Evaluating the potential impact of opencast coal mining on water quality (Groundwater Regulations 1998)

An assessment framework for Scotland

March 2004

Prepared for the Scottish Environment Protection Agency (SEPA)

by

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Executive Summary

The European Directive on Groundwater (80/68/EEC) was fully transposed into Scottish law by the introduction of the Groundwater Regulations 1998. These regulations forbid the introduction of certain substances (denoted as "List I substances") into groundwater, and also place limitations on the extent to which other substances ("List II substances") may be permitted to enter groundwater. Scottish opencast mining, which constitutes an 'activity' under the Groundwater Regulations, poses little risk of introducing List I substances into groundwater, but it has substantial potential to lead to the migration of several 'List II' substances. A code of practice recently published by the Scottish Executive (2003) provides sound advice on the control of pollution from non-mineral pollution sources during opencasting operations. It also identifies the need for the "assessment of the risk of release of List I and List II substances from storage in stockpiles or backfilling, prior to the activity being undertaken". The present document has been prepared to satisfy the need for an assessment framework for List I and II pollutants released by weathering of soils and rocks disturbed by opencast coal mining.

Although this assessment framework assumes the user is familiar with opencast coal mining as it is currently practised in Scotland, a brief overview of opencast coal mining techniques is provided in Annex I in order to ensure the document is accessible to non-specialists. The development of the assessment framework has been founded upon a comprehensive state-of-the-art review of the key issues relating to the potential release of List I and II substances from soils and rocks disturbed by opencasting. This review is presented in full in Annex II. The principal topics which it addresses are:

- The geochemical processes responsible for pollutant release and attenuation in coal-bearing strata affected by opencast mining, including: the oxidation of coal-bearing strata (including pyrite weathering); the role of the two most abundant iron-rich carbonate minerals (siderite and ankerite); neutralisation reactions involving other minerals (including aluminosilicates); and identification of possible sources of List I and List II substances in Scottish opencast drainage.
- Mining hydrogeological controls on pollutant loadings, including: the hydrogeology of unworked coal-bearing strata in Scotland; the hydrogeological behaviour of opencast backfill; and relationships between opencast sites and surrounding groundwater systems.
- Methods for characterising the pollutant release potential of opencast backfill, including consideration of "static tests" (acid-base accounting), "dynamic tests" (leaching columns and humidity cells), and the limitations of both types of test. Modelling approaches which incorporate geochemical kinetics and hydrogeological complexity have also been considered.
- Formal risk assessment techniques, most notably the "source-pathway-receptor" evaluation framework, and its adaptation to opencast systems.
- Pollution prevention and treatment technologies, including: prevention / minimisation of pollutant release processes; active treatment technologies; and passive remediation technologies.
- Environmental regulations which have implications for Scottish opencast coal mining, including both long-established legislation and emerging EU initiatives relating to mining wastes.

Drawing upon lessons learned from the literature review, and taking advantage of significant recent advances in research and practice in the UK and Europe, a six-step assessment framework has been developed. The principal elements of the assessment frameworks are as follows:

Step 1: Outline conceptual model development;
Step 2: Source strength estimation;
Step 3: Initial risk assessment;
Step 4: Conceptual model refinement;

Step 5: Risk mitigation planning;
Step 6: Contingency planning and post-closure monitoring.

Detailed advice is offered on how best to:
- gather the requisite data;
- develop the necessary conceptual models;
- undertake the required risk assessments; and
- plan for effective risk mitigation.

Estimates of the additional costs to opencast developers of implementing this framework have been prepared; these are considered marginal in relation to the overall costs of site exploration and planning.

TABLE OF CONTENTS

List of Tables

List of Figures

1. Introduction

1.1. Terms of reference

This document was commissioned by the Scottish Environment Protection Agency (SEPA), to provide an assessment framework for determining the impact that opencast coal mining operations may have on water quality. The particular focus of this framework is on ensuring compliance with the Groundwater Regulations 1998, though the document also takes into account some of the broader impacts which opencasting can have on both ground- and surface waters. The framework presented in this document builds upon the risk assessment procedure for all types of mineral extraction in Scotland, which was established in a recently-published code of practice (Scottish Executive 2003). In the present document, the focus is more narrow than in the code of practice, in that it considers solely opencast coal mining, but the degree of detail is far greater.

In specifying the scope of this document, SEPA requested inclusion of the following:
1. A hydrogeological summary of the impacts that opencast mining operations can have on water quality, focusing on the quality of water which is likely to leach from the disturbed geological materials. This particular focus logically parallels the consultation on the regulation of non-mineral wastes from mines and quarries which was launched during January 2004 by the Scottish Executive (2004).
2. Specification of the information needed to underpin the assessment of potential impacts that the excavation, storage and backfilling of geological materials from opencast coal sites can have on water quality. Clearly this needs to include information on methods of sampling and testing of strata, together with approximate cost estimates for these tests.
3. Formal proposal of an assessment framework, based on the information gathered in accordance with 2 above, tailored specifically to Scottish conditions. Given the focus of the Groundwater Regulations 1998, it is important that this assessment framework should refer specifically to List I and II substances, while also covering other contaminants known to be associated with coal mine drainage in Scotland. The assessment framework should <u>not</u> be merely a geochemical prediction tool; rather, it must take into account hydrogeological and mining factors which are known to exert important controls on mine water quality in Scotland.
4. Measures for the prevention and / or treatment of polluted drainage from active and closed opencast sites.

In meeting the above requirements, a detailed desk-study has been undertaken, which has principally involved critically reviewing the relevant international literature, and interpreting the suggestions made in that literature in the light of the hydrogeological conditions obtaining in the coalfields of Scotland.

1.2. Structure of this document

To ensure that this document is practical in emphasis, the assessment framework itself is presented up front, in Section 2 below. This is then followed in Section 3 by recommendations on how this assessment procedure can be implemented in practice, together with an introduction to appropriate risk mitigation measures appropriate to Scottish conditions.

Both Section 2 and Section 3 are based upon (and make frequent references to) a rigorous, state-of-the-art review of pollutant release and attenuation processes which are deduced to be operative to some degree in Scottish opencast environments, with particular emphasis on those processes

which could result in the mobilisation of List I and List II substances. The review itself is presented in full in Annex II, which provides in-depth coverage of the following issues:
- the geochemistry of pollutant release and attenuation in coal-bearing rocks
- hydrogeological controls on pollutant release, including both natural hydrogeology and artificial conditions induced by mining and restoration activities
- methods for assessing the pollutant release potential of strata which are to be disturbed by mining
- prevention and treatment technologies appropriate to opencast drainage waters
- the regulatory framework in Scotland, and
- general risk assessment procedures, and how these can be applied to the opencast coal mining context.

1.3. Scope of coverage

The text has been written on the assumption that the reader is basically familiar with opencast coal mining procedures as currently implemented in Scotland. If this is not the case, the reader is urged to first read Annex 1 (and references cited therein) before proceeding to read Sections 2 and 3.

As is made clear in Annex 1, what distinguishes opencast coal mining from other forms of surface mining (see Hartman and Mutmansky 2002) is the immediacy with which waste rock is backfilled ('cast') into the recently-worked void immediately behind the current zone of working, almost always without any intermediate storage stage. The reason for this is economic: double-handling of waste rock is very expensive and there are thus strong financial incentives to undertake stripping and tipping of overburden[1] in a single operation. Therefore, apart from the 'soil store'[1] and the 'baffle banks'[1] which are created during site preparation, there is actually very little *stored* waste rock to speak of in active Scottish opencast coal sites. This is precisely why the assessment framework presented in Section 2 lays such emphasis on the characterisation and management of backfill[1], with proportionately limited discussion of the storage of excavated materials during the working life of an opencast site.

In line with the specification of SEPA, this document restricts its coverage to preventing the migration of pollutants derived from weathering of soils and rocks disturbed by opencasting: List I and List II substances of anthropogenic origin are not considered in any detail. The recent Code of Practice for mineral extraction published by the Scottish Executive (2003) discusses the prevention of pollution by anthropogenic compounds in considerable detail, and the off-site disposal of possible sources of such compounds which it advocates is in harmony with the present consultation document on the regulation of non-mineral wastes from mines and quarries (Scottish Executive 2004).

Table 1 lists the principal List I and List II substances which might *potentially* be mobilised during weathering of natural soils and rocks on Scottish opencast coal sites, together with comments on their likely occurrence in leachates originating in Scottish opencast backfill. A full explanation of the rationale for the comments offered in Table 1 is to be found in Annex II (see Table AII-2 and the accompanying text in Section AII.1.6).

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[1] See Annex I for a definition of this and other opencast terminology.

Table 1: Summary of List I and List II substances which could in theory be mobilised by weathering of soils and rocks disturbed by opencast coal mining. See Annex II (Section A11.1.6) for further explanation.

List I substances	Comments …
Mercury (Hg) and its compounds	**<u>Highly unlikely to be released</u>** during weathering of soils and rocks found in Scottish opencast sites
Cadmium (Cd) and its compounds	**<u>Unlikely to be released</u>** during weathering of soils and rocks found in Scottish opencast sites; limited release is locally possible where hydrothermal veins containing sphalerite (or much less commonly greenockite) cut the local coal-bearing succession
List II substances	**Comments …**
Zinc and its compounds	**<u>May occasionally be released</u>** both by dissolution of disseminated sulphides (traces present in pyrite and chalcopyrite; principal metals in sphalerite) and by desorption from clays etc, most likely at low pH. Never known to exceed 20 mg/l in Scottish opencast drainage waters analysed to date.
Copper and its compounds	**<u>May occasionally be released</u>** in manner similar to Zn, but it is less mobile than Zn and does not often exceed 1 mg/l in Scottish opencast drainage waters analysed to date.
Nickel and its compounds	**<u>May occasionally be released</u>** in manner similar to Zn; some millerite (NiS) is known to occur sporadically in Scottish coal-bearing sequences. Ni is less mobile than Zn, and does not appear to exceed 5 mg/l in Scottish opencast drainage waters analysed to date.
Chromium, Lead, Tin, Barium, Beryllium, Boron, Uranium, Titanium, Molybdenum, Antimony, Arsenic, Selenium, Silver, Tellurium, Thallium, Cobalt and Vanadium	**<u>Highly unlikely to be released</u>** during weathering of soils and rocks found in Scottish opencast sites; occasional traces of As, Co and V at concentrations of up to 50 μg/l have been recorded. None of the other elements have yet been reported above detection limits in Scottish opencast drainage waters to the knowledge of the authors.
Substances damaging to the taste, odour and potability of groundwater	Fe, Mn, Al and SO_4, all of which affect taste and potability, are **<u>commonly released</u>** unless steps are taken to minimise their mobility.
Fluorides	**<u>May occasionally be released</u>** though mineral sources in coal-bearing strata are few; dissolved concentrations normally maintained below 5 mg/l by equilibrium with CaF_2 (fluorite).
Ammonia	**<u>May occasionally be released</u>** at concentrations of several tens of mg/l. Ammonia release is generally restricted to peculiar circumstances, such as in deeply-buried unmined coal seams containing ancient groundwaters rich in Cl, and in waters leaching previously burnt coal-rich zones in backfill.

2. Assessment framework for potential water pollution risks associated with opencast coal mining in Scotland

2.1. Foundational principles of proposed assessment framework

The assessment framework outlined below is based firmly on the state-of-the-art review of pollutant release and attenuation within opencast coal mining environments presented in Annex II of this report. The framework is primarily aimed at ensuring that opencast developments comply with the Groundwater Regulations 1998. Needless to say, it is not acceptable for compliance with the Groundwater Regulations 1998 to be attained in such a way that other regulatory requirements are violated as a side-effect. Hence in developing this framework, attempts have been made at every turn to hold the 'bigger picture' clearly in mind whilst making recommendations.

Respecting this 'bigger picture' demands that potential risks associated with specific sites be examined as seamlessly as possible across the full range of relevant scales (cf Annex II, Section AII.4.1). In terms of **spatial scales**, this means we need to harmonise characterisation from the scale of the individual sample (cf Annex II, Section AII.3.2), through the scale of the individual site (e.g. Annex II, AII.3.4) right up to the full catchment scale[2] (see Younger and Wolkersdorfer 2004). In terms of **temporal scales**, the most appropriate approach is to consider the full life-cycle of any given mining operation: although most attention in the planning process is naturally focused on the operational phase, the duration of this is in all cases dwarfed by the post-closure phase (a point emphasised in Younger and Robins 2002). It is nevertheless necessary to briefly explore the key issues in the operational and post-restoration phases of the opencast site life cycle:

    (i)   <u>operational opencast sites:</u> Key issues relate to both anthropogenic and geogenic List I and II substances. The term 'anthropogenic' is here used to refer to substances which might be released from man-made equipment or substances used in an active opencast site. By contrast 'geogenic' substances are those arising from leaching of disturbed soils and rocks, which are the prime concern of this document. In relation to the anthropogenic List I and List II substances (which are listed in Table AII-2 (Annex II, Section AII.1.6), the precautions necessary to prevent release to groundwater amount to best practice in the operation and maintenance of equipment, fuels, solvents and other man-made substances. The recently-published "Code of practice for the owners and operators of quarries and other mineral extraction sites" (Scottish Executive 2003) provides an appropriate framework for implementing precautions in relation to these anthropogenic List I and List II substances. In relation to geogenic List I and II substances, by far the bulk of disturbed rock in operational opencast sites is backfilled immediately after it is first excavated. The majority of the geogenic pollutant sources are thus the same during the operational phase as in the post-restoration phase, after final completion of backfilling operations. Only the soil store and baffle banks (containing overburden removed during excavation of the initial box-cut) are stored throughout the life of the site (Annex I, Section A1.3). As site operators are only interested in storing good quality topsoil for later use, soil stores normally contain material with no pollution-generating potential. Baffle banks, on the other hand, contain broken rock material which is essentially identical to the remaining backfill on the site, with which it will tend to share a similar pollutant-generating potential. However, both soil stores and baffle banks are invariably constructed at surface, on top of the pre-existing ground surface around the perimeter of the site. This means that any drainage emanating from them can be (and should be) intercepted by surface ditches or land drains, whence it can be sampled to ensure that any necessary treatment is

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[2] This being the focus of the EU Water Framework Directive (2000/60/EC) and thus of the Water Environment and Water Services (Scotland) Act 2003.

implemented prior to discharge to surface streams or to groundwaters. Such discharges are already subject to regulation by SEPA under the terms of the Control of Pollution Act 1974 (COPA) as explained in Annex II (Section AII.6.1). Coal washery fines (typically a mixture of silt and other grades of sediment, which may contain significant quantities of pyrite) are generated during active coaling on those sites where on-site dressing of the run-of-mine coal is undertaken. It is common practice to dispose of washery fines within the backfill. Given that they may contain pyrite, it is prudent to fully assess (and mitigate) any risks associated with their on-site disposal. In addition, the potential effects of dewatering operations on the surrounding groundwater system need to be carefully assessed, especially where dewatering involves abstraction of groundwaters which are already contaminated as a result of former mining activities. Again, as for drainage emanating from soil stores, baffle banks, or as runoff originating anywhere within the site, the discharge of dewatering effluents to controlled waters is already subject to effective regulation by SEPA under the COPA regime. Drawdown of the water table in previously-flooded workings can also restart pyrite oxidation processes where they had previously been effectively prevented by submergence (see Annex II, Section AII.1.5). This risk requires careful consideration during risk assessment for proposed sites.

(ii)   Restored sites: After the site has been restored (see Annex I, Section A1.4 for further details of restoration operations), the key to achieving sustained compliance with the Groundwater Regulations 1998 relates primarily to preventing long-term pollutant release from backfill (which may locally include pockets of washery fines in addition to broken rock). The design of effective preventative measures demands careful conceptual modelling, source characterisation and risk assessment, as will be explained below. In addition, where an opencast site extracted strata below the local water table with the aid of dewatering, the recovery of groundwater levels after restoration can considerably complicate both the characterisation of pollutant release and attenuation[3] and the implementation of mitigation measures. The key to successful risk mitigation in such cases is early diagnosis of groundwater system behaviour, during the planning stages of the opencast development, followed by careful monitoring and (where necessary) interception and treatment of polluted waters.

In reviewing the international literature on 'acid mine drainage prediction' tools (see Annex II, Section AII.3) it was found that the most serious omission in most protocols is a clear focus on understanding where water moves from / to in the subsurface environment which encloses a given opencast site. The literature is replete with thousands of pages of minutely-argued text concerning the intricacies of 'acid-base accounting' and 'humidity-cell tests', and yet almost devoid of serious discussions of the hydrological pathways which are the *sine qua non* for pollution impacts to actually materialise. No matter how sulphidic your mine waste is, unless it is hydrogeologically disposed so that it can both oxidise and transmit its leaching products to an identifiable receptor, then it amounts to nothing more than a geological curiosity. The domination of mine site risk assessment by vernacular approaches to geochemical leaching studies is simply inappropriate for Scotland, where almost all opencasting takes place in ground previously rendered permeable by former deep mining. Rather, a more holistic approach is required, based firmly on a tiered approach to risk assessment recently advocated in the peer-reviewed literature (see Banwart *et al.* 2002, and comments on that paper in Annex II (Section AII.4)). The logic is simple: we need to contextualise every opencast development proposal within its broader hydrogeological setting, identifying potential pollutant pathways and potential receptors. If both of these exist, then we must expend further effort on the characterisation of source risks; where the magnitude of the assessed risks demands it, we must then proceed to design and implement appropriate mitigation

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[3] For instance, it can be difficult to apportion elements of the pollutant load between the recently-worked opencast site itself and the surrounding old workings (which may have been affected by dewatering but were not directly re-worked by opencasting).

strategies (many of which will best be implemented during the operational phase of the site life-cycle).

In setting out the assessment framework below, recourse is made to the notion of a "site conceptual model". It is important to state clearly what is meant by this prior to proceeding to detailed discussions. The term "conceptual model" has a formal meaning in hydrogeology, having been defined by Bear and Verruijt (1987) as 'a set of [*rigorously justified*] assumptions which represent our simplified perception of a real system'. As Rushton (2003) has further explained: "Conceptual models describe how water enters an aquifer system, flows through the aquifer system and leaves the aquifer system". To these hydrogeological definitions we can simply add parallel comments concerning the release, transport and discharge of specific groundwater contaminants.

While conceptual modelling ought always to proceed any attempt to mathematically model a groundwater system (see Rushton 2003), it is by no means always necessary that a conceptual model be converted into a mathematical model. Rather, conceptual models are largely an end in themselves. They represent the current consensus on system behaviour, whether this be informed by direct interpretation of field and laboratory data alone, or whether these data have been further 'inverted' by mathematical modelling. In essence, all mathematical modelling boils down to a formalised, quantitative assessment of the consistency between our concepts of system behaviour and the data upon which these concepts are based (Konikow 1981). Once we have assessed this consistency, we can return to our conceptual model and amend it as appropriate; but it is the conceptual model which remains supreme. As Rushton (2003) rightly comments, the existence of a conceptual model allows others "to assess critically the current thinking and to provide further insights". It is with precisely this intention and understanding in mind that the development of conceptual models for specific opencast sites is proposed in the text which follows.

