Prioritisation of abandoned non-coal mine impacts on the environment

SC030136/R12 Future management of abandoned non-coal mine water discharges

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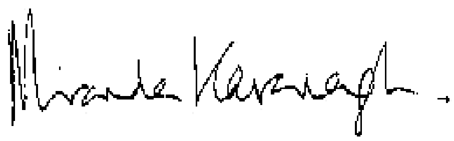

Miranda Kavanagh

Director of Evidence

Executive Summary

The *Prioritisation of abandoned non-coal mine impacts on the environment* project has generated the most definitive evaluation to date of the impacts on the water environment from abandoned non-coal mines across England and Wales. For the first time, an objective assessment has been carried out to prioritise the rivers in England and Wales where pollution from these mines has the highest impact, and where there is the greatest risk that water bodies (river stretches) will fail to meet the objectives of the Water Framework Directive due to abandoned non-coal mines. The specific water bodies which should be the focus of immediate attention in River Basin Management Plans (RBMPs) have been identified, and the work needed to address mining pollution through both research into passive treatment technologies and catchment monitoring investigations is outlined. This Executive Summary details the key outcomes from the entire project, and therefore incorporates elements of the conclusions of all 13 reports that comprise the final deliverables of the project.

Assessing water bodies using water quality, ecological, groundwater and higher impact metrics it has been possible to prioritise *Impacted* and *Probably Impacted* water bodies into ranked lists. The primary focus is the impacts of polluted water discharges from abandoned non-coal mines to surface water courses. Additional information collated in the database also enables assessment of what the other issues are at these sites, such as safety issues, outbreak risk and stakeholder concerns. Taken together this provides a valuable resource to assist in the long-term remediation planning at polluting abandoned non-coal mine sites in England and Wales. The absolute scale of environmental problems and risks associated with abandoned non-coal mines is summarised as follows:

Risk	Number
Water bodies *Impacted by* non-coal mine water pollution	226
Water bodies *Probably Impacted by* non-coal mine water pollution	243
Confirmed mine water discharges	257
Suspected mine water discharges	81
Documented evidence of outbreak risk	19
Mine sites at which there is evidence of diffuse non-coal mine water pollution	112
Definite concerns of airborne pollution, stability, safety, and / or public & animal health	425
Suspected concerns of airborne pollution, stability, safety, and / or public & animal health	275

As well as showing the absolute scale of environmental problems associated with abandoned non-coal mines, additional data collated illustrates the specific nature of these problems, and also the areas of England and Wales in which such problems are most acute (reports III – XI report specifically on problems within individual River Basin Districts of England and Wales). The logical next step is to consider how to actually address these issues, and two of the reports arising from this project are dedicated to precisely that (Reports XII and XIII). *Generic* recommendations on how to manage abandoned non-coal mine drainage problems are provided, with *specific* guidance on aspects that have not previously been addressed elsewhere. In light of

the information provided in these reports, the main overall conclusions and recommendations regarding the future management of abandoned non-coal mines can be summarised as follows:

- It is essential to have a clear understanding of the exact sources of pollution if an effective remediation programme is to be instigated. In some instances a single source of non-coal mine water pollution is clearly the main problem, but in the majority of water bodies there are multiple sources. Diffuse sources of mine water pollution are a major contributor to overall metal flux in abandoned non-coal mine catchments. In very few water bodies is there a clear quantitative understanding of how individual sources of pollution from abandoned non-coal mines contribute to the overall metal flux in that water body. Often it appears that not all of the sources, especially where they are diffuse in nature, have been identified.

- Systematic and consistent scoping studies of water bodies impacted by abandoned non-coal mines, including detailed monitoring of water quality and flow, are therefore recommended. Detailed guidance on the approach to such investigations is provided in this report. The effects on aquatic ecosystems should also be investigated. If remediation measures are implemented without understanding the dynamics of mining pollution in specific catchments, the environmental objectives for water bodies that are set out in RBMPs may not be achieved despite significant expenditure on engineering works and treatment systems.

- Although we sometimes refer to 'priority for remediation' and 'priority for further data collection' for *Impacted* and *Probably Impacted* water bodies respectively, the reality is that additional monitoring programmes will be a necessity at almost *all* of the water bodies in which non-coal mine drainage is identified as an issue. This is because data collection programmes to date have either not been systematic enough to characterise metal fluxes in water bodies, or have not been appropriately targeted to facilitate the design of a treatment system (or both).

- The passive mine water treatment technologies that have been applied with great success to the remediation of coal mine drainage (principally for the removal of iron) will not work to anything like the same degree for the metals in non-coal mine drainage (e.g. zinc (Zn), cadmium (Cd)). These metals are more soluble than iron and so it is more difficult to remove them from the mine water.

- Effective passive treatment of non-coal mine drainage to consistently meet Environmental Quality Standards (EQS), within a practical land area, is a subject of ongoing research. There are many active treatment technologies that could remediate non-coal mine drainage to the standards required to meet EQS, but they come at a high cost, and in many of the locations of major non-coal mine water discharges it appears unlikely that they would be acceptable developments.

- Irrespective of the type of technology, the management of the metal-rich sludge arising from the treatment of non-coal mine drainage remains a problem. Only active treatment technologies currently offer the possibility of recovering metals in sufficient purity that they *might* be recycled, but even for active systems it currently seems unlikely that recycling of metals from abandoned non-coal mine water treatment will be economically viable. There may be reuse options for metal-rich media recovered from mine water treatment systems, but these need further investigation.

- There are other problems associated with former non-coal mining districts besides mine water pollution, albeit in some cases these issues may contribute to problems of water pollution. Stability concerns, safety, airborne pollution, and other human and animal health risks, may be significant, and should therefore be addressed accordingly. The level of detail of information provided with respect to these issues has been very varied. Although there are clearly important specific issues relating to these aspects of abandoned non-coal mines that need to be addressed (e.g. stability concerns at specific sites), the main conclusion of this project is that there needs to be a systematic national approach to the assessment of such problems. As well as identifying the most important problems to address, this will directly serve the requirements in the EU Mining Waste Directive to create an inventory of closed mine waste facilities causing harm to human health or the environment.

- The problems evident at abandoned non-coal mines are multifarious and complex. A chronology of environmental management activities for tackling the problems is therefore proposed in this report. This sets out the specific requirements of investigations of water pollution problems in abandoned non-coal mine districts, whilst also taking into consideration other potential issues that may be present in such catchments. It is estimated that it will take approximately 4.5 years to complete an individual remediation scheme, from commencement of a scoping study to completion of a full-scale treatment system. Detailed discussion is provided on the exact requirements of investigations targeted at identifying appropriate remedial strategies (i.e. scoping and feasibility stages).

- Conducting thorough investigations of environmental problems in abandoned non-coal mining districts can be expensive. This cost is minor, however, compared to the design, installation and operation of systems to remediate such pollution problems. The total cost to remediate all of the water-related environmental problems associated with abandoned non-coal mines that have been identified as part of this project, is estimated to be approximately £370 million over an initial 10 year period, at present day costs, with additional subsequent operating costs. Of this total around 90% is apportioned to mine water treatment, and 10% to mitigation of outbreak risk and diffuse pollution problems. Treatment systems are likely to be required to operate in perpetuity. There are considerable uncertainties regarding the accuracy of this estimate, due in large part to a paucity of quantitative data on abandoned non-coal mine environmental problems (especially relating to mine water discharge flow and volume).

Contents

List of tables

List of figures

1. Introduction

1.1 General introduction

This report discusses options, constraints and preparatory investigations for remediation of abandoned non-coal mine water discharges identified and prioritised during the 'Identification and prioritisation of abandoned non-coal mines' project undertaken by Newcastle University, Atkins Ltd and the Coal Authority. This project was primarily funded by Defra, with contributions from the Welsh Government, the Environment Agency and the Department for Communities and Local Government.

The project (which ran from April 2007 to March 2009) has collated information on some 5,000 non-coal mine sites across England & Wales, with an associated 257 confirmed mine water discharges (and another 81 suspected discharges). The approach to the prioritisation of these sites and discharges is described in detail in the report, *A methodology for identification and prioritisation of abandoned non-coal mines in England and Wales*. The geographical unit of assessment of priorities is the water body, as defined / delineated by the Environment Agency for the Water Framework Directive (WFD; European Community, 2000). In brief, all water bodies in England & Wales have been assigned into one of 4 categories; *Impacted*, *Probably Impacted*, *Probably Not Impacted*, and *Not Impacted*. This categorisation of water bodies is determined by assessing which of a range of issues relating to abandoned non-coal mines and their impacts occur within each water body (as detailed in the Methodology report). By definition, those water bodies that have been categorised as *Impacted* are a higher priority than those *Probably Impacted* and so on. The methodology has allowed for more detailed prioritisation of water bodies falling within each of the *Impacted* and *Probably Impacted* categories, based on a range of metrics described in the Methodology report. Of principal interest in the context of this particular report is the data provided by Environment Agency staff on mine water quality and flow within these water bodies. Due principally to a paucity of data, at the lower end of the prioritisation (i.e. *Not Impacted* and *Probably Not Impacted* sites) large numbers of water bodies are assigned the same priority. However, at the upper end of the prioritisation much more detailed distinctions can be seen between individual water bodies.

The categories of *Impacted, Probably Impacted, Probably Not Impacted*, and *Not Impacted* used to prioritise water bodies are *operationally defined*, in the sense that the categorisation is determined by the occurrence or absence of one or more specified characteristics / features within a particular water body. Specifically, the 4 categories are defined as follows:

Impacted:	Any EQS failure in surface water body with known mine sites
Probably Impacted:	Any EQS failure in surface water body in mining area with no known mine sites **OR** Any EQS failure in surface water body immediately downstream of mining area
Probably Not Impacted:	Mining area with no EQS failures in water body **OR** Mining area with no EQS failures in immediately downstream surface water body

Not Impacted: A non-mining area with or without any EQS failure not already categorised

The scope of the prioritisation exercise has not extended to the investigative lengths described in the paragraphs below for individual water bodies i.e. catchment-scale studies of flow and contaminant load mass balances. Therefore whilst the categorisation of water bodies is a very useful exercise for indicating those geographical areas in which to focus initial efforts in addressing abandoned non-coal mine water pollution, the results are not unequivocal. Rather, these categories reflect *confidence* in whether non-coal mine water pollution is a cause of degradation in a catchment, not *certainty* (in either causality or severity). To one degree or another, therefore, there will be a need to investigate the specific mine water pollution issues in a water body further, albeit with a focus on those in the *Impacted* and *Probably Impacted* categories. The discussion below, of data requirements for mine water pollution assessment and remediation, is therefore pertinent to all water bodies where mine water pollution is an issue.

Notwithstanding the comments in the previous paragraph, those water bodies appearing at the top of the prioritisation list are those in which there is confidence that (a) mine sites / mine water discharges within those water bodies are having an environmental impact on the freshwater environment and, furthermore, that (b) the mine waters are resulting in multiple failures of EQS limits, and a range of other impacts, in the receiving watercourses. In the vast majority of cases it is little coincidence that these are also the mine water discharges for which most water quality data is available. Given that (a) many thousands of sites and ~ 300 mine water discharges have been identified and (b) a reasonable volume of water chemistry data are only available for the highest priority sites, the recommendations for future monitoring and remediation proposed here are based on 20 mine water discharges on the prioritisation list (for reasons explained below). Details of these mine waters are provided in Table 2, and specific reference is made to these mine waters through the course of this document, to ensure that as far as possible this report provides recommendations that are specific to abandoned non-coal mine drainage.

This particular report focuses on the options and data requirements for, and constraints upon, remediation of abandoned non-coal mine water discharges. The legislative catalyst for addressing these issues is the European Union Water Framework Directive (2000/60/EC), which requires Member States of the EU to progress towards “good chemical and ecological status” in water bodies. This project has demonstrated, in the most comprehensive assessment to date, that polluting discharges from abandoned non-coal mines (principally metal mines) are a significant barrier to meeting this broad objective.

However, the WFD is not the only reason for investigating abandoned non-coal mines, and neither is aquatic pollution the only potential impact on the environment of former mine sites. It is for this reason that issues such as airborne pollution risk, outbreak risk (i.e. sudden, catastrophic release of polluted water), stability concerns, contaminated land at former mine sites, and the concerns of stakeholders, have all been considered in assessing the overall hazard associated with abandoned non-coal mine sites (the recently introduced EU Mining Waste Directive (2006/21/EC)

(European Community, 2006) is a key regulatory driver in this respect). For this reason there is a sister report to this one, which specifically considers *Hazards and risk management at abandoned non-coal mine sites*.

This is one of the 13 reports that detail the final results of the implementation of the methodology across England and Wales. The reports are listed below:

I. *A methodology for identification and prioritisation of abandoned non-coal mines in England and Wales*
II. *Prioritisation of abandoned non-coal mine impacts on the environment: The national picture*
III. *Prioritisation of abandoned non-coal mine impacts on the environment in the Dee River Basin District*
IV. *Prioritisation of abandoned non-coal mine impacts on the environment in the Northumbria River Basin District*
V. *Prioritisation of abandoned non-coal mine impacts on the environment in the South West River Basin District*
VI. *Prioritisation of abandoned non-coal mine impacts on the environment in the Western Wales River Basin District*
VII. *Prioritisation of abandoned non-coal mine impacts on the environment in the Humber River Basin District*
VIII. *Prioritisation of abandoned non-coal mine impacts on the environment in the North West River Basin District*
IX. *Prioritisation of abandoned non-coal mine impacts on the environment in the Severn River Basin District*
X. *Prioritisation of abandoned non-coal mine impacts on the environment in the Anglian, Thames and South East River Basin Districts*
XI. *Prioritisation of abandoned non-coal mine impacts on the environment in the Solway-Tweed River Basin District*
XII. *Future management of abandoned non-coal mine water discharges*
XIII. *Hazards and risk management at abandoned non-coal mine sites*

1.2 Introduction to this report

This report addresses the issue of mine water pollution remediation. Water pollution is arguably the single biggest environmental concern at abandoned non-coal mine sites nationally, and has been a primary driver for this project. Essentially the report addresses:

(1) the data that are required to inform the design of remediation schemes
(2) how to gather such data
(3) the current options for remediation of such discharges, and the limitations of such systems with respect to the specific problems associated with abandoned non-coal mines
(4) the recommended chronological approach to environmental management in abandoned non-coal mine water bodies, and
(5) the estimated costs of a national remediation programme.

The scope of the report is therefore broad. The content focuses on provision of guidance on aspects of mine water management that have not been detailed in previous publications, whilst endeavouring at the same time to cover to some degree all relevant aspects of the problem. For this reason, for example, details on water chemistry data requirements and sampling protocols are kept to a minimum, as they have previously been discussed at length by the PIRAMID Consortium (2003). The guidance provided is necessarily generic because the report is national in its scope. However, as far as possible reference is made to specific data collated as part of this project where it is appropriate to do so.

A key section of this report is Chapter 6 (*Recommended approach to managing the environmental impacts of abandoned non-coal mines at the catchment scale)*, within which a phased approach to the investigation of pollution problems and implementation of remediation initiatives for abandoned non-coal mines is proposed. 11 phases are recommended, which broadly fall into 4 categories:

Investigation; chronologically the first phases, comprising:

(1) Scoping phase, (2) Feasibility phase, (3) Conceptual design phase, (4) Pilot-scale testing

Implementation, comprising:

(5) Detailed design phase, (6) Construction phase, (7) Site operational

Land and planning, running concurrently to *Implementation*, and comprising:

(8) Land requirements, (9) Land deal, (10) Planning process

Stakeholder engagement (11), which should run throughout the entire process

This report has a particular focus on the *Investigation* phases, and it is to these issues that Chapters 2, 3 and 4 of the report are devoted. Chapter 2 explicitly considers the scoping phase, whilst Chapters 3 and 4 implicitly relate to the feasibility, conceptual design and pilot-scale testing phases (with specific discussion of the latter). The *Implementation* and *Land and planning* phases are beyond the scope of this report, and are therefore not discussed here. Some issues of *Stakeholder Engagement* are discussed in Chapter 5 (and referred to in earlier chapters), whilst Chapter 7 outlines estimated costs for the investigation and remediation of abandoned non-coal mine water pollution problems in England and Wales. Chapter 8 presents overall conclusions and recommendations.

It is important to note that this report does not cover ecological monitoring and impact assessment as part of the discussion of catchment scale investigations. Ecological quality impacts are unquestionably an important element in the overall assessment of costs and benefits of mine water remediation; the maintenance of ‘good ecological status’ is a primary objective of the WFD, and targets for remediation of abandoned non-coal mine water pollution may well be set specifically to meet objectives for ecological improvements in a catchment. However, the design of an effective remediation scheme is ultimately based on physical and chemical variables. Since the overall objective of this report is to ensure that the design of remediation systems is founded on solid scientific and engineering understanding, the focus here is therefore the physical and chemical characteristics of abandoned non-coal mine waters and the river catchments which they impact.

