at an early stage.

7.4.2. Embankment foundations

All embankment foundations must have adequate strength to support the loading of the embankment under any condition without unacceptable settlement or displacement. It must also provide adequate resistance to sliding. Most soils have sufficient strength to carry the weight of small embankments some materials are unacceptable and these should be avoided. Quite often, unsuitable material may be used as low-grade fill material for landscaping purposes.

Problems with sliding of the embankments on (or at a shallow depth within) its foundation are unlikely if unsuitable material is removed. However, polished surfaces at shallow depths in the foundation, particularly on the sloping ground may he indicative of past ground instability. Excavation, even of limited extent and depth, fill placing or stockpiling of materials may he sufficient to regenerate instability.

These surfaces may also threaten the safe functioning of the completed embankment with raised ground-water levels following filling of the treatment system. If such surfaces are suspected or discovered, specialist advice should be sought. If their locations are reliably known, excavation extending below those depths may be feasible, with careful assessment of the implications on the proposed works. Alternatively, they may be left in-situ, but this will affect other design aspects, and specialist advice must be again sought.

Recently cleared areas of woodland or orchard will have soil with depleted water content as a result of the trees extracting water over a prolonged period. In clay soils, this depletion may take many months to dissipate, whilst in sandy soils dissipation will be quicker. Where a lagoon is to he founded on a wooded site which is on clay, the trees should be cleared as early as possible so that the soil can adjust to the new conditions. During this time the ground will swell and thus it is important that construction is not started until the soil has achieved its new equilibrium moisture content. This is particularly relevant where structures and pipelines are to be constructed.

Existing trees outside the treatment area should normally be retained. Removal of trees that are close to a new structure may be potentially more harmful due to ground swelling in certain clay soils, than leaving them in place.

7.4.3. Seepage control measures

Seepage through permeable foundation soils, which overlie an impermeable layer, can be controlled by constructing a cut-off along the line of the embankment. This is normally a trench of sufficient width excavated down and into the impermeable layer and filled with soil that has a suitable clay content.

An earth embankment founded on rock will be a relatively costly structure arising from the need to excavate a cut-off trench into the rock to control seepage along the fissures in the rock surface. In many rocks only a shallow trench may be needed to extend into the un-fissured rock, but stratified rock will require greater depths. It may be more economical for a small lagoon system to consider lining the base and banks of the lagoon with an impermeable layer.

Many sites will have had land drains installed in the past. These will provide a ready path for seepage and measures must be included to intercept or divert these as part of the construction. Most land drains are within 1m of the ground surface and thus the cut-off trench or general excavation should extend to at least this depth. Any land drains that are encountered should be sealed to stop any future seepage.

Any pipe, culvert, trench or excavation passing through or beneath the lagoon site below top water level must be back-filled with a cohesive material to prevent seepage along the trench. A pipe or culvert should also have anti-seepage collars to prevent the internal erosion of soil by seepage flow along the periphery. These collars, which increase the length of the seepage path, should be made of concrete and be at least 150 mm thick along the line of the pipe and should have a normal minimum spacing of 6 m. It is important that the collars are constructed to the required projection around the entire perimeter of the pipe and extend into the ground beyond the pipeline trench by a minimum of 500 mm.

7.4.4. Construction Materials

The potential construction materials must be reviewed and their adequacy and availability confirmed for the lagoon construction. This should include the following:

- Suitability as embankment fill material
- free from any contamination
- permeability assessment
- available quantity
- ease of winning, transporting and placing
- haul distance.

The locally available material should normally be used for lagoon construction to minimise the length of haulage and avoid the importation of expensive fill from off-site. In many cases, suitable designs can be prepared to incorporate otherwise unsatisfactory fill materials by a range of technical solutions including flattening slopes, providing drainage and other measures. Such designs are outside the scope of this guide and require specialist advice at an early stage. Despite the additional costs of the alternative construction techniques, possibly greater fill qualities and need for technical advice, this approach is still likely to prove more economic than importing better quality fill.

7.4.5. Properties of Main Soil Types

The following properties of the main material types gives an indication of their characteristics and suitability for construction:

Gravels:

- will be of sufficient strength to support embankment.
- will encourage seepage beneath embankment with risk of erosion of fine material and ultimately undermining of embankment. Excavate and replace with low hydraulic conductivity material if of shallow depth, or seek advice if depth in excess of 3 – 4 m.
- will lead to seepage from lagoon with possible loss of fine material and problems at discharge position. Provide liner and control groundwater below liner.
- mix with sand and clay to form stable construction material for shoulders of embankment.
- clean gravels may be suitable for drainage and slope and crest protection purposes.

Sands:

- will normally be of sufficient strength to support embankments, unless material is disturbed or loose tipped. Remove unsuitable material.
- will encourage seepage beneath the dam and from lagoons as gravels, but possible lower flows depending on grading and compactness of material.
- possibly prone to erosion by flowing water and wave action requiring adequate protective measures.
- mix with clay to form stable construction material for shoulders of embankment.

Silts:

- unsuitable for foundation material or construction purposes.

Clays - very soft to soft:

- will be of adequately low hydraulic conductivity beneath the embankment and lagoon.
- will have insufficient strength beneath embankment and should be removed.
- may lead to instability around lagoon perimeter, but unlikely to be significant.
- high plasticity material excluded from exposed locations to avoid cracking in dry summers.

Clays - firm to hard:

- of adequate strength to support embankment if not subject to past landslipping or deformation.
- hard clays may be liable to swelling on saturation from lagoon.
- will form suitably low hydraulic conductivity foundation, provided clays are not fissured.
- will be of adequately low hydraulic conductivity beneath the lagoon.
- may need to be reworked and watered.
- high plasticity material excluded from the shoulders to avoid cracking in dry summers.

Organic clays / peat:

- unsuitable for foundation material or construction purposes.
- possible source of construction material for landscaping purposes or surface soiling.

Made ground:

- unsuitable for foundation and may lead to seepage from the lagoon.
- possible risk of contamination problems with lagoon water.
- possible health and safety problems.
- possible source of fill material if investigated sufficiently and close construction control maintained, but not likely to be economic.

7.4.6. Suitable fill material for embankments

A suitable material to use for the construction of a homogeneous embankment should contain generally not less than 20% or more than 30% of clay, the remainder being well-graded granular material. Such a soil is likely to be stable even when subject to significant changes in moisture content. Embankments made from soil with clay content much less than 20% are likely to be subject to moderate water losses that may necessitate the use of an impermeable liner. Soil with a clay content higher than 30% is likely to shrink and crack on drying and such material should ideally be limited to the centre of an embankment to form the impermeable zone or as a sealing layer under the water surface.

7.5. Lining materials

7.5.1. Introduction

Although it may be possible to use compacted colliery spoil or other *in situ* materials as an impermeable barrier in settlement lagoons and wetlands, it is sometimes necessary to resort to the use of a lining material. This decision must not be taken lightly. On the one hand, the use of a liner throughout a passive treatment system will significantly increase capital costs. On the other, significant toe seepage from a settlement lagoon can affect the structural integrity of an embankment, with potentially catastrophic (and expensive) results. This section of the guidelines briefly outlines the options available and pros and cons of different lining materials.

7.5.2. Types of liner

Lining materials come in three broad types:

1) geomembranes
2) clay
3) bentonite based materials.

All can be used to form an impermeable barrier on a permeable site, but the choice of lining should be made in conjunction with a specialist adviser. Embankment slopes flatter than 1 in 3 may need to be adopted to ensure long-term stability. The underlying bedding material will need to be graded carefully to provide a reasonably uniform surface without undue irregularities, changes in profile or localised voids or protrusions. Measures to avoid the build-up of water or gas under the lining material may be necessary in some circumstances.

Particular care will be needed where the lining is connected to concrete and other structures or pipework and appropriate design and careful construction techniques will ensure that a leakage path is not created. Subsequent works to stop leakage tends to be both difficult, time consuming and expensive.

7.5.3. Geomembranes

A geomembrane is a thin, man-made material that is relatively flexible and elastic. The geomembrane can thus take up irregularities in the underlying material to some extent and can accept limited deformation after placement. Large irregularities and subsequent deformations cannot be tolerated, however, and the membrane is prone to mechanical damage and piercing if not well-placed and protected adequately. Geomembranes may need to be covered to protect against damage, erosion or other detrimental effects. Anchorage and jointing must also be carried out carefully if the membrane is to function satisfactorily in the long-term. It is difficult to regularly desilt ponds containing a liner without significant risk of puncture.

7.5.4. Clay

Where clay is available at a site, it can be used to provide low hydraulic conductivity linings to ponds. The effectiveness of a clay lining depends on a number of factors, which include:

- availability of clay material locally
- nature of available clay material (clay content, plasticity and strength)
- method of excavation and placing
- method of compaction
- thickness of material placed
- post-placement deformation (fluctuating water level, groundwater, cracking, settlement)

The nature of the available clay material is of primary importance where reliance is placed on a relatively thin layer. The clay content should be sufficiently high to ensure low hydraulic conductivity thus reducing seepage. The clay content should not be too high, however, as this leads to shrinkage and cracking on drying if the water level is lowered. The risk of cracking may be reduced by covering the compacted clay with a layer of granular material at least 600 mm thick where it is susceptible to exposure.

An adequate thickness of clay material should be placed and compacted to ensure a low hydraulic conductivity seal to the lagoon. This layer should be at least 500 mm thick when compacted, and should be thicker where the nature of the clay material in the borrow pit area is considered to be variable (particularly silty or granular).

7.5.5. Bentonite

Bentonite has been used for pond lining purposes, either as a loose material (powdered or granular) or within a composite sheet (sandwiched between geotextiles). Loose bentonite can be used in powdered or granular form to reduce the hydraulic conductivity of the lagoon bed. It can be either mixed with *in situ* soil (normally by rotovating) and then compacted to form a layer of lower hydraulic conductivity than the original bed, or placed as a compacted layer of pure bentonite. In both applications the layer should be covered with a layer of selected compacted material.

Composite sheets, comprises a layer of bentonite sandwiched between geotextile materials, can be used to provide a lower hydraulic conductivity barrier, similar in principal to the use of pure loose bentonite. The sheet should be covered with a layer of selected compacted material.

Bentonite gains its impermeable properties due to swelling when it comes into contact with water (due, in particular, to the attraction of positively charged ions in the water). Because of the chemical nature of this reaction, it follows that certain chemicals may impair its functioning. Of particular significance in relation to mine waters is that elevated chloride concentrations may significantly retard the ability of bentonite to form an impermeable seal. Since chloride concentrations may be high in mine waters, especially when water arises from deep mines, care should be taken to ensure that the water quality is compatible with the liner selected.

7.6. Construction plant and equipment

Material has to be transported from the place of excavation and then compacted into an embankment. Economic means of achieving this objective should consider;

- the time available to complete the project
- the transport (haul) distance
- the state of the ground
- the fill type and quantity
- anticipated weather conditions
- the expected response of the fill to the likely weather conditions.

The type of construction plant to be used for a small embankment depends on a number of factors, including:

- type of embankment to be constructed, i.e. homogeneous or zoned
- size of construction
- type of material to be used in construction
- distance of borrow pits from the embankment
- possible available equipment on site and availability over construction period
- time scale for construction
- anticipated weather conditions

Earthworks do not necessarily require the use of specialist plant, and farm equipment may he suitable for the more straightforward or small-scale works. Certain specialist items of plant may be hired for short periods. Much of the construction work can be carried out with similar items of plant, but specialist plant will be required for spreading and compaction of materials. The safe use of construction plant is of paramount importance and particular attention should be paid to safety aspects of operation.

Table 7.1 summarises the available compaction plant typically used for water treatment ponds and lagoons.

Table 7.1. Recommended compaction plant for use in construction of water treatment ponds and lagoons.