2.2 Assessment framework

General comments

This assessment framework operates in a step-wise fashion, with some scope for bypassing certain steps in the event that their contents are not applicable in a particular case. Figure 1 summarises the steps in the framework and how they should be navigated.

As Figure 1 makes clear, an iterative approach to risk assessment is advocated, in which the suitability of data for making reliable judgements is repeatedly questioned until a consensus between developers and regulators is attained. Consensus will be reached only when the following obtains:

- an agreed conceptual model for site behaviour exists

- any risk mitigation measures identified on the basis of the conceptual model have been implemented (or firmly planned for implementation at a more appropriate point in time)

- a long-term contingency plan, incorporating proposals for monitoring, has been accepted by the regulator.
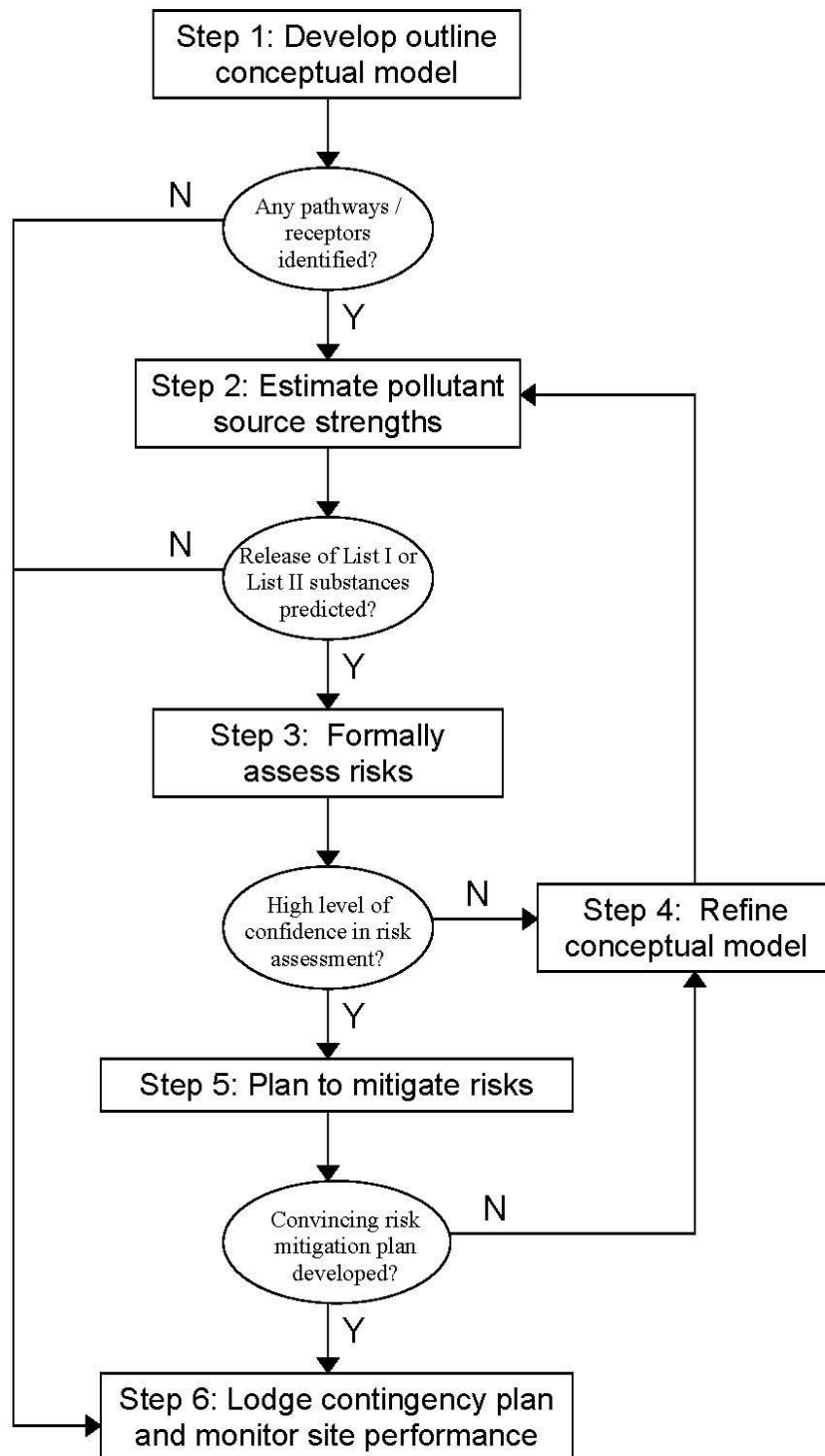
Figure 1: Flow-chart summarising the decision logic of the risk assessment framework proposed for evaluations of the compliance of proposed opencast developments in Scotland with the Groundwater Regulations 1998 and other environmental regulations.

The constituent steps of the framework are outlined below. Notes on how these steps should be implemented are given in Section 3.

Step 1: Outline conceptual model

An initial conceptual hydrogeological model for the site must be developed, with explicit coverage of <u>at least</u> the following key points:

- The hydrogeological setting of the proposed opencast site in relation to the natural base level of drainage in the area (and thus rest water table levels after closure).
- The recharge and discharge areas belonging to the hydrogeological system within which the opencast site will be developed, and the degree to which these will be modified by the proposed extractive activity.
- Any proposals for artificial groundwater lowering to facilitate coaling and related activities with explicit consideration of:
    - Anticipated pumping rates, and the means of pumping (i.e. by sump-pumping within the site and / or by pumping wells or shafts sited outside the immediate site boundary)
    - the extent of any cone of depression and of the total groundwater capture zone for the proposed site
    - the degree to which groundwater lowering within and beyond the site boundaries will lead to drainage of formerly submerged strata / old mine workings containing pyrite
- Identification of all possible migration pathways for leachates generated within the disturbed soils and broken rock on the site (i.e. soil stores, baffle banks, backfill and bodies of washery waste)
- The locations and nature of potential receptors for site drainage, including groundwater, rivers, streams, wetlands and lochs.

At this point, virtually all assessments will proceed to Step 2. However, in the very rare event that <u>no</u> post-closure migration pathways or potential receptors are identified, SEPA may give permission for the evaluation to proceed directly to Step 6. **It is emphasised that a very well substantiated case would need to be made before SEPA would accept that a given assessment could proceed straight from Step 1 to Step 6; a default presumption will always be made in favour of proceeding to Step 2, and this will be maintained in all but the most exceptional of cases.**

Step 2: Pollutant source strength estimation

Where Step 1 has revealed the existence of receptors potentially connected to the proposed opencast site by credible groundwater pollutant transport pathways, it is essential that a robust assessment be made of the possible strength of leachates originating within site backfill. A hierarchical approach to source strength assessment is proposed. As will be seen, the level of evaluation appropriate in a given case is to some degree dependent on the degree of assessed risk. As formal risk assessment is the subject matter of Step 3, there will inevitably be a degree of iteration between Steps 2 and 3, either informally during their initial execution, or else formally where the first-pass risk assessment is adjudged insufficient, triggering a formal re-evaluation of the conceptual model via Step 4 (see Figure 1). The levels of evaluation of source strength can be summarised as follows:

Level 1 evaluation: invocation of site analogues: Where the same seam(s) of coal have been mined nearby by opencasting, and the resulting post-closure site configuration closely resembles that anticipated for the proposed new opencast site, hydrochemical data collected at these 'precedence' analogue sites may be adduced to provide evidence of the likely quality of water to be expected from the proposed site. Where no such analogues exist, or the similarities of the proposed analogues are not sufficiently close to the anticipated configuration

of the proposed site, this approach will not be admissible, and the source strength estimation must proceed in accordance with 'Level 2' below.

Level 2 evaluation: geological screening using sulphur content data. This is likely to be the most appropriate level of evaluation for most low risk sites (see Step 3). It is predicated on two geological premises: (i) the common observation that many of the most prolific acidity-releasing rocks (which are usually pyritic shales and siltstones) occur as seam floor or roof beds and therefore tend to have sulphur contents similar to those of the adjoining coal. This in turn means that sulphur content values for the coal itself (which are routinely measured for commercial reasons, and are therefore well known in many cases) can be used as a proxy measure of overburden sulphur content. (ii) the known association between high sulphur contents and stratigraphic proximity to a 'marine band', i.e. a horizon deposited under marine conditions (see Casagrande 1987; Younger 2000), a factor which has been examined for Scotland in particular by Younger (2001). Advice on how to implement this approach is given in Section 3.3.2 below.

Level 3 evaluation: petrological evaluations of pollution potential. This level of evaluation is most likely to be appropriate for sites of low- to medium-risk (Step 3). Level 3 evaluation requires the development of an applied petrological characterisation of key overburden horizons, which for pollution generation and attenuation purposes means those likely mainly to produce fragments of 4mm diameter or less as a result of opencasting (which in turn means mainly mudstones and siltstones). The evaluation will provide information on the likelihood of the strata releasing List I and List II substances if they are subjected to oxidative weathering and leaching. This provides input to the formal risk assessment (Step 3).

Level 4 evaluation: leaching column experiments to support modelling. This level of evaluation is most appropriate to sites which appear to pose a high pollution risk. Essentially, Level 4 evaluation will only be undertaken where there is a need to specify site-specific reaction kinetics for incorporation into numerical models of site performance. Where this level of detail is necessary, laboratory leaching column experiments may be used to obtain information on leachate generation processes, with the lab-to-field scaling procedures developed by Malmström *et al.* (2000), Younger *et al.* (2002) and Banwart *et al.* (2002) being used to transform the raw lab results to usable field-scale reaction rates suitable for use in site models. Essentially this involves transformation of lab-determined pollutant release rates into equivalent field rates by multiplying the lab rates by a series of factors which account for lab:field contrasts in ambient pH, grain size distribution, temperature, and the ratio of mobile to immobile water (which is often significantly higher in the lab than in the field).

In procedural terms, Level 3 corresponds approximately to 'static testing' as used in the USA (Section AII.3.2), albeit that significant modifications of US practice are advocated (as will be explained in Section 3.3.2 below). Level 4 includes an element of 'dynamic testing' as used in North America (Section AII.3.3), with the added element of explicitly scaling from lab measurements of pollutant release rates to field-scale applications. These modifications are designed to provide a more specific focus on the potential for release of List I and List II substances. Both the Level 3 and Level 4 evaluations require characterisation of rock core material obtained from site investigation boreholes. The design rationale for appropriate coring strategies is outlined in Section 3.3.2, building upon existing practices in the opencast industry.

The decision point at the end of Step 2 may be posed as follows: Is the release to pollutant migration pathways of List I or List II substances predicted? If the answer is "Yes", then proceed to Step 3; if the answer is "No", proceed directly to Step 6.

Step 3: Formal risk assessment

As was mentioned in relation to Step 2, it will very often make sense in practice to develop Step 3 in parallel with some of the characterisation activities under Step 2, so that the level of evaluation deployed under Step 2 is adequately tailored to the picture emerging in Step 3.

The aim of Step 3 is to make an overall assessment of the risk posed by the site, by making a critical analysis of the combination of hydrological data (which will have been collated / collected in support of Step 1) and information on likely pollutant source strengths (Step 2) specific to the site under investigation. The end result will be a clear identification of the degree of risk which a particular site poses. Four categories of site risk are identified, as follows:

> *Very high risk*: Release of List I substances is predicted and possible pathways exist to receptors.
> *High risk*: Release of List II substances is predicted and clear pathways exist to receptors.
> *Medium risk*: Release of List II substances is predicted and possible pathways exist to potential receptors.
> *Low risk*: Release of neither List I nor List II substances seems likely, and there seems little risk of migration of pollutants to any receptors.

In order to arrive at these risk categorisations, the developer will need to further expand upon the conceptual model developed under Step 1. The expansion required will involve making quantitative estimates of the possible scale of pollutant migration to receptors. Given the uncertainties surrounding key hydrological transport parameters it is not reasonable to expect wholly deterministic[4] assessments of pollutant migration: some assessment of the degree of uncertainty inherent in the estimates will therefore be necessary. The magnitude of uncertainty will be an important consideration both in relation to the acceptability of a specific risk assessment, and in relation to the design of suitably robust mitigation measures.

The specific form which pollutant risk estimates take is deliberately not specified here. In some case it will be appropriate to develop quantitative geochemical models in order to evaluate alternative scenarios of pollutant release and immobilisation. Such models may require "Level 4" evaluation tests (as described under Step 2). In other cases it may be possible to develop sufficiently robust risk estimates by manual calculation using *prima facie* reasoning: this is most likely to be acceptable where the hydrogeological setting is relatively simple and well understood.

When the risk estimates have been collated, the developer should meet with SEPA to discuss their findings. If it transpires that the currently-available information is insufficient to support definitive conclusions on the relative magnitude of risk (i.e. "low" versus "high" etc), then the next step will be to proceed to Step 4. If an acceptable risk assessment has been developed, the assessment will pass to Step 5.

Step 4: Conceptual model refinement

This step will often be skipped altogether, and is only invoked where the outcome of a Step 3 risk assessment proved inconclusive. Step 4 is essentially a period of reflection, supported by sensitivity analyses and other forms of uncertainty analysis as appropriate to the specific case under investigation. Key questions which need to be answered during Step 4 are:

- Why was the available site information insufficient to support an acceptable risk analysis?
- Where are weak points / points of maximum uncertainty in the existing conceptual model?
- What further data are required to better assess the hydrogeological framework and / or source strength?

In the course of answering these questions, the following activities will be undertaken:

- the conceptual model will be further refined
- further data collection activities to reduce uncertainty in key elements of the model will be identified and designed

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[4] i.e. single-value answers to simply posed questions

-        the process of risk assessment (Step 3) will be redesigned to ensure that an adequate assessment is achieved hereafter.

Having concluded the above task, the assessment proceeds by returning to Step 2.

Step 5: Risk mitigation planning

At this point in the assessment a thorough risk mitigation plan is drafted, which will include detailed consideration of <u>at least </u>the following items:

- hydrologically "defensive" mine planning measures, aimed at minimising unnecessary ingress of off-site groundwaters and runoff (both during and after working) (see Robins and Younger 2002)
- where the site intersects the pre-mining water table, the design of dewatering operations, specifically addressing:
    - o the mode of dewatering, with advance dewatering always being preferred ahead of sump-dewatering, wherever possible (cf. Norton 1983)
    - o the likely quality of waters to be pumped and their treatment requirements (where appropriate) in order to comply with COPA provisions, and
    - o whether drawdowns beyond the site boundary will restart pyrite oxidation or other pollutant release processes in previously flooded old workings.
- how site design can be formulated to minimise the risk of substantial decant of contaminated leachates in undesirable locations (be this to ground- or surface waters)
- advance planning for active and / or passive treatment operations to minimise aquatic pollution from leachates unavoidably generated within the site (see Sections 2.5.4 and 2.5.5)
- design of restoration measures (e.g. compaction, clay capping of backfill etc) to minimise long-term infiltration (see Section AII.5.3).
- specific planning for compliance monitoring (see Section AII.5.2).

Following agreement of this plan with SEPA and the relevant planning authorities, the plan will be fully implemented during the operational phase of the site life cycle. The final action is to proceed to Step 6.

Step 6: Contingency plan

A contingency plan will be required to ensure the long-term conformity of all sites to the pre-conceptions inherent in the risk mitigation plan (Step 5). This applies equally whether the site in question never posed an identifiable risk, or whether any risks it did pose had been successfully mitigated as a result of actions instigated under Step 5. This plan must be submitted to SEPA at least six months (and not more than 18 months) prior to the closure of the opencast site. It will summarise assessed risks associated with the site and any measures which were implemented to mitigate these. It will then identify any residual risks and any further steps which will be needed to address these. The plan will include measures held in reserve during the site after-care period[5] to be implemented in the event that prior assessment and planning activities prove to have been mistaken in some way. Precautionary monitoring of post-restoration water table recovery and associated changes in groundwater (and where appropriate surface water) quality will typically be required during the after-care period, and in some cases for an agreed period of time following the expiry of other after-care duties.

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[5] planning authorities in Scotland typically require a minimum 5-year after care period for opencast sites, which is occasionally extended to 10 years where assessed risks are high.

3. Practical recommendations on implementation of the assessment framework steps

3.1. Development of outline conceptual model (Step 1)

The five key elements of the initial conceptual hydrogeological model which is proposed to be developed under Step 1 can all be determined by the interpretation of geological data which will be collected during the exploration and design phase of the site life cycle. Much of the necessary data are already collected as part of resource prospecting and evaluation tasks; it is envisaged that the additional data collection needed to support development of the conceptual model can be incorporated into the site exploration strategy. The necessary data interpretation ought to be readily achievable by the same geologists who develop extraction-oriented geological models for opencast developers, albeit they may require some additional professional development in relation to hydrogeological processes and their interpretation. The following notes are intended to facilitate the construction of an appropriate conceptual model.

Opencast sites can be developed in a number of positions within existing hydrogeological systems. Figures 2 and 3 attempt to summarise some of the more common settings likely to be encountered in present-day Scotland. Perhaps the most profound factor requiring determination is the position of the sole of the excavation (and therefore the base of future backfill) in relation to the ambient water table position. Where opencast sites are developed above the water table in previously unmined strata (admittedly an increasingly rare occurrence in Scotland anymore) the backfill will typically have a far higher permeability than the adjoining / underlying intact strata, and thus they most often form perched aquifers, which tend to drain to a spring line corresponding to the lowest topographic point on the former site low wall[6] (Figure 2). Where a supra-water table opencast is in communication with old mine workings, then it can be expected to feed any leachate which it produces into these (Figure 3a-c), whence it will pass to the groundwater flow system of which the lower, flooded portion of the voids form a part. This in turn might eventually flow to surface as a discrete mine water discharge, or else feed groundwater onwards into another part of the groundwater system, perhaps in adjoining un-mined strata. Where opencast workings intersect the water table, they may function as recharge source zones (e.g. Figure 3d), "flow through zones" (Figure 3e) or discharge collection zones (Figure 3f). In the latter capacity, the body of backfill may act more or less like a French drain, collecting groundwater from various subsurface flowpaths (e.g. numerous old deep mine roadways) and channelling it out to the surface environment.

These different settings can be expected to manifest contrasts in their groundwater qualities. While all types of backfilled opencast can generate leachate which infiltrates to depth, groundwater present within the "flow-through" (Figure 3e) and "discharge zone" (Figure 3f) sites will reflect pollutant mobilisation and attenuation processes outside the opencast site boundaries, perhaps as much as ten kilometres away in some cases. The post-working groundwater quality within and down-gradient from such sites will be unlikely to correlate especially strongly with the strength of the leachates produced by the backfill within these sites themselves, for far larger volumes of 'allochthonous' mine water will typically pass through these sites than they would ever themselves generate.