2. Scoping phase: catchment-scale investigations

2.1 Introduction

Historically point sources of mine water pollution have been the focus of attention for remediation. The advent of the WFD, and in particular its emphasis on management of freshwaters at the scale of the river catchment, has raised questions about the importance of diffuse sources of mining-related pollution. Investigations of the importance of such pollution in districts formerly mined for both coal (e.g. Mayes *et al.*, 2008), and base metals (e.g. Jarvis *et al.*, 2006; Mighanetara, 2008), have illustrated the substantial importance of such sources, which may typically account for in the region of 50% of the total mass flux of metals in a catchment under low flow conditions (~ Q90), and in excess of 90% of mass flux during high flow events (~ Q10) (Jarvis *et al.*, 2006). Potential diffuse sources cited include (1) surface runoff from exposed spoil piles, (2) direct inputs of metal-polluted groundwaters via the hyporheic zone, and (3) resuspension / remobilisation of metal-rich stream sediments (Mayes *et al.*, 2008). The proportional importance of each of these various diffuse sources has yet to be accurately quantified, and neither is the exact relationship between point and diffuse contributions clearly understood e.g. bed sediment resuspension / metal remobilisation has been cited as a diffuse source, but the concentration of metals in these sediments may be a result of longstanding point sources, at least in some river catchments.

Irrespective of the need to improve understanding of the dynamics of point and diffuse source pollution, the key issue here is that, in the mining catchments that have been investigated to date, the sum of the mass loads of metals arising from point sources alone is significantly less than the absolute mass of metals measured downstream. This poses a fundamentally important question: will remediation of point sources alone result in sufficient improvements in water quality downstream to meet WFD objectives and / or Environmental Quality Standard (EQS) limits? Even in circumstances in which remediation of point sources alone is deemed the best practical environmental option, understanding the extent of improvements accruing from such initiatives is important if an accurate cost-benefit assessment is to be made during the planning stages of a project.

Driven by the WFD, and in light of the outcomes of investigations of diffuse source pollution, a need has been identified to conduct catchment-wide scoping studies of pollutant sources and impacts in formerly mined districts. The principal objective of such studies is to identify the most important sources of metals to the main river or stream. This is a logical and necessary precedent to remediation planning. However, such investigations can be technically and logistically complex, due to the number of monitoring locations required, the requirement to synchronously monitor flows and water quality, and the need to understand (at least to some degree) the variability in metal fluxes caused by changing hydrological conditions. A three stage approach to these scoping investigations is outlined in Table 1, and explained below.

2.2 Experimental design and implementation for catchment-scale scoping studies

In order to understand the main sources of aqueous metals in a river catchment it is not sufficient to measure only metal concentrations; metal *loads* must be measured, which necessitates synchronous measurement of both concentration and flow-rate. The main objective of a catchment-scale scoping study is essentially to construct a mass balance of metals loads within the catchment. Since, by definition, diffuse sources of pollution defy direct measurement, the mass balance is developed by quantifying flows and (metal) concentrations at all point sources of mine water pollution, tributaries as they enter the main river, and the main river itself at key locations.

Table 1. Recommended three stage approach for conducting scoping studies of catchment-scale metal fluxes

Activity	Output
Stage 1	
1. Collate background information including existing water quality, flow and biology data 2. Reconnaissance surveys to identify point and possible diffuse sources, and potential instream monitoring locations and measurement methods.	In liaison with client: 1. Define study aim, geographic boundaries, and agree monitoring locations and justification; 2. Provide cost estimate for Stage 2; 3. Produce short report (~5 pages), preferably in agreed template.
Stage 2	
1. Conduct preliminary sampling and analysis campaign (during baseflow conditions if feasible); 2. Review flow and contaminant mass balances; 3. Revise and repeat monitoring if flow mass balances outside acceptable margins of error (iterative) or uncertainties regarding metal sources.	1. Check flow mass balances, and review and revise flow monitoring if outside acceptable margins of error (iterative)* ; 2. Calculate metal fluxes and identify main sources, revising monitoring programme if uncertainties remain along ungauged reaches; 3. Provide cost estimate for Stage 3; 4. Produce short report (~5 pages), preferably in agreed template.
Stage 3	
1. Conduct full monitoring investigation, comprising repeat measurements at finalised monitoring stations under a range of hydrological conditions.	1. Check and report balances and metal fluxes; 2. Provide final interpretive report (~20 pages), preferably in agreed template, with all raw data and detailed methods appended (electronically) for future use

*the acceptable margins of error on flow mass balances will be site specific and dependent on the desires of the client.

Given these data requirements it is clear that a study can rapidly become both complex and costly, and therefore an important starting point is to define the geographical boundaries of the investigation. Typically this entails identifying an upstream boundary and downstream boundary on the main watercourse, which in turn requires walkover reconnaissance surveys of the river in question. Reconnaissance surveys may lead to changes to the initially defined study boundary. Once study boundaries are agreed, identification and monitoring of point and diffuse sources, and instream monitoring, can proceed. The sections below are broadly laid out in the order in which an investigator would approach such a scoping study, and include important considerations such as instream flow measurement techniques, and monitoring under varying hydrological conditions. However, it is important to emphasise that this is an iterative process. So, to give one example, even after all monitoring locations are identified results from a first monitoring exercise may reveal a sudden increase in metal loads in an unexpected reach of the river, and therefore the monitoring locations will need revisiting.

In the context of this study it is worth noting that of the mine waters listed in Table 2, for which both flow and water quality data area available, diffuse mining pollution is a confirmed problem in 14 of the 20 water bodies in which these mine waters occur. In part as a consequence, almost all of the mine waters listed in Table 2 either have had, are in the process of having, or require catchment-scale scoping studies, which serves to emphasise the importance of this type of investigation. Management options for diffuse pollution are discussed in more detail in Section 4.6 of this report.

2.2.1. Defining study boundaries

The limiting factor to the absolute size of the study area is often the number of locations that can be physically monitored in a single day. The reason for this is that in order to determine a reliable mass balance all samples and flow measurements need to be made during hydrological conditions that, as far as possible, remain constant during the course of any single sampling campaign (for reasons outlined in Section 2.2.6, below). As an *indication only* because it very much depends on site conditions, an experienced team of 2 people, investigating a 15 – 20 km stretch of river, might manage 10 – 15 sites in a day. Therefore the intensity of sample locations will be determined by the size of the study area, and this in turn will depend on the exact objectives of the study. At one extreme, sampling at 5 – 10 m intervals along a 100 m reach may be required for detailed investigations of groundwater - surface water interactions, whilst at the other, sampling of just the main tributaries of a major river may be required if the objective is a preliminary identification of the main sources of metals to an entire river system. Again, as an indication only, the following might be typical study objectives and the area which they may cover (from small scale to large scale):

- Investigating micro-scale hyporheic processes in areas of vigorous surface water – ground water interaction (~ 10s – 100s m^2)
- Determining impacts of single, discrete abandoned mine (~ 100s m^2 – 10 km^2)
- Identifying main sources of metal flux in small- medium-sized mining catchment (~ 50 – 500 km^2)
- Diagnosing causes of elevated metal loads in major river systems (~ > 1000 km^2)

In the context of delivering the aims of the WFD, the most relevant of these is the third – metal fluxes in small- to medium-sized mining catchments - since this is the category that water bodies usually fall into. The second category (single, discrete mines) is also relevant, but the discussion in Chapter 3 is more pertinent for those cases (because in likelihood concerns will focus on a single discharge, and therefore remediation planning is appropriate). For that reason the remainder of this discussion on catchment-scale investigations focuses on the small- to medium-sized catchment scale of investigation.

Within a small- to medium-sized catchment it is clearly vital that the upstream and downstream boundaries of the scoping study encompass the mine or mining area of concern. In order to ensure this a combination of background research and walkover surveys are required. A number of sources of background information are important:

- Previous reports / publications on the same issue
- Regulatory authority water quality, ecology and flow data
- Groundwater levels and water quality (if available)
- Mine plans (if available)
- Mineral geology maps
- Local information, particularly from local mine enthusiasts

If the driver for the investigation is water quality, then the upstream location (or locations if it is in the headwaters of a catchment) can be quite readily located as the first point moving upstream at which pollutant concentrations are below some predefined level of concern (e.g. EQS limit).

The downstream boundary may be more difficult to identify, though as a guide should be the first point that is downstream of all former underground and surface mine workings and infrastructure. One critical aspect which may determine the location of the downstream boundary is the ease with which it will be possible to measure instream river flow rate. Since the depth and velocity of water at the downstream location may preclude physical entry to the river for health and safety reasons, the options for flow measurement become limited (see Section 2.2.5). Therefore a dedicated flow gauging structure (such as one of the network that the Environment Agency operates) forms an ideal downstream boundary, even if it is not in precisely the location identified from background research as the logical downstream boundary. Other hydraulic structures that are not dedicated gauging stations may also be used if access is possible, at least during low flow conditions. If so, then stage-discharge relationships may be derived, as discussed in Section 2.2.5.

Table 2. Flow and water quality data for the 20 mine waters across England & Wales, all in the *Impacted* priority category, for which flow data are available.

River Basin District	Name	Easting	Northing	Diffuse Pollution?	Mean Flow (L/s)	As	Cd	Cu	Fe	Mn	Ni	Pb	Zn
South West	County Adit	176199	41875	Yes	454	0.19	0.004	0.06	9.83	0.78	0.09	0.004	2.37
South West	Dolcoath Adit	164850	41880	Suspected	150	0.15	0.002	0.0001	8.00	2.25	0.03	0.002	0.90
Northumbria	Rispey Breakout	390900	542900	Unknown	60	nr	0.002	nr	4.65	6.70	0.08	nr	2.40
Western Wales	Frongoch Adit	271237	274259	Yes	60	nr	nr	nr	nr	nr	nr	6.25	5.56
Northumbria	Tail Race Level	391640	542750	Unknown	35	nr	nr	nr	0.25	0.60	0.01	0.05	0.28
Western Wales	Nant y Mwyn Lower Boat Adit	278234	243796	Yes	30	nr	0.03	nr	nr	nr	nr	0.10	10.63
South West	Owlacombe Cross	277226	73589	Unknown	30	0.12	nr	0.03	10.40	nr	nr	nr	0.15
Northumbria	Barney Craig	380400	546700	Yes	20	nr	0.005	nr	nr	nr	nr	0.003	4.00
Western Wales	Cwm Rheidol Adit 6	273022	278330	Suspected	11	nr	0.03	nr	nr	nr	nr	0.75	13.83
Northumbria	Bolts Burn Level	393650	542820	Yes	9	nr	nr	nr	0.50	0.40	nr	nr	0.07
Western Wales	Pughs Adit	280109	274407	Yes	8	nr	0.03	0.01	0.91	nr	nr	0.49	22.40
Western Wales	Nant y Mwyn Upper Boat Adit	278300	244600	Yes	6	nr	0.01	nr	nr	nr	nr	0.50	1.72
Western Wales	Dyffryn Adda adit (or Joint Level)	243807	391218	Yes	6	0.25	0.17	39.43	564.00	19.42	0.19	0.04	57.75
South West	Bridford mine adit	282987	86492	Yes	5	nr	0.04	nr	18.70	nr	nr	2.00	28.00
South West	Wheal Maid Tailings Dam	174995	42386	Yes	5	0.05	0.06	0.70	30.02	nr	0.22	0.01	27.83
Western Wales	East Level	273640	291290	Yes	4	nr	0.002	0.75	0.02	0.01	0.003	0.16	0.04
Northumbria	Saltburn Gill	467337	520137	Yes	3	nr	nr	0.30	1200.00	250.00	1.30	0.001	0.80
Western Wales	Cwm Rheidol Adit 9	272936	278239	Suspected	3	nr	0.10	nr	nr	nr	nr	0.02	69.38
Western Wales	Morfa Ddu adit	242921	390325	Yes	2	0.06	0.04	0.06	12.05	2.52	0.02	0.56	13.83
Northumbria	Grove Rake	389490	544090	Yes	1	nr	nr	nr	0.05	0.02	nr	nr	0.03

Note. Concentrations highlighted in yellow indicate the relevant Environmental Quality Standard (EQS) is exceeded
EQS Values (mg/L): As = 0.05; Cd = 0.00008; Cu = 0.001; Fe = 1.00; Mn = 0.007; Ni = 0.02; Pb = 0.0072; Zn = 0.0078. Note that these are conservative values in the sense that some EQS limits are hardness-related; the values given are for soft waters, in which metals have their greatest toxicity, and therefore these are the lower values for EQS limits. These values were correct at the outset of the project (June 2007) and may have changed subsequently.
nr = not reported (note that this does not necessarily mean that data do not exist)

It will be apparent from this discussion that reconnaissance surveys (Section 2.2.2) are an essential part of this process of defining study boundaries, and they are even more so for the identification of point and diffuse sources of mining-related pollution.

2.2.2. Reconnaissance surveys

Reconnaissance surveys are an essential part of the catchment-scale scoping study. In all likelihood more than one reconnaissance survey may be required to identify the study boundaries, and all the point sources and potential sources of diffuse pollution between. Time and effort spent on these surveys is essential to ensure that the most appropriate monitoring locations are identified and, completed effectively, will save resources during the more detailed monitoring phase. For coal mine water discharges the identification of discharges is comparatively straightforward, for the simple reason that the oxidative dissolution of pyrite results in discharges to surface containing elevated concentrations of iron, which is rapidly deposited on stream beds, and is therefore highly visible. Although some non-coal mine discharges contain iron, in many cases simple visual identification is not possible because the metals of concern remain in solution.

Because many non-coal mine discharges arise from the oxidative dissolution of monosulphide minerals, such as sphalerite (ZnS) and galena (PbS), identification of discharges using field measurements of variables such as pH and conductivity are of less utility than for coal mine water discharges. This is because coal mine water arises from a disulphide mineral (FeS_2), and therefore sulphate concentrations are typically very high compared to background, driving conductivity values up (typically to > 500 μS/cm). The oxidative dissolution of FeS_2 is also a strongly acid-generating process which, in the absence of neutralising reactions, may result in a depressed pH. Neither of these indicators of pollution applies to non-coal mine waters where pyrite is absent: conductivities of non-coal mine waters are often < 200 μS/cm, and it is not uncommon for such waters to have near-neutral pH. Therefore reconnaissance surveys in non-coal mining catchments must rely far more heavily on background information and experience in identifying mining features in the landscape.

It is recommended that spot samples are collected from selected locations during these reconnaissance surveys with analysis limited to key pollutants of concern (e.g. heavy metals). Table 4 sets out the recommended analytical suite for mine water and instream analyses (though see important *caveats* in text).

Although the focus here is on issues of water quality (and quantity) related abandoned non-coal mine problems can also be addressed during reconnaissance surveys. In some instances this may be relevant and influential to the water quality issues, for example where eroding mine waste material is causing direct pollution of water courses. The specific types of issue that need to be identified are discussed at length in the report *Hazards and risk management at abandoned non-coal mine sites*. One issue that is of particular importance is the potential for direct and rapid infiltration of surface water to (usually shallow) mine workings, which may acutely influence the volume of water emerging from point sources at a lower elevation. In some instances it may be possible to divert water away from major infiltration pathways (e.g. open shafts and stopes) with relatively simple engineering works such as cut-off drains. Such interventions are highly beneficial, since they restrict the magnitude of pollution arising at the point source, and thus minimise the requirements for treatment. Though there may not be many instances where such interventions can be made, it is nevertheless worth the minimal effort of inspection during reconnaissance surveys, if only to rule it out as an option (diversion options are not without precedent; such works have been conducted at the Cwm Rheidol mine site in Wales).

2.2.3. Identifying point and diffuse sources

Identification of point sources of pollution is comparatively straightforward. Typically such sources are in the form of abandoned mine levels and adits, and occasionally mine shafts. If mine (abandonment) plans are available then their locations can be identified from these. If local hydrogeology data are also available then it should be possible to identify those which are likely to be free draining, by comparing local groundwater levels with elevations above ordnance datum of mining features likely to transmit water to surface. This is not guaranteed to show all adits that are draining water though, since mine entrances into shallow workings may be conduits for meteoric water entering the workings by infiltration. Again, therefore, walkover surveys are essential. Such surveys will also reveal whether point source discharges are accessible for sampling and analysis. In some cases they are in dangerous locations, and in others debris may have accumulated in front of the drainage feature to the point that the mine water is effectively a diffuse source.

Identification of likely sources of diffuse mine water pollution is far more difficult. Historic maps and mine plans, showing the locations of spoil heaps, washery facilities, ore processing sites, tailings ponds and so forth, will all be of assistance. However, experience of recognising such features on the ground during walkover surveys is unquestionably essential. In a catchment for which no previous studies have been undertaken, identification of diffuse sources such as direct groundwater inputs to the stream, and remobilisation of metals from streambed sediment, is extremely difficult. In such instances the best approach is often to deduce such locations after monitoring the point sources and instream locations first, thereby identifying stream reaches in which metal loading increases, and which therefore may be source areas of diffuse pollution.