Type	Comments	Typical uses
Smooth-wheeled roller Grid roller	Most versatile and normally used for small dam construction. Where vibration facility available, normally used on granular soils only	All zones and materials, except very soft cohesive fill
Grid roller	Most efficient on dry, stiffer cohesive soils and well graded granular materials. Acts in similar way to the sheepsfoot roller using a steel grid instead of projections	Cohesive shoulder fill
Sheepsfoot (tamping) Roller	has a regular array of projecting feet on the roller to knead the soil together. Most suitable on soft cohesive soils when used in conjunction with dozer blade to mix and blend soil, especially if water added. Bonds compacted layers	Core and cut-off trench.. Homogeneous till dams. Cohesive shoulder fill unless very stiff and dry
Pneumatic tyred roller	Suitable for soft cohesive soils and well graded granular materials, less suitable to assist in mixing and blending soils	Core and cut-off trench homogeneous fill dams, cohesive shoulder fill unless very stiff & dry
Vibratory smooth wheeled roller	Used for granular soils, both well and uniformly graded. Efficient in reducing air voids and compaction at depth in previously placed till. Roller should he used initially without vibration to achieve compaction and avoid roller sinking into loose till	All granular materials, little benefit from vibration in cohesive materials unless very stiff and dry
Hand-guided and self-propelled vibrating rollers	Smaller version of vibrating roller. Can have two rollers in tandem. Roller should he used initially without vibration to achieve compaction and avoid roller sinking into loose till.	filling in restricted areas and backfilling adjacent to structures, pipework, etc.
Vibrating plate tampers	Manually guided plant used for compaction of small areas of granular fill, especially in trenches	Compacting fill in localised areas immediately adjacent to structures, pipework, etc. Not suitable for cohesive soils
Power rammers	Manually guided plant used for compaction of small areas of granular fill, especially in trenches	Compacting till in localised areas immediately adjacent to structures, pipework, etc. Not suitable for cohesive soils

NOTE: Self-propelled machines are normally limited to the larger projects beyond the scope of this guide and most small embankments are generally constructed using a smooth roller towed by a suitable dozer.

7.7. Site preparation

7.7.1. General

All structures including fencing, buildings, drainage features, roads, kerbing etc., should be cleared over the entire site area. Where archaeological remains are encountered, the work must cease in the UK in accordance with the Ancient Monuments and Archaeological Areas Act (1998), and the County Archaeological Department must be informed. Depending on the remains, work may be required to cease for a period to allow investigations or recovery work to be carried out.

Trees, scrub, roots, as well as all vegetation and organic matter should be removed below the area of the construction. Topsoil should also be stripped and should be stored in separate temporary stockpiles (see section 6.5.3.2).

7.7.2. Foundation preparation

The interface between the foundation and the pond embankment is a critical area in terms of stability and seepage control. The foundation preparation must include removal of all soft, loose or otherwise unsuitable material. The exposed foundation should be scarified by ploughing or disc harrowing along the line of the embankment to provide a key between the foundation and the embankment material. This enhances the embankment stability and lessens the risk of seepage. A cut-off trench may be required to intercept any land drains, other shallow features or permeable layers. This should be excavated into sound material once the general foundation preparation has been carried out.

Where a watercourse crosses the foundation, the unsuitable materials in the channel should be dug out to a sufficient depth to expose suitable foundation material and any slopes into the excavation battered back to slopes of 1 in 6 or flatter. A cut-off trench should be excavated below the foundation level, which is exposed at the bottom of the excavation.

7.7.3. Unsuitable material

The use of unsuitable material in embankment construction or its presence in the foundation below the embankment can lead to stability or settlement problems and seepage, erosion or other difficulties with the completed structure. Unsuitable materials include:

- topsoil, including subsoil
- peat-rich or highly organic materials, including logs and tree stumps
- very wet materials
- material from borrow areas which is very wet or soft
- material with a liquid limit in excess of 90% or a plasticity index greater than 65
- frozen soil
- colliery spoil containing significant combustible materials
- material having hazardous chemical or physical properties
- hardcore, concrete or other building materials.

Unsuitable material is normally utilised as low-grade fill for landscaping purposes or disposed of in a borrow pit. In certain instances it may be possible to incorporate it within the areas of landscaping or infilling following specialist advice.

Polluting or hazardous materials should be excluded from any aspect of the construction and should be removed to a recognised licensed tip. Other unsuitable materials should be disposed of in the borrow pit or used as fill for landscaping purposes elsewhere on site. The material should be placed in layers not more than 300 mm thick and compacted by earthmoving plant.

7.7.4. Trenches and excavations

Excavations should be carried out in such a manner as to avoid damage or deterioration to the formation of the excavation or trench to minimise disturbance to the adjacent ground. Slopes of shallow excavations should normally be no steeper than 1:1, but flatter slopes will be required in poor ground when water is present or as required for access on other constraints. Excavation in excess of 2 - 3 m requires special consideration and flatter slopes may be necessary. Where the ground cannot be battered back, some means of temporary support such as trench sheets must be employed. Where access into the excavation is required and the depth is in excess of 1.2 m, protective measures should be taken to support trench sides. Soft material should be removed from the bottom of excavations and replaced with concrete beneath structures or bedding material beneath drains or pipes.

7.8. Excavation and placement of subsurface media in PRBs

7.8.1. Introduction

In essence, constructing a PRB involves excavating the aquifer material and replacing it with the reactive fill. The main problem during construction is the stabilisation of trench walls. This section contains a brief description of construction methods. They are summarised in Table 7.2. Much more thorough descriptions are presented by Gavaskar *et al.* (1998).

Table 7.2. Stabilization and construction methods for permeable reactive barriers (after Gavaskar *et al.*, 1998)

Method	Type of material	Comments
WITH STABILIZATION	Best for non-cohesive materials	
- Sheet piles	Must not contain boulders	The most widely extended method Depth down to 40 m
- Wooden piles	Must not contain gravel	Do not prevent water inflow Shallow barriers (less than 6m)
- Slurry	Should contain some fine material	For permeability, slurry must be biodegradable.
- Caissons	Must not contain boulders	Indicated for funnel-and-gate systems.
WITHOUT STABILIZATION	Best for cohesive materials	Can be used for non-cohesive materials by lowering phreatic level
- Conventional excavation	Low resistance materials (can be combined with ripper or pneumatic hammer)	Appropriate for very shallow barriers
- Continuous trenching	Side sheets make it appropriate for loose material	Depth up to 10 m
- Injection (jetting)	Big size pores	Little experience
- In situ mixing	Loose material without boulders	Little experience

7.8.2. Excavation with sheet piles

Piles are used for protecting the trench during excavation and filling. The most conventional method consists of using steel sheet piles, but one could also use wooden piles. In essence, the method consists of:

1. Excavating the soil surface down to near the phreatic level so as to reduce the depth of the actual reactive trench.
2. Driving the sheet piles into the ground, usually with the help of a drop or a vibrating hammer, so that an enclosed area is created.
3. Excavating the interior of the enclosure. This is done with a backhoe down to depths of 10m or with a clamshell, which is more expensive but may reach down to 50m. Intermediate beams may be needed to retain the side walls.
4. Filling the enclosure with reactive material. Care must be paid to avoid segregation and for a uniform settling. In the case of granular fillings, such as iron, this can be done under water, possibly with the help of a pipe or sloping plane. However, in the case of mixtures, such as organic matter and gravel, the trench should be dewatered.
5. Extracting the sheet piles and covering the upper excavation. Extraction is also done with the aid of a vibrating hammer which is likely to cause some settling of the reactive filling. Therefore, one should have left some overfill material to compensate settling (a 5 to 10% settling is to be expected). Otherwise, one must be prepared to supply additional filling during the extraction of the sheet piles.

This method is relatively simple because it is based on technologies that are widely available.
Total excavation cost fluctuates around €60/m^2, but increases with depth. The system works in most materials, but becomes extremely difficult if boulders are present. Wooden piles cannot be used in gravels or high hydraulic conductivity materials.

7.8.3. Excavation with a slurry head control

This is similar to excavation with sheet piles, except that trench walls are stabilized by means of a slurry. Slurry head is kept above ground by building two small walls with a height of 0.2 – 0.5 m on the sides of the trench. This, together the fact that the slurry density is relatively high, creates a flux of slurry towards the aquifer and compensates the pressure of ground against the trench. Slurry flux into the aquifer is limited by clogging, which also tends to stabilize the trench walls by increasing the cohesiveness of the natural material.

It will be apparent from the description above that the method is not applicable to significant layers of clean gravel (thickness greater them, say, 1m). In fact, the method is difficult to apply to non cohesive granular media. In these cases, the walls tend to collapse, thus increasing excavation and filling volume.

Slurry is often comprised of bentonite. However, since this tends to make the side walls impervious, it is only used for creating impermeable barriers to divert contaminated groundwater e.g. the side barriers (funnel) of a funnel-and-gate system.

Biodegradable slurry can be used for PRBs. This type of slurries consists of cellulose polymers, the degradation products of which flow away with groundwater. A positive side-benefit of this degradation is that it may encourage reducing conditions. While there is some experience with this method for well excavation, use in actual barriers is limited). Therefore laboratory or pilot testing, to confirm that biodegradation is indeed going to occur under the conditions expected for the designed barriers, is recommended.

7.8.4. Excavation with caissons

A caisson is a cylinder (normally circular but sometimes rectangular) that is water tight and can withstand external water and soil pressures. It is frequently used for construction below the water table.

The caisson is driven into the ground in a manner not entirely different to that of sheet piles. The main difference is that the whole enclosure is driven at once. This places an upper limit on the size of the caisson, which rarely exceeds 2m (much larger sizes can be achieved by simultaneous excavation and driving, but the cost increases significantly). Because of this, caissons are used for funnel-and-gate systems, where the caisson is used for the gates and sheet piles (which can be driven in with the same hammer) or slurry walls make up the funnel.

7.8.5. Open trenches

When the aquifer material is consolidated or displays some cohesiveness, one may not need any stabilization method. This reduces construction difficulties and results in capital savings. Even if the aquifer material is not resistant, one may excavate provided that the water level is lowered. In this case, the construction sequence would proceed as follows:

1. Excavate soil surface down to the phreatic surface using conventional equipment.
2. Install dewatering method. A well may suffice for a short PRB. A line of well points may be more appropriate for long barriers. In this case, one must make sure that the top of the pumping section is below the bottom of the trench. In either case, one should anticipate to pump contaminated water, so that a temporary treatment or diversion method should be planned.
3. Excavate trench. This can be done with conventional methods. If the material displays some cohesion, a backhoe is more appropriate because it reduces over excavation.
4. Fill trench. If the trench sides are inclined with a low slope, reactive filling must be emplaced at the same time as the excavation

A particular method that deserves singular mention is continuous trenching. This is based on using a continuous trenching machine, which cuts through the soil using a chain saw type mechanism with large scoops to extract material. The trench walls are supported by means of two steel plates located on the sides of the machine. Filling material can be fed to the bottom of the excavation from a compartment on the back of the trencher.

7.8.6. Other methods

In situ injection (jetting) involves the injection of reactive material (e.g. finely ground zero-valent iron) into a porous medium suspended in high pressure water. The concept has not been tested, but its possible utility is discussed by Gavaskar *et al.* (1998). Such technology is frequently used in geotechnical projects e.g. cement injection to create low hydraulic conductivity barriers, or to improve the resistance properties of soils.