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[6] See Annex I; the low wall is the lowest rim of un-mined strata surrounding a present / former opencast void.
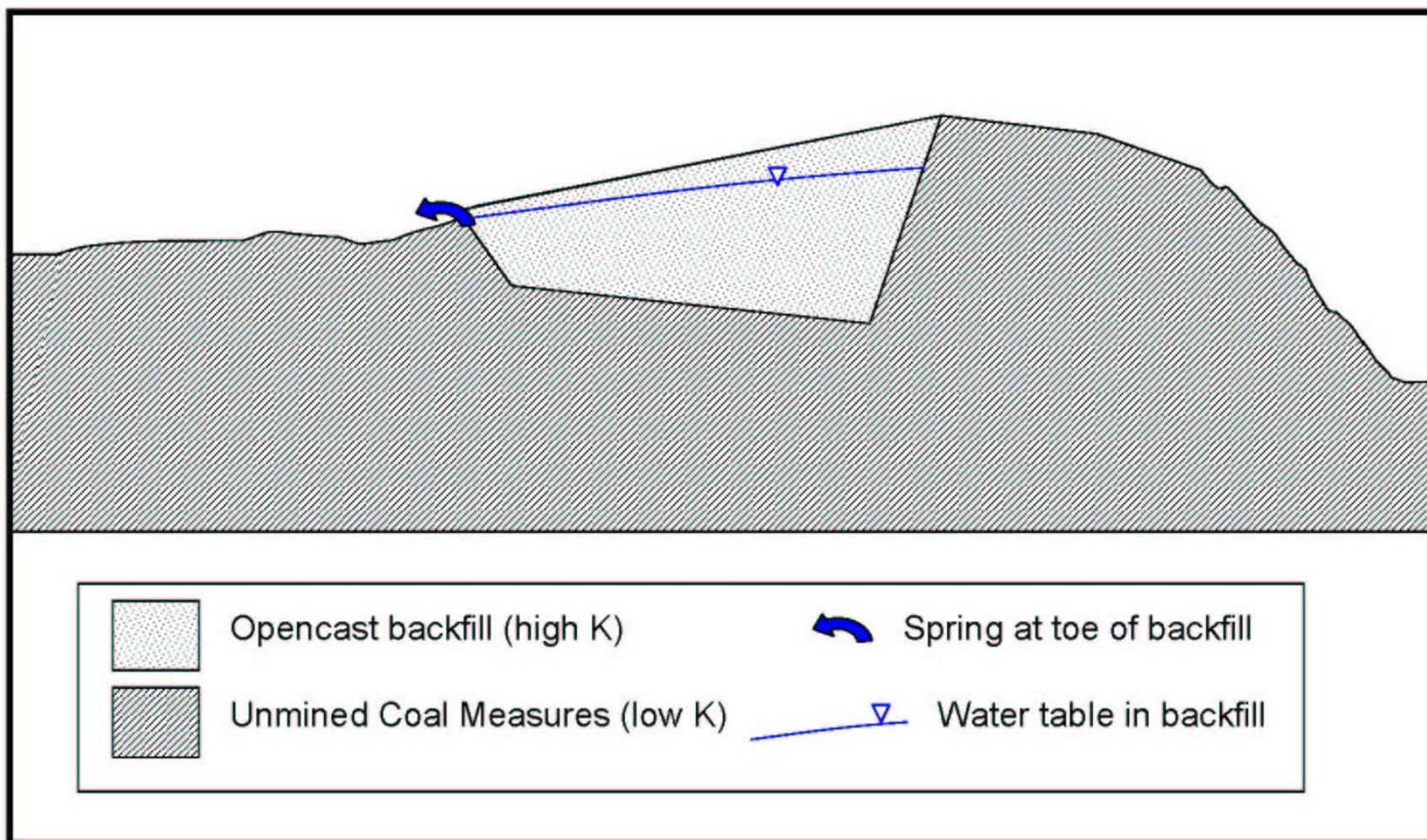
Figure 2: Schematic cross-section illustrating how the contrast in hydraulic conductivity (K) between a restored opencast site and unmined strata can lead to development of a perched aquifer draining to a spring located at the lowest point (backfill 'toe') of the low wall.

Legend:
- Opencast backfill (high K)
- Unmined Coal Measures (low K)
- Spring at toe of backfill
- Water table in backfill
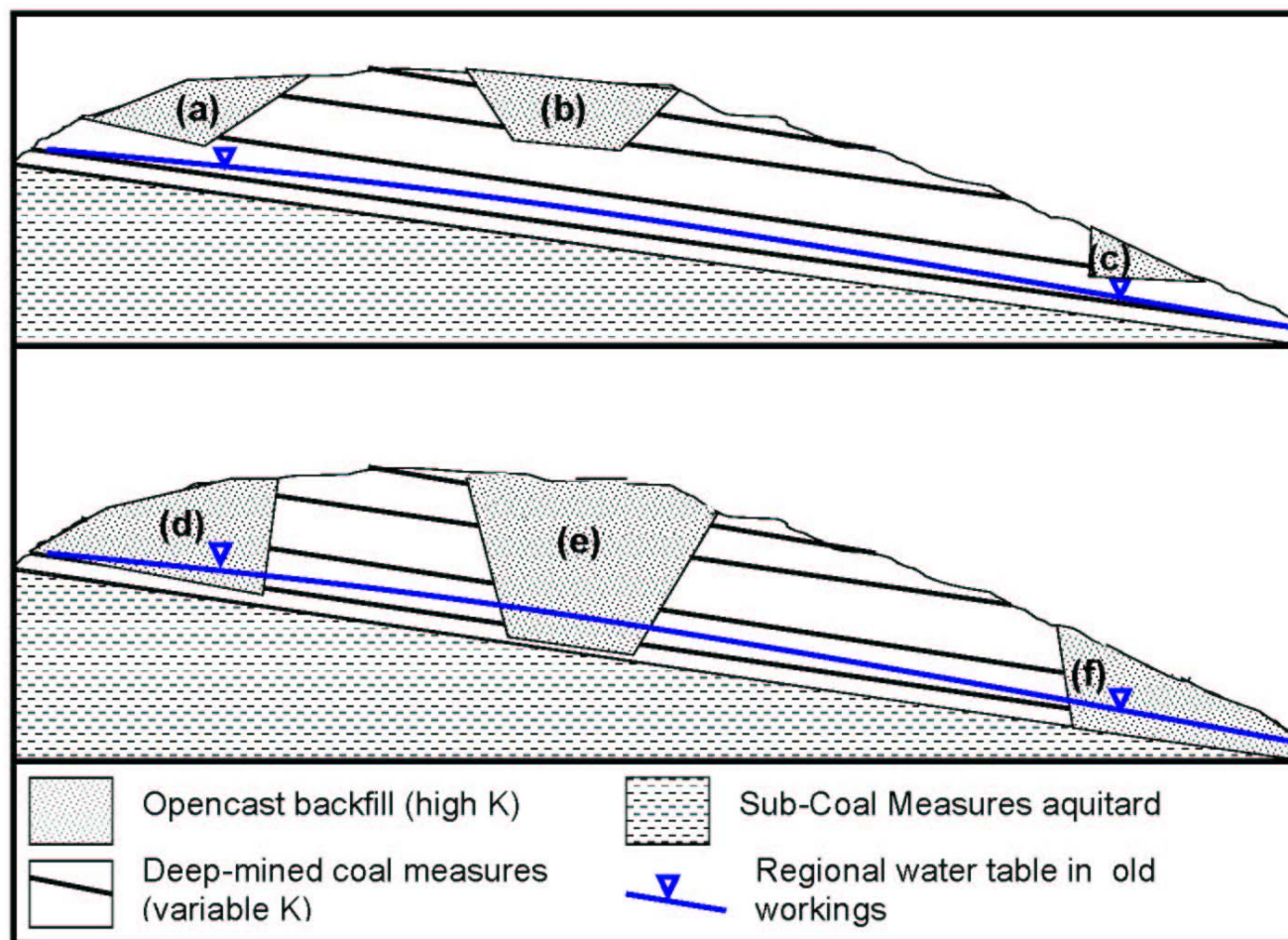
Figure 3: Sketch cross-section illustrating the possible locations of backfilled opencast sites with respect to the water table within a regional groundwater flow system. Supra-water table settings: (a) opencast backfill occupying former natural recharge area in seam outcrop area; (b) opencast backfill offering a 'window' for further recharge to centre of flow system; (c) supra-water table backfill in the vicinity of a groundwater discharge zone. Sub-water table settings: (d) opencast backfill in recharge area providing permeable pathways all the way to the water table; (e) "flow-through" backfill in centre of flow system; (f) "discharge zone" body of backfill acting as a 'French drain' to flooded underground mine workings up-dip.

Conscious quantification of the risks associated with dewatering, working and backfilling sites which are in communication with regional groundwater flow systems in this way is an essential element of conceptual model development, and demands a good appreciation of former mining methods (Younger *et al.* 2002). Proposals for artificial groundwater lowering to facilitate coaling and related activities will require explicit consideration of anticipated pumping rates and the likely 'radius of influence' of drawdown caused by the pumping. This in turn should allow quantification of the degree to which groundwater lowering beyond the site boundaries will lead to drainage of formerly submerged strata / old mine workings containing pyrite and other possible pollutant source / sink minerals, including siderite and ankerite (see Sections AII.1.3 and AII.1.4).

Wherever possible, opencast sites should be 'advance dewatered' by means of pumping boreholes or old mine shafts sited outside of the working area (Norton 1983). This has a number of key advantages, including minimisation of water contact with recently-oxidised materials within the site, and avoidance of entrainment of suspended solids. For smaller sites, sump pumping within the site will be the only realistic option, but here again a little care in the routing of site drainage and the locations of sumps can yield short- to medium-term dividends in terms of minimising any deterioration in water quality.

The general principles of mine dewatering have been discussed in some detail by Younger *et al.* (2002). In order to establish a net decline in groundwater levels, it will be necessary at first to pump at far greater a rate than that at which water enters the system. Once maximum drawdown has been achieved, a lesser pumping rate will suffice to maintain drawdown at a constant level. The design of such operations for the case of opencast coal mines which will intersect flooded old underground workings have been established by Norton (1983) (who worked extensively on Scottish opencast sites) and Minett *et al.* (1986; see also Minett 1987) (who developed test-pumping interpretation and drawdown-prediction methods for proposed opencast operations in formerly mined areas, and tested them for performance on advance-dewatering sites in Northumberland). Key elements of the approaches advocated by these authors include:
-    quantifying pre-opencasting outflow rates (Q_O) from the groundwater system into which the surface excavations will be sunk, by means of flow measurement at any surface outflows of mine water (for the *a priori* identification of which, see below)
-    estimating the 'drawdown volume' (V_D), i.e. the volume of water lying between the present water table and the projected water table at the time of maximum dewatering; this represents a discrete volume which must be removed from storage in order for sufficient groundwater lowering to be achieved
-    quantification of rates of surface water entry to void (Q_S), comprising direct rainfall into the workings plus any surface runoff which may enter it (despite all efforts to divert this away wherever possible).

One of the most difficult components to estimate is V_D, as this reflects the total volume of flooded voids (both natural porosity and mined voids) in the area to be dewatered. Various simplifying assumptions are typically applied in order to quantify V_D, and one of the more successful approaches is that described by Minett *et al.* (1986), in which an equivalent 'specific yield' is effectively calculated for the old workings (see Younger and Adams 1999, p. 54, for further discussion on this point).

Having quantified these three factors, then the predicted dewatering requirements are calculated as follows. It is assumed that the initial pumping rate will need to be such as to remove V_D within a reasonable time frame. Thus if maximum dewatering must be achieved within one year of the commencement of pumping, one can estimate that the required daily pumping rate must exceed $V_D / 365$. This might imply pumping rates which exceed the pre-mining outflow rate (Q_O) by six or seven times. Subsequently, the steady pumping rate to maintain the achieved drawdown will equal: $F \cdot Q_O + Q_S$, where F is some factor which allows for the fact that the steady inflow rate to the voids will be greater where driving heads are high than it was pre-mining, when hydraulic gradients

are low. Typical values of F apparently range between about 1.2 (e.g. around 1.25 at Blindwells) to around 2 (e.g. about 1.8 at Westfield).

Prediction of drawdowns beyond the boundaries of the proposed opencast site is challenging. It is, of course, possible to use standard radial flow groundwater models to predict how drawdown varies with distance from the pump locations. However, complications arise in practice due to:
- the presence of faults, which commonly displace Scottish coal-bearing sequences on a range of scales, and
- the hydrogeological behaviour of flooded deep mine workings. Of particular note is the prevalence of turbulent flow in old mine roadways, especially where these occur in close proximity to the opencast void. Explicit account of flow in old roadways can be incorporated into numerical models of mine dewatering operations, as has recently been successfully demonstrated by Nuttall *et al.* (2002), who were able to accurately model observed pumping rates and drawdowns in a coalfield in Lancashire using a hybrid numerical simulation code in which a 3-D variably saturated porous medium is coupled to a 3-D pipe network representing open mine voids. In many cases it is likely to prove more cost-effective to make an approximate evaluation of likely drawdown effects, plan to mitigate any possible negative consequences, and then simply monitor their development during site development.

Guidelines for the numbers and locations of monitoring boreholes to facilitate effective drawdown observation and management for mines and quarries of all types have recently been published by the European Commission's 'ERMITE' RTD project. Table 2 customises the recommendations made in the ERMITE guidelines for the particular case of opencast coal sites in Scotland.

Table 2: Recommended frequencies and locations of ground water monitoring wells around proposed opencast dewatering operations in Scotland.

Anticipated rate of dewatering (10^3 m^3/d)	< 0.5	0.5 - 3.0	> 3
Minimum no. of radial lines of monitoring wells	1	2	3
Distance[1] to innermost monitoring wells (m)	20	30	50
No. of intermediate monitoring wells per line	1	2	3
Distances[1] to intermediate wells (km)	0.2	0.3, 3	0.5, 2, 5
Distance[1] to outermost monitoring wells (km)	0.8	6	10

[1] from outer boundary of mined area.

Delineation of potential migration pathways and receptors is based on interpretation of the wider hydrogeological system within which a proposed opencast will sit (see Figures 2 and 3). In determining the position of a given opencast site relative to the water table (Figure 3) one will already have identified most of the boundaries of the enclosing hydrogeological system, including recharge areas, lateral and vertical limits to the inter-connected body of groundwater, and the system discharge areas. The mined hydrogeological system may discharge to the surface or into an adjoining aquifer composed of natural (unmined) sediments (see Robins 1990 for examples of natural aquifers in Scotland). Surface discharges from mined hydrogeological systems in Scotland almost always correspond to relict mined features, such as old drainage adits or overflowing shafts in relatively low topographic positions (e.g. near rivers or on the coast, if this is nearby). Younger and Adams (1999) developed specific guidance on how to locate likely points of surface discharge from large systems of inter-connected mineworkings. The methodology which they proposed is as follows:
- determine the outcrop patterns of the major worked seams (typically using British Geological Survey maps where available)
- scrutinise Coal Authority mining records to ascertain the locations of old mine entrances (adits, shafts etc) on or near the outcrops of the seams
- determine which of the old entrances lies in the lowest topographic position on the outcrop; this will most likely form the principal system discharge point

-     determine the distribution of other old mine entrances which lie a little higher than the most probable discharge point; if roof falls or other obstructions prevent discharge at the latter point, these higher outflow routes might become active.

Performing the same sort of exercise in order to accurately identify possible migration pathways to adjoining aquifers may be rather more difficult. As Younger and Adams (1999) note, the presence of a migration pathway to an aquifer (e.g. a shaft or adit which has a damaged lining where it passes through an aquifer overlying the Coal Measures) is not in itself a sufficient condition for decant to occur; the requisite driving head must also be present. Furthermore, in the best-studied example of aquifer pollution following mine flooding (relating to the Magnesian Limestone south of the Butterknowle Fault in County Durham, NE England) recent detailed analysis has revealed the presence of widespread cross-aquitard upflow from the mine workings to the overlying aquifer (Williams 2001; Neymeyer 2003). Overall migration pathways in such cases are very diffuse and difficult to attribute to any one pollutant source zone (i.e. deep mine or body of opencast backfill).

Using reasoning along the above lines, it is possible to identify possible post-closure migration pathways for leachates generated within bodies of opencast backfill. This in turn allows identification of the locations and nature of any possible receptors for migrating leachates, including both groundwaters and surface waters (burns, rivers, wetlands and lochs), which in addition to their possible supply uses may also host important aquatic ecosystems. Given the requirement of the Water Framework Directive that there be "no deterioration" in the status of such ecosystems, virtually any surface water course likely to receive opencast leachate ought to be regarded as being potentially at risk.

3.2. Estimating pollutant source strengths (Step 2)

In those few cases where the Step 1 investigations fail to identify any credible migration pathways to receptors, then Step 2 will not be implemented. For the vast majority of other cases, it will be necessary to make some evaluation of the potential of the opencast site to release List I or List II substances. Table 1 lists the substances of relevance.

In relation to **List I substances** (see Table 1), Hg and Cd *might* be anticipated to be released due to weathering of sulphide-rich strata (cf. Scottish Executive 2003). However, on the basis of a careful review of the literature and experiences in Scotland to date (see Annex II, Section AII.6.1), it has been concluded that disturbed geological materials in Scottish opencast sites are highly unlikely to leach either of these metals. In the case of Hg, not only is there a complete absence of likely source minerals, but the sorptive behaviour of the metal is such that it is almost wholly immobilised by electrostatic adhesion to ochre or other mineral surfaces wherever the pH exceeds 5 (see Younger *et al.* 2002, pp. 124 - 125). Cd is rather more mobile than Hg: throughout the ambient pH range typical of Scottish mine waters, Cd can be expected to be mobile, provided sources for it exist within the mine wastes. However, mineralogical records thankfully record only one occurrence of CdS (greenockite) in veins cutting Scottish Coal Measures rocks (Livingstone 2002; see also Annex II, Section AII.1.6). Nevertheless, the known propensity for trace quantities of Cd to occur in solid solution within sphalerite does mean that the abundant thin sphalerite-bearing quartz veins which cut coal-bearing strata in many parts of Scotland might locally provide a source of Cd. While sweeping assumptions of doom are not justifiable, where quartz veins carrying minor sphalerite are observed it would be wise to assay for the possible presence of Cd in future opencast leachates by means of laboratory column tests (i.e. a 'Level 4' evaluation as described below).

In relation to **List II substances** (see Table 1) there is no doubt that several "substances deleterious to the taste and potability of groundwater" (specifically Fe, Mn, Al and SO_4) are present in many opencast leachates. In some cases Zn, Ni, NH_4 and (occasionally) Cu may also be present. As the Groundwater Regulations 1998 require that "the input of List II substances to

groundwater … must be limited to avoid pollution of groundwater", there is no question that steps must be taken to evaluate the likely degree of mobilisation of these substances in backfill leachates. Step 2 of the assessment framework envisages operators taking a hierarchical approach to such evaluations.