2.2.4. Instream monitoring locations

With upstream and downstream boundaries of the study area determined (Section 2.2.2), and all point and potential diffuse sources identified, the selection of instream monitoring points becomes possible. The purpose of the instream monitoring locations is to enable calculation of metal loads in the receiving watercourses, which can then be compared to the input metal loads from the point source mine waters in the catchment. Essentially, where instream metal loads exceed the total load due to the combined point sources then diffuse sources of some form must be contributing. Conversely, if the cumulative load due to point sources exceeds the instream load, then attenuation of metals within the stream must be occurring. In reality of course both may be happening simultaneously i.e. diffuse sources are adding to the total instream load simultaneously to metals being attenuated within the stream channel. Quantitative resolution of this type of problem (perhaps by use of geochemical modelling to estimate rates of attenuation) has not been accomplished to date. However, from an operational point of view the key issue to appreciate is whether sources other than the point discharges are making a significant contribution to the instream load. The exact interrelationship between attenuation processes and diffuse source additions is unlikely to be critical to answering this central question.

Figure 1 is a useful illustration of the positioning of instream monitoring locations. It is an actual example from the PhD study of Gozzard (2008), which demonstrated the importance of diffuse sources of mining pollution in the River West Allen, Northumberland, which was formerly mined for lead and zinc. Zinc is the main contaminant of concern.

Figure 1 illustrates a short section of the River West Allen (approximately 200m) that formed part of the study by Gozzard (2008) (which investigated a 20 km stretch of this river). It is in this reach that the most important point source of mine water pollution occurs (marked as Point C). Downstream of the mine water discharge, on the right-hand bank, there is exposed waste spoil material which, during rainfall events, is a source of diffuse surface runoff (Point B). In addition there are suspected to be diffuse groundwater and metal-contaminated sediment sources of pollution downstream of the main site of the former mine (Point A).

In order to quantify the impacts of the various sources a number of instream locations were monitored in addition to the main point source, and these are illustrated on Figure 1. To reiterate an earlier crucial point, at all of these locations both metal concentrations and flow-rate must be synchronously monitored, and all points must be monitored under the same hydrological conditions, if a true appreciation of metal fluxes is to be gained.

The reasons for the selection of the monitoring points shown in Figure 1, moving from upstream to downstream, are as follows:

- The upstream study boundary was determined by identifying the river reach meeting two conditions: (1) the first point moving upstream where zinc was at an acceptable concentration in the river and (2) a reach in which flow-rate could be measured both safely and accurately (see Section 2.2.5, below).
- An ephemeral tributary stream enters from the left bank between the upstream monitoring point and the point mine water discharge, and this must therefore be monitored, if for no other reason than to rule it out as a source of zinc.
- Clearly the metal load emanating from the point source of mine water pollution must be monitored, and to assess its impact on the river an instream monitoring point is located in the river immediately downstream of the mixing zone of this mine water with the river.
- Reconnaissance surveys and background information suggested significant potential for diffuse pollution from the area below the point mine water discharge, and therefore another downstream monitoring point is located below the area believed to be the main source of these diffuse pollutants. For the purposes of this illustration, this second downstream monitoring point represents the downstream boundary (in reality, in the full study the actual

downstream boundary was some 15 km further downstream, at a hydraulic structure which facilitated flow measurements).

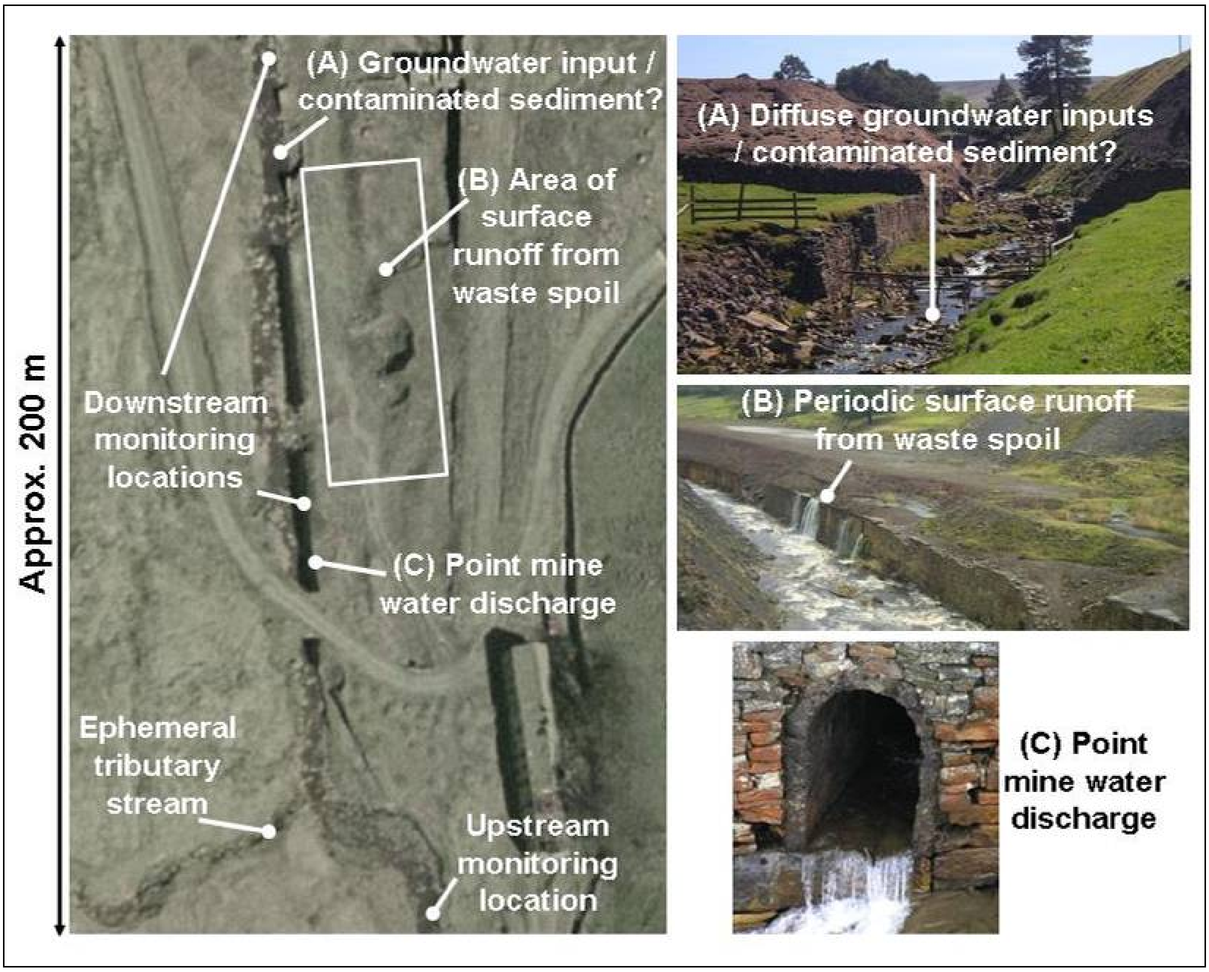

Figure 1. Point and diffuse non-coal mine water pollution in the River West Allen, Northumberland, and monitoring locations to quantitatively assess metal fluxes (see text for further details).

In this way it is possible to begin to quantify the importance of the various sources of mine water pollution within this reach of river. This process can then be extended to cover longer stretches of river, and indeed entire catchments, albeit the logistical difficulties become greater, as discussed in Section 2.2.2. The key point is that concentrations and flows need to be measured at all point sources and at instream points above and below both point sources and suspected or known diffuse sources.

2.2.5. Instream monitoring requirements and methods

Methods for monitoring flow-rates from point sources of mine water pollution, and sampling and analysis requirements for them, are discussed in Chapter 3 of this report, and are therefore not repeated here. The analytical suite for samples from instream locations is broadly the same as that shown in Table 4, with the same *caveats* as outlined in Chapter 3 regarding 'fine tuning' of the list of determinants over time. Importantly, however, the list of variables for analysis for instream samples may be expanded depending on the particular objectives of a study. Of particular importance is dissolved organic carbon (DOC), which would not ordinarily be determined in a mine water discharge. However, it may be very significant with respect to the fate of metals in receiving watercourses, and is therefore an advisable additional determinant for instream samples.

Monitoring instream flow-rates can be one of the most challenging aspects of any study of catchment-scale metal fluxes. As the size of the river increases, so do the challenges, since physical access becomes either difficult or impossible. An exhaustive review of the options and method details are not provided here as they are present in many standard hydrology texts (e.g. Gordon *et al.*, 2004; Shaw, 1996). However, the main methods, and some comments about them, are encapsulated in Table 3, and illustrations of some of them are shown in Figure 2.

In the absence of a gauging station the reality in practice is that it is sometimes difficult to identify a suitable method of flow measurement at a suitable location in a river catchment. In part as a consequence of this flow-rate values can be subject to substantial errors, particularly where the velocity-area method is used in reaches that do not lend themselves to it (i.e. boulders, turbulent flow). Difficult decisions therefore sometimes need to be made about monitoring locations in light of (a) the capital investment required to ensure accurate and reliable flow measurements and (b) the reliability and utility of the data if they have been collected using a method that may be subject to very substantial errors. That said, the flow-rate does not have to be measured with the same accuracy as is adopted for water resource investigations.

Though it is difficult to be definitive because of the site specificity of such issues, as a general rule if it is not feasible to measure flow-rate with appropriate confidence at a particular site, then it is better to use an alternative downstream / upstream site, review the data collected, and then make a decision as to whether it is worth pursuing accurate flow measurement at the original monitoring site (thus, it becomes an iterative process).

Once flows have been measured at what are deemed to be appropriate locations (point sources and instream locations) the data should be compiled and reviewed. Specifically, all of the flows should balance i.e. the flow-rate at a given downstream location should be the sum of the flow-rates of sources contributing to it. However, a certain amount of expert interpretation and iteration is required here, since if diffuse sources are significant they may contribute to flow-rates at the downstream location. Notwithstanding this important *caveat*, if flow balances are not within acceptable predefined margins of error, then monitoring locations and methods should be reviewed to evaluate potential sources of error.

Table 3. Common methods for measurement of flows in streams and rivers

Method	Comments
Engineered gauging stations	• Ideal solution if available • Facilitates continuous measurement • Data usually publicly available if station operated by environmental regulator • Very expensive to install compared to other options (engineering costs, planning approval etc)
Hydraulic control structures	• Installed to allow development of flow rating curves from stage-discharge relationship (see Shaw, 1996 for details) • Lower cost alternative to gauging station • Appropriate for streams in which substantial bed load movement e.g. upland streams
Acoustic Doppler Current Profiler (ADCP)	• Require personnel on both sides of river, but no physical access to river required • In absence of gauging station only feasible option for high flows • Cost of equipment ~ £10,000
Velocity-area method	• Not suitable for turbulent reaches containing boulders • May underestimate flows where significant hyporheic flows • Requires physical access to river • Relatively low cost (~ £3,000 for flow impeller)
Dilution gauging (slug injection and continuous injection)	• Appropriate for turbulent flows or in streams with significant hyporheic flows • Continuous injection in combination with water quality testing very appropriate for diagnosing diffuse sources (e.g. Kimball *et al.*, 2005) • Slug injection appropriate for gauging low- to medium- flows

Figure 2. Instream methods of measuring flow-rate

(A) Flume with V-notch throat, (B) Compound Crump profile weir with flat-V centre section, (C) Acoustic Doppler Current Profiling (ADCP) and (D) Velocity-area method: measuring mean velocities at measured intervals across a stream cross-section

Flow-rates and water quality of diffuse pollution sources are, by the very nature of diffuse pollution, very difficult to characterise. It is for this reason that diffuse inputs are derived in the approach described above, rather than directly measured. As illustrated in Figure 1 ((B) Periodic runoff from surface waste spoil), occasionally it is possible to collect samples of some diffuse sources for water quality analysis, but measurement of flow-rates, in order to calculate metals loads, remains problematic.

2.2.6. Metal fluxes under varying hydrological conditions

Previous catchment-scale investigations have repeatedly demonstrated that absolute fluxes of metals change substantially with changing hydrological conditions. Specifically, both Mayes *et al.* (2008) and Gozzard (2008) have shown that the contribution of diffuse pollution to total mass flux of metals may be less than 50% under low flow conditions (~ Q90), but higher than 90% under high flow conditions (~ Q10). In general terms the reason for this appears to be that pathways for diffuse pollutants become active during storm events (e.g. surface runoff transports fine waste spoil material into streams), and therefore metal flux due to diffuse sources increases markedly. In contrast, because many point mine water discharges arise from extensive areas of subsurface mine workings that effectively form a ‘reservoir’, the amplitude of flow-rate fluctuations of point discharges tends to be far less. Thus, metal load fluxes associated with point discharges are far less prone to variation than diffuse sources.

Quantifying this type of variation necessitates conducting the type of monitoring campaign described above repeatedly, under varying flow conditions (with obvious implications for flow measurement during storm events). To ascertain the range of metal fluxes occurring during as many flow conditions as possible, ideally such investigations should span a full calendar year. The cost of such investigations is clearly higher than that of just one or two campaigns, but it is necessary if a complete understanding of metal dynamics in a catchment is to be gained. In circumstances in which funding or time simply does not permit an investigation spanning a year, and therefore a compromise has to be reached, conducting catchment scale studies under baseflow (low flow) conditions is recommended. However, the conclusions of any such report must emphasise the uncertainties regarding metal flux under other hydrological conditions.

2.3 Conclusions and implications for remediation planning

Though an understanding of metal fluxes in river catchments may in some circumstances be an aim in itself, from an operational viewpoint the key purpose is to inform subsequent remediation planning. As stated earlier, a critical question is whether remediation of point sources alone will result in sufficient water quality improvements to meet predefined chemical and ecological status objectives (*cf* WFD). In instances where diffuse mining-related pollution is an important contributor to absolute metal loads in rivers, the answer to this question may in fact be no, at least during periods in which diffuse pollution is particularly prevalent i.e. high flow conditions. Therefore appreciating the overall flux of metals in a formerly mined catchment is important for a number of reasons (in the order in which they would be resolved):

(1) The main sources of metals under different flow conditions can be identified
(2) A quantifiable understanding of the dynamic importance of diffuse pollution can be ascertained
(3) An assessment of the water quality and ecological benefits likely to accrue from treatment of a point source(s) can be made
(4) The water quality and flow data gathered for point sources can be used directly for treatment system design purposes at a later stage (Chapter 4)
(5) The requirement for other types of intervention, to enhance the level of water quality and ecology improvements, can be assessed e.g. spoil capping works to limit surface runoff
(6) An overall cost-benefit evaluation of proposed engineering works and predicted environmental improvements in a river catchment (or water body) can be undertaken in an informed manner.

In order for data from such cost-benefit analyses to feed back into national prioritisation exercises such as the one reported upon here, and to assist the national regulator in delivering its commitment to the WFD in a consistent and effective way, catchment-scale investigations of metal mass fluxes in former mining catchments (and indeed other forms of pollution assessment and control) should be conducted on a nationally consistent basis. Thus, a single guidance document, and arguably a single reporting format, is recommended for such investigations, irrespective of the specific location.

The cost, time, and expertise required for catchment-scale non-coal mine water pollution investigations will be substantially greater than that needed for characterisation of a single point source discharge. Because the experimental design for such monitoring requires a detailed understanding of the background of the catchment in the first place, it is virtually impossible to accurately estimate the cost of an investigation without prior knowledge of the specific catchment. Furthermore, even with an understanding of the background (e.g. mine adit locations, gauging stations), the process of identifying monitoring locations is iterative, and therefore the monitoring locations and techniques may well only be determined once the project has actually started. Thus, a pragmatic, staged, approach appears to be required to effectively deliver projects of this type. The recommended approach is shown in Table 1.

There are of course certain disadvantages to this approach, of which the following may not be an exhaustive list:

- Procurement is likely to become more complicated and protracted if this approach necessitated 3 separate contracts (albeit there is logic in awarding to a single investigator)
- It becomes difficult for the funder to allocate resources on a catchment by catchment basis
- The overall time taken from the commencement of Stage 1 to the conclusion of Stage 3 is likely to be longer than if a single contract was let (in part because of the reasons above)

However, drawbacks such as these (that may in any case be resolvable) are far outweighed by the problems arising from inadequate investigations of catchment-scale metal fluxes, which fail to answer the critical questions that need answering, and therefore essentially need repeating (at cost) if effective environmental improvement schemes are to be instigated.

3. Water quality and flow data requirements for remediation of point sources of pollution

3.1 Introduction

In water bodies where individual point sources of mine water pollution are identified as the major causes of downstream degradation, or even where they are one of several causes, benefits of treatment of the discharge may outweigh the costs, and therefore planning for remediation will proceed. This section provides details of the information that is required to design a treatment system for a non-coal mine water discharge, and briefly reviews currently available data for non-coal mine waters appearing in the *Impacted* and *Probably Impacted* categories of the prioritisation database. For clarity the discussion here is restricted to consideration of the availability of data for the design of a treatment system for a *single point source of mine water* pollution.