7.9. Embankment construction

7.9.1. General

Embankment construction should start at the earliest opportunity to avoid degradation of the prepared foundation. The embankment should be raised at a uniform rate and in general kept longitudinally level so far as is practical under the prevailing weather conditions, and consistent with the progress of other work on the embankment. The rate of construction for large pond structures should be limited to not more than 1 m in height per week. The fill should be spread in layers and fully compacted. Each layer of soil should be placed and compacted along the entire length of the embankment in a continuous process and any openings for access should be kept to a minimum. This is to avoid creating discontinuities that could lead to differential settlement and areas of weakness and potential leakage. Where openings are left, these should be infilled at the earliest opportunity by removing any dried material on the fill surface, cutting back any longitudinal slopes to not steeper than 1 in 6 and ensuring the fill is adequately keyed into the previously placed material.

An embankment should be constructed with a surface crossfall of 1 in 20 to shed surface water. The surface should be left sufficiently even to prevent the ponding of rainwater in ruts and holes and should be rolled smooth to encourage drainage at times of inclement weather. Prior to further fill placing the surface layer should be ripped to key new material into that previously placed. Embankment fill must never be placed into standing or running water and fill placing should cease when the material is likely to become softened during and after inclement weather. Any fill that has softened should normally be removed, although it may be possible to dry the fill by surface harrowing or ripping with a dozer. No fill should be placed when either the fill or the placing surface is frozen.

The embankment should be overfilled beyond the required profile and then trimmed to the required slopes and levels. If a section of embankment is below design level and additional fill is required, a section of embankment should be cut back and additional fill built up as a series of layers. Fill should not be placed as a sloping layer on the side of the embankment; as this will subsequently tend to soften and may lead to shallow slope instability.

Materials should not be stored on the embankment and care should be taken to avoid contamination of previously placed fill materials. Diesel spillages can damage the fill, whilst granular materials spilt on the core and not removed can lead to seepage through the embankment when in operation.

7.9.2. Fill placing and compaction

Soil compaction is the process whereby soil particles are packed more closely together through a reduction in air content. The objective of compaction is to modify the behaviour of tipped soil to produce a fill that has the desired properties for the required application i.e. generally to decrease the hydraulic conductivity to the required values and increase the strength of the fill. Some compaction arises from the plant transporting, placing and spreading the fill, but specialist compaction plant is normally used.

The mechanical means by which fill is compacted may be either of the following, dependent primarily on soil type:

- rolling for cohesive materials.
- rolling with vibration for granular materials.

Within the scope of these guidelines, the compactive plant is likely to be a smooth drum roller towed by a dozer. For granular fill, the roller is normally used with the vibration facility, built into the roller, to vibrate the individual fragments into a dense state. On larger projects, a sheepsfoot roller may be used for the cohesive material. This has a series of protrusions or feet over the surface of the drum, which kneads the material and produces a dense fill.

Where structures are included in the fill, care should be taken to ensure that the standards of fill placing and compaction around structures are not significantly different from elsewhere. This will minimise future settlement. The use of heavy plant may need to be limited around structures and smaller hand operated plant may be required. Filling around and above pipework also needs particular care and normally a pipe trench should not be excavated nor the pipe laid until the fill level is not less than 500 mm above the intended level of the crown.

Moisture in the fill will influence the effect of compaction. There is a particular moisture content, dependent on the nature of the fill material and the compaction plant, at which material can be compacted to a maximum density. This is termed the optimum moisture content and values for various soil types can be obtained from testing. The hydraulic conductivity will increase dramatically with decreased moisture content below the optimum value. The general implication is that the material should be close to its optimum value to produce the most satisfactory fill, particularly for impermeable clay layers.

In practice, clay used at maximum density in an impermeable zone should contain sufficient water to remain intact when rolled in the hand to form a thread of 3 mm diameter. If it does not, then the soil is classified as dry and the addition of water is required. Also, if the measured moisture content is lower than desired, the fill should be watered slightly to increase the moisture content for placing. This must be added sparingly and excess moisture must be avoided, as compaction plant will not be able to operate on the material. There may be a requirement to monitor the quality of the compaction during the works. This may involve the use of *in situ* nuclear density meters, or *in situ* density assessments by using sand replacement tests. Monitoring of the moisture content at which the fill is placed should also be carried out. These *in situ* methods of measuring the density should be compared with laboratory tests to ensure that the materials are adequately compacted.

7.9.3. Choice of construction method

All spoil deposited wholly or mainly in a solid state must be placed in layers. The three methods given in Table 7.3 have been widely used with success in the UK mining industry for the construction of spoil heaps and water treatment lagoons.

In general, Method A or B should be used where stability is the main requirement. Method C is useful where hydraulic conductivity is the main requirement. Where material may be liable to spontaneous combustion(i.e. some colliery spoils), Method A or B should be used.

7.9.4. Soil Density

The density to which a particular soil can be compacted depends upon:

1. Moisture content
2. Type of compaction equipment
3. Thickness of layer to be compacted
4. Number of passes of the roller
5. Roller speed.

Table 7.3 Common methods for spoil compaction

Method	Description of placement
A	In layers not exceeding 300 mm thick.
B	In layers not exceeding 300 mm thick and compacted with a minimum of 4 passes of a towed smooth wheeled roller having a weight not less than 5 tonnes/m width, or its equivalent.
C	In layers not exceeding 500 mm thick. The thickness must be comensurate with stability requirements and practical considerations.

The compaction requirements that should be observed for the different soil types to achieve optimum compaction at the specified moisture contents are given in Table 7.4 based on the standards required by the UK Department of Transport. This gives guidance on the maximum thickness of the layers and the minimum number of passes. If these guidelines are adopted, a relatively well-compacted fill material should result. If there are any doubts regarding fill material or compaction requirements, specialist advice should be sought.

Sheepsfoot rollers are normally used on relatively soft cohesive soils in conjunction with a bulldozer blade to spread, mix and compact the soil. Subsequent passes of the roller, in conjunction with the blade, rework and compact the soil. Granular soils normally require vibratory compaction, especially if they are not well graded.

Table 7.4 Compaction requirements for typical soil types

Type of compaction plant	Category	Max depth of compacted layer (mm)	Minimum number of passes	Soil type
Smooth wheeled roller (or vibratory roller without vibration)	over 2100 kg up to 2700 kg	125	8	cohesive
	over 2700 kg up to 5400 kg	125		or
				granular
Mass/metre width of roll:	over 5400 kg	150	4	
Grid roller	over 2700 kg up to 5400 kg	150	10	stiffer
Mass/metre width of roll:	over 5400 kg up to 8000 kg	150	8	cohesive or
	over 8000 kg	150	4	granular
Sheepsfoot (tamping) roller Mass/metre width roll	over 4000 kg	150	4	
Pneumatic-tyred roller	over 1000 kg up to 1500kg	125	6	softer
Mass per wheel:	over 1500 kg up to 2000 kg	150	5	cohesive
	over2000kgupto2500kg	175	4	or
	over 2500 kg	200	4	granular
Vibratory smooth—wheeled roller	less than 700 kg over 700 kg	100	unsuitable	granular
Mass per metre width of a vibratory roll;	up to 1300 kg over 1300 kg	125		
	up to 1800 kg over 1800 kg	150	12	
	up to2300kg	175	8	
			4	
	over 2300 kg	200	4	

8. SELECTION OF MATERIALS FOR PASSIVE SYSTEMS

8.1. Pipework

8.1.1. General

Pipework can be placed in two categories:

- that passing over or around the treatment site and
- that passing through or beneath the treatment ponds.

The former need to be installed at a depth great enough to avoid damage, and must be sufficiently flexible either at the joints or in the pipe itself to accommodate movement of the supporting ground. Pipes passing beneath the ponds must not only be flexible, but also sufficiently strong to withstand the high loads produced by differential settlement as a result of the embankment loading. It is prudent in the latter case to adopt the most suitable type of pipe, normally spun or ductile iron or concrete. The high cost of replacing or repairing a pipe that is experiencing problems will far outweigh any initial supply costs.

Where pipework enters or leaves a structure, allowance for settlement and differential movement should be included by the provision of short lengths of pipe adjacent to the structure. This is often termed a ‘rocker pipe’.

Whatever type of pipe is used, the pipe and joints must be capable of withstanding the internal pressures in the pipeline. Thrust blocks are likely to be needed on changes in horizontal or vertical direction of the pipework. Manufacturers' catalogues or publications from their Associations should give information on the use of various types of pipe. Whatever type of pipe is adopted, the quality and care of construction are critical to the long-term functioning of the pipework.

8.1.2. Hydraulic considerations

Pipelines may run full or part-full, and the flow characteristics are different in each case. When running full they act as pressure mains and when part-full as gravity drains. The rate of flow is a function of the pipe diameter, the hydraulic gradient (which for gravity drains equals the pipeline gradient) and the frictional resistance of the pipeline to flow.

The rate of flow in a pipeline, can be found by use of tables based on the Colebrook-White equation. Table 8.1 contains data for the range of pipeline conditions likely to be encountered in passive treatment applications. The table can be used to determine the appropriate pipe diameter to accommodate a certain flow depending on the pipe gradient. The tabular values are for pipe-flow conditions with a frictional resistance factor appropriate to the conveyance of surface water. To allow evaluation of flows when the pipe is not flowing full, Table 8.2 gives the ratio of pipe-full discharge and velocity for partial flow conditions.

Other points to keep in mind when undertaking the hydraulic design of a pipeline or pipe network include:

- At peak flow-rate the velocity should exceed 0.7 m/s to prevent deposition of inert solids, but should not exceed 3 m/s to avoid scour.
- Reducing the gradient of a pipe by one half only results in a velocity reduction of 30%.
- The velocities of flow in a pipe flowing full, and one flowing half-full, are equal.

As an example of how to make use of Tables 8.1 and 8.2, a hypothetical 1000 m pipeline, with a fall of 20 metres, is required to discharge 35 L/s when it is running one third full. From Table 8.2 it can be seen that the proportionate discharge for a pipe flowing one third full is 0.24, and therefore the full-bore flow of a pipe, which carries 35 L/s at one third full, is 145.8 L/s. The hydraulic gradient of the pipe is 0.02 (20/1000). By following the relevant row along Table 8.1, it can be seen that a 300 mm diameter pipe is capable of carrying 157.4 L/s, and therefore this is the appropriate diameter of pipe required for the design flow. Returning to Table 8.2, the proportionate velocity of a pipe flowing one third full is 0.82, and therefore the velocity in the design pipe will be 2.23 m/s (from Table 8.1) multiplied by 0.82 i.e. 1.83 m/s.

It should be noted that for gravity pipelines the head causing flow is the difference in level between the pipe invert level and the point of discharge, over the relevant length of pipeline. However, for pressure pipelines it is the head above discharge level due to a pump or storage tank water level.