At least one of the two lowest levels of evaluation (Levels 1 and 2) **should be applied in all cases without exception**. *Level 1 evaluation* can only be applied where hydrochemical data can be obtained from former opencast sites nearby, which worked the same seam(s) of coal within a closely similar hydrogeological setting (cf Figure 3). Where the similarities are sufficiently close, data from these sites can be regarded as reasonable analogues upon which to base a *prima facie* evaluation of the pollutant release potential associated with further opencasting, backfilling and restoration activities. One would expect the hydrochemical data to comprise samples of any surface discharges of backfill leachate, plus samples of leachate collected from boreholes completed within (or in the strata underlying) the backfill, wherever feasible. In the absence of such data, the alternative base-level evaluation of pollutant release potential is the 'Level 2' evaluation as described below.

Geological screening of pollutant release potential (Level 2 evaluation) can be applied in its own right, or as a means of corroborating a tentative Level 1 evaluation. Given that ideal analogues for proposed opencast sites will not always exist, Level 2 evaluation is likely to be the default option in most cases. The first task in undertaking *Level 2 evaluation* is to review site stratigraphy, and indeed the stratigraphy of any surrounding mined ground in which the water table might be lowered due to site dewatering (see 3.3.1). Fortunately the coalfields of Scotland are extraordinarily well mapped, and hence stratigraphic review of a proposed site will often be readily accomplished using published maps and memoirs of the British Geological Survey[7]. The aim of the review is to determine whether:

(a) any known 'marine bands' (i.e. horizons containing fossils of marine or brackish-water associations) outcrop within the site boundaries or within the zone predicted to be affected by site dewatering, and

(b) whether any of the seams to be mined at the site lie in relatively close stratigraphic proximity ($\pm$ 100m in the local stratigraphic sequence) to a regionally-recognised marine band (even if structural and erosional history has removed the marine band itself from the local outcrop).

Where either or both of (a) and (b) are answered in the affirmative, it is concluded that the proposed opencast site is likely to generate acidic leachates. Such acidic leachates are invariably rich in Fe and SO_4 (which are released to solution by dissolution of pyrite, the principal acid-generating reaction; see Annex II, Section AII.1.1). Once waters are acidic, dissolution of other sulphides is accelerated and desorption of a wide range of other metals (Zn, Ni etc) from mineral surfaces is expedited. Where acidic leachates are generated, therefore, there is no doubt that List II substances will be released into solution also. Where either of (a) or (b) are true, therefore, it is essential that the source strength estimation proceeds with a Level 3 evaluation.

In the unlikely absence of reliable mapping of marine bands, or where (a) and (b) above are answered in the negative, the next recourse within Level 2 evaluation is to examine seam sulphur data. As was noted previously, it is commonly found that the most prolific acidity-releasing rocks in Scottish coal-bearing sequences occur in close proximity to the seams themselves. This is not surprising in view of the fact that buried organic matter, which was on its way to becoming coal, created highly reducing conditions in the subsurface which favoured precipitation of pyrite in the enclosing sediments during burial diagenesis. For present purposes, these common field relations mean that sulphur content values for the coal seams within a site can be used as a proxy measure of overburden sulphur content. Furthermore, substantial quantities of pyritic seam-split material

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[7] 1:50,000 scale maps suffice for a preliminary evaluation, although 1:10,000 scale maps will be more useful for detailed site-scale evaluations.

tend to be removed from the coal by washing on site and the residues backfilled along with waste rock. Thus although the coal itself is taken off site, there is still some value in knowing the sulphur content of the seam *per se*.

Because of its commercial importance, the sulphur content of coal is routinely measured by opencast developers. At present, the Scottish coal market is dominated by the demand from the Longannet and Cockenzie power stations, which still lack flue-gas desulphurisation (FGD) facilities, and which therefore insist on coal feedstock containing less than 1 wt% S_{tot}[8]. Some Scottish opencast sites supply coal to the Alcan plant in Northumberland, which at the time of writing has a slightly higher total sulphur limit of 1.2 wt% S_{tot}. Enactment of the full provisions of the EU Large Combustion Plant Directive[9] would result in the emission limit values for the coal-burning plants not fitted with FGD being lowered to as little as 0.2 wt% S_{tot}; were such a low limit to become the market norm, virtually the entire UK coal industry would become obsolete (and thus the present document would become obsolete also). As matters presently stand, it seems unlikely that many future opencast sites in Scotland will work coals containing much more than 1 wt% S_{tot}. Given the degree of avoidance of high-sulphur strata which these market pressures are promoting, the scope for future opencast developments to lead to violation of the Groundwater Regulations 1998 is clearly limited. Ironically, if FGD is fitted to Longannet and Cockenzie, so that they can accept higher sulphur coals in future, the risk that sites will work more acid-generating coal-bearing sequences will increase again, and the provisions of this document will become ever more relevant.

Given that opencast developers are keenly interested in seam sulphur contents, most of them tend to maintain their own databases of relevant data. In addition, both TES Bretby and (to a lesser extent) the Coal Authority also hold substantial databases which include such information. It must be stressed that wt% S_{tot} varies not only between seams, but also spatially within seams. Although the spatial distribution of sulphur is sometimes described as utterly random, this is not in fact the case, for it tends to display geostatistical behaviour typical of regionalised variables. This does mean, however, that proximity to the site under consideration is an important consideration when considering what can be gleaned from archival records of seam sulphur content.

While the most useful data are <u>pyritic sulphur</u> contents, these are nowhere like as abundant as simple <u>total sulphur</u> content data. The relationship between total sulphur and pyritic sulphur is rather complex (Casagrande 1987). Below about 0.5 wt% S_{tot}, virtually all of the sulphur present in a coal seam will be organic, which does not contribute to the generation of acidic leachates. As the wt% S_{tot} climbs towards 1, the proportion of the total accounted for by pyrite increases markedly, and at higher wt% S_{tot} values 50% or more of the sulphur is likely to be pyritic. There is thus a direct, if rather fuzzy, correlation between wt% S_{tot} and acidity production potential. Younger (2000, 2001) has explored some of the practical implications of this correlation for deep-mine drainage in Scotland. One would expect the implications to be similar for opencast sites, given that they are even more accessible to atmospheric oxygen than abandoned deep mines. Table 3 provides a summary of the deduced correlation between total sulphur content and the concentrations of total iron in site drainage waters.

As Fe is rather less mobile than Mn and SO_4 at all pHs, and is less mobile even than Al when pH is less than 4.5, the key conclusion to be drawn from Table 3 is that **release of List II substances is to be anticipated wherever the total sulphur content of the worked seams exceeds about 1 wt% S_{tot}.** This in turn implies that any Level 2 evaluation revealing the presence of seams with wt% $S_{tot} > 1$ ought to trigger Level 3 evaluations.

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[8] wt% S_{tot} = percentage by weight total sulphur
[9] Which will not be fully activated as long as the current UK "Plan for the Reduction of Emissions into the Air of Sulphur Dioxide and Oxides of Nitrogen from Existing Large Combustion Plants" remains in force.

Table 3: Relationship between total sulphur content of worked coal seams and anticipated iron concentrations in backfill leachates (adapted after Younger 2002b).

Total sulphur content (wt %)	Predicted Fe concentrations (mg/l)	
	Maximum[a]	Long-term average[b]
< 1	0.01 - 0.5	< 0.02
1 - 2	0.5 - 100	0.02 - 10
2 - 3	100 - 350	10 - 35
3 - 4	350 - 1200	35 - 120
4 - 5	1200 - 1500	120 - 150

[a] as anticipated in leachates developing within oxidised backfill (of the indicated sulphur content) which has recently been submerged for the first time.
[b] anticipated from mature bodies of opencast backfill for which mitigation measures as described in Section 3.5 have either not yet been implemented, or have not been implemented successfully.

Except where the Level 2 assessment has 'screened sites out' on the grounds that they pose no appreciable risk to groundwater quality (i.e. by concluding that wt% S_{tot} will nowhere exceed a threshold value of 1% within the backfill), a *Level 3 evaluation* will be required. Although it shares some philosophical grounding with US-style acid-base accounting (see Annex II, Section AII.3.2), the methodology proposed here is more explicitly mineralogical in focus. The reasons for this are twofold: (i) to develop reproducible, scientific data in support of a formal procedure of conceptual model development (as opposed to *ad hoc* analyses which are difficult to contextualise beyond site boundaries), and (ii) to avoid false 'over counting' of either acidity or alkalinity, which is inherent in laboratory titrations using reagents which are far more concentrated than those encountered in real opencast environments. For instance, the US literature is replete with extended discussions on how to adapt titrations so that they account for the possibility of 'false' alkalinity readings arising from the presence of siderite (see Kleinmann 2000). However, the stoichiometric arguments upon which such 'corrections' for siderite are based have great potential to be confounded by the presence of ankerite, which is occurs widely as a fracture wall lining in Scottish coal-bearing strata (Younger *in press*). As was made clear in Section AII.1.3, ankerite does possess net-neutralisation capacity, so that 'correcting' it out of the alkalinity quotient of a rock sample can result in a significant over-estimation of acidity production potential. As more and more 'corrections' are added to titrimetric methods in order to account for uncertainties in sample mineralogy, so the complexity and costs of the lab work escalate. It is far from clear whether the additional effort and expense are worthwhile, for as Kwong (2000) has rightly remarked: "Some of the modified methodologies have gone into such detail that the intent of ABA analysis as a screening tool, and the practicality of the procedure, is frequently overlooked".

A protocol for acid-base accounting suited to Scottish conditions has been developed by Schmolke (2003), building on two existing British Standards. This involves determination of total sulphur and pyritic sulphur according to BS1016 (1996, Section 106.5.1), and determination of acid neutralising capacity (ANC) according to BS1377 (1990, Part 3: C16). The maximum potential acidity (MPA) which a given rock could yield (i.e. the total amount of acidity which would be released to solution in the highly unlikely event that all of the pyritic sulphur which it contains were to be thoroughly oxidised and leached into solution) can be calculated by reference to the overall stoichiometry of pyrite oxidation and hydrolysis of ferric iron (i.e. the combined effects of equations A1, A3 and A4 in Annex II). The literature on how to translate pyritic sulphur contents into equivalent acidities has been reviewed in a manual edited by Kleinmann (2000, pp 74 - 79). The current consensus is that for every 0.1 wt% of pyritic sulphur in one kilogramme of a given rock, 3.125 grammes of MPA (expressed as $CaCO_3$ equivalent) would be produced were this rock to be completely oxidised and thoroughly leached. This in turn means that the same rock would need to possess more than 3.125 g/kg of ANC if it were to produce a non-acidic leachate under laboratory conditions. It must be emphasised that **the simple comparison of ANC with MPA is not in itself a diagnostic**

criterion for whether a given body of backfill will produce acidic leachates. This is because pyrite oxidation only occurs at significant rates above the water table, whereas the dissolution of the minerals which contribute to the overall ANC can occur anywhere in the system, above or below the water table (see Annex II, Section AII.1.5.). Flow system characteristics are therefore every bit as important as the ANC/MPA balance. Whilst not discouraging attempts to perform ABA according to the British Standards listed above, therefore, it is here proposed that ABA is <u>not</u> used in isolation. Rather it should be seen as simply one of a number of optional assays which can be deployed in the pursuit of obtaining a rational overview of the likely pollutant release behaviour of a given body of broken rock: the development of a well-constrained conceptual model remains the overall objective, and all source strength estimation techniques must be the servant of this objective.

It is possible to avoid the many problems besetting titrimetric approaches to the estimation of acidity production potential by making direct determinations of sample mineralogy. For the fine-grained rocks most likely to release acidity to opencast leachates, mineralogical determinations cannot be achieved by optical means (neither with naked eye nor using standard petrological microscopes). While scanning electron microscopy can be used, this requires significant expert judgement in addition to elaborate sample preparation. It is also possible to digest samples, chemically analyse the digestates and reconstruct the original mineralogy by back-calculation. However, for routine mineralogical determinations on samples of potentially acid-generating mudstones, siltstones or fine-grained sandstones, the most appropriate technique is widely acknowledged to be Quantitative X-Ray Diffractometry (QXRD). In recent years, QXRD, using the "standard-less" Rietveld Method, has improved greatly in precision, with the proportions of component minerals now being measurable to within $\pm$ 1wt% (which is sufficiently precise for ABA purposes. (This is a marked improvement on the "5 - 20%" error values previously quoted in Kleinmann (2000, p. 79) for the previous generation of XRD machines used in the USA prior to the year 2000). QXRD is now offered commercially by a number of highly-rated labs in the UK[10]; furthermore, it requires far less sample (only about 15g) and costs no more than far more complicated titrimetric ABA analyses. Direct determination of the quantities of key pollutant source and sink minerals within samples of overburden material provides the basis for making an unequivocal judgement of the likelihood that these materials will release List II substances when they are comminuted and oxidised during opencasting. QXRD has the advantage that it is a non-destructive method of analysis, so that further geochemical testing can be carried out on the same samples if the mineralogical determinations reveal, for instance, the presence of sphalerite (which might host the List I substance Cd). Having determined the weight percentage composition of the various minerals in a sample, it is relatively easy to use published weathering rates for the various pollutant-generating and pollutant-attenuating minerals (Younger *et al.* 2002 pp 98 - 112) to make forward predictions of the likely fate of List II substances in the rock in question, depending on whether it will be lodged above or below the water table in the long-term (see Section AII.1.5).

It has been found that only particles less than about 4mm in diameter are actually reactive enough to contribute materially to pollutant release under field conditions (Annex II, Section AII.3.4). In the context of opencast coal mining in Scotland, this observation has the important practical consequence that virtually all pollutant release will be associated with those lithologies which tend to break up into fragments smaller than 4mm diameter as a result of the physical attrition to which the are subjected during stripping and backfilling. This in turn means that the lithologies most demanding our attention are the mudstones and siltstones. It happens that other lithologies (coals and sandstones) in typical overburden sequences are routinely cored in any case during site exploration (Annex I, Section AI.2).

It is here recommended that, in order to support the direct evaluation of acidity production potential on mineralogical grounds, cores be taken also in a representative sample of the mudstones and siltstones on the site, focusing especially on those which give the highest natural gamma readings

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[10] including a number of Scottish university labs and the central labs of the British Geological Survey

(as these are typically the most pyritic; see Thomas 2002, p. 206). The number of boreholes already used in typical opencast exploration is more than adequate for pollution prediction purposes; all that is required is that mudstones and shales be sampled as well as the sandstones and coals which are already cored in most cases. Sub-samples of some of the sandstones (especially the most friable ones, which are therefore most likely to yield particles $\leq$ 4mm in diameter during mining) and of seam roof, floor and split materials should also be sent for QXRD along with the mudstones and shales. As the costs of making these extra determinations must be borne by the operator, it is proposed that they be free to propose to SEPA a QXRD sampling / analysis density in proportion to the findings of the Level 2 evaluation. Obviously core retrieval and analysis will most likely be undertaken during the early stages of site exploration, so operators need to make early contact with SEPA to ensure sampling densities will be acceptable.

In most cases QXRD will provide sufficient information to allow an assessment of pollution production potential within the overall site conceptual model developed under Step 1. However, where the QXRD reveals that there is sphalerite or (far less likely) greenockite present in a particular sample, this should be analysed for Cd. This can be readily done on the very same samples (usually in the same labs) using X-Ray Fluorescence (XRF) spectrometry. If Cd is detected, a Level 4 evaluation will be necessary. Otherwise, this level of sample investigation will suffice, save where the operator themselves or SEPA (following receipt of the Step 3 results) recommends Level 4 evaluations to support numerical modelling of the site.

Leaching column experiments (*Level 4 evaluations*) are proposed only in cases where numerical site modelling is deemed necessary in order to reduce predictive uncertainties for sites which are still favoured for development, despite initial indications of a high pollution risk. Given the state of the coal market in Scotland, it is not considered likely that many sites will fall into this category. However, cases in which local authorities and / or SEPA favour opencasting as a means of cleaning up existing polluted mine sites could fall into this category. Where this level of detail is necessary, laboratory leaching column experiments may be used to obtain information on leachate generation processes.

An example of an appropriate laboratory leaching column is shown in Figure AII-2 (Annex II). Based on exhaustive experimental work recently concluded by the second author, the use of a leaching column such as that shown in Figure AII-2 is recommended in preference to the use of a more elaborate 'humidity cell' apparatus (e.g. Figure AII-1). Scientifically, this recommendation reflects the finding that the so-called 'primary' release rate obtained from humidity cell tests depends significantly upon which of the many variations of the methodology are actually used, also on the way in which the data are interpreted. Which of the 'release rates' measured by humidity cells (if any) corresponds logically to the actual 'primary' release rate of the same material under field conditions remains unknown (and is probably unknowable). It therefore follows that running humidity cell tests in order to directly predict actual mine site pollutant release rates is probably futile. While humidity cell testing can help to assess the differences in reactivity between various waste rock / tailings materials under the standard conditions of one particular testing regime, there is no evidence to suggest that it is any better than a Level 3 evaluation in reaching this conclusion. Considering the difficulties in interpreting data from humidity cell tests and the large financial investment involved in running them for periods of many months, it is probably more prudent to abandon this type of test work completely in favour of simpler evaluations of weathering rates which can be scaled-up to simulate field weathering rates using the procedures developed by Malmström *et al.* (2000), and expanded upon by Younger *et al.* (2002) and Banwart *et al.* (2002). To this end, the use of leaching columns such as that shown in Figure AII-2 are preferable. Further details on how to set up and run an effective laboratory leaching column are provided by Gandy and Evans (2002) and Banwart *et al.* (2002). Given that they are unlikely to be used with any great frequency, these details are not reproduced here.

3.3. Initial assessment of risks (Step 3)

It will often be sensible to initiate the Step 3 risk assessment while the Step 2 characterisation activities are still under way, so that the latter can be adapted if necessary to better reflect the emerging assessment of risk posed by the site.

The key task in Step 3 is to make quantitative estimates of the possible scale of pollutant migration to receptors. These assessments must include some evaluation of the magnitude of the attendant uncertainties. A number of alternative approaches to quantifying key elements of the overall conceptual model may be deployed, depending on the nature of the site in question and the gravity of the consequences of any mis-judgement.

In some case it will be appropriate to develop quantitative geochemical models in order to evaluate alternative scenarios of pollutant release and immobilisation. The great advantage of this approach is that it allows direct and explicit allowances to be made for natural attenuation processes which tend to operate over time-scales appropriate to the time it takes leachates to follow migration pathways in the field (months or years) even though they are rarely evident in even the longest laboratory tests. This type of assessment can be made using publicly-available geochemical modelling software, such as the US Geological Survey's "PHREEQC" package. Examples of the application of this and allied codes to predictive analysis of mine water quality evolution in Scotland (albeit for deep mine systems) have been published by Younger *et al.* (1995) and Chen *et al.* (1999). In all such applications, whether to deep or opencast mines, the key requirement is the definition of a flow path (which will have been done under Step 1 in this framework) followed by the specification of initial water quality (which may simply be rain water) and of any reactants which are deemed likely to affect water quality. The most challenging problem in all cases is specifying the 'reactive surface area' of the minerals involved in the various pollutant release / attenuation reactions. It is possible to estimate this parameter at field scale by back-calculation of observed tracer release from minerals with well-constrained weathering rates (see Banwart *et al.* 2002, p. 143). However, in many case it will be necessary to work with assumed values, and to account for the uncertainties which this inevitably introduces by means of sensitivity analyses (e.g. Younger *et al.* 1995; Gandy and Evans 2002).