The design of any mine water treatment system (in fact any water treatment system period) is critically based on the contaminant *load* i.e. the absolute mass of contaminant discharged per unit time, calculated by multiplying flow-rate by concentration. Flow-rate and concentration *must* be measured simultaneously to derive a reliable load measurement, and since in many cases flow-rates and (metal) concentrations in mine water discharges vary, repeat measurements need to be made under differing hydrological conditions (preferably over the course of at least 1 full year). It is important to note that if a thorough catchment-scale investigation has been conducted (as detailed in Chapter 2), this information should already be available.

In relation to the availability of metal load data, it is important to note that of all the data collected as part of this project there are only 23 mine waters (out of a total of 257 confirmed discharges) for which both water quality <u>and</u> flow data are available. It should be emphasised that this does <u>not</u> imply that these variables are all that are required for the design of a treatment system. This is certainly not the case; the data shown were collected solely for the purposes of this national impact assessment, not as a basis for a case-by-case evaluation of treatment requirements. Actual data requirements for treatment system design are discussed below.

3.2. Data requirements for treatment system design

Guidance on the data requirements for mine water treatment system design has been provided in previous publications by the PIRAMID Consortium (2003) and Younger and Wolkersdorfer (2004) (Chapter 2 and Appendix 1 of these publications respectively). The reader is referred to these sources for details of sampling protocols, analysis and QA/QC procedures. The variables requiring determination for treatment system design are discussed here, even though they are covered in the publications cited above. This is because, in the experience of these authors, unfortunately it remains all too common for analyses of mine water discharges to be incomplete, severely restricting the utility of the data for purposes of diagnosing mine water provenance and / or using it for the design of a treatment system.

Table 4 summarises the analytical requirements for a point mine water discharge being investigated for possible treatment. The list is generic in the sense that it is unlikely that all of the contaminant metals will require determining on every sampling occasion – some, for example, may prove to be below detection limits on a consistent basis. However, if nothing is initially known of the quality characteristics of a mine water, then a broad list of analytes should be determined in the first instance.

The reasons for inclusion of the variables shown in Table 4 are as follows:

- *Design variable*: either essential for the design of an appropriate treatment system, or occasionally of importance on a case-by-case basis e.g. for contaminant metals, where it is the dominant pollutant of concern
- *Indicates salinity*: although rarely an issue with discharges from abandoned metal mines, this can occasionally be significant since at high salinity concentrations healthy growth of the reeds typically used in wetland systems is unlikely.
- *Charge balance*: Often overlooked, but an essential quality control protocol. To calculate the charge balance all of the major anions and cations must be determined (may also highlight important variables that have been overlooked in the analysis).
- *Provenance*: The provenance of a mine water may be very relevant for design purposes, for example if the chemical characteristics indicate a surface- or ground-water fed mine water discharge, with consequent implications for flow variability.

It is impossible to be entirely prescriptive about the analysis suite to be adopted at any particular mine water. The critical issue is to ensure that the concentrations of all the potentially important variables are determined. This can be largely informed from an understanding of the hydrogeology and mineralogy of the abandoned workings, and the sensitivity of the receiving watercourse. Notwithstanding the importance of an informed approach to analysis selection, modern analytical equipment is such that the analysis of 10 metals is not much more time-consuming or costly than the analysis of just one metal. Therefore at least on the first few sampling and analysis exercises a rather liberal approach to the analysis suite selection can be an advisable precautionary approach.

As noted above, the measurement of flow-rate for discrete point sources is relatively straightforward. The most common approaches are:

- 'Bucket-and-stopwatch' measurements: perfectly acceptable and accurate where flow-rates are low and physical conditions permit i.e. a bucket of known volume can be used to capture all of the flow in a period of at least 10 seconds (repeated 3 times, with the mean value taken).
- The installation of a sharp-crested weir of some variety (e.g. V-notch, rectangular notch), sometimes with a self-contained pressure transducer and logger behind it to facilitate semi-continuous measurements, is the preferred approach for small

channels or adits. Simple hydraulic equations are used to calculate the flow-rate from a measurement of the depth of water above the weir. Such structures will yield accurate results as long as the simple rules for installation, outlined in all good hydrology texts, are followed.

- For discharges from pipes, where the pipe is running partially full but the flow is too great for measurement by bucket-and-stopwatch, the Chézy, Strickler or Manning formulae can be used to determine flow. To do so it is necessary to know the diameter of the pipe, its angle, and the maximum depth of water within it.

Table 4. Variables to be determined during design of a mine water treatment system

Variable	Typical measurement unit	Principal reason(s) for inclusion
Flow-rate	L / second or m^3/d	Design variable (always)
pH	-	Design variable
Total acidity	mg/L as $CaCO_3$	Design variable
Total alkalinity	mg/L as $CaCO_3$	Design variable
Conductivity	µS/cm or mS/cm	Indicates salinity
Suspended solids	mg/L	Design variable (occasional)
Sulphate	mg/L	Salinity; charge balance; provenance
Chloride	mg/L	Salinity; charge balance; provenance
Nitrate	mg/L	Design variable (occasional)
Ammoniacal nitrogen	mg/L	Design variable (occasional)
Calcium	mg/L	Charge balance; provenance
Magnesium	mg/L	Charge balance; provenance
Sodium	mg/L	Salinity; charge balance; provenance
Potassium	mg/L	Charge balance; provenance
Total iron (Fe)*	mg/L or µg/L	Design variable
Ferrous iron (Fe^{2+})	mg/L or µg/L	Design variable
Aluminium*	mg/L or µg/L	Design variable (occasional)
Arsenic*	mg/L or µg/L	Design variable (occasional)
Cadmium*	mg/L or µg/L	Design variable (occasional)
Copper*	mg/L or µg/L	Design variable (occasional)
Manganese*	mg/L or µg/L	Design variable (occasional)
Nickel*	mg/L or µg/L	Design variable (occasional)
Lead*	mg/L or µg/L	Design variable (occasional)
Silver*	mg/L or µg/L	Design variable (occasional)
Zinc*	mg/L or µg/L	Design variable

Note. Adapted from PIRAMID Consortium, 2003.
*Collection and analysis of both total and filtered samples advisable

It is worth noting that in-line flow meters, whilst highly effective for domestic water supply purposes, are typically beset with problems when applied to mine waters, due to a combination of corrosion and mineral encrustation problems.

3.3. Current data availability

In collating the data required for the primary purpose of the current project – prioritising abandoned non-coal mines – results for all of the variables identified in Table 4 were not requested. It is therefore difficult to make a conclusive assessment of the current availability of complete datasets for non-coal mine waters around the country. It is certainly the case that for specific mine waters such data are available; the Dyffryn Adda adit on Anglesey (Table 2) is one such example. What is unquestionably in short supply is consistent, reliable, synchronous flow-rate and water quality data, gathered at sufficient frequency to have confidence in the variations in contaminant load of individual mine waters. To give an indication of the overall shortage of such information, 20 mine waters are listed in Table 2, covering the whole of the Dee, Northumbria, South West and Western Wales RBDs. These were in fact the only mine waters for which flow-rate data were held by the Environment Agency, out of a total of 257 mine waters identified. Data from other sources may be available to improve the record, but the shortage of synchronous flow and water quality measurements is nevertheless clear. Experience suggests that there will be even fewer records of point discharges for which reliable flow and water quality measurements *covering a range of hydrological conditions* are available. A discussion of the type of investigations that are required to effectively address these shortfalls in data quality has already been presented in Chapter 2 of this report.

4. Treatment options for non-coal mine waters

4.1 Introduction

Mine water treatment technologies are usually divided into two categories: Active treatment and Passive treatment. Active treatment entails the use of energy and / or chemicals to facilitate treatment. Passive treatment systems, in contrast, rely on gravity and naturally-occurring biogeochemical reactions to effect treatment. Both approaches have advantages and disadvantages, as discussed below, and both have an important role to play in the overall management of mine water pollution (in a global sense).

In reality many operational treatment systems in the UK (the vast majority operated by the UK Coal Authority) combine elements of active and passive treatment. For example mine water is typically pumped from abandoned mine shafts at many Coal Authority sites, both to prevent uncontrolled discharge to surface and facilitate gravity drainage through the treatment system. This is therefore ‘active’ because it utilises energy, but subsequent treatment is often by passive means, such as settlement lagoons and aerobic wetlands. There may also be scope in the future for effective treatment using a variety of ‘hybrid’ treatment systems, which employ elements of active treatment processes in passive systems, keeping the operating costs to a minimum whilst maximising treatment performance. These issues are discussed further in the text below. The key point here is that the division of treatment technologies into active and passive should not preclude consideration of options that harness aspects of both.

The most common contaminants of non-coal mine discharges in England and Wales are metals. Acidity (low pH) is less commonly a pollutant of non-coal mine drainage and, although it is a critical treatment system design variable, it is rarely a cause of failure of EQS limits at downstream points because of simple dilution effects. Sulphate can also be a pollutant in mine drainage. However, because non-coal mine drainage tends to arise from the oxidative dissolution of monosulphides it is less of an issue than for coal mine waters. Additionally, in the UK it is unusual for it to be present in receiving watercourses at concentrations high enough to be a hazard to aquatic ecology, and dilution effects typically mean it does not exceed EQS limits at downstream locations. The focus of the active and passive treatment sections of this report is therefore on metal contaminants, albeit with considerable reference to pH for reasons that will become obvious.

Whilst ‘end of pipe’ treatment is a common approach to remediation of mine water pollution problems, source or pathway control is arguably a preferable solution where it is feasible, and is likely to be a very important approach with respect to efforts to address diffuse pollution remediation in the future (see Section 4.6, below, for a brief review). The focus of this section, however, is mine water treatment (of point sources). The underlying mechanistic principles of mine water treatment are outlined first, as this informs the subsequent discussion of the possible treatment options for non-coal mine waters in England and Wales. An overview of active treatment technologies and passive treatment technologies is provided. In the passive treatment section there is an emphasis on the limitations of these technologies for application to non-coal mine water discharges, which serves to inform subsequent discussion of the possible options for passive treatment of non-coal mine drainage, and research needs to that end (Section 4.4).

4.2 Biogeochemical principles of mine water treatment

Where the removal of sulphate to statutory limits is not a requirement the primary objective of non-coal mine water treatment is almost always the elevation of pH, where necessary, and the immobilisation of metals within the confines of the treatment system. Even where pH is not outside regulatory limits close attention needs to be paid to it, since pH is a crucial control on the geochemical behaviour of metals in aqueous environments, as detailed below. Where pH is low, generation of alkalinity is a critical first step in treatment. Typically an abiotic increase in pH is accomplished by dissolution of alkaline materials such as calcite, or the addition of alkali chemicals, most commonly sodium hydroxide (caustic soda) or hydrated calcium hydroxide (lime). The relevant reactions are as follows:

Dissolution of calcite at pH < 5 (Reaction 1) and pH > 5 (Reaction 2):

$$CaCO_3 + 2H^+ \leftrightarrow Ca^{2+} + H_2O + CO_2 \qquad (1)$$
$$CaCO_3 + H_2CO_3 \leftrightarrow Ca^{2+} + 2HCO_3^- \qquad (2)$$

Increasing pH using sodium hydroxide (3) and calcium hydroxide (4):

$$NaOH + H^+ \leftrightarrow Na^+ + H_2O \qquad (3)$$
$$Ca(OH)_2 + 2H^+ \leftrightarrow Ca^{2+} + 2H_2O \qquad (4)$$

Using alkali chemicals in stirred tank reactors pH can be very rapidly elevated (minutes). In passive treatment systems, in which calcite dissolution is the main source of alkalinity, reaction times are longer; Hedin *et al.* (1994a) suggest optimal contact time for generation of alkalinity in limestone reactors is 14 hours.

A key process in the remediation of mine water pollution is the generation of solid metal (oxy)hydroxides (and sometimes carbonates) under oxic conditions (in fact this is a key process in *every* full-scale mine water treatment system currently operating in the UK). The prevalence with which precipitation will occur depends upon both the specific metal and the pH. Specifically, the rate of the precipitation of hydroxides increases with pH[1], which is precisely why the elevation of pH, by one of the means identified above, is required. This is the principal mechanism by which iron is removed from coal mine drainage: mine water is aerated to generate oxidising conditions in which ferrous iron is rapidly converted to ferric iron (5), and the ferric iron then forms a solid hydroxide precipitate which is typically retained in settlement lagoons or wetlands (6).

$$Fe^{2+} + \tfrac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \tfrac{1}{2}H_2O \qquad (5)$$

[1] It is important to note that some metals are amphoteric, and therefore become more soluble at very high pH.

$$Fe^{3+} + 3H_2O \leftrightarrow Fe(OH)_3 + 3H^+ \quad (6)$$

Iron is readily removed in this way because its hydroxide solubility product is very small, as indicated in the right hand column of Table 5. Thus, it will form readily in the presence of even low concentrations of hydroxyl ions (OH^-), and this is reflected in the pH values for formation of hydroxides shown in Table 6 i.e. Fe^{3+} will begin to form at relatively low pH. Furthermore, as pH increases, and more hydroxyl ions become available, the rate of removal will increase.

What is also clear from Tables 5 and 6 is that other metals behave differently; the hydroxide solubility products of all the metals shown are greater than that of iron, and the target pH for their removal is higher. This characteristic of differing metal solubility is used to advantage in some (active) mine water treatment technologies which, through careful pH control, facilitate selective precipitation of metals, in order to recover them for possible reuse. However, in passive treatment systems the very high pH required to immobilise some metals as hydroxides (e.g. manganese) represents a significant problem, and this is taken up further in the discussion below.

Table 5. Solubility products (log K_{sp} at 25°C) of sulphides and hydroxides of metals

Metal	Sulphide	Metal	Hydroxide
Cu^{2+}	-35.9	Fe^{3+}	-38.6
Cd^{2+}	-28.9	Pb^{2+}	-19.9
Pb^{2+}	-28.1	Cu^{2+}	-19.8
Zn^{2+}	-24.5	Fe^{2+}	-16.3
Ni^{2+}	-21.0	Zn^{2+}	-16.1
Fe^{2+}	-18.8	Ni^{2+}	-15.3
Fe^{3+}	-	Cd^{2+}	-14.3
Mn^{2+}	-13.3	Mn^{2+}	-12.7

Note. Arranged in order of readiness with which the precipitate will form (from top to bottom) (data from Diaz et al., 1997).

Table 6. Target pH values for removal of various metals as their hydroxides (from Younger *et al.*, 2002).

Metal	pH
Fe^{3+}	3.5
Cu^{2+}	6.8
Pb^{2+}	6.8
Zn^{2+}	8.2
Fe^{2+}	8.5
Cd^{2+}	9.8
Mn^{2+}	10.2

Note that optimal pH values for rapid removal may be higher than the values indicated.

pH is not the only variable that requires consideration with respect to the immobilisation of metals. Oxidising conditions are required for precipitation of metals as their hydroxides. Redox potential (Eh) is therefore also an important control on the state of metals in aqueous environments, and the pH-Eh relationship can be used to predict the behaviour of metals. A pH-Eh diagram for zinc is shown in Figure 3. What can be seen here is that the solid phases $ZnCO_3$ (smithsonite) and $Zn(OH)_2$ are generally favoured in oxidising conditions in which pH is ~ 8 for $ZnCO_3$ and > 8 for $Zn(OH)_2$. Note the sharp contrast to Fe^{3+}, the main contaminant in coal mine drainage, which will begin to form a solid hydroxide, $Fe(OH)_3$, at pH > 3.5 under oxic conditions (Table 6).

The discussion thus far has focused on abiotic mechanisms for increasing pH and removing metals as hydroxides under oxic conditions. However, pH may also be increased under anoxic conditions by an important biotic process, which may simultaneously immobilise divalent metals as sulphides. Dissimilatory bacterial sulphate reduction (BSR) is a much cited process in mine water treatment, particularly in relation to passive remediation. The principle is that sulphate, which is invariably present in elevated concentrations in mine waters, can be reduced under anoxic conditions by sulphate reducing bacteria. The reduction of sulphate by BSR can consume protons (Reaction 7), generate alkalinity (Reaction 8), and simultaneously release sulphide to form a precipitate with divalent metal ions (Reaction 9).

$$2CH_2O + SO_4^{2-} + 2H^+ \rightarrow 2CO_2 + H_2S + 2H_2O \qquad (7)$$
$$SO_4^{2-} + 2CH_2O \leftrightarrow H_2S + 2HCO_3^- \qquad (8)$$
$$H_2S + M^{2+} + 2HCO_3^- \rightarrow MS + 2H_2O + 2CO_2 \qquad (9)$$

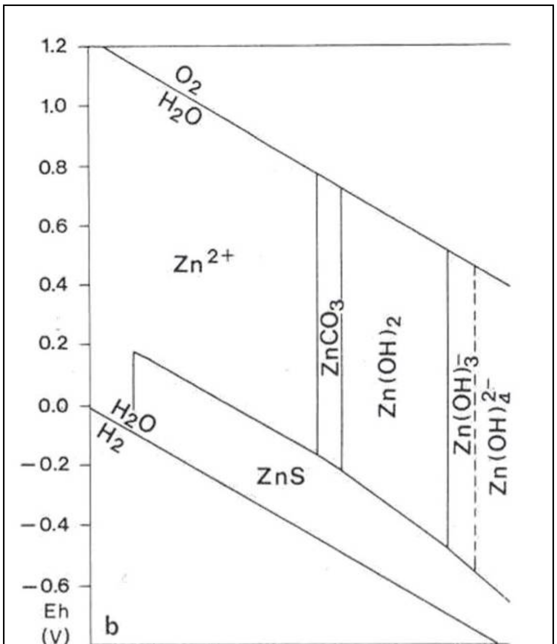

Figure 3. Eh/pH diagram for the Zn-O-H-S-C system at 25ºC and 1 atm (Salomans & Förstner, 1984)

In order to promote anoxic conditions, and provide a source of carbon to enable sulphate reducing bacteria to metabolise, the substrate for passive treatment systems harnessing BSR is some form of organic medium, such as compost. To further enhance the generation of alkalinity, limestone is often mixed with the organic substrate, resulting in calcite dissolution. Sulphate reduction is harnessed in both active and passive systems. In the UK the focus has been utilisation of BSR in passive systems, with the principal objective of raising pH and immobilising at least a portion of the metal contaminants within the organic substrate. In other countries sulphate reducing bioreactors have been developed as active systems, with the specific objective of lowering sulphate concentrations to regulatory limits. This has been particularly the case in semi-arid and arid countries, in which mine water may be required to be treated to standards appropriate for subsequent potable use.