Table 8.1. Pipe full flow capacity for discharge of surface water

Pipe Diameter (mm)		100		150		225		300		375	
Hydraulic Gradient		Velocity m/s	Q L/s	Velocity m/s	Q L/s	Velocity m/s	Q L/s	Velocity m/s	Q L/s	Velocity m/s	Q L/s
0.0010	1/999	0.23	1.84	0.31	5.45	0.40	16.1	0.49	34.6	0.56	62.2
0.0011	1/909	0.25	1.93	0.32	5.73	0.42	16.9	0.51	36.2	0.59	65.3
0.0012	1/833	0.26	2.02	0.34	5.99	0.44	17.6	0.54	37.8	0.62	68.3
0.0013	1/769	0.27	2.11	0.35	6.25	0.46	18.4	0.56	39.4	0.64	71.1
0.0014	1/714	0.28	2.19	0.37	6.49	0.48	19.1	0.58	41.0	0.67	73.9
0.0015	1/666	0.29	2.27	0.38	6.73	0.50	19.8	0.60	42.4	0.69	76.5
0.0018	1/555	0.32	2.50	0.42	7.39	0.55	21.7	0.66	46.5	0.76	83.9
0.0020	1/500	0.34	2.64	0.44	7.80	0.58	22.9	0.69	49.1	0.80	88.6
0.0024	1/417	0.37	2.90	0.48	8.56	0.63	25.2	0.76	53.9	0.88	97.2
0.0028	1/357	0.40	3.14	0.52	9.27	0.68	27.2	0.82	58.3	0.95	105.1
0.0032	1/312	0.43	3.36	0.56	9.92	0.73	29.1	0.88	62.4	1.02	112.4
0.0036	1/278	0.46	3.57	0.60	10.5	0.78	30.9	0.94	66.2	1.08	119.3
0.0040	1/250	0.48	3.77	0.63	11.1	0.82	32.6	0.99	69.9	1.14	125.9
0.0044	1/227	0.50	3.96	0.66	11.7	0.86	34.3	1.04	73.3	1.20	132.1
0.0050	1/200	0.54	4.23	0.71	12.5	0.92	36.6	1.11	78.2	1.28	140.9
0.0055	1/182	0.57	4.44	0.74	13.1	0.96	38.4	1.16	82.1	1.34	147.9
0.0065	1/154	0.62	4.84	0.81	14.2	1.05	41.8	1.26	89.3	1.46	160.9
0.0070	1/143	0.64	5.02	0.84	14.8	1.09	43.3	1.31	92.7	1.51	167.0
0.0080	1/125	0.68	5.38	0.90	15.8	1.17	46.4	1.40	99.2	1.62	178.7
0.0090	1/111	0.73	5.71	0.95	16.8	1.24	49.2	1.49	105.3	1.72	189.6
0.100	1/100	0.77	6.03	1.00	17.7	1.31	51.9	1.57	111.0	1.81	199.9
0.011	1/91	0.81	6.32	1.05	18.6	1.37	54.5	1.65	116.5	1.90	209.8
0.013	1/77	0.88	6.89	1.15	20.3	1.49	59.3	1.79	126.7	2.07	228.2
0.015	1/67	0.94	7.40	1.23	21.8	1.60	63.7	1.93	136.2	2.22	245.2
0.017	1/59	1.00	7.89	1.31	23.2	1.71	67.9	2.05	145.1	2.36	261.2
0.020	1/50	1.09	8.57	1.42	25.2	1.85	73.7	2.23	157.4	2.57	283.4
0.024	1/42	1.20	9.39	1.56	27.6	2.03	80.7	2.44	172.5	2.81	310.6
0.028	1/36	1.29	10.2	1.69	29.8	2.19	87.3	2.64	186.5	3.04	335.6
0.032	1/31	1.38	10.9	1.81	31.2	2.35	93.3	2.82	199.4	3.25	358.9
0.036	1/28	1.47	11.5	1.92	33.9	2.49	99.0	2.99	211.6	3.45	380.8
0.040.	1/25	1.55	12.2	2.02	35.7	2.63	104.4	3.16	223.0	3.63	401.5
0.050	1/20	1.73	13.6	2.26	40.0	2.94	116.8	3.53	249.5	4.07	449.1
0.060	1/17	1.90	14.9	2.48	43.8	3.22	128.0	3.87	273.4	4.45	492.1
0.080	1/12	2.20	17.3	2.86	50.6	3.72	147.9	4.47	315.9	5.15	568.5
0.095	1/10	2.39	18.3	3.12	55.2	4.05	161.2	4.87	344.3	5.61	619.6

Table 8.2 Proportion of flow and velocity in pipes running part full

PROPORTION OF FULL BORE VALUE			
Depth	**Velocity**	**Discharge**	**Remarks**
1.00	1.00	1.00	Pipe full
0.95	1.09	1.07	Max discharge
0.80	1.14	0.97	Max velocity
0.75	1.13	0.91	3/4 depth
0.50	1.00	0.50	1/2 depth
0.33	0.82	0.24	1/3 depth
0.25	0.70	0.14	1/4 depth
0.17	0.55	0.07	1/6 depth
0.05	0.25	0.01	1/20 depth

8.1.3. Pipe bedding and surround

The appropriate bedding and surround to pipework must be provided to ensure long-term reliable function and durability of the pipework, and should not have any adverse effects on other aspects of the lagoon embankments. The bedding must be suitable to allow the required strength of the pipe to be developed and this may often mean that the excavated material cannot be used as bedding and surround. A granular bedding is required in many instances and such materials should be compatible with the type of pipe. In certain instances, a concrete bedding or surround may be required.

Where there is no alternative to pipework passing under a treatment pond, the bedding must comprise a cohesive or clay backfill with anti-seepage collars, as seepage along granular bedding cannot be tolerated. Alternatively, a concrete surround could be used, but this can be expensive, and must include provision for settlement and differential movement by the inclusion of flexible discontinuities in the concrete surround at the pipework joints.

Granular bedding or backfill to a pipe may allow seepage of groundwater along the line of the pipe, but this can be mitigated by a series of clay or concrete cut-offs to inhibit flow.

Table 8.3 Minimum trench width for pipes

Nominal pipe size (mm)	Overall trench width (m)	Nominal pipe size (mm)	Overall trench width (m)
100	0.55	600	1.35
150	0.60	675	1.45
225	0.70	750	1.50
300	0.85	825	1.60
375	1.05	900	1.90
450	1.15	975	2.00
525	1.20	1050	2.10

A combination of these various requirements dictates that, for buried pipes, the trench width required will be substantially greater than the pipe diameter itself. Table 8.3 provides indicative values for trench widths.

8.1.4. Manholes

Where pipes change direction or intersect, it is good practice to provide a manhole to allow the pipes to be cleaned if they become blocked. Simple arrangements can readily be incorporated into standard manhole constructions. The structure may settle slightly after construction and to ensure some flexibility, short lengths of pipe are incorporated into the drainage run to allow accommodation of movement at the additional pipe joints.

8.2. Channels

8.2.1. General

In passive treatment systems surface channels offer some advantages to pipelines. Principally, where waters have a high suspended solids load, channel cleaning is much simpler than equivalent cleaning operations for pipes. Conversely, because flow velocities tend to be higher in pipes, the likelihood of settlement of suspended solids is lower in pipes, and therefore maintenance requirements are less frequent. Channels also offer simpler access if, for example, flow monitoring devices (e.g. V-notch weirs) are to be installed. However, depending on construction materials, channels tend to be more expensive to install.

8.2.2. Hydraulic considerations

The design of channels can be carried out using the Colebrook-White formula. However, because of the variety of cross sections used in channel design, it is difficult to tabulate suitable values. A simpler method for channel design is to use the Manning formula:

$$V = (M^{0.67} \cdot S^{0.5}) / n$$

where,

where,	V	=	velocity (m/s)
	M	=	hydraulic mean depth (m)
	S	=	gradient
	n	=	roughness coefficient

and the hydraulic mean depth (M) is the cross-sectional area of water (m^2) divided by the wetted perimeter (m).

Thus, if the capacity of a concrete channel of 1.0 m width and 0.5 m depth, and gradient of 1 in 1000, is to be determined:

$$\begin{aligned} M &= \text{cross-sectional area / wetted perimeter} \\ &= 0.5 / 2.0 \\ &= 0.25 \end{aligned}$$

$S = 1 / 1000$
$= 0.001$

$n = 0.013$

(typical values for n, for different materials, can be found in standard hydraulics text books)

and therefore, $V\ (m/s) = (0.25^{0.67} \times 0.001^{0.5}) / 0.013$
$= 0.96\ m/s$

and since V = Q x A, $Q\ (m^3/s) = 0.96 \times (1.0 \times 0.5)$
$= 0.48\ m^3/s$

8.3. Drainage systems and materials

Drainage materials for an efficient drainage system must be durable and fully compatible. Drains must retain the ability to discharge sufficient quantities of flow but must be constructed such that material does not clog the drain and reduce its efficiency. A combination of permeable fill, geotextiles and pipes is often the preferred solution to seek to ensure longevity of operation. Permeable fill is generally used in conjunction with porous pipes, and more recently geotextiles, to assist the flow of water into the drainage pipes. It also functions as the granular bedding for the drainage pipework. Washed gravel and crushed stone are generally used as permeable fill materials. Other materials that may be available locally include blast furnace slag and clinker.

Toe drains should be constructed in straight lines or regular curves, with careful excavation to minimise the disturbance to the adjacent ground. Where a pipe is included in the drain, the pipe and bedding should be carefully placed. The trench fill material should be placed carefully into the drain from a height of not more than 1.5 m to avoid segregation. The use of transverse horizontal sight rails to form a uniform gradient along the trench is also recommended.

Adequate precautions should be taken to prevent damage to or contamination of the drain by the movement of plant or from material adjacent to the drain. Any section of drain or constituent material that is damaged during construction should be replaced.

8.4. Inlet and outlet structures

Within settlement lagoons and wetlands the flow should be spread as evenly as possible across the width of the pond, to avoid "streaming" and short-circuiting. Although the practice is somewhat more complex than the theory implies, extreme sophistication in achieving this objective is not required. In passive treatment systems there are basically two methods of flow distribution:

a) full length inlet and outlet weirs
This system appears to give the greatest potential for even distribution of flows through a pond. However, it is a relatively costly system to construct, requiring civil engineering skills in compaction of ground to mitigate differential settlement, carpentry, reinforcement and concreting. A common

alternative configuration to a full length is a crenelated weir. By design such weirs and channels reduce the velocity of the influent to a minimum. However, in doing so it promotes sedimentation of solids in the channel, which may reduce distribution efficiency.

The siltation can be readily overcome by regular maintenance, although experience shows that at many remote treatment sites this does not readily occur. The maintenance of full length weirs at the toe of steep inner slopes of ponds and adjacent to deep water brings health and safety issues for employees carrying out such tasks.

b) multiple pipe outlet systems

This system uses multiple pipe inlets or outlets, over the width of the pond, to achieve flow distribution. For example, a pond 30 metres wide could have between 4-8 inlets / outlets to spread the flow reasonably over the pond width. This is a relatively cheap system to construct and it involves only basic civil engineering skills. It is also amenable to adjustment as pipework systems can be disconnected and re-connected to a revised layout if required. The system is based upon 'twin wall' pipework, incorporating 'tees' at intervals to allow adjustment of inflow / discharge rate over the width of the pond.

There is often an understandable resistance to the use of pipe inlet / outlet systems because of the risk of blockages caused by sediments in the mine water. However, if the systems are designed in accordance with normal 'foul drainage' practices the velocity will tend to maintain adequate pipeflow conditions. In the event of maintenance requirements, pipework capacity can be restored by 'rodding' to remove accumulated sediments. This can generally be achieved from a safe location, away from deep water, which will reduce health and safety risks.

Both types of inlet / outlet structures have their advantages and disadvantages. Weir channels area easier to maintain, but the frequency of maintenance is greater. Engineering weir channels is more complicated, and the cost tends to be higher. Also, in the UK at least, they tend to be constructed from concrete, which may detract from amenity value. Complete blockage of pipes tends to be more of a risk, especially at the outlet end of wetlands. This can be mitigated against to an extent by avoiding planting reeds in the vicinity of the outlet pipes, but plants have a habit of colonising any shallow-water areas.

Wetland plants are themselves an excellent means of distributing flow, and wherever possible their growth should be encouraged.

8.5. Selection of wetland plants

8.5.1. Appropriate plant species

It is essential to consider the indigenous species when choosing plant species for use in wetlands. Local species will not only be intrinsically suited to the local climate and environmental conditions, but the resulting system will also blend into the surrounding area more easily. The suitability of local species will ensure that growth of the plants in the wetland will be maximised. It should be borne in mind, however, that locally thriving wetland plants may grow in conditions comprising saturated ground whereas when in use in the wetland, there will probably be a submerged water depth of up to 300 mm over the plant.

Table 8.4. Growing conditions of species used in constructed wetlands (at latitudes in the range 50 - 60º).