In cases where the hydrogeological setting is relatively simple and very well characterised, less rigorous (and thus less demanding) mass balance calculations may yield sufficiently robust risk estimates, thus obviating the need for proceeding to full-blown geochemical modelling. The type of approach which may be appropriate is typified by the "performance model" for iron mobilisation in deep mine workings presented by Younger (2000). The details of that particular model would require adaptation prior to application to opencast environments, but the type of logic employed is transferable. However, at the present time no such generic performance model exists; its development might be fostered by the activities promoted this document. This does not mean that there is any reason why developers should not derive and apply site-specific "source-pathway-receptor" analyses in order to assess the risks attached to their proposals. As long as the analyses are based on scientific arguments, and the algorithms are open to scrutiny, a wide range of possible approaches is potentially defensible.

Whatever method of analysis is used, the end-point of the assessment must be the categorisation of a given site <u>as a whole</u> according to whether it poses a low, medium, high or very high risk, as defined in Step 3. Having reached this point, the way is clear for identifying and implementing mitigation strategies (Step 5). However, in the event that the currently-available information remains insufficient to support definitive conclusions, the assessment enters an iteration loop (Figure 1), beginning with Step 4.

3.4. Refining the conceptual model (Step 4)

This step does not demand any great logistical definition beyond that stated in the framework itself: it is essentially a period of reflection and uncertainty analysis, in which the weaknesses of the

existing conceptual model and risk assessments are systematically identified, and plans are laid for addressing them.

3.5. Planning for risk mitigation (Step 5)

Fortunately, the planning of risk mitigation for opencast coal mines is greatly facilitated by the recent publication of guidance by the European Commission's 5th RTD Framework project 'PIRAMID', entitled "Engineering guidelines for the passive remediation of acidic and/or metalliferous mine drainage and similar wastewaters" (PIRAMID Consortium 2003). In addition, comprehensive catchment management guidelines for the control of mining impacts on the freshwater environment have also been published by a second European Commission 5th Framework project 'ERMITE' (Wolkersdorfer and Younger 2004). Table 4 identifies which of the recommended elements of the Step 5 risk mitigation plan can be designed by reference to these two sets of guidelines

Various measures which might be taken to minimise the risk of long-term pollutant release from the opencast backfill can be identified. One obvious suggestion is to ensure that coal washery fines (which are often significantly enriched in pyrite compared to the coal itself) are "buried deep and drowned", i.e. emplaced low down in the void, below the eventual post-mining water table level. It may in some case be feasible to ensure a high water table position within the backfill (which is to be encouraged from the point of view of pollution prevention, all other things being equal (such as satisfaction of ground stability requirements)) by either leaving unmined pillars of rock between the highwall and any nearby old workings, or by deliberately plugging any connections to old workings with mass concrete plugs (e.g. Younger and Wolkersdorfer 2004). The feasibility of such approaches on any given site will depend on local hydrogeological conditions, and (given the irretrievability of deeply buried plugs) on a risk analysis of the consequences of future failure of any man-made structures.

Other suggestions which are occasionally made include the addition of limestone to the backfill. The costs involved in achieving this will often be prohibitive in even the most favourable geological circumstances. In Scotland this is even more of a non-starter than usual because there is a general shortage of limestone in the country: dolomites and dolomitic limestones abound, but very few calcite-rich limestones occur north of the English border. Of the few which exist, these are mainly unavailable for quarrying (e.g. the Durness Limestone of the northwest Highlands, which hosts innumerable karstic and floral SSSIs). In fact there are a number of other 'possible' mitigation measures which cannot be implemented at a cost compatible with keeping the opencasting operation economically viable. For instance, an obvious postulate is to selectively handle known high sulphur overburden beds, so that they are stored temporarily outside of the void and to be backfilled in the very deepest part of the void when the most opportune time arrives. In practical terms, this would mean lifting the material from the void in much the same way that coal is lifted, which would be prohibitively expensive in virtually all cases. (Exceptions might be where high sulphur materials are present mainly in the box cut, which will in any case be removed from the void and can in principle be backfilled at some opportune juncture).

In reality the most promising measures for preventing post-closure migration of polluted leachates all relate to the process of restoration. Compaction of backfill helps to reduce permeability (e.g. Rimmer and Younger 2003). Capping the backfill with clay can help minimise infiltration, thus arresting pollutant transport. It has recently been shown that even partial capping of pyritic spoil, where the clay is emplaced only over known "hot spots" in which pyrite oxidation is known (from borehole sampling) to be occurring, can be very effective at reducing the overall pollutant load leaving a body of coal mine waste (Gandy and Younger 2003). If the backfill is predicted to decant at surface (e.g. Figure 2, Figure 3f), it ought to be possible to manipulate the final restoration contours so that a passive treatment wetland system can be located strategically where it can easily capture toe leachates.

Table 4: Sources of practical guidance on the design of risk mitigation measures.

Risk mitigation measures	Hydrologically defensive mine planning	Design of dewatering operations	Minimise pollutant migration[c]	Planning for active or passive treatment	Design of restoration measures
Purpose	Minimising unnecessary water ingress	Minimise potential for water contamination	Minimise migration of pollutants to receptors	Prevent migration of pollutants to receptors	Minimise potential for water contamination
When to be implemented?	During mining and restoration	Before and during mining	Throughout mine life cycle	During and after mining	During mining and restoration
Covered in ERMITE Guidelines?[a]	Y	Y	Y	Y	N
Covered in PIRAMID Guidelines?[b]	N	N	Y	Y	Y
Other sources of guidance	Robins and Younger (2002)	Norton (1983); Minett *et al.* (1986)	IPPC Bureau (2003)	Younger *et al.* (2002)	Rimmer and Younger (1997)

[a] i.e. Younger and Wolkersdorfer (2004). [b] i.e. PIRAMID Consortium (2003).
[c] i.e. for List I substances, <u>prevent</u> migration altogether; for List II minimise as far as possible

Given the abundant availability of up-to-date guidance (Table 4), risk mitigation planning ought not to be the most onerous of the Steps in this assessment framework. The sooner it is commenced, however, the better, as relatively inexpensive measures implemented during mining can often save fortunes post-closure.

3.6. *Contingency planning and monitoring (Step 6)*

Having completed all other steps in the risk assessment framework, the preparation of a contingency plan ought to be one of the simplest tasks to fulfil. It essentially comprises a listing of residual risks of low likelihood, which cannot be eliminated altogether but which can be allowed for by means of preparing responses in advance for problems which might arise. Given that planning authorities in Scotland typically require a minimum 5-year after care period for opencast sites, in any case (occasionally extended to 10 years where assessed risks are high) there will typically be a substantial period of time within which opencast operators can reassure themselves that residual risks are unlikely ever to become manifest.

Given all of the uncertainties in predicting the hydrological and geochemical behaviour of opencast backfill post-closure, it is recommended that precautionary monitoring of post-restoration water table recovery and associated changes in water quality (surface and subsurface) be undertaken during the after-care period (and beyond, in the event that residual risks emerge rather than vanish). In addition to monitoring any boreholes used to observe drawdown and recovery outside of the site boundaries (as determined using Table 2) it is recommended that sample points within the restored opencast be monitored as follows:

(i)   all surface outflows of leachate to be monitored monthly for flow rate, and six-monthly for water quality in accordance with the current parameter suite (see Table 5).

(ii)   water levels to be monitored monthly, and water quality to be determined every 6 months, for a network of boreholes sunk through backfill to the water table[11]. At least

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[11] where no water table is present within the backfill, monitoring will be in at least two boreholes located down hydraulic -gradient in the groundwater flow system into which leachate is believed to migrate.

three boreholes will be required (one of which ought to be up-gradient of the site to allow identification of background groundwater quality, unaffected by the site in question), and where the site exceeds 30 ha in total area, one additional borehole should be added for every further 10 ha (or part thereof) of site area.

Table 5: Parameter suite for groundwater quality monitoring on opencast sites to allow assessment of compliance with the Groundwater Regulations 1998

Name of analyte suite	Comprising …	When measured?
1. Field determinands	pH, temp., conductivity	Until monitoring finally ceases
2. Key List II metals	Fe, Mn, Al, Zn, Ni, Cu	Until borehole water levels and concentrations of all metals been stable[b] for at least 12 months; thereafter monitoring can be reduced to annual unless a 20% increase in one of the List II metals or non-metals is detected.
3. List II non-metals	F, NH_4^+	
3. Selected anions	SO_4, [a]HCO_3	
4. List I metals	Cd, Hg	Cd: ONLY if Zn > 5 mg/l in last sample Hg: ONLY if pH < 4.5 and Cd > 50 μg/l in last sample

[a] only measured if pH > 4.5, and specified only because it is essential to determining the acidity-alkalinity balance of the waters. [b] i.e. varying within $\pm$ 20% of mean value, with no discernible trend of rise or fall over time.

In terms of the analytical suite, it is not necessary to monitor for the full catalogue of List I and List II substances unless other evidence (which will have been unearthed in Steps 1 - 3) makes this desirable. Rather, a time-variant **parameter suite** is proposed in Table 5 which will allow ready identification of any changes in key geochemical and hydrological processes, whilst also allowing for a gradual decline in the number of monitored parameters over time, as confidence in the site risk assessment grows. Provided that no significant risk indicators manifest a resurgence, it is envisaged that monitoring may be allowed to lapse altogether after 5 or 10 years (as determined by the local authority in liaison with SEPA).

3.7. Cost implications for opencast operators

The following additional costs can be anticipated by operators due to implementation of this risk assessment framework:

(i)   Staff training costs, to ensure they are demonstrably capable of fulfilling the extra site characterisation duties arising from Steps 1 and 3 of the assessment framework; based on known costs of established CPD courses, these are unlikely to exceed a one-off cost in excess of £2500 for any one company.

(ii)   Staff costs for time spent implementing the assessment steps. It is difficult to estimate these costs, as they may vary from a few thousand pounds for a small site, up to a few tens of thousands of pounds for very large sites.

(iii)   Rock sample analysis costs, particularly those relating to Level 3 evaluations (and, in rare cases, Level 4 evaluations): these are estimated at £150 / sample for either ABA or QXRD, and £300 if both techniques are used[12]. Operators could reasonably expect to incur

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[12] Based on quotations from BGS laboratory, Keyworth, and Bodycote Materials Testing Laboratory, Edinburgh, in line with the protocol developed by Schmolke (2003).

additional costs on the order of £500 per hectare of site in order to implement the recommendations of Section 3.2.

(iv)   <u>Monitoring borehole installation costs</u>, including
        a. those installed before extraction commences to monitor for the off-site effects of dewatering; these costs are estimated at around £7500 per Ml/d pumped, and
        b. those installed within the site backfill during the restoration phase, to facilitate water monitoring post-closure; these are estimated at around £1500 per hectare of site area.

(v)   <u>Costs of post-closure water sample analysis</u>, which except for the most problematic sites are unlikely to exceed around £2500 per annum per site, or about £12500 in total during a 5 year post-closure after-care period.

Of course in some cases companies will prefer to outsource some of the services required to implement this framework; in this case the cost elements (i) and (ii) above will be replaced by sub-contracting costs, which can be expected to cost somewhere in the range £7,500 to £75,000 per site (depending on site size and complexity).

4. Conclusions

The Groundwater Regulations 1998 fully transpose into Scottish law the requirements of the European Directive on Groundwater (80/68/EEC). They forbid the introduction of certain substances ("List I") into groundwater, and placing strict limitations on the extent to which other substances ("List II") may be permitted to enter groundwater. Scottish opencast mining poses little risk of introducing List I substances into groundwater, but it has substantial potential to lead to the migration of several 'List II' substances. A code of practice recently published by the Scottish Executive (2003) provides sound advice on the control of pollution from non-mineral pollution sources during opencasting operations, and identifies the need for the "assessment of the risk of release of List I and List II substances from storage in stockpiles or backfilling, prior to the activity being undertaken". It is to provide an assessment framework to assist in the fulfilment of this need that the present document has been prepared.

- To inform the development of an appropriate assessment framework, a state-of-the-art review of the key issues surrounding opencasting and groundwater quality has been prepared (Annex II).

Drawing upon lessons learned from the literature review, and taking advantage of significant recent advances in research and practice in the UK and Europe, a six-step assessment framework has been developed. The principal elements of the assessment frameworks are as follows:
    Step 1: Outline conceptual model development
    Step 2: Source strength estimation
    Step 3: Initial risk assessment
    Step 4: Conceptual model refinement
    Step 5: Risk mitigation planning
    Step 6: Contingency planning and post-closure monitoring.

Detailed advice is offered on how best to: gather the requisite data; develop the necessary conceptual model; undertake the required risk assessments; and plan for effective risk mitigation. Estimates of the additional costs to opencast developers of implementing this framework have been prepared; these are considered marginal in relation to the overall costs of site exploration and planning, reflecting the desire to develop a framework which has proportionate benefits and costs.

Annex I: Brief summary of opencast coal mining operations as typically undertaken in Scotland

A1.1. Opencast mining - overview and terminology

Opencast extraction now accounts for all coal production in Scotland. Its success is in part predicated on the relatively favourable overburden-to-coal ratios typical of many Scottish coal basins, and to the evolution of reliable, tried-and-tested methods of workings which were pioneered and later perfected under nationalisation by the former British Coal Opencast Executive (BCOE) and the many contractors who worked the individual opencast coal sites (OCCSs) on its behalf.

Notwithstanding its mature state of development, openly-published accounts of opencast coal mining procedures are surprisingly difficult to find. Hartman and Mutmansky (2002, pp. 197 - 208) provide an engineering description of the principal activities involved in opencasting, though their description is based largely on USA experiences. Grimshaw (1992) has written a history of British opencast coal mining which provides lots of interesting examples of real sites, although it is not written as an engineering text and is frustratingly devoid of references for further reading. A rather different history of opencasting in Britain, documenting the controversies which have best many OCCS developments, is provided by Beynon *et al.* (2000), who also reference much of the unpublished literature of the BCOE itself, and documents relating to from numerous OCCS public enquiries. Finally, Younger *et al.* (1992, pp. 32 - 35 & 46 - 48) have briefly reviewed opencast mining procedures from an hydrogeological perspective.

The brief summary presented here is no substitute for first-hand experience of opencast coal mining, and any reader who does not possess such experience is counselled to make amends. The following review is based primarily on the author's own experience of a range of Scottish OCCSs.

Most OCCS operations in Scotland have been (and will be) undertaken in the following sequence:
- (i)   Exploration and reserve quantification
- (ii) Site preparation and creation of separate 'long-term' stores of topsoil and of overburden removed from the 'box-cut' (see Table AI-1).
- (iii) Coaling and backfilling, which are essentially undertaken simultaneously in virtually all cases.
- (iv) Site restoration and after-care, which involves re-emplacement of the box-cut overburden and of the stored topsoil. Where the restoration is to agricultural land or forest (as is very often the case) the previously stored soil may well require reconditioning prior to cultivation. In many case, restoration is undertaken in phases while coaling is still in progress elsewhere on the site, so that "restoration" is not necessarily a stand-alone activity which occurs only at the end of the active life of an OCCS.

Each of the above activities is discussed in further detail below.

Like most other forms of mining, opencasting has generated a vernacular of its own. Some of the more commonly-used terms (which are used in the main body of this document) are defined in Table AI-1 below.

A1.2. Exploration and reserve quantification

Given the scale of investment required to develop a profitable opencast operation, developers invariably invest significantly in prior exploration of proposed OCCSs. A typical site investigation for a proposed OCCS in Scotland at the present day tends to involve the following two-step process:
- (i)   An array of open-hole drilled boreholes will be installed across the entire site on approximately 100m-spacings. (Spacings as small as 10m were formerly used by the BCOE, but in the present market such fine-resolution drilling would tend only to be employed under special circumstances, such as where the cost of working are attractively low (e.g. where the seams are shallow and will thus require little overburden stripping) but the coal quality is subject to considerable uncertainty (e.g. sulphur

contents are in excess of current market norms, so that careful planning for blending would be necessary)).

(ii)   In each of these open-holes, geophysical logging will be undertaken (normally for calliper, natural gamma and short- and long-spaced gamma-gamma density logs) in order to identify the positions of the target coal seams, and also to identify sandstones which may need geotechnical testing to determine "rippability" etc.

(iii)   Using the information obtained from logging of these open holes, intervals for coring will be identified (i.e. seams, immediate roof and floor strata and any indurated sandstones). Secondary boreholes will then be drilled immediately alongside between 25% to 35% of the original boreholes, with open-holing to the tops of the previously-identified coring intervals followed by coring of the interval, with open-holing commencing thereafter to the top of the next coring interval. (Again, in special circumstances such as those outlined under (i) above, all of the boreholes on the site may be fully cored, in which case step (ii) may be dispensed with).

The information obtained from investigations of this type are then used to produce quantitative models of coal yield, overburden stripping requirements and design elevations for pavement seam elevations and post-restoration contours. It is during this phase of OCCS planning that data collection to support Step 2 of the site assessment framework can be achieved most economically. Having designed the OCCS in this manner, it is as this point that the necessary planning applications will be made, followed by applications for relevant permits (discharge consents etc). The site is now ready for development.

Table AI-1: Opencast coal mining terminology - a selective glossary.
(Terms illustrated in Figure AI-1 below are marked with an asterisk).

Term	Definition …
Advance dewatering	Prevention of groundwater ingress to an OCCS by means of pumping boreholes or shafts located outside of the working area.
Backfill *	Broken waste rock material occupying an opencasted void.
Baffle banks *	A mound of tipped overburden or stored soil placed around the perimeter of an OCCS to limit external visibility and / or provide a partial barrier to site noise and dust.
Bench *	A more-or-less horizontal surface of rock, developed within an OCCS, as the highwall retreats in response to excavation.
Box-cut	The first void sunk on an opencast site after topsoil stripping, creating the space for subsequent highwall retreat with synchronous dumping of backfill.
Coaling	The extraction of coal from an opencast void.
Dewatering	Removal of water from an OCCS or its environs to prevent flooding of the void.
Highwall *	The artificially-cut cliff of intact strata which gradually retreats during the working of an OCCS.
Loose wall *	The frontal slope of the main mound of backfill in an opencast mine which faces onto the highwall.
OCCS	Common abbreviation of "opencast coal site"
Overburden	Rocks other than coal which must be removed in order to gain access to the coal deposits to be worked in an OCCS
Pavement (seam)	The lowest seam worked in a given OCCS
Restoration	Completing the transformation of the upper surface of a body of backfill to its intended after-use.
Soil store	A pile of topsoil stored during the working of an opencast site for later spreading during restoration.
Sump	The lowest point in a site (or part of a site), which is often used to collect water which is then removed by dewatering pumps.
Topsoil	Biologically-productive soil (comprising the A through C horizons) naturally present on a site before opencasting.