Other mechanisms may also serve to immobilise metals within treatment systems. Such processes include adsorption, co-precipitation and bioaccumulation. Whilst treatment systems are not explicitly designed on the basis of such processes, they may be significant in certain situations, as briefly discussed below. Like precipitation processes, many are pH-dependent, with implications for the rate of removal. As an example, Figure 4 illustrates the pH-dependency of sorption of various metals onto hydrous ferric oxide (HFO).

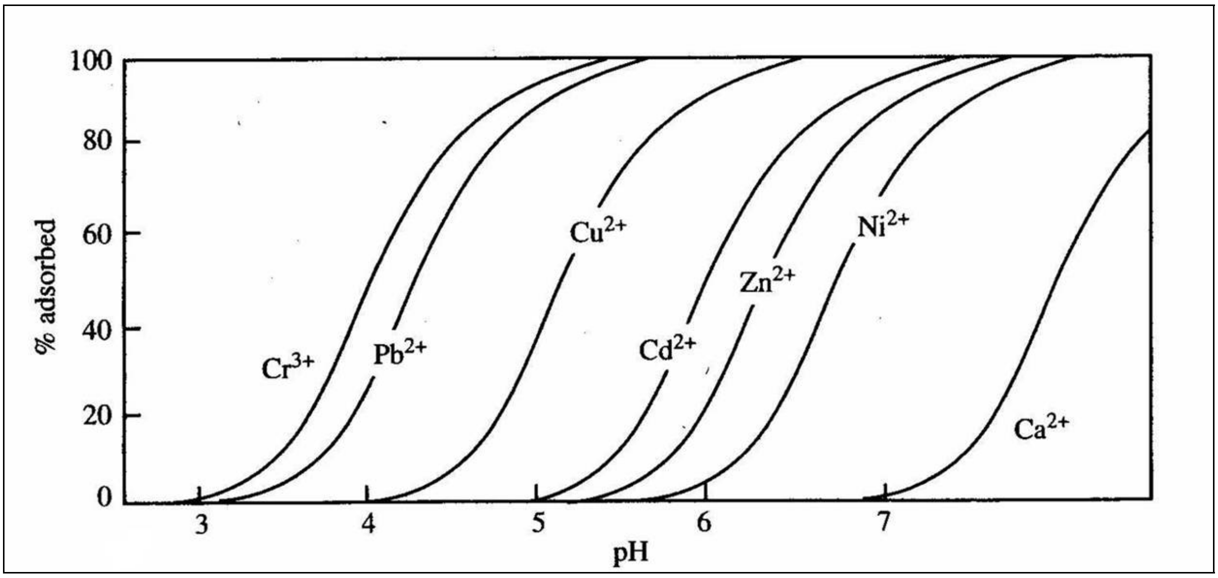

Figure 4. pH specific adsorption edge profile (for adsorption onto Hydrous Ferric Oxides) for a range of cations (from Gaillardet *et al.*, 2003)

4.3 Active treatment options

Active treatment entails the use of conventional wastewater treatment units, and necessitates the ongoing input of power and chemical reagents. Although such processes require close process control, and have high operating costs, in most cases active technologies allow for the rapid removal of contaminants from mine water. Consequently the size of units is substantially less than equivalent passive systems, and the overall land area requirement accordingly much smaller. They therefore generally find application in cases where contaminant load and flow-rate is very high and / or land area available is restricted.

In the many instances where the removal of acidity and iron are the objective active treatment classically comprises *o*xidation, *d*osing with *a*lkali and *s*edimentation (ODAS) (Younger *et al.*, 2002):

- Oxidation, to convert ferrous iron to the less soluble ferric iron, and also to liberate dissolved CO_2, which may otherwise contribute to acidity. In the UK oxidation is most commonly achieved using simple aeration cascades, but chemical oxidants, such as hydrogen peroxide, can also be used to good effect, albeit at a greater cost.
- Dosing with alkali, to lower the solubility of most metals due to the increase in pH, and thus encourage the precipitation of metals as their hydroxides because of the ready availability of hydroxyl (OH^-) ions. Hydrated lime [$Ca(OH)_2$] and caustic soda [NaOH] are most often used for this purpose.
- Sedimentation to remove metal hydroxide precipitates. The rate of sedimentation is often accelerated by using lamella plate thickeners, clarifiers and chemical flocculants.

At the most basic level alkali and mine water are simply blended in a mixing tank, from where it discharges directly to sedimentation ponds. The volume of alkali dosed is controlled by continuous monitoring of pH at some point below the mixing tank. However, in many cases a more sophisticated approach is adopted, for example by using lamella plate clarifiers to accelerate the removal of the ferric iron sludge. Increasingly typical practice is to return an aliquot of the thickened sludge to the influent of the sedimentation unit to increase the density of the final sludge (to yield sludge with 25-30% solids by volume). This is known as the “high density sludge” (HDS) process, and successfully operating examples can be seen at Horden, County Durham, and Wheal Jane, Cornwall. The Wheal Jane system is a classic example of a high volume, high metal load discharge. The treatment system now in place at that site treats in the order of 400 L/s of mine water, with approximate pH 3.5, and iron and zinc concentrations of around 200 mg/L and 50 mg/L respectively. Further details of the active treatment plant at Wheal Jane tin mine are provided by Coulton *et al.* (2003).

The ODAS approach is satisfactory in many instances, and simply by adjusting the dose-rate of alkali any or all of the metals listed in Table 5 can be effectively removed. Thus, such approaches could be used to successfully remediate non-coal mine drainage across England and Wales, albeit at very great capital investment (see Chapter 7). To date it has not been deemed necessary to address the removal of sulphate in mine water treatment systems, but if salinity removal is required then more advanced technologies are required. As noted above, this is a more likely scenario where water resources are at a premium, such as the semi-arid climates of southern Africa and Australia.

The reduction of sulphate to sulphide species is one approach to metal removal that has been harnessed in both passive and active treatment systems. Active treatment varieties of these “sulphidisation” processes are discussed by Dill *et al.* (1994), Dvorak (1996), Hammack *et al.* (1994), and Rowley *et al.* (1994) among others. Such technologies are capable of lowering metal ion concentrations to very low levels in treated waters. A further development along these lines is the possibility of mixing waste-streams containing high concentrations of dissolved sulphur species with metal-rich mining effluents, and thus simultaneously removing both metals and sulphur. The development of such a process, in which tannery waste is mixed with mine water, is described by Rose *et al.* (1998) and Boshoff (1999). Such technologies have yet to find any application in the UK for abandoned mine water discharges and, as noted above, significant capital investment would be required to do so.

In a technological sense membrane processes, such as reverse osmosis, micro-, ultra-, and nano- filtration, could all find application to abandoned mine water treatment. However, in most cases they are not viable, principally because of the high costs associated with the membranes themselves and the high operating pressures required (Younger *et al.*, 2002).

The implementation of other active treatment technologies suffers from similarly prohibitive costs in most mine water treatment cases. However, for completeness, such technologies include (Younger and Wolkersdorfer, 2004):

- Solvent extraction
- Sorption / ion exchange
- Electrochemical extraction
- Biochemical extraction
- Barium sulphide process
- Biological trickling filters

Further information on active treatment technologies can be found in Younger *et al.* (2002), and the International Network for Acid Prevention's (INAP) Global Acid Rock Drainage (GARD) guide, which will be available from INAP's website in late 2009 (www.inap.com.au).

4.4 Passive treatment options and their application to non-coal mine water discharges

The principles and design of passive treatment systems for mine water treatment have been discussed in detail by the PIRAMID Consortium (2003) and Younger *et al.* (2002) among others. Both volumes are widely available and exhaustive details are therefore not repeated here other than where it is pertinent to the discussion. This section focuses on the potential application of these treatment systems to the significant problem of non-coal mine drainage in England and Wales.

The UK has a track record of implementation of passive treatment systems for *coal* mine drainage remediation that is unrivalled anywhere in Europe. More than 50 full-scale systems are currently in operation, the majority installed by the UK Coal Authority. However, with the exception of the HDS plant at Wheal Jane, all of them have the primary objective of elevating pH (where necessary) to facilitate the removal of only *iron* within the treatment system. It is apparent from Table 2 that for non-coal mine drainage, iron is not the only cause for concern (and even where it is present the Coal Authority's track record is evidence enough that it can be dealt with effectively). From the discussion in Section 4.2 it will also be clear that the removal of other metals, such as zinc and cadmium, requires different geochemical conditions to that needed for the removal of iron, particularly with respect to the pH.

In principle at least, zinc, and other metals present in non-coal mine drainage, can be removed via precipitation as their hydroxides within aerobic passive treatment systems e.g. settlement lagoons and aerobic wetlands (*cf* Figure 3). In addition, Nuttall (1999) illustrates the possibility of removing zinc as its carbonate (smithsonite) in anoxic limestone drains (ALD) (*cf* Figure 3). Adsorption and co-precipitation processes may also be an effective means of attenuation of metals such as zinc, using high surface area media under appropriate geochemical conditions (*cf* Figure 4). The treatment performance data of a range of passive treatment systems in which the authors of the publications report removal of zinc are provided in Table 7 (it should be noted that whilst the majority were designed for metal removal, not all of them were explicitly designed to do so). Whilst Table 7 clearly does not provide an exhaustive list of performance data for passive systems which receive zinc as a contaminant, it nevertheless highlights some very illuminating data with respect to the potential application of some current passive treatment technologies to the remediation of non-coal mine drainage.

Of all the data shown in Table 7 column 10 is arguably the most significant. This reports the area-adjusted removal rate for zinc within each of the systems. The area-adjusted removal rate shown in this column is the mass of metal removed per unit area of treatment system per unit of time. It is calculated by dividing the metal load (in g/d) removed within the treatment system by the area of the treatment system (in m^2). Measurements of the area-adjusted removal rate in wetland systems treating coal mine drainage have revealed that on average iron is removed at an area-adjusted removal rate of 10 $g/d/m^2$ in aerobic wetland systems receiving water with circum-neutral pH (Hedin *et al.*, 1994b). This figure of 10 $g/m^2/d$ remains the basis for design of coal mine drainage treatment wetlands for removal of iron under aerobic conditions, using the following formula:

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

where: A = Required wetland area (m^2)
Q_d = Average daily flow-rate to wetland (m^3/d)
C_i = average daily influent iron concentration (mg/L)
C_t = target iron concentration (mg/L)
R_A = Area adjusted contaminant removal rate ($g/m^2/d$)

The formula above clearly shows that (a) the contaminant load (the numerator) must be known to calculate wetland area (*cf* comments in Section 3.1 about the importance of this variable) and, most importantly in the current context, (b) the value of the area-adjusted removal rate (the denominator) has a major bearing on the overall size of the wetland.

Returning to column 10 of Table 7 it can be seen that the area-adjusted removal rate of zinc in the passive treatment systems is with only one exception less than 0.5 $g/m^2/d$, and in a number of cases at least an order of magnitude lower. Thus, from the data shown in Table 7, an aerobic wetland for attenuation of zinc would need to be *at least* 20 times the size of such a system for equivalent remediation of iron pollution (and a similar principle applies to settlement lagoons, for which hydrolysis and precipitation of metals are also the main *modus operandi*).

To take just one example from Table 2, Pugh's adit in Western Wales has a flow-rate of 8 L/s and zinc concentration of 22.4 mg/L. Even using an area-adjusted removal rate for zinc of 0.5 $g/m^2/d$ the area of wetland calculated is in excess of 30,000 m^2. Given the upland locations, and consequently steep topography, of many non-coal mine discharges around England and Wales, the likelihood of identifying areas of land sufficient to accommodate such large wetland systems is slim at best. Quite apart from this consideration, the capital costs of construction of such large systems are very high, principally due to the earth-moving / import costs.

Alternative approaches to passive remediation of zinc and other metals include harnessing high rates of sorption onto various types of media (*cf* Figure 4), and / or co-precipitation at high pH. The pilot-scale ochre drain investigated by Mayes *et al.* (2009) is one such example, the performance data for which are shown in the first row of Table 7. This particular system exhibited a comparatively good area-adjusted removal rate of 3.7 $g/m^2/d$ over the 10 month duration of operation. The principal mechanisms of removal were sorption to the ochre pellets used in the system, and co-precipitation with calcite-dominated secondary minerals at high pH. The cement used as a binder to create the ochre pellets was cited as a key source of alkalinity; pH > 11 was recorded at some points during operation. Although such high pH values clearly encourage the removal of metals, rapid precipitation of calcium carbonate minerals can cause problems of armouring of the ochre, and the costs of full-scale implementation may be prohibitive given the cost of the cement used as a binder. Perkins *et al.* (2007) report on an alternative approach which utilises dealginated seaweed as a sorbent for metals such as Zn, Cd. Removal rates are initially high for comparatively short residence times, but sorption capacity of the material appears to become exhausted quickly, raising issues of how such systems might be maintained over the medium- to long-term.

Table 7. Treatment system performance of different types of passive treatment technology, using various assessment metrics

1	2	3	4	5	6	7	8	9	10	11
Wastewater type	**System type**	**Monitoring period (months)**	**Area (m^2) Volume (m^3)**	**Flow (L/s)**	**Residence time (days)**	**Zn influent / *effluent* (mg/L)**	**Load removed (g/d)**	**Treatment efficiency (%)**	**Area-adjusted removal rate ($g/m^2/d$)**	**Volume-adjusted removal rate ($g/m^3/d$)**
Mine water[1]	Pilot-scale Ochre drain	10	0.35 m^2 0.21 m^3	0.03	0.02	1.7 / *1.2*	1.30	32	3.71	6.19
Mine water[2]	Pilot-scale Aerobic wetland / algal mat	48	420 m^2 240 m^3	0.04 – 0.17	16 - 79	16 / *1.2*	191.8	91	0.47	0.80
Mine water[2]	Full-scale aerobic wetland / algal mat	24	13,200 m^2 6,000 m^3	8.5	8.2	14.4 / *9.9*	3,300	31	0.25	0.55
Mine water[3]	Pilot-scale Anoxic Limestone Drain	1.5	4.5 m^2 2.25 m^3	0.02	0.65	6.91 / *5.74*	2.02	17	0.45	0.90
Not specified[4]	Subsurface flow wetland	-	- -	-	-	2.50 / *0.73*	-	71	0.11	-
Urban runoff[5]	Full-scale Surface flow wetland	12	8,250 m^2 -	18	-	0.065 / *0.057*	12.4	12	0.002	-
Domestic wastewater[6]	Full-scale Gravel subsurface flow wetland	9	650 m^2 325 m^3	6.25	-	0.113 / *0.014*	53.5	88	0.08	0.16
Zn-containing wastewater[7]	Lab-scale Hydrosoil surface wetland	2	0.896 m^2 0.224 m^3	0.003	1.00	1.76 / *0.34* 1.76 / *0.52*	0.319 0.278	81 70	0.37 0.31	1.42 1.24
Smelter water[8]	Lab-scale subsurface wetland	4	0.231 m^2 0.069 m^3	8.3×10^{-5}	7.00	0.140 / *0.043*	7×10^{-4}	69	0.003	0.01

Note. Figures derived where not explicitly given in text of references (from Mayes et al., 2009; citations therein referenced again here for convenience. Sources of data: [1]Mayes *et al*. (2009); [2]Kalin (1998); [3]Nuttall (1999); [4]Kadlec and Knight (1996); [5]Scholes *et al* (1998); [6]Lesage *et al.* (2007); [7]Gillespie *et al.* (1999); [8]Song *et al.* (2001)

Nevertheless, further development work of such passive technologies may offer promise for the future, especially where there is a potential to utilise waste ochre from coal mine drainage treatment systems for the remediation of non-coal mine water discharges. Particular development needs, given that the capacity of any media for sorption is finite, are to develop technologies that either regenerate media within such treatment units, or to engineer systems such that removal (and disposal) and replacement of exhausted media can be undertaken straightforwardly during operation.