Species	Maximum altitude (m)	pH range	Maximum water depth (m)
Phragmites australis	400	3.5-10	0.50
Typha latifolia	500	3.5-10	0.75
Iris pseudacorus	300	6-9	0.25
Scirpus lacustris	300	4-9	1.00
Juncus effusus	800	4-6	0.25

Knowledge of local species can be gained in many cases from local environmental bodies or by site visits. However, experience has revealed those species that are most amenable to the unusual quality conditions of mine waters, and typical growth conditions for these plants is given in Table 8.4. It is important to note that the majority of species cannot stand large fluctuations in water depth. It is therefore essential, when engineering a system, that variations in water level will be minimal.

In most treatment schemes, selection of plant species will be made with final discharge water quality as the main objective. However, where wetlands may be located in situations close to local housing and communities, it is beneficial to use a variety of species rather than a monoculture. This will provide a more aesthetic treatment system and attract wildlife.

The different species show some variation in their ability to tolerate metals. The most tolerant are *P. australis* and *T. latifolia* , with *I. pseudacorus* being the least tolerant of high concentrations of heavy metals. Iris is best used at the margins of constructed wetlands, which have additional aesthetic and wildlife benefits. It is also important to note that plant growth may be restricted in wetlands which receive very high levels of metals (e.g. [Fe] > 100 mg/L), particularly close to the inlet of a wetland, and during the first few years of growth.

8.5.2. Sources of reeds

There may be advantages if plants are purchased from a nursery local to the area of planting, in order that the plants are already suited to the local climatic conditions. However, quite often plants are purchased for sites from other parts of the country and, except in extreme conditions, growing success remains likely. Plants should be purchased as 9-cm pot grown or 400-ml container grown, which should be well established at the time of planting, with 1 or 2 shoots/rhizomes formed.

Several successful wetlands have been constructed using plants transplanted from adjacent areas. The removal of 'clumps' of plants using excavators, to be replanted in the new wetland, can be more expensive than purchase and planting of nursery specimens. In addition there is a risk of compaction of the new wetland by excavators during the replanting exercise.

8.5.3. Planting and cultivation techniques

The formation of the wetland should be constructed level and it is wise to carry out any necessary re-grading to correct discrepancies. The corrections necessary can be ascertained by flooding the wetland area to determine any high and low areas.

The whole of the surface area of the wetland should be thoroughly broken up by suitable means to produce a reasonably fine tilth. Care shall be taken not to raise underlying material to the surface. Cultivation must not be carried out when the soil is excessively wet or in conditions likely to damage the soil structure. Any vegetable matter or debris brought to the surface during this operation should be removed.

Planting is typically at a density of 4 plants / m^2, although significant growth and cross-colonisation should occur in the first year of growth, and therefore a density of 3 plants / m^2 is often acceptable. Planting should be in staggered rows into damp soil with 15 to 20mm soil cover over the top level of the plant roots. The plants should be set upright and firm. Plants should always be kept moist and should not be stored on site.

8.5.4. Water depth

The plants should be maintained at the correct 'wet' condition during the whole period of planting. It is important not to subject first-year seedlings to sustained submergence as this will lead to the death of the seedlings. The outlet weirs should be set so as to maintain the appropriate minimum depth of water on the wetlands following completion of planting, to allow the plants to become established. When plants become established, it is appropriate to increase the water depth to a working depth of 200-300 mm. *Iris pseudacorus*, however, requires a water depth <100 mm during its establishment, and raising of the level of the bed in areas to be planted with this species should be contemplated.

The optimum planting season for wetland plants in the UK is May-July, which allows the plants to become established before the dormant winter period. This timing will vary slightly with latitude and altitude.

8.5.5. Maintenance and management

Any contract for wetland planting should include for maintenance for a period of one year, with an obligation to replace failed plants written into the contract.

Wetland systems will need some management during their lifetime. There is a natural tendency, particularly where *Phragmites australis* is used, for silting up to occur in the wetland and for vegetation more suited to drier environments to invade. In order to prevent this, engineering of the wetland should include suitable freeboard. Also, occasional thinning of the plants may be necessary to maintain water flow through the system.

The die-back of plants every year results in significant production of leaf matter, essential for the maintenance of a carbon source in the wetland. However, some of this material may be transported out of the system, blocking pipes and drains from the wetland. Preventative measures to avoid blockages should be part of the wetland design. If not, a system of clearance should be included in the management scheme. *Scirpus lacustris* is less invasive than many other species and is therefore a good species to use adjacent to inlet / outlet weirs to reduce the risk of blockage.

9. CONSTRUCTION CONTRACTS AND RELATED MATTERS

9.1. Contractual matters

Contracts for the construction of civil engineering projects are normally made between two parties. The party who commissions the work is normally the landowner and the future undertaker (often referred to as the 'Client', 'Promoter', or the 'Employer' in contractual terminology). The party who constructs the work or provides other construction services is usually referred to as the 'Contractor' and may be a civil engineering contracting firm or a local plant/earthmoving firm. A contract is also required between the parties when various aspects of work forming the site investigation are carried out, or when any specialist construction or other service is required during the construction from a further party.

The contractor undertakes to carry out the specified works for a sum of money in accordance with the employer's instructions, usually within a stated period of time. The employer has an obligation to pay the contractor for the work he carries out and to give possession of the site to the contractor for the duration of the contract. The success of the contract is dependent, to a large degree, on the quality of the employer's instructions to the contractor, normally contained within the contract documentation. This is of legal significance and should include the following distinct elements:

- **form of contract (or agreement)** — forms the contract by reference to the required works and the other documentation
- **conditions of contract** — are the conditions and terms under which the contract is formed
- **specification (including the drawings)** — sets out how the work is to be executed
- **method of measurement** — sets out the basis for the financial arrangements

It is important that careful consideration is given to selecting the most appropriate form of contract documentation. Amendments to standard documentation should be included where necessary, but such changes should be kept to a minimum to reflect the particular requirements and minimise the potential consequences of inappropriate, incorrect or incompatible amendments and additions.

The place where the contract is made binding by communication and acceptance will be defined by the laws of the country that apply.

Civil engineering contracts are usually made in writing in standard forms. These have many advantages and their use is recommended. It is important to be aware that the making of a contract requires no formality. A verbal agreement is not recommended, as there is no record of the terms of the contract.

The basis of the contract is agreement. Agreement comprises an offer and acceptance. An offer must be distinguished from an attempt to negotiate. If accepted, an offer becomes a binding contract. The acceptance of an offer must be unconditional and it must be communicated to the person who makes the offer. The terms of acceptance must correspond precisely with the terms of the offer to ensure that acceptance is unconditional. The words ‘I accept your offer’ constitute an unconditional acceptance and bring a binding contract into existence.

The terms of the contract do not have to be set out in full in the documents, letters or conversations that constitute the offer and acceptance. Terms contained in another document, a set of standard conditions for example, may be incorporated within the contract by reference.

An essential prerequisite of binding contract is that the Agreement must be supported by ‘consideration’. In civil engineering contracts the consideration for the promise made by the contractor to carry out the works will usually b the promise by the employer to pay the price for the works. Reference to the financial arrangements should be clearly defined in the terms of the contract and the details specified or reference made to a standard method of measurement and payment.

Clarity and precision are essential requirements when drafting a contract. Standard forms of civil engineering contract clearly define the rights and duties of the contracting parties and their use is strongly recommended.

If the parties enter into a contract, but there is an error due to omission or misstatement in the contract documentation, the contract may be rectified by the parties given certain provisions. These are that both parties agree the contract is inadequate and agree on the form of rectification to be applied.

The contract should also allow for changes necessitated by conditions revealed during construction or in the undertaker’s requirements. An adequate site investigation should minimise the former whilst the undertaker should clarify his requirements prior to the completion of the design insofar as is possible. Changes to the works agreed with the contractor should be conveyed clearly by means of a written instruction and amended drawings.

The insurance requirements should be covered fully in the conditions of contract, together with the contractor’s obligations to rectify faulty workmanship or outstanding works, both during the construction and for a specified time thereafter. Provisions for the settlement of disputes arising from the contract should also be included in the conditions of contract.

A contract is also recommended with any specialist advisers who may be required to provide advice on any aspect. This should refer to a brief or list of the employer’s requirements in sufficient detail to enable the required services to be performed effectively.

9.2. Contract options

9.2.1. General

Two main options can be adopted by the employer to detail the conditions under which a contract is formed:

- The employer may negotiate with one or more contractors to carry out the required works shown on the drawings and described and defined in the specification. In some cases the contractor(s) may be involved in discussions at the design stage or carry out this work for the employer. Normally, the contractor will offer to carry out the required works to standard terms and conditions, although the employer may amend these by negotiation.

- The employer may prepare contract documentation, normally specifying standard conditions of contract, and including the drawings and specification. Details of the method of measurement and payment will also be included. Several contractors are then requested to price the works.

The use of a non-standard or a standard form of contract is feasible for small constructions. The choice should be the employer's, based on the size, scope and complexity of the proposed works, the degree of control that he is able or willing to exert and the financial arrangements for paying for the works. The contractor(s) should not be able to exert any pressure in the choice of contract and, ideally, the employer should clarify his preferred approach prior to any liaison with the contractor(s). Specialist advice should be taken where necessary. A non-standard form of contract is normally based on a written quotation from a contractor and the contractor's standard terms and conditions.

9.2.2. Standard Form of Contract

Most EU jurisdictions have their own nationally-standardised forms of contract for civil engineering works. For instance, in the UK[24] the standard form of contract recommended is the 'Conditions of Contract, Agreement and Contract Schedule' for use in connection with 'Minor Works of Civil Engineering Construction' (First Edition). Alternatively, the more comprehensive 'Conditions of Contract and Form of Tender, Agreement and Bond' for use in connection with 'Works of Civil Engineering Construction' (Sixth Edition) may be used. Where a ground investigation is to be carried out using a specialist contractor, the standard UK form of contract recommended is the 'Conditions of Contract for Ground Investigation' (First Edition). Should the undertaker wish to amend a standard form of contract, he should seek specialist advice when preparing the contract documentation and before requesting contractors to price the works. These documents may be used as models for contracts in jurisdictions which lack specific national standards (which may be the case in accession countries etc).

9.3. Specification

9.3.1. General

The specification forms part of the contract documents under the ICE standard forms of contract. It should be prepared in conjunction with drawings of the proposed work and should be concise and complete. The specification should describe the workmanship and materials required as well as

[24] Copies of these documents may be obtained from The Institution of Civil Engineers (ICE), Great George Street, London.

indicating the position in the works of the various items if not indicated on the drawing.

The specification should also set out clearly any constraints on the contractors freedom to do the work as, and in the order, he thinks fit.

9.3.2. Drawings

The contract drawings should detail all the contract work. They must give sufficient information to enable all parties to understand the requirement of the project. They provide the most accessible method to convey information. The drawing should be referenced with the title of the project, preferably the name of the employer and a specific title. Each drawing should be separately numbered and, if revised, should be given a letter suffix to delineate the latest version. Whenever information is updated the drawings should be amended, and copies should be circulated to all parties to the contract.

The drawings should include:

- a large-scale plan of the works at a scale of 1:200 (or larger). This should have the main construction works marked on, together with immediately local access routes and any constraints on the works.
- cross-sections of the proposed earthworks, including details of slopes, internal zoning, excavation depths, etc.
- larger scale drawings with details of specific features of the works including pipework, structures, slope protection, drainage, crest works.
- landscaping drawings illustrating the planting scheme across the site.

Additional drawings may be required to show other specific requirements of the project.

Drawings are also required for approval of projects by regulatory authorities. These authorities may require drawings at minimum scales. Where required, advice on the preparation and requirements of contract drawings should be sought from the specialist adviser.