A1.3. Site preparation and creation of 'long-term' stores

The first step in developing an opencast site is to selectively remove topsoil. This biologically productive soil will be stored on a part of the site not intended to be mined. (During restoration, this soil store will be used as the primary source of 'soil-forming materials' for emplacement over the backfill and re-conditioning to achieve fertile conditions; see A1.5). For sites which intersect the pre-mining water table, advance dewatering measures will typically be installed before or during topsoil stripping. In order to create the space for simultaneous coaling and backfilling during the remaining active life of the site, the initial box-cut (see Table AI-1) will then be sunk (usually to the pavement seam) creating a void within which bench working can proceed (see A1.4). The box-cut materials will typically be emplaced circumferentially around the perimeter of the OCCS to form a 'baffle bank', which not only acts as a store for the box-cut materials but also serves a number of desirable ends typically specified in planning conditions imposed by permitting authorities, such as screening the site to minimise noise, dust and visual intrusion impacts on the surrounding area. The soil store and the baffle banks are generally the only significant temporary stores of earth materials on an OCCS, for economic coaling is generally predicated on simultaneous casting of overburden in the part of the void most recently worked, which lies immediately behind the active working bench (see below).

Figure AI-1: Sketch cross-section of an active opencast site illustrating how backfill emplacement occurs simultaneously with coal production at the toe of the highwall.
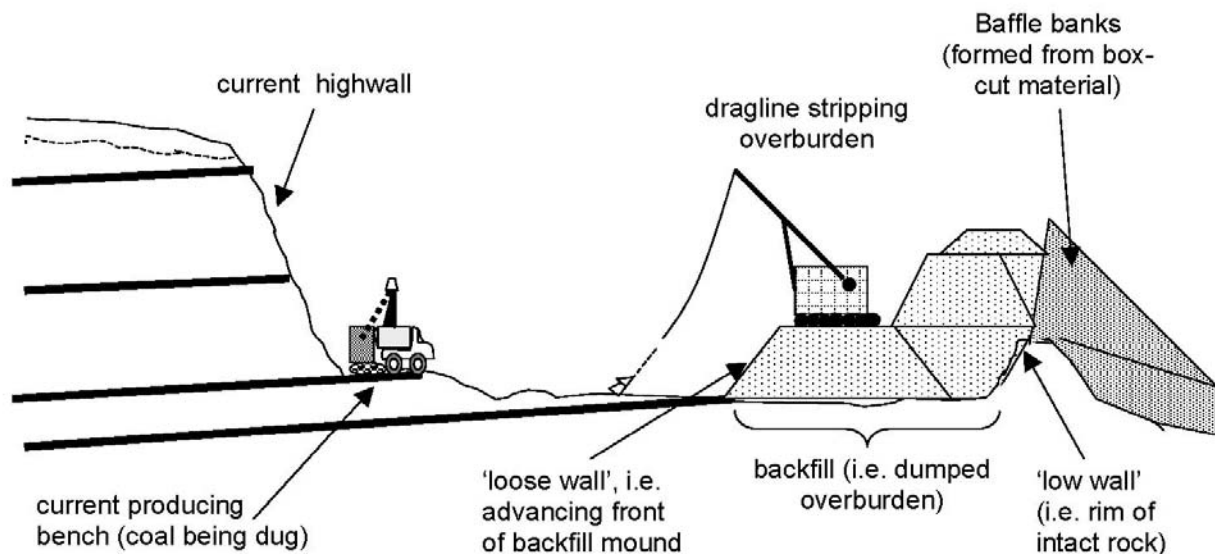
A1.4. Coaling and backfilling

In Scottish opencast coal mining, profitable site operations are achieved by eliminating temporary storage of excavated overburden. The goal of mine design is essentially to avoid expensive double-handling of broken rock materials by ensuring that overburden can be stripped from the intact seams and dumped behind the working bench in a single operation. The manner in which this is achieved is illustrated in Figure AI-1. With the box-cut material usefully deployed to form the site baffle banks, all later overburden stripping is followed immediately by dumping onto the last-worked area. The advancing front of dumped material is known as the 'loose wall', and between

this and the ever-retreating 'cliff' of soon-to-be-excavated strata lies the current producing bench. An appreciation of the short space of time between lifting overburden and dumping it to advance the loose wall is essential to appreciate the focus of the assessment framework presented in Section 2 on problems associated with backfill. The only other category of material meriting special attention is coal washery wastes. These are often disposed within the backfill (sometimes in purpose-dug settling ponds within the fill material), and they tend to be preferentially enriched in pyritic sulphur relative to the worked seams and are thus often highly geochemically reactive.

A1.5. Site restoration and after-care

In most cases, phased restoration of an OCCS will commence even as coaling continues elsewhere on the site, for there is no merit (financial or otherwise) in delaying restoration until all coaling has ended. The three principal activities comprising restoration are as follows (Rimmer and Younger 1997):

(1) Preparation of the final landform: Most OCCS have sufficient broken overburden material to completely fill the voids which they generate. This is because the gross porosity of backfill is significantly greater than that of the unmined strata. It should thus be possible to resotre most OCCS so that the pre-mining contours are closely reproduced. Only where the overburden-to-coal ratio is in single figures is a residual void likely to be left. Where the floor of such a void lies below the water table, or where it is lined with clay or compacted mudstone, such a residual void may form a pit lake. (Few pit lakes currently exist in Scotland, and few are likely to be formed in future). Within the backfill itself, drainage infrastructure is often incorporated during final restoration, in order to achieve optimum soil moisture conditions for revegetation.

(2) Soil establishment: Generally, the material in the soil store will be used for this purpose. It may require substantial re-conditioning before it is suitably productive to support cultivation. Where insufficient topsoil is available to cover the final landform, it may be necessary to either import more topsoil, or else to use mudstones already present on the site as 'soil-forming materials'. Some of the geochemical implications of soil importation and / or in situ soil formation from SFMs are described by the PIRAMID Consortium (2003).

(3) Long-term maintenance of the revegetated area, which will normally be carried out by farmers or foresters who have little background in the intricacies of spoil as an earth material. Relatively little attention is devoted to this topic in the literature, beyond documentation of geotechnical stability problems which can arise during later agricultural use of restored spoils (see Groenewold and Rehm 1982).

To these three activities, a further one is now often added in cases where the objective is not agricultural after-use but the establishment of areas of high nature conservation value, namely ecosystem establishment. Still in its relative infancy, to date this has most often involved attempts to create species-rich grasslands. Although such efforts can obviously draw upon transferable skills from the agricultural and forestry sectors, the aim of achieving significantly greater species diversity (20 - 25 species / m^2) than one would expect in an agricultural sward (2 - 3 species / m^2) is challenging in practice. The fact that most soils formed from mine spoil tend to contain less N than natural soils might be considered a benefit in this context, as it exerts ecological pressures which potentially open the soil system to colonisation by a range of plants which are better adapted such conditions. However, the N content can be so low that many of these plants struggle to become established, leading to dominance by clover and other plants capable of fixing atmospheric N.

Planning authorities in Scotland typically require a minimum 5-year after-care period for former opencast sites. This will occasionally be extended to 10 years where assessed pollution / stability risks are high. After-care plans typically include contingency measures to be implemented in the event that prior assessment and planning activities prove to have been mistaken in some way.

34

Annex II: State-of-the-art review of potential pollutant release and attenuation processes associated with opencast coal sites in Scotland

AII.1. Pollutant release and attenuation: geochemical processes in coal-bearing strata affected by opencast mining

AII.1.1. Oxidation of coal-bearing strata and pyrite weathering

It is impossible to rationally assess the environmental risks associated with mining a given sequence of coal-bearing strata unless the assessor has a good grasp of the mineralogy and geochemical behaviour of those strata. To provide a summary of the key issues, the following paragraphs have been adapted from a recent, comprehensive review of the geochemical impacts of coal mining (Younger *in press*). For a more thorough appreciation of the complexities of mine water geochemistry, the reader is encouraged to refer to the various papers cited below, especially the comprehensive account of mine water chemistry given by Younger, Banwart and Hedin (2002).

In essence, all of the environmental problems associated with opencast coal mining (including the disposal of the lithic wastes to which it gives rise) can be attributed to a single cause: the incompatibility between naturally 'reduced' coal-bearing strata and the strongly oxidising surface / near-surface atmosphere. A state of 'dynamic redox equilibrium' has typically developed over geological time at the interface between the atmosphere and unmined coal-bearing strata (cf. Strömberg and Banwart 1994): opencast coal mining fundamentally disturbs this equilibrium, introducing oxygen to formerly anoxic rocks, and thus prompting the degradation of coal and of various redox-sensitive minerals. This in turn leads to the mobilisation of a number of redox-sensitive elements, including carbon itself (principally in the gaseous phase), nitrogen (often in the form of the dissolved ammonium ion), iron and manganese.

Coal itself is a naturally-reduced geochemical entity which undergoes substantial changes in the presence of atmospheric O_2 (Thomas 2002). The various coal macerals vary in their susceptibility to oxidation, but vitrinite and inertinite are both known to oxidise significantly. Oxidation of the coal leads to breakdown into fine-grained particles, which are then prone to hydration. In some cases, oxidation leads inexorably to the spontaneous combustion of the coal, which occurs because the sorption of O_2 gas onto fresh coal surfaces is a strongly exothermic process. Where the supply of O_2 is great enough for extensive heating of the coal surface to occur, but still sufficiently limited that the heat is not dissipated safely into the atmosphere, the coal can reach ignition point (Thomas 2002). This is the root cause of uncontrolled fires in opencast backfill, which are themselves testament to the continued intrusion of oxygen to rather surprising depths within the waste rock material. The *in situ* burning of coal within backfill results in substantial changes in the mineralogy, geochemistry and mechanical properties of the waste rock material. Visually, the most obvious change is that burnt opencast backfill is red in colour (as opposed to the greys and blacks of unburnt backfill), due to the high-temperature ($\leq 1000^{o}C$) oxidation of all Fe-bearing minerals in the strata to form hematite (Fe_2O_3). Large fused masses of oxidised rock up to 10m in diameter are frequently found in burnt backfill (Rimmer and Younger 1997), and these clearly have far greater mechanical strength than the pre-existing loose waste rock. Gaseous emissions from burning zones in coal comprise mixtures of noxious oxides of C, S and N, as well as smoke particles containing polycyclic aromatic hydrocarbons (PAHs) and other potentially toxic organic compounds (Lee *et al.* 1981). Condensates containing these compounds often accumulate either within, or above, burned masses of backfill, and they can be released into water during subsequent wetting of the backfill. Though rarely encountered in practice, it is perhaps worth noting that elevated ammonium concentrations are sometimes associated with backfill-fire quench waters, being derived from the dissolution of sal-ammoniac (NH_4Cl).

The risk of spontaneous combustion of coal is known to be exacerbated by the presence of pyrite and marcasite[13] (both of which have the chemical formula FeS_2), the oxidation of which is strongly exothermic. Pyrite swells on oxidation (Bell 1996), promoting the disintegration of the coal and shale clasts within which it occurs, and rendering them ever-more permeable to O_2. The first step in pyrite oxidation is the seven-electron oxidation reaction which raises S from its average redox state within pyrite of -1 up to the +6 state which it maintains in the SO_4^{2-} molecule. With such a profound shift in oxidation state, the maintenance of electroneutrality demands hydrolysis, which results in the release of protons (H^+) into solution:

$$FeS_{2(s)} + 3.5\ O_{2(g)} + H_2O \rightarrow Fe^{2+}_{(aq)} + 2SO_4^{2-}_{(aq)} + 2H^+_{(aq)} \qquad \text{(A1)}$$
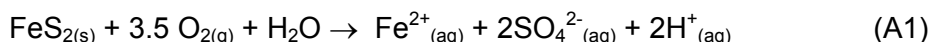
This reaction (along with others in the overall chain of pyrite oxidation) result in this process being one of the most acid-generating in Nature (e.g. Appelo and Postma 1993; Backes et al. 1993).

Since pyrite oxidation has been the subject of numerous in-depth reviews in recent years (e.g. Backes et al.. 1986; Evangelou 1995; Younger et al.. 2002; Rimstidt et al. 2003) no detailed exposé is warranted here. However, a few words on the occurrence of pyrite in coal-bearing sequences are appropriate in this context. It is well-known that pyrite is present at greatest concentrations in those coals (and adjacent shales) which were deposited and /or underwent early diagenesis in the presence of marine or brackish, tidal waters (Williams and Keith 1963; Caruccio and Ferm 1974; Casagrande 1987; Morrison *et al.* 1990; Younger 1994, 2000). High total S contents therefore correlate quite closely with stratigraphic proximity between a given coal seam and the nearest bed in the sequence containing marine fossils (most notably bivalves of *Lingula* sp.). Total S contents are of economic importance in the marketing of coal, and hence data on this parameter are widely available, typically being measured as part of a suite of determinations termed 'ultimate analysis' (Thomas 2002). Total S contents in coals vary within rather restricted limits: few coals will have less than 0.1 percent by weight (wt%) total S, and contents in the range 0.5 - 1.5 wt% S are typical of most marketable coals. "High S" coals typically have total S contents between 3 and 5 wt%, with some extreme examples reaching 10 wt% S (for instance, in marine-influenced coals of Jurassic age at Brora in Sutherland; Younger 2001). Given that a considerable component of the total S in any coal is actually "organic S" bound within the coal macerals (which does not contribute to acidification of mine waters) it is important to determine what proportion of the total S is actually attributable to the presence of pyrite (i.e. what is the wt% of 'pyritic S'). Although direct determinations of pyritic S are occasionally made, pyritic S content data are far less common than total S data. It is therefore desirable to be able to estimate the likely pyritic S content of a sample of coal-bearing strata for which only the total S content is known. The following 'rules' (derived from Casagrande 1987) are useful in this regard:
   - organic S is a major component of total S in all coals, and where total S is less than about 0.5 wt%, organic S probably accounts for nearly all of the S present
   - as the total S rises significantly above 0.5 wt%, pyrite comes to gradually account for more and more of the total S content, though even in the highest-S coals, pyritic S seldom accounts for more than 50% of total S

Armed with the above information, it is possible to make a preliminary assessment of the pollution potential of specific coal seams and associated strata simply on the basis of their total S content (see Section 3 for further discussion).

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[13] In this document, the discussion will refer primarily to pyrite, though much of what is stated is also applicable to marcasite.

AII.1.2. Storage of acidity in efflorescent salts

Equations such as (A1) above imply that the oxidative dissolution of pyrite is congruent, directly liberating Fe^{2+}, SO_4^{2-} and H^+ to solution. However, in the common circumstance that water is insufficiently abundant to immediately transport the oxidation products away from the mineral surfaces, pyrite oxidation more commonly results initially in the accumulation of various hydroxysulphate evaporite minerals. These minerals form efflorescent crusts, typically white and yellow in colour, on the surfaces of pyrite-rich coals and mudstones, and they effectively 'store' the oxidation products in a readily-soluble form until some hydrological event delivers sufficient water to dissolve and transport them away. Because pyrite often occurs in mudstones, where Al-bearing clay minerals are in contact with acidic pyrite oxidation waters, Al is frequently released from the clays and is also stored in these hydroxysulphate phases. When these minerals finally dissolve, they result in abrupt and extreme increases in dissolved acidity. For this reason they have been termed "acid generating salts" (AGS) (Bayless and Olyphant 1993). A wide range of AGS have been identified temporarily storing acidity on both surface exposures of opencast backfill (e.g. Bayless and Olyphant 1993), whence they release dissolved acidity whenever a significant rainfall event follows a dry spell during which the AGS have been accumulating. They are also commonly found in underground coal workings (e.g. Younger 2000), where they typically store acidity until the dewatering pumps are finally withdrawn and the workings containing the AGS flood. Table AII-1 lists some of the AGS commonly encountered in coal mine settings.

AII.1.3. The role of iron-rich carbonate minerals: siderite and ankerite

Pyrite is not the only Fe^{2+}-bearing mineral which undergoes oxidative weathering when coal-bearing strata are exposed to the atmosphere through mining. The Fe-rich carbonate minerals siderite ($FeCO_3$) and ankerite ($Ca(Mg,Fe)(CO_3)_2$) are also important reservoirs of Fe^{2+}.   Siderite is a relatively common mineral in the shales and sandstones of Carboniferous coal-bearing sequences in Scotland, in which it usually occurs in the form of ellipsoidal concretions orientated with their long axes parallel to bedding. In some localities in central Scotland the density of siderite concretions in specific mudstone horizons is so high that they have historically been worth mining as Fe ores. (The fact that these siderite masses could often be won from the same mines which produced the coal needed to smelt them meant that the capital costs of developing them were relatively modest, partly explaining why these low-grade ores played such an important role in the development of the Scottish iron and steel industries).

In terms of the environmental impacts of coal mine drainage, siderite plays an ambiguous role (see, for instance, Morrison *et al.* 1990, and works cited therein). On the one hand, the acidity released by pyrite oxidation can be transported into the presence of siderite, which will dissolve to neutralise it:

$$FeCO_{3(s)} + H^+_{(aq)} \rightarrow Fe^{2+}_{(aq)} + HCO_3^-{}_{(aq)} \qquad\qquad (A2)$$
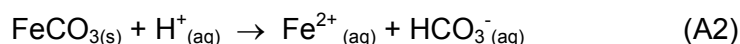
Further neutralisation (i.e. consumption of H^+) occurs when the Fe^{2+} released by reaction (A2) is oxidised to Fe^{3+} (e.g. Singer and Stumm 1970):

$$Fe^{2+}_{(aq)} + 0.25\ O_{2(g)} + H^+_{(aq)} \rightarrow\ Fe^{3+}_{(aq)} + 0.5\ H_2O \qquad\qquad (A3)$$

However, when the Fe^{3+} produced in reaction (A3) undergoes hydrolysis, three moles of proton acidity (H+) are released for every mole of Fe^{3+} hydrolysed:

$$Fe^{3+}_{(aq)} + 3\ H_2O \rightarrow\ Fe(^{3+})(OH)_{3(s)} + 3\ H^+_{(aq)} \qquad\qquad (A4)$$

If the overall budget of protons consumed and released by reactions (A2) through (A4) is calculated, it is at once evident that siderite dissolution has no net neutralising capacity for acidic waters (see also Morrison et al.. 1990 and Younger 1998). Thus siderite dissolution can only

37

provide local neutralisation, and that only where O_2 is absent from the system. Moreover, reaction (A2) suggests that siderite can be a net source of dissolved Fe in coal mine drainage waters, a point which is corroborated by evidence from geochemical modelling of certain neutral-pH coal mine and ironstone mine waters in northern England (e.g. Younger 1995; 2002b).