Nuttall (1999) and Nuttall and Younger (2000) have investigated the use of Anoxic Limestone Drains (ALD) as a technology for immobilisation of zinc. The principle is that at approximately pH 8.2 zinc will form its carbonate mineral, smithsonite, as shown on the phase diagram in Figure 3. Table 7 (row 4) shows that whilst residence time in the pilot-scale system installed was reasonable, area-adjusted removal rate was still low (0.45 $g/m^2/d$), and treatment efficiency was comparatively poor at 17% (probably a function of the residence time). Nevertheless, this approach is currently being investigated further at a pilot-scale system on the Red River, Cornwall. Although data for this system have yet to be thoroughly interrogated, first indications are that the system is remediating zinc to good effect. The key issue will be whether the rate of the process is sufficiently fast to make this is a logistically feasible option for full-scale deployment.

Current passive treatment options also include a number of systems which work under anoxic conditions with the aim of encouraging BSR in organic substrates, to simultaneously generate alkalinity and immobilise metals as sulphides, as illustrated in Reactions 7 - 9. Compost wetlands and Reducing and Alkalinity Producing Systems (RAPS) are the most common configurations for treating mine water where it emerges at surface. It is evident from the solubility products for metal sulphides (shown in Table 5), that some of the key metals present in non-coal mine drainage are removed preferentially over the iron which is ubiquitous in coal mine drainage. Thus, in theory at least, the order of removal of metals as sulphides, under appropriate geochemical conditions, is Cu > Cd > Pb > Zn > Ni > Fe^{2+} > Fe^{3+} > Mn. In principle therefore the use of compost-based passive systems, operating under anoxic conditions, appears more promising as a means of removal of metals such as zinc than alternative systems operating under aerobic conditions.

There have been numerous studies of the potential of such compost-based systems for the remediation of mine drainage. However, there are a number of key problems that need addressing before such units could be installed with confidence at full-scale:

- The vast majority of studies[2] are undertaken at laboratory-scale, and therefore do not account for issues of different environmental conditions at actual mine sites, or of the very substantial differences in performance that may result by increasing the scale of operation by several orders of magnitude (due, for example, to non-ideal hydraulic performance).

[2] In preparing this report a brief review of compost-based mine drainage treatment systems was undertaken, using a widely available publications database. Of 130 recent publications found on the subject, only 2 were conducted at pilot- or full-scale under field conditions.

- Notwithstanding the point above, the duration of some experiments reported in the literature is simply too short to give any confidence at all that the unit would operate effectively over the long time periods necessary for a full-scale system (e.g. 14 days operation in the case of Jong and Parry (2003)).
- Performance in terms of zinc removal is often poor, or very unpredictable (e.g. Wildemann *et al.*, 1990). This in part appears to relate to the initial SRB content of the organic substrate.
- Where very high removal rates are reported it is often in systems in which the residence time of waters within the treatment system is very high (e.g. > 5 days), and such units are therefore most unlikely to be a practical solution given the size requirements.

In addition to the issues of engineering scale, one of the key barriers to development of reliable compost-based systems appears to be the development, and *maintenance*, of an active population of sulphate reducing bacteria, especially under field conditions (e.g. Johnson and Hallberg, 2005). Both Johnson and Hallberg (2005) and Mayes and Jarvis (2008) noted that a period of maturation of the organic substrate (4 – 10 months) prior to use appears to improve subsequent performance, presumably due to the development of microbial populations. In an effort to improve initial microbial populations, and maintain active colonies, mine waters and / or substrates may be inoculated with microbial populations at the outset to encourage their establishment (Pereyra, 2008), or carbon additions may be made during operation to ensure continued activity of SRB populations over the medium- to long-term (Mayes and Jarvis, 2008). The results of such experiments suggest that there may be an important place for *enhanced passive treatment* in the addressing non-coal mine drainage in England and Wales. Such systems might operate in the same way as conventional passive compost-based units, but with a requirement for periodic dosing with a carbon source or microbial inocula to ensure sustained performance. Given that passive systems require maintenance in any case, such requirements should not incur substantially increased operating costs, particularly if low-cost sources of carbon / microbial inocula can be identified e.g. carbon-rich waste streams, such as dairy waste.

There seems little doubt that in general terms the deployment of compost-based passive systems offers more promise for the remediation of non-coal mine drainage than aerobic wetland systems. Nevertheless, further development work is required to provide the confidence in performance needed before full-scale installation proceeds, not least due to the capital costs involved, as discussed in Chapter 7, below. Particular issues that require resolution include:

- Pilot-scale experimentation to establish performance under field conditions, determine modes of metal removal, and begin to quantify scale-dependence issues
- Identification of potential carbon additives and sources of microbial inocula
- Assessment of the robustness of SRB communities under field conditions
- Determination of area-adjusted removal rates in pilot-scale systems, to carry forward to full-scale design

To this end the installation and monitoring of pilot-scale systems will provide the most useful answers, since these would enable some of the problems of engineering scale to be addressed, would operate under ambient environmental conditions, but would not require the capital commitment of a full-scale system. The ALD and wetland system at the Red River in Cornwall is one such example, as are the ochre drain investigated by Mayes *et al.* (2009) and the ALD system reported by Nuttall and Younger (2000). Plans by the Environment Agency Wales to install a compost-based bioreactor with carbon addition at Cwm Rheidol, Wales, (designed by Newcastle University) will offer significant insight to the potential of such systems, and provide crucial area-adjusted removal rate data as noted above. In selecting sites for the location of such pilot-scale units it is not necessarily recommended that those mine waters at the top end of the priority list of abandoned non-coal mine sites are the focus of attention. The choice should be primarily based on the ease with which the principles of operation of a system can be demonstrated. To this end important considerations are:

- ease of access to the site, because regular visits will be required
- security, because monitoring equipment will need to be left at the site
- ease with which mine water can be transferred to the pilot system, because a consistent supply is essential. To achieve this pumping mine water to the system should be considered, even if the long-term plan is to have gravity drainage
- The water quality of the mine water. Mixtures of metals will complicate interpretation of subsequent results, and elevated suspended solids concentrations may result in problems of pipe clogging.

4.5 Sludge management and metal recovery and reuse

4.5.1. Introduction

A critical problem for the operators of all mine water treatment systems is that of how to manage the metal-rich sludge generated during treatment. For active treatment systems management of metal-rich sludge is a routine operational requirement. For passive treatment systems it tends to be a longer term issue, crucially arising at the point at which the treatment system becomes exhausted. In both active and passive systems the problem is ultimately the same though: metals do not degrade, at least not within timescales that are of any practical use, and therefore metals are retained in the treatment system, and must be managed. In principle there are 3 management options:

1) Dispose of the sludge to landfill
2) Reuse the sludge by first immobilising the metals to a degree that is acceptable to regulatory authorities
3) Recover the metals from the sludge for reuse or resale

4.5.2. Disposal of sludge to landfill

This is the current practice for iron-rich sludge generated at the Coal Authority's coal mine water treatment systems around the UK. The costs involved are substantial. At present the Coal Authority generates approximately 20,000 tonnes of iron sludge (ochre) per year. In 2008/09 landfill disposal costs were £30 / tonne, and therefore the annual disposal cost is approximately £600,000. In 2009/10 landfill costs will increase by approximately 25%, and if this trajectory is maintained disposal of 20,000 tonnes of ochre will cost around £1.2M by 2011/12.

How might this translate into disposal costs for sludge generated from non-coal mine water treatment systems? A key issue will be whether these sludges are classed as *hazardous* or *non-hazardous* wastes. A preliminary estimate for a *non-hazardous* waste can be calculated by using the data shown in Table 2, and in light of current sludge production and disposal costs at the Coal Authority's Horden treatment system. The Horden plant is a High Density Sludge system, and is therefore highly efficient in terms of the volume of sludge produced per unit volume of water treated. The plant treats approximately 100 l/s of water with an iron concentration of 120 mg/L. Effluent iron concentration is routinely less than the consent condition of 10 mg/L, and therefore the daily iron load removed from the water approximates 1 tonne / day. This translates into a sludge mass of 5.5 tonnes / day (and therefore at £30/tonne the annual landfill disposal cost at Horden was ~ £60,000 in 2008/09).

Using the metal loads of the 20 mine waters shown in Table 2 an estimate can be made of the sludge disposal costs if all were to be treated using High Density Sludge systems. Table 8 summarises the daily mass of metals that would be generated. Only the metal with the highest concentration is used for the calculation of metal load in each mine water. Thus, in Table 8 only the iron load is calculated for the County Adit and Dolcoath adit (Table 2), and the loads of other metals in these discharges are not calculated or included in the totals in Table 8. The reason for doing this is that the absolute mass of metal sludge generated is unlikely to be significantly greater in cases where there is a mixture of metal contaminants compared to those where there is a single metal contaminant. Thus, the estimate is conservative in this sense.

Assuming the ratio of mass of metal removed to mass requiring disposal to landfill is the same as that at Horden (i.e. 1:5.5), then the total mass of metal-rich sludge generated at the 20 systems listed in Table 2 would be ~2,600 tonnes / year, and at £30/tonne the disposal cost would be £78,000 / year, rising to £152,000 / year by 2011/12 at 25% annual increase in disposal cost.

Table 8. Total mass of metals for the mine water discharges shown in Table 2

Metal	Mass (kg/day)
Iron	1133
Zinc	96
Manganese	37
Lead	32
Copper	1
Total	**1299**

Note. For each mine water only the metal with the highest concentration is selected for calculation of the total mass; see text for further details.

This may be a conservative estimate because, as illustrated in Tables 2 and 8, in many non-coal mine water discharges there are metals other than iron present, and the landfill charges that these attract may be greater than those for iron, depending on total concentration in the sludge. It is difficult to make any usefully realistic estimate of how this might influence overall costs. Furthermore, this cost estimate does not include transport of the sludge from treatment plant to landfill; this is potentially significant given the remote nature of many of the non-coal mine sites.

It is also difficult to extrapolate these metal masses and landfill costs upwards to cover the potential sludge disposal costs in a scenario in which all non-coal mine waters discharges were remediated. One of the main reasons for this is that there are so few data regarding the flow-rates of these discharges, which is an essential requirement to calculate metal loads.

4.5.3. Sludge reuse

There have been a number of investigations of the possible reuse options for iron-rich sludge from coal mine drainage treatment systems. Hedin (2002) reports on the use of iron oxide for pigment production, one of a number of possible reuse options being explored by Iron Oxide Recovery Inc (*personal communication*, Dr Bob Hedin, Iron Oxide Recovery Inc, 2008). The key aspects of the viability of such an option appear to be twofold: (1) the iron oxide must be of a suitable quality for the pigment manufacturer and (2) the overall economics of the initiative must be beneficial. In regard to the first of these points the physicochemical characteristics of the ochre produced by the Coal Authority's 50 or so treatment plants does vary. A thorough investigation of how this would affect possible reuse options as a pigment has not been conducted, but it seems likely that not all sludges would be appropriate. More importantly in the context of non-coal mine drainage, Kairies *et al*. (2005) report that reuse options become much more limited where trace metals are present in the ochre. In light of the discussion above it is clear that sludges produced from non-coal mine treatment systems will inevitably contain such metals, and in some cases in high concentrations. It seems very likely that this will severely limit options for reuse of non-coal mine drainage solids as a pigment.

Mayes *et al.* (2009) and Heal *et al.* (2005) have investigated the possible utility of iron ochre as a sorbent material in water treatment systems. The use of the material for removal of metals from non-coal mine drainage is discussed in Section 4.4, above. Heal *et al*. (2005) have explored the potential for its use to sorb phosphorus in tertiary sewage treatment, and the results are encouraging. Furthermore, preliminary experiments suggest that the phosphorus-laden ochre may then be applied to agricultural land where it may slowly release the phosphorus as a fertiliser.

In the absence of a direct reuse, it is recommended that the potential for stabilisation of metals should also be explored further. This is particularly the case for non-coal mine drainage sludge, which in all likelihood will contain higher concentrations of potentially eco-toxic metals than coal mine drainage sludge. Stabilisation technologies may be used simply to reduce costs of landfill (because of the reduced leaching of metals into aqueous phase) or possibly to produce a useable product, such as building materials, if the metal(s) can be shown to be suitably immobile. Recent reviews of cement-based stabilisation (Chen *et al*., 2009) and geopolymerisation (Komnitsas, 2007) suggest that such technologies may be some way from being accepted by industry, in part due to some outstanding questions regarding the stability of the materials depending on the metal content of the waste.

Clearly therefore further work would be required to establish whether such approaches are of any potential for managing waste from mine water treatment systems.

With all of these reuse options, and particularly those where the material in which the metals are bound is used in the environment (e.g. as building materials, applied to agricultural land), the barriers to implementation may relate as much to regulatory issues as technical development needs.

4.5.4. Metal recovery

Cost-effective and environmentally acceptable direct recovery of metals from aqueous solutions, or subsequent recovery of metals from waste sludge following treatment, would be the ideal solution to problems of environmental management of abandoned mine effluents. For metal hydroxides or sulphides to be suitable for processing in a smelter they must be of high purity (at least 90%). Many non-coal mine discharges contain a mixture of metals, and the need for this level of purity of metal salt means that each metal must be recovered from aqueous phase separately (usually in a particular sequence). The geochemical properties of metal hydroxides and sulphides make this technologically feasible. Specifically, because the solubility products of individual metal salts are different (as shown in Table 5) careful control of pH and S^{2-} concentrations permits selective removal of metals. There are a variety of means of accomplishing this, but they are, without exception, active treatment technologies, and therefore attract the high costs of such operationally intensive systems. The reason no passive technologies are capable of selective recovery of metals at high purity is that the careful control of biogeochemical conditions required is simply not possible.

At its simplest, selective recovery of metals comprises careful addition of alkali (lime or caustic soda) to raise pH to the level required of the target metal, removal of that metal by sedimentation, and then elevation of the pH again to target the next metal, which is removed in a separate sedimentation tank (and so on for each metal). Various other chemical processes have been investigated (and used) for the same purpose, including (individually or in combination with other units) membrane processes, flotation, sorption, electrochemical techniques, reverse osmosis and various ultra-filtration techniques. However, all are very capital intensive. For this reason there has been an increasing interest in the potential application of bioprocesses for metal recovery. A comprehensive review of the various configurations has recently been completed by Kaksonen and Puhakka (2007). Some of these have been both patented and applied at mine water sites. These include the BioSulphide process® (see Rowley *et al*., 1997), and the Thiopaq® process (Boonstra *et al.*, 1999). Potentially interesting new developments in this area, both only at lab-scale to date, include the biosulfidogenic selective recovery of metals at low pH described by Johnson *et al.* (2006), and the application of fuel cell technology to mine water treatment, which may offer the potential to generate electricity and treat mine water simultaneously (e.g. Cheng *et al.*, 2007).

Whilst some of these processes have been applied to the remediation of abandoned mine water pollution, and recovery of metals therein, in many instances they are actually targeted for use by the mining and metallurgical industries, where ongoing revenue streams are more likely to make the installation of cost-intensive water treatment and metal recovery systems feasible. Not only is the labour to operate such systems already available at active sites, but in many cases there are also smelting facilities available to process the recovered metals. This raises one of a number of potential problems with the application of metal winning technologies to the abandoned non-coal mine sites of England and Wales:

1) *Availability of suitable processing facilities for metals recovered*: the last zinc smelter in the UK, the Britannia zinc works at Avonmouth, near Bristol, closed in 2003, and the only lead smelters in the UK are for secondary lead i.e. batteries recycling. Thus, there are no smelting facilities available in the UK.
2) *The mass of metal recovered from individual sites*: although a significant environmental problem, the amount of metal potentially recovered from abandoned non-coal mines is very small in metallurgical industry terms.
3) *Location of mine sites*: many of the most significant abandoned non-coal mine waters discharge in remote, upland areas of the UK, many of which are protected by conservation designations. Even excepting the costs of transport of recovered metals to a metal smelter (overseas), the installation of a power supply to operate the active treatment processes required would be costly in itself, and locating an active treatment system in such areas would likely meet with substantial opposition.

On this basis the prognosis for the potential of metal recovery at abandoned non-coal mine sites is clearly not promising. However, this should not preclude a more thorough assessment of the possibilities. A complete evaluation should take account not just of the economics of metal recovery from mine water discharges, but also of the environmental benefits (in financial terms if possible) of *not* disposing of waste treatment sludge to landfill sites. Neither should such an assessment necessarily focus just on point sources of mine water; the total flux of metals from formerly mined catchments, including that in temporary storage in stream bed sediments, may be substantially greater than the sum of point sources, therefore improving the viability of a metal recovery strategy.