9.4. Safety

9.4.1. Public safety

The design of all structures and equipment, and the construction of a water treatment system must be in accordance with the general safety provisions of relevant health and safety at work legislation, and any associated regulations. Such legislation is aimed primarily at ensuring the safety of persons and others affected by the work activity. Consideration must also be given to the safety of the public who might have free access, or may gain unauthorised access, to the site and adjoining land.

Owners of water treatment systems, or those responsible for their management, must be aware of the dangers associated with water. Whilst there is no legal requirement or necessity to fence off every pond or lagoon it is recommended that where access is readily available fences and warning signs should be erected. This is particularly necessary where there is an abrupt drop to the surface of the water and around inlet or control structures. Where public access is anticipated, warning signs that comply with national requirements should be displayed where they are clearly visible. Lifebelts should also be provided close to the water body and maintained in good operating condition, notwithstanding vandalism. Lagoons lined with HDPE, or similar plastics, may be particularly hazardous, because a person falling into such a lagoon will have extreme difficulty gaining purchase on the lagoon sides to extricate themselves from it.

The presence of chambers, manholes, voids or other restricted areas within any of the structures will be hazardous in terms of access, maintenance and use. They may also collect gases or other noxious substances (particularly in coal mining areas). Statutory constraints apply to the entry into such areas, but these should be minimised in the design whenever possible. The design should always include consideration of the construction and operation of the site to minimise potential hazards wherever possible.

The inlets or outlets of pipes and other structures will prove attractive to children. Where possible, such features should be avoided by maintaining high water levels or providing lockable screens. Other confined spaces, e.g. manholes, valve chambers, etc., should be securely locked at all times and designed to have a free flow of air if possible.

The provision of shallow gradients around the perimeter of wetlands, where easy access can be gained, will be of considerable value as a safety measure. Transitions from shallow to steeper slopes below water should be gradual to avoid a hazardous sudden change in bed level. It is also recommended that provisions be made to assist a person to climb out if steep slopes are present. A suitably inexpensive arrangement would be a 'ladder' of vehicle tyres, securely tied together and anchored to the top of the slope. Where vertical walls or structures abut deep water, hand holds and/or rungs may be necessary.

Areas of soft mud or silt may be exposed if the water level drops or is lowered for any reason. These may be dangerous and may require temporary fencing and signing. Pollution, or the development of various algal growths, may be a public health risk, and may require temporary fencing or signing.

9.4.2. Construction Safety

The construction of a treatment system must be undertaken in accordance with national rules and regulations. For instance in the UK, works have to be implemented in accordance with the Health and Safety at Work Act and the Construction (Design & Management) (CDM) Regulations. Similar controls are in place across Europe. Prior to construction, the undertaker should take steps to become aware of the hazards which may exist, so that the operations can be planned in such a way as to eliminate risks by design, if possible, and to minimise and control any residual risks. Some of the more common hazards that are encountered on sites involving earthworks are:

- The failure of temporary slopes including failure of slopes in cuttings, trenches and embankments. An appraisal of the stability of such slopes should be made if failure could cause a potential hazard and, if necessary, the slope cut to a safe angle or shoring installed. Special care should be taken where groundwater and/or soft or variable soils are encountered, as this will substantially reduce the stability.
- Unless specifically allowed for in the design of an excavation, heavy plant should not be allowed to approach, nor excavated material placed near the edge of, slopes, whether cut or filled, or near trenches or other excavations.
- Collapse of material when excavating from the base of a working face. It is important to ensure that the face is not overhanging or excessively high. If a localised failure occurs, there should be

no risk to the driver and excavator or any other operatives or plant.

- Undermining where bench working is being carried out, such that the higher benches are undermined and there is a risk of material falling or rolling from the upper levels.

9.4.3. Safety related to construction plant

The following general guidelines should be adhered to during construction:

- Heavy plant should be routed along distinct haul roads, preferably separate from pedestrians and light traffic. Wherever possible, the haul road should be formed so as to maximise sight lines by avoiding or smoothing bends and any high sections or humps along the route.
- Haul roads should be well maintained and kept in good condition as this will maximise vehicle control and minimise the braking distance. Gradients should be kept to a minimum and in dry weather the surface of the haul road should be watered to minimise dust clouds.
- Visibility from many large machines is poor, particularly when reversing, and it is advisable to keep clear of such machines. If this is difficult, such as when setting out or checking the fill, the drivers should be warned verbally or by roadside signs.
- Overhead obstructions such as cable bridge soffits, or temporary works should be indicated clearly by signs or tapes.
- Drivers and operators must be trained and fully competent in the use of the plant.
- Plant should be regularly maintained and not operated beyond its capability or capacity or as described by the manufacturer.
- Plant should be operated with the appropriate guards in position and operatives should wear all necessary protective equipment.
- Tipping of fill on embankments should be carried out short of the edge of the advancing layer of fill and then the additional fill dozed forward.
- Vehicles that travel on public roads must, by law, be in a roadworthy condition. Where mud is spread on public roads by the wheels of site plant, road-cleaning / wheel washing equipment should be used to minimise the possibility of accidents involving the public. This is particularly important in the winter months.
- Wheeled vehicles working on slopes can slide out of control. Tracked vehicles are safer on steeper slopes but damage to the ground surface is more severe.
- Grass covered slopes are intrinsically hazardous and more likely to lead to sliding, the danger increasing with steeper slopes.

10. MAINTENANCE AND RENOVATION OF PASSIVE SYSTEMS

10.1. Regular inspection requirements

10.1.1. Introduction

Visual observations and monitoring should be carried out at regular intervals after completion of the water treatment system to ensure problems associated with the structures and adjacent areas are identified at an early stage in their development. These problems are generally associated with the following:

- instability
- seepage
- erosion
- blockage (of channels and flow structures)
- theft, vandalism and damage

Regular monitoring during the first filling of the system is particularly important, as many problems will only become apparent at this stage. The embankments should he checked visually, at least daily, during this time and action taken if any seepage's or other matters of possible concern become apparent.

10.1.2. Inspection Regime

It is important that all water treatment systems are regularly inspected to ensure that they remain in a safe and secure condition as well as operationally efficient throughout their lifetime. Most treatment systems will be unmanned, receiving only regular visits for maintenance purposes by mechanical / electrical operatives. It is important that a person with an understanding of water engineering and geotechnical analysis visits the site at regular intervals to review the safety and stability of the ponds, embankments and all associated structures. Consideration of the safety of the site in respect of unauthorised entry and facilities for rescue from water bodies will also be carried out. Remedial works programmed if found to be necessary.

10.1.3. Frequency of inspection

The frequency of inspection is related to the complexity of each treatment system and the potential risk that it may pose. The inspections that would become routine for each site are in addition to those made by maintenance operatives, although the latter are a valuable source of regular information on the site.

A typical inspection schedule for various types of treatment installation is:
Daily: Where a treatment system has daily attendance, a visual check of the full extent of the installation should be carried out and this should be recorded in a log-book.
Weekly: A formal inspection by the site foreman should be made using a form designed to cover all operational and safety aspects of the site. Any factors giving cause for concern should be reported to the site manager.

6 Monthly: An inspection by a civil engineer experienced in construction of earth structures and geotechnical appraisal to ensure that there is no deterioration of the standards of stability and security of the site. A typical formal report is shown within this chapter as Figure 10.1. All treatment systems that incorporate significant water bodies should have this inspection as part of the routine scheduled for the site.

10.2. Long-term maintenance and renovation

Definitive guidance on the long-term maintenance, and particularly the wholesale renovation, of passive treatment systems in Europe is difficult to provide, for the simple reason that it has not yet had to be undertaken. True, some passive treatment systems have had to be abandoned, due to inadequate design or factors outside the control of the designer, but there is no experience of renovating a successful system that has simply reached its design capacity.

Thus, the following issues are those, in the opinions of the authors of these guidelines, that are likely to be important in terms of long-term maintenance and renovation of passive treatment systems. Some of the points below are based on actual experience (e.g. metal sludge removal), and others are likely maintenance issues, based on observations of sites that have been monitored for 5 – 10 years (i.e. the oldest passive treatment systems in Europe).

Sludge removal is without doubt the most important long-term maintenance issue for passive treatment systems. Without doubt, the removal of ochreous sludge from settlement lagoons will be the single biggest long-term maintenance cost, unless lagoons are massively over-sized in terms of the metal load they receive. Typically sludge removal entails the use of mobile 'sludge gulpers', which suck the wet sludge from the base of lagoons. It is then necessary to dispose of the sludge. Despite on-going research efforts to find novel re-uses for ochre, it remains the case that supply far out-strips demand. Invariably landfill is the final destination for ochreous sludge. However, the costs of disposal to landfill are high, and only likely to rise in the future as international legislation increasingly pushes waste minimisation policies. Landfill costs are particularly high where sludge contains metals that are classified as hazardous, such as Pb, Zn, Cd. It is therefore recommended that sludge is dewatered prior to disposal, to reduce the total volume and cost of disposal. Common practice in the UK has been to construct sludge drying beds on the treatment site. Sludge is simply transferred from the settlement lagoon(s) to the sludge drying bed, where it gradually dewaters due to gravity drainage and evaporation. There are four main mechanical alternatives to sludge drying beds (Younger *et al.*, 2002):

- Vacuum filtration
- Continuous pressure dewatering
- Frame-and-plate pressing
- Centrifuge separation

It is important to enlist specialist advice in the application of such technologies, since the operating conditions of these mechanical devices must be correct if efficient dewatering is to be accomplished.

In terms of renovation of passive treatment systems the main issues will be associated with the removal and disposal of (metal-enriched) compost media, soils, reeds, and inorganic / inert media. It is most unlikely that a reuse option will be available for any of these treatment media, and therefore they too will need to be disposed of to landfill. In planning such renovation, due consideration must be given to how to prevent any sudden release of metal-contaminated solids or liquids from the media during their removal. The main costs of such a renovation operation will be:

- plant for removal and transport to landfill
- landfill charges
- plant for replacement of media / reeds
- material costs for whatever has been removed e.g. soils, reeds, liners (if they have been removed, or irreparably damaged during excavation of media)

Although difficult, it is important to endeavour to build these long-term maintenance and renovation costs into a full cost-benefit analysis of the most appropriate treatment option, right at the outset of a passive treatment system design project.

Figure 10.1. Typical format of a 6 monthly engineer's inspection form for a mine water treatment system.

MINE WATER TREATMENT SITE
ENGINEERS INSPECTION REPORT

Date of this report	Date of last report	Weather during inspection

Any parts of the tip excluded from this inspection ?

Item	No	Yes	Item	No	Yes
1 Any incidents/evidence of intrusion or vandalism or inadequate security on the site?			10 Are all manholes, covers & ditches in satisfactory condition?		
2 Any excavation since the last inspection?			11 Is the final discharge arrangement clear of obstructions & in satisfactory condition?		
3 Any slumping, bulging, cracks or fissures indicating movement of lagoons or wetland?			12 Is the lifesaving equipment in satisfactory condition?		
4 Any seepage from the ponds or wetland?			13 Does the pond effluent quality appear satisfactory?		
5 Any slumping, bulging, cracks or fissures indicating movement of earth slopes on the site?			14 Are fences & warning notices around the treatment ponds adequate and satisfactory?		
6 Any slumping, bulging, cracks or fissures indicating movement of the Gabion Wall?			15 Are fences/barriers around the pumphouse adequate and satisfactory?		
7 Any erosion or undercutting of slopes?			16 Are fences/barriers around the adit adequate and satisfactory?		
8 Is the lagoon drainage arrangements into & between ponds satisfactory?			17 Any indication of leakage from pumping mains?		
9 Is the lagoon freeboard at least 750mm?			18 Are the management rules & specification being complied with?		

19 Comments on all entries in heavy-lined boxes,

20 Details of any features giving cause for concern.(*Immediately inform the Project Manager*)

21 Maintenance or remedial works required

22 Any other remarks.

Signature Countersignature

....Date................. ....Date ....