Table AII-1: Acid generating salts' (AGS) commonly found in and on sulphur-rich opencast backfill. These AGS effectively store the products of pyrite oxidation in solid form until later submergence and dissolution. (adapted after Younger *et al.* 2002).

Mineral Phase	Formula
(i) Iron-rich minerals	
Aluminocopiapite	$Fe(II)Fe(III)_2Al_2(SO_4)_6(OH)_2 \cdot 20H_2O$
Copiapite	$Fe(II)Fe(III)_4(SO_4)_6(OH)_2 \cdot 20H_2O$
Coquimbite	$Fe(III)_2(SO_4)_3 \cdot 9H_2O$
"ferrosic hydroxide"[a] ("Interlayered Green Rust"[a])	$Fe(II)_4Fe(III)_2(OH)_{12}(SO_4 \cdot 3H^2O)$

Jarosite
  Hydronium Jarosite
  Natrojarosite
  Jarosite
	
	$(H_3O^+)Fe(III)_3(SO_4)_2(OH)_6$
	$NaFe(III)_3(SO_4)_2(OH)_6$
	$KFe(III)_3(SO_4)_2(OH)_6$

Kornelite	$Fe(III)_2(SO_4)_3 \cdot 7H_2O$
Melanterite	$Fe(II)SO_4 \cdot 7H_2O$
Rhomboclase	$(H_3O^+)Fe(III)(SO_4)_2 \cdot 3H_2O$
Römerite	$Fe(II)Fe(III)_2(SO_4)_4 \cdot 14H_2O$
Rozenite	$Fe(II)SO_4 \cdot 4H_2O$
Schwertmannite	$Fe_8O_8(OH)_6SO_4$
Szomolnokite	$FeSO_4 \cdot H_2O$
(ii) Al-rich minerals	

Alunite
  amorphous
  crystalline
	
	$KAl_3(SO_4)_2(OH)_6$ (am)
	$KAl_3(SO_4)_2(OH)_6$ (c)

Alunogen	$Al_2(SO_4)_3 \cdot 17H_2O$

Basaluminite
  amorphous
  crystalline
	
	$Al_4SO_4(OH)_{10} \cdot 5H_2O$ (am)
	$Al_4SO_4(OH)_{10} \cdot 5H_2O$ (c)

Halotrichite	$Fe(II)Al_2(SO_4)_4 \cdot 22H_2O$
Pickeringite	$MgAl_2(SO_4)_4 \cdot 22H_2O$

[a] note: these are not formal mineral names, but are used in the literature to describe this poorly-defined phase

In contrast to siderite, ankerite does not commonly occur in any great quantities in the mudstones and sandstones of coal-bearing sequences (Hawkes and Smythe 1937; Smythe and Dunham 1947). Rather, ankerite tends to occur in the form of patchy films lining the surfaces of the many cleats (i.e. micro-joints) which cut the coal itself (Figure AII-2b, Annex II). Cleat is considered to develop relatively late in the post-burial history of coal-forming peats (Thomas 2002), and thus

ankerite is typically a product of late-stage diagenesis. As Thomas (2002) has recently pointed out, most sedimentological studies of coal-bearing sequences focus on the depositional fabrics of the clastic sediments rather than the coals themselves. Furthermore, insofar as diagenesis is considered at all in such studies, the focus is on its early stages, and there is thus a tendency to ignore post-cleat mineralisation phases. For both of these reasons, ankerite occurrence tends to be under-reported in the scientific literature relating to coal-bearing sequences. However, the preferential location of ankerite on coal cleat surfaces means that it is ideally located to participate in rock-water interactions, and it therefore often makes a marked contribution to the hydrochemical evolution of ground waters flowing through coal-bearing strata, most notably in the elevation of dissolved Mg2+ concentrations.

Given that siderite dissolution has no overall neutralisation effect on acidic waters, it is important to assess whether the same applies to ankerite. The dissolution of one mole of stoichiometric ankerite ($Ca(Mg,Fe)(CO_3)_2$) consumes two moles of protons:

$$Ca(Mg,Fe)(CO_3)_{2(s)} + 2H^+_{(aq)} \rightarrow Ca^{2+} + 0.5Mg^{2+} + 0.5\ Fe^{2+}_{(aq)} + 2HCO_3^-_{(aq)} \qquad (A5)$$

When the 0.5 moles of Fe^{2+} liberated in (A5) are oxidised to form $Fe^{3+}_{(aq)}$, a further 0.5 moles of protons are consumed (according to reaction (A3)). The ultimate hydrolysis of this Fe^{3+} to form solid $Fe(OH)_3$ (according to reaction (A4)) will yield 1.5 moles of H^+. Thus the overall proton balance ($\Sigma(H^+)$) can be summarised as: $\Sigma(H^+) = -2 - 0.5 + 1.5 = -1$. Hence, for every mole of stoichiometric ankerite dissolved, there is a net consumption of one mole of proton acidity. Given that most ankerites contain rather less Fe^{2+} than Mg^{2+} (Smythe and Dunham 1947), in the majority of cases, the net consumption of protons will be even greater. Thus, in contrast to the case of siderite, ankerite does indeed possess net-neutralisation potential for acidic waters.

AII.1.4. Mineral sources of neutralisation

Besides these Fe^{2+}-rich carbonate minerals, those carbonate minerals which are most common in the geological record as a whole, i.e. calcite and dolomite, tend to be rather under-represented in Scottish coal-bearing sequences. There is a lack of these minerals both as primary deposits (i.e. in the form of discrete beds of limestone or dolostone) and as secondary phases (i.e. as intergranular cements or post-depositional mineral veins). Contrary to common assumptions, this is so even in the so-called 'Limestone Coal Group', which is distinguished by the presence of beds of dolostone (mis-named limestones in the literature) above and below the coal-bearing sequence, but which lacks limestone units in close proximity to any of the widely-worked coals. (This lack of calcite and dolomite contrasts significantly with conditions in the Appalachian coalfields of the USA where limestones are relatively frequent in the coal-bearing succession (Kleinmann 2000), a point which needs to be borne in mind when considering the merits of various mine water quality prediction techniques originating there).

Besides the carbonates, it is important not to overlook the geochemical importance of the many silicate minerals which are abundantly present in coal-bearing sequences in Scotland. Quartz is one of the most common minerals in the sedimentary rocks typically associated with coal. A relatively inert mineral, quartz does not significantly affect the acid-alkali balance of pore waters, and to that extent it is 'environmentally neutral'. The mudstones and sandstones associated with coals contain many aluminosilicate minerals of geochemical significance, most notably clays (predominantly kaolinite and illite; Thomas 2002) and muscovite. The various clay minerals are generally rather benign in their environmental geochemistry, acting as powerful sorbents which serve to limit the mobility of ecotoxic metals at circum-neutral pH (Stumm and Morgan 1996). Furthermore, in the presence of acidic waters the clay minerals and many other aluminosilicate minerals (notably feldspars and amphiboles) will dissolve, providing valuable buffering capacity, especially in circumstances where no neutralising carbonate phases are available (Banwart *et al.* 2002; Younger *et al.* 2002). It should be noted, however, that dissolution of these minerals in very

acidic mine waters is the principal source of dissolved Al, which is strongly ecotoxic. Since dissolved Al is trivalent, it hydrolyses in a manner similar to Fe^{3+} (reaction (A4) above):

$$Al^{3+}_{(aq)} + 3H_2O \rightarrow Al(OH)_{3(s)} + 3 H^+_{(aq)} \qquad\qquad (A6)$$
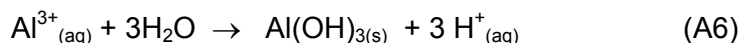
The solid phase resulting from this reaction typically forms a white precipitate, which on account of its low density often forms impressive masses of white foam where the water from which it is precipitating is subjected to turbulent eddies. Generally, this white precipitate is X-ray amorphous, though if it is allowed to settle for periods in excess of six months crystalline forms of Al(OH)3, most notably gibbsite, are often identified. (Other Al-rich secondary minerals are (hydroxy-) sulphates, formed by evaporation of mine waters which have undergone reaction (A6); see Table AII-1 for several examples).

AII.1.5. "Location is everything"

It is essential to emphasise that the neutralisation of water by the dissolution of carbonate and silicate minerals can occur <u>anywhere</u> in the subsurface, both above and below the water table, whereas significant dissolution of pyrite only occurs in micro-oxic and oxic pores above the water table (including, in season, those pores which lie within the zone of water table fluctuation and are therefore submerged following periods of groundwater recharge). In other words, acidity can only be generated in significant quantities in the unsaturated zone, whereas alkalinity can be generated in both the unsaturated and saturated zones. For this reason it is fair to assume that water which experiences a very long retention time in the saturated zone is far more likely to have a circum-neutral pH and excess alkalinity than unsaturated zone drainage in the same strata.

Because of this, issues of 'order of encounter' and hydraulic residence time in different portions of the subsurface have great importance for the likely quality of water in a body of opencast backfill. Some of the ramifications of this point are explored in further detail in Section AII.2 below.

AII.1.6. Possible sources of List I and List II substances in Scottish opencast drainage

The overall scope and implications of the Groundwater Regulations 1998 are discussed in Section AII.6.2 below. One of the most important elements of the Regulations is their categorisation of various pollutants into two lists. In this section a brief review of these pollutants is offered, with comments on their known occurrence and possible sources in opencast drainage. These are presented in Table AII-2, which distinguishes between:

  i.   Substances likely to leach from disturbed geological materials, which are thus specifically associated with opencast coal mining, and
  ii.   Substances of anthropogenic origin which occur in many industrial settings and are thus not peculiar to opencast coal mining.

Guidelines for managing the risks associated with (ii) above either already exist (Scottish Executive 2002) or are the subject of current consultations (Scottish Executive 2003). As was stated in Section 1.1, the substances corresponding to (i) above are the particular focus of this document.

As Table AII-2 makes clear, **there is very little risk of List I substances being released from strata disturbed by opencast mining in Scotland**. Although both mercury and cadmium <u>have</u> been reported at very low concentrations in coal mine waters elsewhere in the world, such occurrences tend to reflect the presence of hydrothermal veins containing cinnabar (HgS) or Greenockite (CdS). For instance Loredo *et al.* (2002) describe Hg contamination in the Coal Measures of northern Spain associated with vein mineralisation. The most comprehensive mineralogical compendium ever produced for Scotland (Livingstone 2002) reveals that veins containing Hg and Cd minerals are very few in Scotland. Although one such occurrence (at the

Table AII-2: Possible sources of List and List II substances in opencast coal mine drainage

Substance	Likely to leach from disturbed geological materials? [Y/N] (commentary)	Possible anthropogenic sources in opencast coal mines …
List I Substances:		
Organohalogen compounds	N	Electrical equipment; certain solvents (de-greasing agents); certain wood preservatives and pesticides.
Organophosphorous compounds	N	Selected pesticides
Organotin compounds	N	**None**
Other carcinogens, mutagens aqnd teratogens	N	**None**
Mercury (Hg) and its compounds	N (very remote chance of leaching, in conditions never yet encountered in Scotland; see text for further discussion)	Electrical equipment
Cadmium (Cd) and its compounds	N (small chance of leaching, under conditions not yet reported in Scotland; see text for further discussion)	Certain types of battery
Mineral oils and hydrocarbons	N	Spills and leaks from heavy machinery; spills and leaks from storage tanks on site
Cyanides	N	**None** (given that cyanide-based herbicides are no longer used)
List II substances:		
Zinc and its compounds	Y (found locally at ≤ 20 mg/l; see text)	Various vehicle components; roofing materials.
Copper and its compounds	N (but rarely noted at ≤ 1 mg/l; see text)	Electrical components.
Nickel and its compounds	Y (found locally at ≤ 5 mg/l; see text)	Few. Some may be released from batteries, and from abrasion of galvanised steel.
Other metals and metalloids[a] and their compounds	N (not impossible for small amounts (≤ 50 μg/l) of Cr, Pb, B, As, Se, Th and Co)	Few. Some may be released from electrical components, steel plating and (for Pb) roofing materials.
Biocides not listed in List I	N	Herbicides and pesticides
Substances damaging to the taste, odour and potability of groundwater	Y (Fe, Mn, Al and SO_4 are very common and affect taste and potability)	Some Fe from weathering of iron and steel components in plant.
Organic compounds of silicon	N	Some sealants used on plant.
Fluorides	Y (rarely present at ≤ 5 mg/l; see text)	Few. Possibly associated with welding wastes.
Ammonia	Y (can occur ≤ 50 mg/l; see text)	Few. Associated with certain types of explosives; fertilisers used during restoration.
Nitrites	N	Few. Associated with certain types of explosives; fertilisers used during restoration.

[a] those specified in List II are: Chromium, Lead, Tin, Barium, Beryllium, Boron, Uranium, Titanium, Molybdenum, Antimony, Arsenic, Selenium, Silver, Tellurium, Thallium, Cobalt and Vanadium.

former Hilderston silver mine, 3 km NE of Bathgate) does adjoin formerly-mined Coal Measures (Livingstone 2002, p. 127), all other occurrences are associated with non-coal-bearing strata. Furthermore, of those water analyses from Scottish opencast sites which include both Hg and Cd in the analysis suite (some of which relate to highly acidic waters), the first author has yet to find an instance where either metal was present above detection limits[14]. It would thus be unreasonable to expect widespread leaching of Hg or Cd from opencast backfill in Scotland. Nevertheless, the widespread occurrence of sub-economic Pb-Zn vein mineralisation in the Scottish coalfields (Livingstone 2002) does give pause for thought in relation to Cd: it is well known that Cd substitutes for Zn within the sphalerite (ZnS) crystal lattice, which explains why dissolution of this mineral can lead to the presence of several $\mu g/l$ of Cd in certain circum-neutral mine waters in Northern England (Nuttall 1999). It is therefore not possible to absolutely dismiss the possibility that, where the local strata are cut by small Pb-Zn veins, some backfill leachates in Scotland *may* contain potentially detectable Cd.

The risk of occurrence of List II substances is rather more tangible. Evidence from water analyses at numerous opencast sites around the country reveal that the named metals Zn and Ni are occasionally present in opencast drainage waters at modest (though non-negligible) concentrations. Although little studied, it seems likely that trace quantities of their primary sulphides (sphalerite (ZnS) and millerite (NiS)) are widespread in Scottish coal-bearing strata. Millerite certainly occurs in close association with siderite nodules in the Coal Measures of Ayrshire (Anderson and Smythe 1942), in the Durham Coalfield of north-eastern England (Anderson and Smythe 1942; Dearman and Jones 1967) and (in greatest abundance) in the South Wales coalfields (North and Howarth 1928 ; Bevins 1994). Sphalerite has been proven to accompany millerite in the South Wales occurrences (North and Howarth 1928), and may also co-occur thus in Scotland. Clearly, dissolution of millerite and sphalerite in oxic waters could explain the occurrence of Ni and Zn in some opencast drainage waters. In addition, trace quantities of Zn and Ni are probably naturally present within the lattices of pyrite crystals in the Scottish coal-bearing sequences, and are thus released to solution during the oxidative dissolution of pyrite. Furthermore, the lowering of pH associated with pyrite oxidation tends to favour the mobilisation into solution of any Zn and Ni formerly held on mineral surfaces by sorption. Similar factors likely control the occurrence of copper, which is occasionally detected in acidic opencast drainage, though rarely at concentrations which would cause any problems in relation to drinking water supplies or for groundwater-fed freshwater ecosystems. All of the other metals and metalloids associated with opencast drainage tend to occur below detection limits in virtually all cases. Occasional detection of arsenic is reported, though never at very high concentrations, no doubt due to the efficiency of natural immobilisation of arsenic through sorption and / or co-precipitation reactions associated with freshly-precipitated ferric hydroxide.

One of the less explicitly defined categories of List II substances are those "which have a deleterious effect on the taste, odour and potential for potable use of groundwaters". In the case of opencast drainage waters, the principal substances in this category include the two principal products of pyrite oxidation, i.e. Fe and SO_4, plus two other very common metals: manganese (Mn) and aluminium (Al). Manganese is often rendered highly mobile in the low- to medium-pH and low Eh conditions typical of deeper zones of backfill at many opencast sites. Removal of Mn from polluted mine waters is an area of active research (e.g. Johnson 2002). To date there is a lack of 'proven' dependable *in situ* processes for Mn immobilisation. Given the present state of technology, therefore, once Mn is in solution there are no affordable options exist for halting its transport through the subsurface[15]. Although Al is a very common constituent of acidic coal mine drainage (as was mentioned in Section AII.1.4) it is far less likely than Mn to enter drinking water supplies simply because it becomes highly immobile once pH rises above 5. Any natural processes or

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[14] Although it is essential to bear in mind the old adage that "absence of evidence is not evidence of absence"

[15] Unaffordable options, such as destruction of the permeability of the ground to prevent <u>all</u> water movement do exist but would never withstand scrutiny during a cost-effectiveness appraisal.

engineering interventions which tend to raise the pH of opencast drainage above this threshold will definitely halt the transport of Al. Careful monitoring of Al mobility is nevertheless important, because it is extremely ecotoxic in its dissolved form, and can therefore be highly damaging to groundwater dependent terrestrial eco-systems and surface water bodies, which are defined as key groundwater receptors under the Water Framework Directive.

Given the very low maximum acceptable concentrations (MACs) for Mn and Fe in drinking waters (both < 0.3 mg/l), it is very often the case that opencast drainage waters would tend to render receiving aquifers unsuited for potable supply use with regard to these two parameters. This is a major issue in relation to implementing the key Article 4 objective of the Water Framework Directive to achieve Protected Area standards. This is because <u>any</u> derogation of a Drinking Water Protected Area (DWPA)[16] that would result in an increase in the level of treatment required prior to use as a drinking water supply would be a breach of this key objective. The situation with regard to sulphate (SO_4) is even more extreme: the MAC for SO_4 is 250 mg/l, whereas the average SO_4 concentration in Scottish coal mine waters is around 890 mg/l.

Amongst the non-metals in List II, fluorides are not very often found in opencast drainage in Scotland, their solubility typically being limited to a few mg/l by equilibrium with fluorite. Far more common is ammonia (in the form of the ammonium ion, NH_4^+) which has frequently been reported at concentrations ranging from below detection limits to several tens of mg/l, with concentrations in the range 1 - 10 mg/l being fairly common. As in the case of aluminium, the subsurface mobility of ammonium is greatly hindered by rising pH, for as the availability of the hydroxide ion increases, so too does de-protonation of the ammonium ion, leading to conversion of the remaining NH_3 to the gaseous phase, whence it is readily lost to the atmosphere upon aeration:

$$NH_4^+{}_{(aq)} + OH^-{}_{(aq)} \rightarrow H_2O + NH_{3(g)} \qquad\qquad (A5)$$
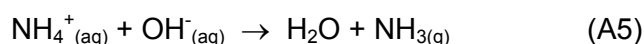
However, ammonium also resembles aluminium in being highly damaging to groundwater-dependent terrestrial ecosystems and surface water bodies, and therefore demands careful monitoring.