4.6 Diffuse pollution remediation

Diffuse pollution has already been identified as a significant problem in abandoned non-coal mine districts of England and Wales (see *Identification and prioritisation of abandoned non-coal mines – the national picture* report). Diffuse pollution is a confirmed or suspected contributor to total metal flux to streams and rivers at 187 sites across England and Wales. In 117 of these cases exposed spoil heaps / tailings are cited as the likely source of this diffuse pollution. In a further 38 cases no clear source is evident, but water quality or metal loading data is used as evidence of diffuse pollution i.e. changes in water quality or loading cannot be attributed to known point sources of pollution. For the mine waters listed in Table 2, 14 out of 20 have confirmed diffuse pollution problems. In all likelihood there will actually be more sites than those identified here at which diffuse pollution is a problem, and this is simply a function of the difficulty of quantitatively diagnosing the problem, as discussed in Section 2.2 of this report. Dedicated investigations of diffuse mining-related pollution (e.g. Gozzard, 2008; Mayes *et al.*, 2008) clearly indicate that diffuse sources may be a major contributor to absolute metal loads to rivers (> 90% of the metal load under certain hydrological conditions). Therefore diffuse sources of pollution cannot be disregarded if the overall aim of environmental management of river catchments is to achieve 'good chemical and ecological status'.

Much has been written on approaches to the remediation of diffuse pollution, though the focus to date has principally been in the context of urban drainage and agricultural diffuse pollution. Arguably the most comprehensive text on the subject is that of Novotny (2003), though many others exist. Mining-related diffuse pollution has its own unique features, of which it is clearly necessary to be cognisant when addressing possible remediation options. Nevertheless, some of the technologies applied to treatment of diffuse sources of urban and agricultural pollution may be transferrable to mining pollution remediation.

Given that some 60% of all cases of diffuse pollution reported here arise from spoil heaps, it is reasonable to conclude that the transport of contaminants, likely as metals associated with sediment, is either as surface runoff, erosion of waste material forming artificial river banks, or shallow subsurface flow to the toe of the spoil heap (or some combination of the three). In the case of surface runoff two broad options are available for control or treatment:

1) Capping of spoil heaps with impermeable material, to prevent surface runoff
2) Installation of some form of riparian interception and treatment system

Capping spoil heaps, usually with a clay layer and topsoil cover to encourage vegetation growth, can be an effective means of preventing polluting runoff, limiting infiltration and therefore generation of polluted groundwater, and concomitantly rehabilitating the heap following any necessary re-grading and stabilisation works. It can therefore be a highly effective remediation strategy. Although it may be feasible at some abandoned non-coal mine sites there are a number of potential barriers:

- *Cost*: spoil capping is a major earth moving and import exercise, which is consequently highly costly
- *Topography*: as indicated in Figure 5, the very steep topography of mine waste material in many locations may render re-grading works prohibitively expensive, and / or present insurmountable engineering difficulties with respect to capping operations
- *Geological / mineral / heritage interests*: sites such as Parys Mountain attract considerable interest from mineral enthusiasts and mining historians, bodies which might strongly oppose such capping works. In addition, many spoil heaps have conservation designations such as *Site of Special Scientific Interest (SSSI)* which may preclude capping.

Figure 5. Steeply inclined and unstable mine waste at (A) Cwm Rheidol and (B) Cwm Ystwyth, both in Ceredigion, Wales

In some abandoned non-coal mining districts waste material immediately abuts streams and rivers, effectively forming the river bank. The result is potential diffuse pollution of the water course due to bank erosion. Figure 6 illustrates engineering works targeted at limiting such problems. Along this stretch of the River Nent, Cumbria, rock-filled gabions and additional rock slabs have been used to stabilise the river banks, and the waste heaps immediately adjacent to the river have been capped and vegetated. These works, which extend for a distance of approximately 1 – 2 km, had a reported capital cost of approximately £3 million. Although, unfortunately, no information is available about the resulting improvements in water quality, visual inspection by the authors indicate the intervention has been effective in minimising bank erosion.

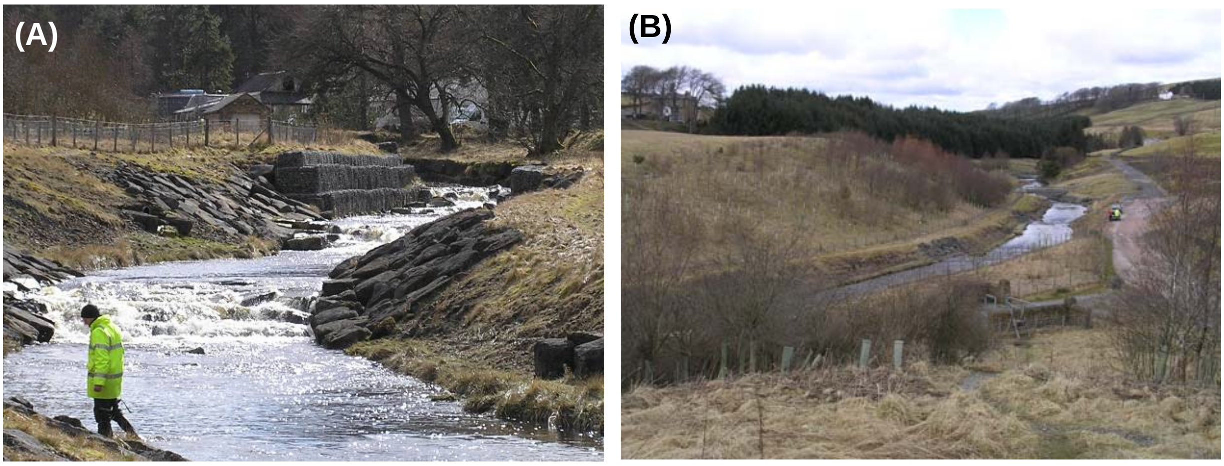

Figure 6. Use of gabions to stabilise banks of the River Nent, Cumbria (A) and re-grading, capping and vegetation of spoil material along the same stretch of river (B)

Where surface runoff is confirmed as the principal pathway of pollutants to water courses, there are a variety of interception and treatment technologies that may, probably with modification, be effective in remediating diffuse non-coal mine water pollution. A variety of filter strips, swales, grassed waterways and infiltration trenches have been proposed and / or applied for other types of diffuse pollution (Novotny, 2003). Essentially all serve the same purpose insofar as they are installed in riparian zones parallel to water courses, to intercept surface runoff, and in some cases render some degree of treatment. In principle surface runoff intercepted by an infiltration trench, or similar, could allow collection of diffuse surface runoff such that it could then be treated as a point source, but gravity treatment of drainage in riparian zones is clearly problematic, and therefore pump-and-treat options might need to be considered.

Appropriately engineered, grassed swales may offer an opportunity to simultaneously intercept and treat surface runoff from mine waste. Figure 7 shows a schematic layout of such a system. Surface runoff is intercepted in a linear depression in the riparian zone, and water then drains into a drain pipe contained within a permeable gravel bed. The ‘active soil layer’ indicated in Figure 7 could be constructed so that it encourages the metal attenuation processes discussed in Section 4.2 of this report. The potential of such systems would require further investigation. In particular, the difficulties of passive remediation of metals such as zinc and cadmium, discussed earlier, would need to be addressed, as they would apply equally here as to ‘conventional’ passive systems. In addition, a clear understanding of the volumes of water being intercepted would be required, in order to design systems of an appropriate size.

The coarse nature of the material that makes up mine waste is such that infiltration of water to the subsurface is a common occurrence. Therefore polluted shallow groundwater is an issue at some mine sites, and clearly the possible options for remediation described above are inappropriate in such cases. Permeable Reactive Barriers (PRBs) have been applied to good effect to remediate acidic and metal-rich shallow groundwaters (e.g. Jarvis *et al.*, 2006), and Environment Agency guidance is available on the design of such systems (Carey *et al.*, 2002). Such systems are only likely to be cost-effective where the polluted groundwater is relatively shallow (e.g. 3 – 4 m maximum) due to the substantial excavation requirements for anything deeper. To ensure that all water passes through the PRB it needs to be keyed into impermeable material, such as clay, at the base of the system. If low cost manures can be identified at sites for use as the reactive media, then continuous wall configurations are likely to be more cost-effective than funnel-and-gate PRB systems (see Carey *et al.*, 2002 for a detailed discussion of the configurations of PRBs). It will be clear from these requirements and comments that investigations of the lateral and vertical extent of the polluted groundwater plume, as well as characterisation of the aquifer material, are required to establish the exact engineering requirements.

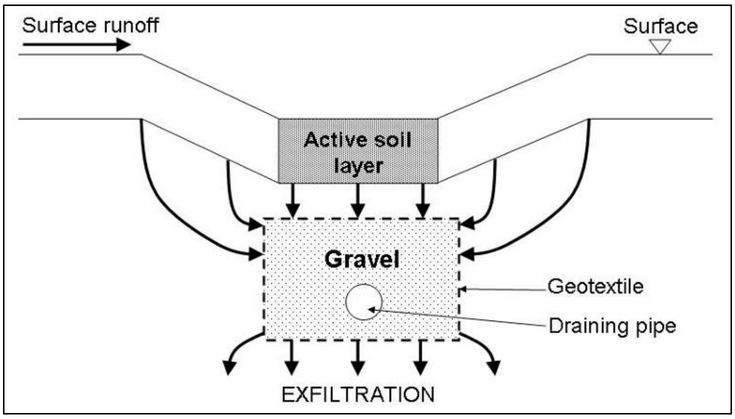

Figure 7. Schematic illustration of a grassed swale, which potentially could be applied to the remediation of diffuse surface runoff from abandoned non-coal mine waste spoil heaps (after Novotny, 2003)

Characterisation of diffuse pollution in water bodies affected by abandoned non-coal mine water pollution (see Section 2.2) is an essential prerequisite to decision-making on the type of engineering solution to address such problems. Even where such characterisation exercises have been undertaken, a specific objective of such investigations has not been a detailed consideration of the approaches to, and constraints upon, remediation of diffuse pollution. It is therefore difficult to be anything other than generic about the remediation options in the discussion above. For this reason it is recommended that more detailed feasibility / research studies are undertaken of diffuse pollution remediation options at some of the key sites at which this is considered an issue e.g. Frongoch, Wales; Barney Craig, Northumbria (see Table 2). This in turn should inform wider application of such approaches at other sites.

5. Stakeholder engagement

Reference has already been made, in Section 2.2.2, of discussions with stakeholders to assist with scoping studies. In the overall pursuit and delivery of a remediation strategy there is no question that regular liaison with stakeholders is critical. However, such involvement needs to be balanced against the specific objectives of the line of enquiry being made. The authors of this report have reviewed a number of reports for the Environment Agency which purport to resolve mass flux issues at abandoned non-coal mine sites, but in fact fail to do so. One of the reasons for this appears to be that in many cases the organisation conducting the investigations are asked to undertake stakeholder liaison within the same brief (e.g. the locations of conservation and heritage sites). The result is that clear, concise, guidance on the sources and fate of pollutants is at best lost in amongst a lengthy discussion of stakeholder issues, or at worst is totally absent (the inclusion of discussions of stakeholder issues is not the only cause of this type of problem, but our experience suggests it certainly contributes to it).

Stakeholder engagement is essential, but it is strongly recommended that such liaison is appropriately targeted / phased during investigations. During scoping studies the primary objective is to clearly identify and quantify the source and fate of pollutants, and such matters are complex enough without adding additional deliverables. Far from being cost-effective (insofar as multiple issues at the same site are addressed at once), the evidence from reports to date is that this approach is actually counter-productive, since the work on the source and fate of pollutants simply has to be repeated if it is not done properly.

It is therefore recommended that stakeholder engagement occurs at a number of points during the overall timeframe of a remediation project, as appropriate to the task at hand. These are identified in the overall timeline shown as Figure 8, and are summarised in Table 9.

Table 9. Recommended phased approach to stakeholder engagement

Stakeholder engagement activity	Timing
Consultation with local mining experts (individuals and groups) on source and nature of non-coal mining pollution.	Scoping phase
Investigation of diverging and converging issues e.g. heritage, conservation concerns	Feasibility phase
Discussions with planners and local residents of likely configuration of treatment system	Conceptual design phase
Presentation of system design and operation to local residents and planners	Detailed design phase

Note. See Figure 8 for exact details of timing.

6. Recommended approach to managing the environmental impacts of abandoned non-coal mines at the catchment scale

The recommended approach to the environmental management of abandoned non-coal mines across England and Wales is presented here. Based on the long mine water management experience of this consortium, Figure 8 presents an indicative chronology of activities aimed at remediating non-coal mine drainage. The Gantt chart shown illustrates a typical timeframe, from initial scoping studies aimed at defining problems at a site right through to completion of a full-scale treatment system. The timeline is based upon that currently used by the Coal Authority, and is informed by that organisation's long experience: construction of some 50 full-scale treatment systems over the last 15 years. However, it has been modified somewhat in response to the particular problems at non-coal mine sites.

The focus of the previous 3 chapters of this report is intentionally the first 4 tasks on the Gantt chart (scoping phase through pilot-scale testing), and Chapter 5 has briefly discussed Stakeholder engagement (task 11). Although complex issues in themselves, detailed design and construction, and land and planning requirements, will be very similar to those required for design of coal mine water remediation systems once a decision has been made about the actual system type. Such matters are therefore not discussed here.

It is critically important to appreciate that the durations of each phase given are *indicative only*. In practice, for a specific site, some phases may be longer or shorter due to particular issues at the site. The scoping phase may be shorter for sites at which mine water dynamics are comparatively straightforward (e.g. a single point source), or longer at particularly complex sites which are subject to significant variations under varying hydrological conditions. The duration of pilot-scale testing may be significantly shortened if there is confidence in the technology being applied, but even with established processes such as high density sludge (HDS), pilot-scale testing to optimise treatment efficiency is recommended.

Notwithstanding these *caveats*, the Gantt chart usefully serves the purpose of illustrating the chronology of tasks that require addressing during the management of abandoned non-coal mine water pollution. Importantly, it also reveals that the overall duration from commencement of an investigation to completion of a full-scale treatment system may typically be around 5 years in total.

Figure 8. Indicative timeline for investigation, design, and construction of treatment systems for abandoned non-coal mine discharges (see text for important *caveats* regarding these timescales)

			Year 1				Year 2				Year 3				Year 4				Year 5			
ID	Task name	Duration	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	Scoping phase	15 months																				
2	Feasibility phase	8 months																				
3	Conceptual design phase	6 months																				
4	Pilot-scale testing	12 months																				
5	Detailed design phase	7 months																				
6	Construction phase	6 months																				
7	Site operational	-																				
8	Land requirements	3 months																				
9	Land deal	18 months																				
10	Planning Process	10 months																				
11	Stakeholder engagement	-		A				B			C							D				

Notes:

1. **Scoping phase**: Detailed requirements of the 3 stage approach recommended in this report are discussed in Section 3 of the report.
2. **Feasibility phase**: Critical to collect sufficient reliable data for purposes of design. See Sections 3 and 4 of the report for discussion
3. **Conceptual design phase**: Expertise in mine water treatment biogeochemistry and engineering essential. A discussion of the underpinning principles and options is provided in Section 4.
4. **Pilot-scale testing**: It is strongly recommended that pilot-scale testing is undertaken for any treatment option, and it is essential for novel passive systems. Note that issues of engineering scale will need to be addressed when moving to full-scale design (see Section 4.4 for further discussion). Again, appropriate expertise is crucial.

11. **Stakeholder engagement**: See Section XX for discussion.

    A. Consultation with local mining experts (individuals and groups) on source and nature of non-coal mining pollution.

    B. Investigation of diverging and converging issues e.g. heritage, conservation concerns

    C. Discussions with planners and local residents of likely configuration of treatment system

    D. Presentation of system design and operation to local residents and planners

7. Indicative costs for remediation of non-coal mine water discharges in England and Wales

In preparing these reports the project consortium was asked to include an estimate of the costs for remediation of abandoned non-coal mine water pollution across England and Wales. Providing an estimate of total costs for remediation of a widespread problem such as mining pollution, for an entire country, is fraught with very real potential for extremely large errors. To give an indication of the difficulty of such estimations, the Coal Authority typically spends 18 – 24 months conducting detailed investigations of a *single discharge* before accurate predictions of construction costs for a full-scale treatment system are determined. In contrast, in the tables below we estimate the total cost of remediation of *more than 250 discharges*, with some information about approximately 10% of these discharges, and virtually no information about the remaining 90%. The estimates below must therefore be viewed with considerable caution. Some more specific qualifications are listed in the paragraphs below.

The estimates are based on the substantial practical experience of this consortium in design, construction and operation of mine water treatment systems, completion of scoping and feasibility studies of mining-related pollution, and execution of a large variety of civil engineering projects. Having been tasked with providing such an estimate, the costs have been broken down to a reasonable level of detail, partly because there is greater confidence in some of these figures than others (see below) and partly for the simple reason that a single 'bottom line' figure, without any justification, is of no practical use if the figures are to be refined, and the estimate improved, over time. It is certainly anticipated that these figures will be amended over the coming years, not least as the abandoned non-coal mines database is populated with more data.