IMC Consulting Engineers Ltd.

REFERENCES

Ackers, P., White, W.R., Perkins, J.A., and Harrison, A.J.M. (1978) *Weirs and flumes for flow measurement.* John Wiley and Sons, Chichester. 327pp.

Ackman, T.E. (2000) Feasibility of lime treatment at the Leviathan Mine using the In-Line System. *Mine Water and the Environment*, **19**, 56-75.

Ackman, T.E. and Kleinmann, R.L.P. (1993) An in-line system for the treatment of mine water. *International Mine Waste Management*, **1**(3), 1-4.

Ackman, T.E. and Place, J.M. (1987) *Acid mine water aeration and treatment system*. US Patent No. 4,695,378, Sept 22.

Amos, P. and Younger, P.L. (2003) Substrate characterisation for a subsurface reactive barrier to treat colliery spoil leachate. *Water Research*, **37**, 108 - 120.

Appelo, C.A.J. and Postma, D. (1993) *Geochemistry, Groundwater and Pollution*. A.A. Balkema Publishers, Rotterdam. 536 pp.

APHA (1998) Standard Methods for the Examination of Water and Wastewater. 20th Edition. American Public Health Association, American Water Works Association and the Water Environment Federation. Washington D.C.

Arnesen, R.T., Bjerkeng, B., and Iversen, E.R. (1997) Comparison of model predicted and measured copper and zinc concentrations at three Norwegian underwater tailings disposal sites. In *Proceedings of the Fourth International Conference on Acid Rock Drainage* (held Vancouver, Canada, May 31 - June 6, 1997). Volume IV, pp. 1833 - 1847.

Aubertin M., Chapuis, R.P., Bouchentouf, A. and Bussière, B. (1997) Unsaturated flow modelling of inclined layers for the analysis of covers. *Fourth International Conference on Acid Rock Drainage*, Vancouver, B. C., Canada, May 31 – June 6, 1997.

Baacke, D. (1999) *Geochemisches Verhalten umweltrelevanter Elemente in stillgelegten Polysulfiderzgruben am Beispiel der Grube Freiberg/Sachsen*. Unpublished thesis, TU Bergakademie Freiberg, Germany.

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

Bell, A.V., Riley, M.D. and Yanful, E.K. (1994) Evaluation of a composite soil cover to control acid waste rock pile drainage. *International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage*, Pittsburgh, PA, April 1994.

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 Restoration*, **17**(4), 99-107.

Brassington, R. (1998) Field hydrogeology. (Second edition). Wiley, Chichester. 248pp.

Brown, M.E. (1997) *The amelioration of contaminated mine water by wetlands.* Unpublished PhD Thesis, Camborne School of Mines, University of Exeter, UK.

Brown, M.E., Atkinson, K., Wilkins, C., and Norton, P.J. (1994) Contaminated mine water amelioration - study of a long-established natural wetland. *Proceedings of the 5th International Mine Water Congress.* International Mine Water Association. Nottingham, UK, 18 - 23 September 1994. Volume 2 pp. 499 - 519.

Building Research Establishment (1994) *BR 255 – Performance of Building Materials in Contaminated Land.*

Building Research Establishment (2001) *BRE Special Digest 1 – Concrete in Aggressive Ground.*

Carrera, J., Alcolea, A., Bolzicco, J., Bernet, O., Knudby, C., Manzano, M., Saaltink, M., Ayora, C., Domenech, C., de Pablo, J., Cortina, J., Coscera, G., Gilbert, O., Galache, J., Silgado, A. and Mantecon, R. (2001) An experimental geochemical barrier at Aznalcóllar. In Oswald, S.E., and Thornton, S.F., (editors), *Preprints of the 3rd International Conference on Groundwater Quality.* University of Sheffield, UK, 18 - 21 June 2001. pp. 407 - 409.

Casiot, C., Morin, G., Bruneel, O., Personné, J.C., LeBlanc, M., Dusquesne, C., Bonnefoy, V. & Elbaz-Poulichet, F. (2003) Bacterial immobilisation and oxidation of arsenic in acid mine drainage (Carnoulés creek, France). *Water Research* (in press)

Chamberlain, E.J., Erickson, A.E. and Benson, C.H. (1997) *Frost resistance of cover and liner materials for landfills and hazardous waste sites*. US Army Corps of Engineers, Special Report 97–29.

Cohen, R.R.H. and Staub, M.W. (1992) *Technical manual for the design and operation of a passive mine drainage system*. Report to the US Bureau of Land Reclamation. Colorado School of Mines, Golden, CO. December 1992. 69 pp.

Cohen, R.R.H. (1996) The technology and operation of passive mine drainage treatment systems. In US EPA (Editors) *Managing environmental problems at inactive and abandoned metal mine sites*. EPA Seminar publication No EPA/625/R-95/007. pp. 18-29.

Cortina, J.L., Holtermann, I., de Pablo, J., Cama, J., & Ayora, C., (2003) Passive in-situ remediation of polluted water with caustic magnesia. *Environmental Science and Technology* (in press)

Cravotta, C.A. and Trahan, M.K. (1999) Limestone drains to increase pH and remove dissolved metals from acidic mine drainage. *Applied Geochemistry*, **14**, 581-606.

Dawson, R.F. and Morin, K.A. (1996) *Acid mine drainage in permafrost regions: issues, control strategies and research requirements*. MEND Project 1.61.2. Department of Indian and Northern Affairs Canada, Ottawa, Ontario.

Department for Environment, Food and Rural Affairs / The Environment Agency (2002) *The Contaminated Land Exposure Assessment Model (CLEA): Technical Basis and Algorithms, CLR 10*. (See also CLR 7, 8, 9 and associated TOX and GV publications).

Dey, M., Sadler, P.J.K., and Williams, K.P. (2003) A novel approach to mine water treatment. *Land Contamination and Reclamation* **11,** (2), 253 - 258.

Durham, A.J.P., Wilson, G.W., and Currey, N. (2000) Field performance of two low infiltration cover systems in a semi arid environment. *Proceedings of the Fifth International Conference on Acid Rock Drainage*, (ICARD 2000), Denver, Colorado, May 21-24, 2000. Volume II, pp 1319 – 1326.

Eger, P., Wagner, J.R., Kassa, Z., and Melchert, G.D. (1994) Metal removal in wetland treatment systems. *Proceedings of the International Land Reclamation Conference and the Third International Conference on the Abatement of Acidic Drainage*. Pittsburgh, PA, 24-29 April 1994. US Bureau of Mines, Pittsburgh, PA. Vol. I. (US Bureau of Mines Special Publication SP 06A-94), pp 80 - 88.

Elberling, B., Schippers, A. and Sand, W. (2000) Bacterial and chemical oxidation of pyritic mine tailings at low temperatures. *Journal of Contaminant Hydrology*, **41**, 225 – 238.

Ettner, D.C. (1999) Pilot scale constructed wetland for the removal of nickel from tailings drainage, southern Norway. *Proceedings of the Congress of the International Mine Water Association,* Sevilla, Spain, 13-17 September 1999. Volume I, pp. 207-211.

European Commission (2003) *Proposal for a Directive of the European Parliament and of the Council on the management of waste from the extractive industries*. Commission of the European Communities, Brussels. COM(2003) 319 final (2.6.2003). 56pp.

Evangelou, V.P. (1994) Potential microencapsulation of pyrite by artificial inducement of ferric phosphate coatings. *Journal of Environmental Quality*, **24**, 535-542.

Evangelou, V.P., McDonald, L.M. Jr and Seta, A.K. (1998) Acid mine drainage. In Meyer, R.A. (Editor) *Encyclopedia of Environmental Analysis and Remediation*. John Wiley & Sons, New York. pp. 1-17.

Freeze, R.A. and Cherry, J.A. (1979) *Groundwater*. Prentice Hall, Englewood Cliffs, NJ, USA. 604 pp.

Furugård, G. (1985) Utvärdering av tjäldata från olika marktyper i ett riksomfattande observationsnät. Samarbetsorganisationen för fordon-markforskning. *SFM Meddelande*, **35**, 151-172 (in Swedish).

Gammons, C.H., Drury, W.J. and Li, Y. (2000) Seasonal influences on heavy metal attenuation in an anaerobic treatment wetland, Butte, Montana. *Proceedings from the Fifth International Conference on Acid Rock Drainage*. Volume 2. pp. 1159 – 1168.

Gavaskar, A.R., Gupta, N., Sass, B.M., Janosy, R.J. and O'Sullivan, D. (1998) *Permeable barriers for groundwater remediation: design, construction and monitoring*. Batelle Press, Columbus, OH. 176 pp.

Gusek, J.J. (1998) Three case histories of passive treatment of metal mine drainage. *Proceedings of the 19th Annual West Virginia Surface Mine Drainage Task Force Symposium*. West Virginia University, Morgantown, WV, USA.

Hartman, H.L. and Mutmansky, J.M. (2002) *Introductory Mining Engineering*. (Seconf edition). Wiley, New York. 570pp.

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

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

Hem, J.D. (1992) *Study and Interpretation of the Chemical Characteristics of Natural Water* (3rd Edition). U.S. Geological Survey water-Supply Paper 2254. United States Government Publishing Office, Washington DC. 263 pp.

Huntsman, B.E., Solch, J.G. and Porter, M.D. (1978) Utilization of Sphagnum species dominated bog for coal acid mine drainage abatement. *Proceedings of the Geological Society of America*, 91st Annual Meeting (Abstracts), Toronto, Ontario, Canada, pp 322.

Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL) (1987) *Guidance on the Assessment and Redevelopment of Contaminated Land*. ICRCL 59/83. 2nd edition.

Jarvis, A.P. (2000) *Design, construction and performance of passive systems for the treatment of mine and spoil heap drainage*. Unpublished PhD Thesis, Department of Civil Emgineering, University of Newcastle-uopn-Tyne, UK. 231 pp.

Jarvis, A.P. and England, A. (2002) Operational and treatment performance of an unique Reducing and Alkalinity Producing System (RAPS) for acidic leachate remediation in Lancashire, UK. *Proceedings of the International Conference Uranium Mining and Hydrogeology III and the International Mine Water Association Symposium*. Freiberg, Germany, 15-21 September 2002. pp. 1003-1010.

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

Johnson, K.L. (2002) *Manganese in mine water and its removal by passive treatment*. PhD Thesis, Department of Civil Engineering, University of Newcastle. 260pp.

Johnson, K.L. (2003) The importance of aeration in passive treatment schemes for manganese removal. *Land Contamination and Reclamation* **11**, (2), 205 - 212.

Kalin, M. (1998) Biological polishing of zinc in a mine waste management area. In Geller, W., Klapper, H. and Salomons, W. (Eds*) Acidic mining lakes: Acid Mine drainage, Limnology and Reclamation*. Springer, Heidelberg. pp. 321-334.

Kalin, M., Smith, M.P. and Fyson, A. (1997) The use of natural phosphate rock to reduce acidic drainage from pyritic waste rock. *Proceedings of the ICARD Conference*, Vancouver, May 30 – June 6, 1997.

Kennard, M.F., Hoskins, C.G. and Fletcher, M. (1996) *Small embankment reservoirs*. Construction Industry Research and Information Association (CIRIA) publication No. R161.

Kepler, D.A. and McCleary, E.C. (1994) Successive Alkalinity Producing Systems (SAPS) for the treatment of acidic mine drainage. *Proceedings of the International Land Reclamation and Mine Drainage Conference and the 3rd International Conference on the Abatement of Acidic Drainage*. (Pittsburgh, PA; April 1994). Volume 1: Mine drainage, pp. 195-204.