Table 10 provides a generic cost estimate for the remediation of an abandoned non-coal mine water discharge. Because of uncertainties relating to (a) the complexity of preliminary investigations required, (b) the type of treatment that would be required (e.g. active or passive), and (c) the system's absolute size and engineering requirements specific for a particular discharge, lower and upper estimates are provided. The same approach has been adopted in Table 11. In Table 10 costs have been broken down into (i) investigation, (ii) construction and (iii) operation. 'Investigation' refers to scoping studies through to detailed design (tasks 1 – 5 in Figure 8). In essence the lower limit assumes a simple scoping study (e.g. for a single point source discharge), remediated with a modest passive treatment system, whereas the upper limit would be more in line with a complex, multi-source scoping phase, treated by a fully active treatment system. The costs indicated in Table 10 are based on actual capital investment figures for scoping investigations completed to date (by the Coal Authority and Environment Agency), and on the construction and operation costs of actual coal mine water remediation systems operated by the Coal Authority. Although the actual mechanics of treatment of abandoned non-coal mine water discharges may be different to coal mine water systems, the major elements of construction cost are likely to be similar. Therefore although the differences between upper and lower estimates in Table 10 may be substantial, there is reasonable confidence that these are genuinely indicative figures, at least when calculated at present day costs, as they are here.

One noteworthy point on the issue of present day costs is the operation costs of mine water treatment systems. In Table 10 we have calculated costs for a 10 year life cycle, extrapolating using the present day cost of operation shown for all 10 years. This figure could therefore be subject to variation if some form of economic model was used to project actual future costs. A 10 year life cycle has been selected as this is a conservative estimate of the lifetime of a passive treatment system (see Jarvis and Younger (2006) for further discussion). The figure given for a 20 year life cycle of a passive system does not include any costs for rehabilitation of the system, though arguably it should. In the worst case scenario the system would require total replacement, and therefore the 20 year life cycle cost would be £1.7M rather than the £1.2M as indicated (i.e. an addition of the original construction cost). In reality this is not usually necessary, and rehabilitation of a passive system does not generally necessitate a complete re-build. However, the figure for the 20 year life cycle for the lower limit cost should still be viewed as conservative.

At abandoned non-coal mine sites diffuse pollution, and a variety of other hazards may be present. In Table 11 we have endeavoured to estimate the costs of addressing a number of these issues. These costs will be subject to very substantial variation on a site-by-site basis.

Tables 10 and 11 present costs for a single issue e.g. a single mine water discharge, a particular hazard in a water body. In order to extrapolate to a national cost estimate from these figures some key data from the *Hazards and risk management at abandoned non-coal mine sites* report are reproduced in Table 12. Using these numbers, Table 13 shows the calculation of the estimated total cost of remediation of abandoned non-coal mine water pollution across England and Wales: ~ £370M for an initial 10 year period. Clearly there is potential for very substantial margins of error in this figure, though derivation of such a number by any other means of calculation would also be open to such criticism.

In addition to the qualifications about the accuracy of these figures already detailed, the following comments and *caveats* should be borne in mind when reviewing Tables 10 – 13:

- An economic model has not been used to adjust costs into the future; all costs shown are at present day costs.
- Landfill costs are likely to increase substantially in coming years, as are reagent costs for active treatment. Therefore operating costs in general, and sludge disposal costs in particular, are likely to increase in coming years.
- The upper limit of (active) treatment system construction cost of £2.5M is not actually the highest recorded for a UK mine water treatment system; the Wheal Jane (Cornwall) and Dawdon (Durham) system costs were substantially higher (in part due to local planning requirements at Dawdon), but it is not anticipated that any treatment system for a future abandoned non-coal mine discharge would be at this scale.

- No costs are included in Table 13 for remediation of problems relating to airborne pollution risk, stability and safety, and / or animal and public health. The reason is that there are deep uncertainties about the reliability of the data returns on these issues. Specifically, some respondents appear to have defaulted to an affirmative answer.
- There is enormous potential variation in both investigation and remediation measures for outbreak risks (Table 11), and therefore the figures should be treated with extreme caution. The figures shown are based on 3 examples:
  1. The outbreak of coal mine water at Sheephouse Wood, Yorkshire, which cost the Coal Authority £150,000
  2. Dealing with the outbreak risk at Parys Mountain in 2003[3], which cost approximately £500,000.
  3. Estimated costs for addressing the outbreak risk at Force Crag, Cumbria, which are more than £600,000.
- The costs in Table 13 are generated from the numbers in Table 12. However, the figures in Table 12 are likely to change over time, as the database will be updated in the future. In particular, we have not included in Table 12 the additional 81 sites at which mine water discharges are suspected. Changes to number of discharges will of course result in a change to the cost estimates for remediation.
- We have estimated that 10 discharges out of the current total will require substantial investment in active treatment systems. Given the very high cost of such systems, clearly even small changes to the number of such systems will have a large impact on the bottom line cost.
- Diffuse pollution remediation cost estimates in particular are highly uncertain, largely due to the great paucity of quantitative data about the scale of the problem.
- Some figures in Tables 10, 11 and 13 are rounded up or down for convenience.

[3] The works involved removal of an underground dam and pumping and discharge of 270,000 m^3 of highly acidic mine water to surface

Table 10. Indicative lower and upper estimates of the costs of preliminary investigation, design, construction and operation of an abandoned non-coal mine water treatment system

Item	Lower estimate (£)	Upper estimate (£)
Scoping phase	15,000[A]	50,000[B]
Feasibility phase	25,000[A]	50,000[B]
Conceptual design	10,000	30,000
Pilot-scale testing	50,000	100,000
Detailed design	35,000	75,000
Sub-total (investigation):	**£135,000**	**£305,000**
Construction	500,000[C]	2,500,000[D]
Sub-total (construction):	**£500,000**	**£2,500,000**
Annual operation	30,000[C]	750,000[D]
Sub-total (operation):	**£30,000**	**£750,000**
First life cycle (10 years)	**£935K**	**£10.3M**
Two life cycles (20 years)	£1.2M	£17.8M

Notes:

A: Assumes a single point source

B: Assumes multiple sources / pollution issues

C: Assumes modest sized passive system (figure based on Coal Authority data for passive system costs)

D: Assumes substantial active treatment system (figure based on Coal Authority data for Horden HDS plant, County Durham), and includes sludge disposal to landfill

Table 11. Preliminary indicative estimates of costs associated with engineering works at abandoned non-coal mine sites to address problems of outbreak risk, diffuse pollution, and other hazards

Item	Lower estimate (£)	Upper estimate (£)
Mitigation of outbreak risk	150,000[A]	750,000[B]
Bank stabilisation to mitigate diffuse pollution	160,000[C]	800,000[D]
Diffuse pollution remediation e.g. PRB, swale	150,000[E]	1,000,000[F]
Waste capping works	500,000[G]	2,000,000[G]

Notes:

A: Actual costs of mitigation following Sheephouse Wood coal mine water outbreak

B: Based on 3 examples (see text) but still an estimate only since very much dependent on scale of outbreak e.g. mitigation / remediation of outbreak of severity of Wheal Jane would certainly run to cost measurable in millions

C: Calculated on basis of £80,000 per 100 m length, a cost calculated based on civil engineering experience of Atkins Ltd. Lower limit assumes 200 m length requiring remediation

D: Calculated as above but assumes 1 km stretch requires remediation

E: Small treatment area with limited engineering works

F: Large treatment area with substantial engineering works

G: Very rough, indicative estimates only

Table 12. Abridged summary of numbers of environmental impacts and risks from abandoned non-coal mines across England and Wales

Risk	Number
Confirmed mine water discharges	257
Documented evidence of outbreak risk	19
Water bodies in which there is evidence of diffuse non-coal mine water pollution	112
Instances where there are concerns about airborne pollution, stability and safety, and / or public and animal health	425

Note. See *Hazards and risk management at abandoned non-coal mine sites* for more complete information.

Table 13. Indicative estimated cost, over a 10 year life cycle, to remediate water-related environmental problems at abandoned non-coal mines in England and Wales (*see very important qualifications in text about these figures*)

Item	Number	Unit cost	Total[A]
Mine water treatment – worst discharges	10	£10.3M[B]	£103M
Mine water treatment – other discharges	247	£935,000[C]	£231M
Outbreak risk mitigation	19	£150,000[D]	£3M
Diffuse pollution remediation	112	£310,000[E]	£35M
Total			**£372M**

Notes:

A: Figures are rounded
B: Upper estimate for first life cycle cost in Table 10
C: Lower estimate for first life cycle cost in Table 10
D: Lower estimate for mitigation of outbreak risk, shown in Table 11
E: Sum of lower estimates for bank stabilisation to mitigate diffuse pollution (£160,000) and diffuse pollution remediation (£150,000), shown in Table 11.

8. Conclusions and recommendations

The *Prioritisation of abandoned non-coal mine impacts on the environment* project has generated the most definitive evaluation to date of the impacts of, and risks to the water environment from abandoned non-coal mines across England and Wales. Application of the methodology developed to prioritise water bodies has, for the first time, provided the environmental regulator with an objective assessment of where pollution from these mines has the highest impact, and where there is the greatest risk that water bodies (river stretches) will fail to meet the objectives of the Water Framework Directive due to abandoned non-coal mines. The specific water bodies which should be the focus of immediate attention in River Basin Management Plans (RBMPs) have been identified, and the work needed to address mining pollution through both research into passive treatment technologies and catchment monitoring investigations is outlined.

Having rolled-out the abandoned non-coal mine prioritisation methodology to all the River Basin Districts of England and Wales it is clear that the underpinning logic and operation of the methodology is sound. The details of the methodology itself are provided in Report I: *A methodology for identification and prioritisation of abandoned non-coal mines in England and Wales*.

By assessing water bodies using water quality, ecological, groundwater and higher impact metrics it has proved possible to prioritise *Impacted* and *Probably Impacted* water bodies into ranked lists. The primary focus of the project throughout (at the request of the project sponsors) has been the impacts of polluted water discharges from abandoned non-coal mines to surface streams and rivers. Nevertheless, additional information collated and stored in the database enables environmental managers to assess what the other issues are at these sites, such as safety issues, outbreak risk and stakeholder concerns. Taken together this provides a valuable resource to assist in the long-term remediation planning at polluting abandoned non coal mine sites in England and Wales.

The absolute scale of environmental problems associated with abandoned non-coal mines has been thoroughly assessed for the first time as part of this project, and Table 14 summarises these impacts and risks.

Table 14. A summary of the impacts and risks associated with abandoned non-coal mines in England and Wales, and the frequency of their occurrence

Risk	Number
Water bodies *Impacted* from non-coal mine water pollution	226
Water bodies *Probably Impacted* from non-coal mine water pollution	243
Confirmed mine water discharges	257
Suspected mine water discharges	81
Documented evidence of outbreak risk	19
Water bodies in which there is evidence of diffuse non-coal mine water pollution	112
Instances where there are concerns about airborne pollution, stability and safety, and / or public and animal health	425

This report on the *Future management of abandoned non-coal mine water discharges* provides *generic* recommendations on how to manage abandoned non-coal mine drainage problems, with *specific* guidance on aspects that have not previously been addressed in published guideline documents on mine water pollution (such as the passive treatment guidance by the PIRAMID Consortium (2003)).

The main overall conclusions and recommendations regarding the future management of abandoned non-coal mines, and specific recommendations arising from this particular report, can be summarised as follows:

- To manage water bodies impacted by abandoned non-coal mine water pollution it is vital to have a clear understanding of the exact sources of pollution if an effective remediation programme is to be instigated. In some instances a single source of non-coal mine water pollution is clearly the main problem, but in the majority of water bodies there are multiple sources, and diffuse sources may play an important role in the overall flux of contaminants to receiving water courses. If remediation measures are implemented without understanding the overall pollution dynamics in the catchment the environmental objectives for water bodies that are set out in RBMPs may not be achieved despite potentially substantial expenditure on engineering works and mine water treatment systems.
- Previous research, and additional information presented here, suggests that diffuse sources of mine water pollution are a major contributor to overall metal flux in abandoned non-coal mine catchments.
- Although we sometimes refer to 'priority for remediation' and 'priority for further data collection' for *Impacted* and *Probably Impacted* water bodies respectively, the reality is that additional monitoring programmes will be a necessity at almost *all* of the water bodies in which non-coal mine drainage is identified as an issue. This is because data collection programmes to date have either not been systematic enough to characterise metal fluxes in water bodies, or have not been appropriately targeted to facilitate the design of a treatment system (or both).
- There are actually very few water bodies for which there is a clear quantitative understanding of how individual sources of pollution from abandoned non-coal mines contribute to the overall metal flux in that water body. In many instances it appears that all of the sources, especially where they are diffuse in nature, have not even been identified.
- A national programme of systematic scoping studies of water bodies impacted by abandoned non-coal mines is therefore recommended, and detailed guidance on how to carry out such investigations is provided in this report. It is recommended that a 3-stage approach to scoping studies is adopted, and that such studies should be conducted over at least a 12 month period in most water bodies.
- The problems evident at abandoned non-coal mines are multifarious and complex. A chronology of investigative activities for tackling the problems is therefore proposed. This sets out the specific requirements, in a logical chronological order, of investigations of water pollution problems arising in water bodies in abandoned non-coal mine districts. According to this

chronology it is estimated that it will take approximately 4.5 years to complete an individual remediation scheme, from commencement of a scoping study to completion of a full-scale treatment system.

- The chronology is broadly based upon that currently used by the UK Coal Authority. A significant addition is the inclusion of pilot-scale mine water treatment (over a recommended period of 12 months), which reflects the developmental nature of passive treatment systems for remediation of metals such as zinc, cadmium and lead.
- Stakeholder engagement is an essential aspect of the overall management of environmental problems in abandoned non-coal mine water bodies. It is recommended that this liaison is undertaken in a phased approach.
- The passive mine water treatment technologies that have been applied with great success to the remediation of coal mine drainage (principally for the removal of iron) will not work to anything like the same degree for the metals in non-coal mine drainage (e.g. Zn, Cd). These metals are far more mobile than iron and so it is more difficult to remove them from the mine water. Such systems will fail entirely if the loadings of metals such as those found in non-coal drainage are simply substituted into the equations used for the successful design of systems for the removal of iron.
- Effective passive treatment of non-coal mine drainage to consistently meet Environmental Quality Standards (EQS), within a practical land area, is a subject of ongoing research. There are many active treatment technologies that could remediate non-coal mine drainage to the standards required to meet EQS, but they come at a high cost, and in many of the locations of major non-coal mine water discharges it appears unlikely that they would be acceptable developments.
- Passive treatment technologies therefore require further investigation. Particular avenues of enquiry include:
    - Pilot-scale experimentation to establish performance of BSR-based systems under field conditions, to determine modes of metal removal, and begin to quantify scale-dependence issues
    - Identification of potential carbon additives and sources of microbial inocula for such systems
    - Investigations of the potential for deployment of *enhanced passive treatment* systems in general
    - Assessment of the robustness of SRB communities under field conditions
    - Determination of area-adjusted removal rates in pilot-scale systems, to carry forward to full-scale design
    - Further investigation of other passive systems that rely on attenuation processes other than BSR, and in particular the potential for the use, and especially regeneration, of sorbent materials

- Irrespective of the type of technology, the management of the metal-rich sludge arising from the treatment of non-coal mine drainage remains a problem. Only active treatment technologies currently offer the possibility of recovering metals in sufficient purity that they *might* be recycled, but even for active systems it currently seems unlikely that recycling of metals from abandoned non-coal mine water treatment will be economically viable. There may be reuse options for metal-rich media recovered from mine water treatment systems, but these need further investigation.

- There are other problems associated with former non-coal mining districts besides mine water pollution, albeit in some cases these issues may contribute to problems of water pollution. In some cases issues such as stability concerns, safety, airborne pollution, and other human and animal health risks, may be significant, and should therefore be addressed accordingly. The level of detail of information provided with respect to these issues has been very varied. Although there are clearly important specific issues relating to these aspects of abandoned non-coal mines that need to be addressed (e.g. stability concerns at specific sites), the main conclusion of this project is that there needs to be a systematic national approach to the assessment of such problems. As well as identifying the most important problems to address, this will directly serve the requirement in the EU Mining Waste Directive to create an inventory of closed mine waste facilities causing harm to human health or the environment.
- Conducting thorough investigations of environmental problems in abandoned non-coal mining districts can be expensive. This cost is minor, however, compared to that of the design, installation and operation of systems to remediate such pollution problems. The total cost to remediate all of the water-related environmental problems associated with abandoned non-coal mines that have been identified as part of this project is estimated to be approximately £370 million over an initial 10 year period, at present day costs, with additional subsequent operating costs. Of this total around 90% is apportioned to mine water treatment, and 10% to mitigation of outbreak risk and diffuse pollution problems. Treatment systems are likely to be required to operate in perpetuity. There are considerable uncertainties regarding the accuracy of this estimate, due in large part to a paucity of quantitative data on abandoned non-coal mine environmental problems (especially relating to mine water discharge flow and volume)

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