Kepler, D.A. and McCleary, E.C. (1997) Passive aluminum treatment successes. In *Proceedings of the 18th Annual West Virginia Task Force Symposium*, held Morgantown, West Virginia, 15 - 16 April 1997.

Knutsson, S. (1984) *Tjäle och frost*. Handboken Bygg, G21. Liber, Stockholm (In Swedish).

Kruseman, G.P., and de Ridder, N.A., 1990, *Analysis and evaluation of pumping test data.*. (Second edition). International Institute for Land Reclamation and Improvement (ILRI), Wageningen, The Netherlands. ILRI Publication No. 47. 377pp.

Laine, D.M. and Jarvis, A.P. (2002) Engineering design aspects of passive in situ remediation of mining effluents. *Proceedings of the National Conference: Mine Water Treatment: A Decade of Progress*, University of Newcastle upon Tyne, UK, 11-13 November 2002, pp. 25-40.

Lamb, H.M., Dodds-Smith, M. and Gusek, J. (1998) Development of a long-term strategy for the treatment of acid mine drainage at Wheal Jane. In Geller, W., Klapper, H. and Salomons, W. (Eds) *Acidic mining lakes: Acid Mine drainage, Limnology and Reclamation*. Springer, Heidelberg. pp. 335-346.

Lindvall, M., Eriksson, N. and Ljungberg, J. (1999) Decommissioning at Kristineberg mine, northern Sweden. *Proceedings of Sudbury '99 Mining and the Environment II*, September 13 – 17, 1999, Sudbury, Canada, pp 855 – 862.

Lundgren, T. (1995) *Sluttäckning av avfallsupplag* (Closure of waste deposits). Swedish Environmental Protection Agency, Report 4474, (In Swedish).

McCrae, C.W.T., Blowes, D.W. and Ptacek, C.J. (1999) In situ removal of arsenic from groundwater using permeable reactive barriers: a laboratory study. *Proceedings of the Sudbury '99 Conference "Mining and the Environment II"*, held Sudbury, Ontario, Canada. Volume 2. pp. 601-609.

McGregor, R., Blowes, D., Ludwig, R., Pringle, E., Choi, M., Booth, R., Duchene, M. and Pomeroy, M. (1999) Remediation of heavy-metal contaminated groundwater using a porous reactive wall within an urban setting. *Proceedings of the Sudbury '99 Conference "Mining and the Environment II"*, held Sudbury, Ontario, Canada. Volume 2, pp. 645-653.

MEND (1994) *Evaluation of alternate dry covers for the inhibition of acid mine drainage from tailings*. MEND (Mine Environment Neutral Drainage Program) Project 2.20.1.

Merkel, B., Hartsch, J., Pinka, F. and Peters, H. (1997) *Chemische Veränderungen nach Auflassen von Uranbergwerken unter besonderer Berücksichtigung von Radionukliden* (on website at http://www.geo.tu-freiberg.de/~merkel/gruflu.pdf).

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.

MMSD (2002) *Breaking new ground. The Report of the Mining, Minerals and Sustainable Development project.* Earthscan Publications Ltd, London. 441pp.

Mueller, R.F. (2001) Microbially mediated thallium immobilization in bench scale systems. *Mine Water and the Environment*, **20**, 17-29.

Nairn, R.W. and Hedin, R.S. **(**1993**)** Contaminant removal capabilities of wetlands constructed to treat coal mine drainage. In G.A. Moshiri (Ed) *Constructed wetlands for water quality improvement*. Lewis publishers, Boca Raton. pp. 187-195.

Naturvårdsverket (1993) *Gruvavfall från sulfidmalmsbrytning* (Mine waste from sulphide ores). Swedish Environmental Protection Agency, Report No. 4202, (In Swedish).

NCB (1982) *Technical Management of Water in the Coal Mining Industry*. National Coal Board Mining Department, London. 130 pp.

Novak, P. (1994) Improvement of water quality in rivers by aeration at hydraulic structures. In Hino, M., (editor), *Water Quality and its Control*. Balkema, Rotterdam. pp 147 – 168.

Nuttall, C.A. (2003) Testing and performance of a newly constructed full-scale passive treatment system at Whittle Colliery, Northumberland. *Land Contamination & Reclamation*, **11** (2), 105 - 112.

Nuttall, C.A. and Younger, P.L. (2000) Zinc removal from hard circum-neutral mine waters using a novel closed-bed limestone reactor. *Water Research*, **34**, 1262-1268.

Otwinowski, M. (1994) *Quantitative analysis of chemical and biological kinetics for the acid mine drainage problem*. Report for Mine Environment Neutral Drainage Program and the British Columbia Acid Mine Drainage Task Force.

Parker, K. (2003) Mine water management on a national scale - experiences from the Coal Authority. *Land Contamination and Reclamation*, **11**, (2), 181 - 190.

Phillipps, P., Bender, J., Simms, R., Rodriguez-Eaton, S. and Britt, C. (1995) Manganese removal from acid coal-mine drainage by a pond containing green algae and microbial mat. *Water Science and Technology*, **31**(12), 161-170.

Rasmuson, A. and Eriksson, J.C. (1987) *Capillary barriers in covers for mine tailings dumps*. Swedish Environmental Protection Agency, Report No. 3307.

Ricard, J.F., Aubertin, M., Firlotte, F.W., Knapp, R., McCullen, J. and Julien, M. (1997) Design and construction of a dry cover made of tailings for the closure of Les Terrains Aurifères Site, Malartic, Quebec. *Fourth International Conference on Acid Rock Drainage*, Vancouver, B. C., Canada, May 31 – June 6, 1997.

Sen, A.M. and Johnson, D.B. (1999) Acidophilic sulphate-reducing bacteria: candidates for bioremediation of acid mine drainage. In Amils, R. and Ballester, A. (editors) *Biohydrometallurgy and the Environment: Toward the Mining of the 21st Century*. Process Metallurgy 9A. Elsevier, Amsterdam. pp. 709-718,

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

Sjöblom, Å. and Håkansson, K. (2003) Repartitioning of river trace metals following the addition of minor amounts of ferrous iron. Submitted to *6th International Conference on Acid Rock Drainage*, Cairns, Australia, 2003.

Stoltz, E. and Greger, M. (2001) Wetland plants reduce metal content in drainage water from submersed tailings. *Proceedings of Securing the Future, International Conference on Mining and the Environment*, June 25 – July 1, 2001, Skellefteå, Sweden, Vol. 2, pp 806 – 813.

Stumm, W. and Morgan, J.J. (1996*) Aquatic Chemistry: Chemical equilibria and rates in natural waters*. (3rd edition). John Wiley & Sons Inc., New York.

Tarutis, W.J., Stark, L.R. and Williams, F.M. (1999) Sizing and performance estimation of coal mine drainage wetlands. *Ecological Engineering*, **12**, 353-372.

Thompson, L.C. (1996) Cyanide biotreatment and metal biomineralization in spent ore and process solutions. In US EPA (Editors) *Managing environmental problems at inactive and abandoned metal mine sites*. EPA Seminar publication No EPA/625/R-95/007.

Tremblay, R.L. (1994) Controlling acid mine drainage using an organic cover: The case of the East Sullivan Mine, Abitibi, Quebec. *Proceedings of the International Land Reclamation and Mine Drainage Conference and Third International Conference on the Abatement of Acidic Drainage*, Pittsburgh, PA, April 24 – 29, 1994, U. S. Department of the Interior, Bureau of Mines Special Publication SP 06B-94.

Vick, S.G. (1983) *Planning, design and analysis of tailings dams.* Wiley, New York. 369pp.

Walton-Day, K. (1999) Geochemistry of the processes that attenuate acid mine drainage in wetlands. In *The Environmental Geochemistry of Mineral Deposits. Part A: Processes, Techniques and Health Issues*. Eds G.S. Plumlee and M.J. Logsdon. Reviews in Economic Geology Volume 6A, Society of Economic Geologists, Inc., Colorado, USA. pp. 215-228.

Wang. Y. and Reardon, E.J. (2001) A siderite / limestone reactor to remove arsenic and cadmium from wastewaters. *Applied Geochemistry*, **16**, 1241-1249.

Watzlaf, G.R. (1997) Passive treatment of acid mine drainage in down-flow limestone systems. *Proceedings of the 2000 National Meeting of the American Society for Surface Mining and Reclamation*, Tampa, Florida, June 11-15, 2000.

Watzlaf, G.R., Schroeder, K.T. and Kairies, C. (2000) Long-term performance of alkalinity-producing passive systems for the treatment of mine drainage. *Proceedings of the 2000 National Meeting of the American Society for Surface Mining and Reclamation*, Tampa, Florida, June 11-15, 2000.

Watzlaf, G.R., Schroeder, K.T., Kleinmann, R.L.P., Kairies, C.L. and Nairn, R.W. (2003) *The Passive Treatment of Coal Mine Drainage*. National Energy Technology Laboratory, U.S. Department of Energy, and University of Oklahoma, OK, U.S.A. In Press.

Waybrant, K.R., Blowes, D.W. and Ptacek, C.J. (1998) Selection of reactive mixtures for use in permeable reactive walls for treatment of mine drainage. *Environmental Science and Technology*, **32**, 1972-1979.

Wieder, R.K. and Lang, G.E. (1982) Modification of acid mine drainage by a freshwater wetland. In: McDonald, B.R. (Editor*), Proceedings of the Symposium on Wetlands of the Unglaciated Appalachian Region*, West Virginia University, Morgantown, West Virginia, USA.

Willow, M.A. and Cohen, R.H.H. (1998) pH and dissolved oxygen as factors controlling treatment efficiencies in wet substrate, bio-reactors dominated by sulfate reducing bacteria. In *"Tailings and Mine waste '98"* (Symposium Volume of the American Society for Surface Mining and Reclamation).

Young, C.A. and Jordan, T.S. (1996) Cyanide remediation: current and past technologies. *Proceedings of the 10th annual Conference on Hazardous Waste Research*. pp. 104-129.
Younger, P.L. (1997) The Longevity of Minewater Pollution: A Basis for Decision-Making. *Science of the Total Environment*, **194/195**, 457 - 466.
Younger, P.L. (1998) Design, construction and initial operation of full-scale compost-based passive systems for treatment of coal mine drainage and spoil leachate in the UK. *Proceedings of the International Mine Water Association Symposium on "Mine Water and Environmental Impacts"*, Johannesburg, South Africa, 7-13 September 1998. (Volume II), pp. 413-424.
Younger, P.L. (2000a) Holistic remedial strategies for short- and long-term water pollution from abandoned mines. *Transactions of the Institution of Mining and Metallurgy (Section A),* **109**, A210-A218.
Younger, P.L. (2000b) Predicting temporal changes in total iron concentrations in groundwaters flowing from abandoned deep mines: a first approximation. *Journal of Contaminant Hydrology*, **44**, 47 - 69.
Younger, P.L. (2002) Coalfield closure and the water environment in Europe. *Transactions of the Institution of Mining and Metallurgy (Section A: Mining Technology)*, **111**, A201 - A209.
Younger, P.L., Banwart, S.A. and Hedin, R.S. (2002) *Mine Water: Hydrology, Pollution, Remediation*. Kluwer Academic Publishers, Dordrecht, Netherlands. 442 pp.
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., Jayaweera, A., Elliot, A., Wood, R., Amos, P., Daugherty, A.J., Martin, A., Bowden, L., Aplin, A.C., and Johnson, D.B. (2003) Passive treatment of acidic mine waters in subsurface-flow systems: exploring RAPS and permeable reactive barriers. *Land Contamination and Reclamation*, **11**, 127-135
Ziemkiewicz, P.F., Skousen, J.G., Brant, D.L., Sterner, P.L. and Lovett, R.J. (1997) Acid mine drainage treatment with armored limestone in open channels. *Journal of Environmental Quality*, **26**, 1017-1024.

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