It is thus in relation to those List II substances which "have a deleterious effect on the taste, odour and potential for potable use of groundwaters" that the greatest challenge for opencast operators arises, for the "no deterioration" principle established by the Water Framework Directive may well be difficult to achieve unequivocally, given that groundwaters occurring within the coal-bearing strata themselves are considered to be 'groundwaters' within the meaning of the Groundwater Regulations 1998.

AII.2. Mining hydrogeological controls on pollutant loadings

AII.2.1. Hydrogeology of unworked coal-bearing strata in Scotland

In their natural state, most of the coal-bearing strata in Scotland do not form notable public supply aquifers (Robins 1990). However, given that geological units exist within coal bearing strata that are locally important as aquifers, and which have properties consistent with the current definition of DWPAs[16], coal-bearing strata are currently classified as aquifers under SEPA's Water Framework Directive characterisation process. Most of the Carboniferous strata in the Scottish coalfields were probably originally characterised by low hydraulic conductivities ($10^5 \le K \le 1$ m/d), with the high degree of sedimentary stratification and the combined effects of geological structure and geomorphology resulting also in a high degree of hydraulic confinement of the groundwater,

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[16] The current definition of a DWPA is a groundwater body providing > 10 m^3/d of drinking water as an average, or any groundwater body serving more than 50 persons; most of Scotland will fall within this definition, largely on account of the scale adopted for delineated groundwater bodies.

resulting in low values for the storage coefficient (typically $< 10^{-5}$). However, given the long history of deep mining in Scotland, there are very few situations where utterly native hydraulic properties have been preserved. Fracturing of super-incumbent strata due to collapse of historical mine voids at depth has resulted in many "un-mined" sequences of coal-bearing strata actually possessing considerably greater hydraulic conductivities and storativities than they probably had in prehistory. In reality, therefore, very few sequences of coal-bearing strata in Scotland can truly be regarded as being utterly unaffected by mining.

AII.2.2. Hydrogeological behaviour of opencast backfill

Where they have been directly disturbed by mining, the coal-bearing strata of Scotland often form significant 'man-made aquifers' (e.g. Robins and Younger 1996). A considerable literature exists on the hydrogeology and geochemistry of deep-mined coalfields in Scotland (see Younger 2001 for a review); however, relatively little has appeared in the public domain in relation to the hydrogeological impacts of opencast mining (Norton 1983; Robins 1990). It is nevertheless clear that open-cast coal mining profoundly changes the hydraulic properties of coal-bearing strata and also greatly alters their propensity for releasing pollutants to natural waters. For instance, interpretation of slug / bail test data from boreholes completed in opencast backfill in Fife indicate hydraulic conductivities ranging from around 1 m/d to more than 100 m/d, i.e. typically three to four orders of magnitude greater than the adjoining unmined strata. Although field measurements of storage coefficients in backfill are generally lacking, there is strong evidence that most bodies of backfill are hydraulically unconfined. Taken together with their generally high porosities, this suggests that storage coefficients in backfill are likely to exceed 0.01 and may locally attain values as high as 0.25.

Studies of the physical hydrology of opencast backfill are relatively rare worldwide. Nevertheless, recent years have seen some major advances in the our overall understanding of the hydrology of backfill and other deposits of waste rock (Younger *et al.* 2002), which can be summarised in the following four statements:
1. Bodies of opencast backfill often host subsurface flow systems of their own, functioning effectively as "perched aquifers" above the regional water table present in underlying / adjoining rocks (mined and un-mined).
2. Even where they do not host perched aquifers, bodies of backfill above the regional water table frequently function as zones of preferential recharge to deeper groundwater systems.
3. The heterogeneous nature of backfill gives rise to highly preferential subsurface flowpaths.
4. The physical instability of backfill is reflected in relatively rapid changes in surface and subsurface hydrological pathways

The existence of discrete, 'perched' water table aquifers within backfill is commonly observed. The most obvious cause of perching is simply impoundment of infiltrating water within permeable backfill above lower-permeability un-mined strata below. Depending on the local distribution of hydraulic conductivities, perched aquifers in backfill can play a significant role in catchment storage/flow attenuation processes, often feeding waters to underlying/adjacent groundwater systems over the longer term (between storm events and during periods of negative effective rainfall). However, multiple-level perching can also arise within the backfill, both due to inherent heterogeneities within the waste rock (see below) and also through the operation of some of various geochemical processes discussed in Section AII.1. In particular, the fact that iron is often readily available in the pore waters of backfill means that it can oxidise and precipitate along any interface between nearly-saturated fine-grained backfill and still-aerated coarse grained backfill. This gives rise to layers of ferric hydroxide which are commonly termed "hard pans". Besides providing a sink for iron and other metals within the waste rock pile, these hard pans frequently clog the pores, reducing permeability. Eventually, the hard pans become so lowly permeable that they begin to function as aquitard horizons, impeding further unsaturated flow downwards, and promoting locally saturated flow conditions above them, leading to lateral flow of perched groundwater towards the margins of the backfill, where (depending on site topography) the positions of the hard pans may be evidenced by the presence of spring lines and seepage faces.

Like all shallow aquifer systems, perched aquifers in backfill are highly sensitive to seasonal variations in groundwater recharge, and are thus prone to exhibit rapid fluctuations in water table elevation in the wake of rainfall events, even during summer periods when recharge to Scottish bedrock aquifers is usually minimal. Seasonal changes in flow direction within backfill are to be expected, reflecting the uneven variations in rates of water table rise across the entire body of backfill. The inhomogeneity of water table response to rainfall reflects the influence of preferential flowpaths which become active at different water table elevations.

The development of preferential groundwater flowpaths within backfill is a direct consequence of the waste rock disposal practices commonly used in opencast mining. In the broadest terms, the heterogeneity in sediment fabrics resulting from end-tipping and dragline spreading is reflected in widely ranging values of saturated hydraulic conductivity. The higher values of hydraulic conductivity tend to correspond to rubbly zones (i.e. the toes of successive loose wall positions during the progradation of the backfill), whereas lower values are more typical of what were originally running surfaces for draglines and dumper trucks on the temporary upper surface of the backfill. Consequently, hydraulic conductivity in backfill is often rather structured (cf. Newman *et al.*, 1997), which accounts for the demonstrably preferential flow pathways detected in many perched aquifers in these materials.

It is crucial to note that the predominant flow paths for water differ markedly between the unsaturated and saturated zones of the backfill. Since unsaturated hydraulic conductivity is a function of moisture content (e.g. Freeze and Cherry, 1979) the hydraulic conductivities of the moisture-retaining fine-grained layers will often be higher than that of the well-drained coarse layers above the water table. This means that *the fine-grained layers are the preferential flowpaths in the unsaturated zones of backfill* (e.g. Diodato and Parizek, 1994; Newman *et al*, 1997). For this reason, detailed numerical simulations of unsaturated flow through waste rock piles have often had to incorporate preferential flow pathways to accurately reproduce observed drainage and solute transport behaviour (e.g. Strömberg and Banwart, 1994; Eriksson and Destouni, 1997; Erikkson *et al.*, 1997). In a comparison of model simulations of lab and field tracer experiments with saturated and unsaturated waste rock, Erikkson *et al*. (1997) found that preferential flow pathways accounted for between 55% and 70% of the total water content of the backfill, with the remaining water moving only slowly. As a practical rule of thumb, it has been found that preferential flow through the fine-grained layers in the unsaturated zone of backfill is to be expected wherever the recharge rate (expressed in units of equivalent depth per unit time) is numerically less than the saturated hydraulic conductivity of the backfill (expressed in consistent units of length per unit time) (Wilson *et al.* 2000).

By contrast, below the water table, the saturated hydraulic conductivity of the coarse, rubbly zones is usually several orders of magnitude greater than that of the fine-grained layers, so that *the coarse layers are the preferential flowpaths in the saturated zones of backfill* (e.g. Aljoe, 1994; Aljoe and Hawkins, 1994). This latter observation explains why the hydraulic conductivities needed to satisfactorily model flow beneath the water table in backfill tend to be towards the higher end of the overall range. For instance, in simulating the effects of large bodies of surface mine backfill on regional ground water flow, Schwartz and Crowe (1985) found it appropriate to assign K values in the range 9×10^{-2} to 90 m.d^{-1} to the backfill. When simulating flow within a single body of backfill, it has sometimes been found necessary to locally assign hydraulic conductivities as high as 1000 m.d^{-1} in order to satisfactorily reproduce observed flows and water levels (Hawkins, 1994). In such settings, Aljoe (1994) noted that the aquifer operates like a "pseudokarst" system, with rapid transmission of water in a few erosional soil pipes (the formation of which is described by Groenewold and Rehm, 1982), draining water from the less permeable backfill mass which encloses them. As test pumping wells nearly always intercept the backfill mass rather than erosional pipes, borehole values for K will usually not be high enough to account for the flow rates occurring in the soil pipes. Insistence on using only field-measured values of K can lead to underestimates of backfill drainage rates by as much as 80% (Aljoe, 1994).

The contrasting modes of preferential flow above and below the water table have profound consequences for pollutant release and transport, since much of the most reactive pyrite is found in the fine-grained backfill, while coarse-grained fractions are often poor in buffering minerals. Hence acid generation is favoured above the water table, and its rapid transport with only limited buffering is favoured below the water table.

Settlement of backfill is the norm for a decade or so after infilling of an opencast mine void. The process of settlement is due to closer packing of the grains in response to self-loading by the sediment body. Closer grain packing means tighter pore necks and therefore a decrease in hydraulic conductivity. Disintegration of mudstone clasts in the presence of fresh water also leads to the release of fine-grained sediment, which can block pores and therefore serve to decrease the overall hydraulic conductivity. The net result of these processes has been investigated by several researchers. For instance, Groenewold and Rehm (1982) undertook borehole permeability tests in a suite of boreholes drilled into freshly re-contoured opencast backfill in the north-central USA, and then repeated the measurements one year later after settlement had occurred. In nearly all of the boreholes, hydraulic conductivity dropped by an order of magnitude over the year. From analysis of data obtained in various European countries, Skarzynska and Michalski (1999) reported that decreases of two orders of magnitude are by no means uncommon in opencast backfill. On the other hand, where rubbly zones impart a higher initial hydraulic conductivity to the backfill, water can flow so rapidly that it entrains silt and even sand grains, leading to the development of erosional pipes. These are the very features which impart "pseudokarstic" hydraulic properties to some backfills (e.g. Aljoe, 1994; Hawkins, 1994). When such pipes collapse, they form surface depressions which serve to intercept surface runoff on top of the backfill thus diverting even more of the effective rainfall into the subsurface (Groenewold and Rehm, 1982).

AII.2.3. Relationships between opencast sites and surrounding groundwater systems

The Scottish Ministers have recently been granted powers to introduce regulations to control water abstraction in Scotland[17]. Under these powers, it is possible that opencast dewatering operations will in future be more closely regulated than at present. As such, the assessment of pumping and drawdown regimes in the vicinity of active and proposed opencast sites will require rather more detailed consideration than it has hitherto been accorded. However, the literature on this topic is surprisingly sparse (for the few exceptions see Norton 1983; Robins 1990; Robins and Younger 2002). Hence much of the following text is necessarily presented here for the first time, with few citations of earlier work. It draws extensively upon the experience of the first author, with especial emphasis on Scottish conditions wherever possible.

Whatever the internal hydrogeological functioning of bodies of opencast backfill, it is often far more important from the regulatory perspective to consider how they interact with the surrounding hydrogeological systems. Given that avoidance of excessive groundwater ingress to working opencast sites is in the interests of the operator, consideration of the hydrogeological setting of an opencast coal site is frequently an important element of the planning for a new development. For instance, avoidance of excessive water make will often depend on leaving barriers of unworked strata between the main extraction zone and any natural aquifers or flooded old deep mine workings. Such "stand off" zones in otherwise workable mineral directly affect the economics of opencast sites, and therefore need to be carefully designed. When all is said and done, however, numerous situations exist in which the opencast site cannot be worked economically without intersecting flooded old workings. In such cases they are usually subjected to "advance dewatering" (Norton 1983; Minett *et al.* 1986): the water level within the old workings is lowered

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[17] See the Water Environment and Water Services (Scotland) Act 2003

long before the opencast excavation approaches them, by pumping boreholes located outside of the intended opencast extraction zone.

Less commonly, opencast workings may directly intersect aquifers in adjoining coal-less strata. In the past this sometimes happened accidentally, typically due to mis-judgement of the position of a fault where there was an insufficiency of accurate 3-D geological information. Vigilance must be maintained to avoid any such incidents in future. Wherever mining leads to the establishment of direct connections between an opencast site and adjoining non-coal bearing water-bearing strata, or previously-flooded old workings, interactions between the opencast site and the surrounding groundwater systems will likely be vigorous both during and after working. The particular details of the interactions will always need to be investigated on a site-specific basis. Some recent experiences in Scotland exemplify some of the complications which can arise locally from such interactions:

(i)   At Mains of Blairingone, in the westernmost extremities of the Fife coalfields, the dewatering and working of an opencast site led to a temporary (albeit lengthy) reduction in outflows from adjoining abandoned underground mine voids. After the cessation of opencasting, these outflows were re-established, though with much poorer water quality than they had displayed prior to the operation of the nearby opencast site (Marsden *et al.* 1997). In this case, the old underground mine workings act as a drain to perched groundwater which accumulates within opencast backfill.

(ii)   To the west of Falkirk, numerous old workings for fireclay and coal were intersected by a series of deep opencast workings immediately south of the Union Canal. Water emerging from a hollow (known locally as the "Orange Hole") at the toe of the lowermost body of opencast backfill is both voluminous and of very poor quality. Given that the Orange Hole has recently been appropriated to form the holding basin for the Falkirk Wheel, the polluted drainage flowing from this hollow has gained a heightened public profile over the last few years. Water balance calculations demonstrate that infiltration over the surface area of the known area underlain by opencast backfill is far too small to explain the quantities of water emanating from the Orange Hole. In this case, the opencast backfill is effectively acting as a 'French drain' which gathers mine water from a vast network of underground fireclay and coal mine voids.

Even where opencast workings do not come into direct contact with natural aquifers or flooded underground workings, it is still possible for opencasting to significantly affect external groundwater systems. At least two further scenarios of this nature are frequently encountered in practice:

A.   Where an opencast site is developed up-dip from old mine workings, in which the water table rests well below the sole of the opencast excavation. Given that opencast sites can only be sited where coal is shallow, many sites are in fact located in relatively elevated positions, whence they can feed substantial recharge to flooded deep mine systems. In such cases, the increase in permeability associated with stripping and backfilling native sandstones and mudstones can promote a marked increase in down-dip flows of water to the old mine workings at depth. This in turn renders flow through these voids all the more vigorous, perhaps leading to enhanced surface discharges of mine water some distance away. This set of circumstances explains a number of cases in which the establishment of lakes as part of post-opencast restoration has been frustrated by the inability of the voids to retain the water fed to them by local surface runoff: infiltration of water through the pit floor feeds down-dip migration of waters to flooded deep workings several kilometres away.

B.   Where an opencast mine changes the permeability of an area of ground which was formerly a runoff source zone for a stream. Given that opencasting can raise hydraulic conductivity by several orders of magnitude, it is not surprising that more infiltration occurs over restored opencast sites than over unmined ground.   Depending on the dip of the strata, the enhanced infiltration can divert water away from one surface catchment to another. One surface water catchment which formerly received significant surface runoff from the site now occupied by the opencast backfill may now receive rather less flow than an adjoining catchment down geological dip, into which groundwater emerging from a perched water table system within the

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backfill now seeps perennially, enhancing its baseflow. Where the depleted surface watercourse provided indirect recharge to a natural aquifer downgradient, this long term loss of runoff can lead to depletion of groundwater resources in strata which have no direct connection to the opencasted rock sequence. (A recent example of this phenomenon has been documented by the first author in north-eastern England, and similar cases are likely to exist in several parts of Scotland).

In relation to A above, given that opencast leachates are more likely to be neutralised the longer they spend in the saturated zone, and since there is an inverse proportionality between pH and the mobility of most pollutants of concern (see chapter two of Younger *et al.* 2002), there may be circumstances in which encouragement of deep infiltration of opencast leachates is the most sustainable long-term management option. Circumstances in which this is likely to apply will be explored later in this document.

It is nevertheless generally true that the hydrogeological configurations of opencast sites can profoundly affect their long-term pollutant release behaviour. This is because pyrite oxidation cannot continue at any significant rate below the water table (see Section AII.1.5 above). Hence, in the long-term, only those portions of the backfill which lie above the minimum annual water table position within the backfill can contribute to the acidification of groundwater within the footprint of the opencast site. Where an option exists, there is much to be gained from engineering an opencast site such that the post-closure water table will lie as near ground surface as possible. At the very least, the lack of pyrite oxidation in the saturated zone strongly suggests that mine plans which allow for deep burial of the most sulphur-rich waste rock are likely to lead to the least post-closure water treatment liabilities.

There is nevertheless an important *caveat* to add to the above generalisation: the initial flooding of a restored opencast site will almost always lead to a temporary *increase* in pollutant concentrations in the groundwater within the backfill. This is simply because of the presence of acid-generating salts (Table AII-1) within the pyritic portions of the backfill. The initial flooding of a body of backfill unavoidably results in the dissolution of all AGS present in waste rock between the sole of the excavation and the climax water table position. As long as this deterioration in water quality is anticipated it need not cause especial problems, for provision of treatment facilities can be accommodated within the overall site restoration and after-care arrangements. Where a body of backfill would tend to give rise to surface discharges, treatment can be implemented where the leachates reach surface. For bodies of backfill which feed their waters to greater depth, time-limited pump-and-treat operations can be used to reduce pollutant concentrations until the AGS dissolution products have all been removed from the system. It is only when the likelihood of such a deterioration in quality is overlooked, or disregarded, that problems of uncontrolled pollution are likely to arise.

The good news in relation to water quality deterioration during the initial flooding of backfill is that the most severe levels of contamination tend to be rather short-lived. The so-called "first-flush" phenomenon, which was originally described in relation to deep mines (Glover 1983; Younger 1998, 2000) applies equally to opencast coal sites. Essentially, the peak concentrations of iron (and to a lesser extent other contaminants) can be expected relatively rapidly after the completion of water table rebound within the spoil (usually within weeks of decant of water from the site). Thereafter, an exponential improvement in quality invariably ensues, until the water quality attains some asymptotic, long-term residual level of pollutant release (which is typically around $1/10^{th}$ of the peak concentration). Recent examples of this genre in Scotland include:
- the rebound at Blairingone opencast, which led to a peak iron concentration of about 120 mg/l, which has since declined to less than 20 mg/l
- the rebound at Westfield, where Fe concentrations exceeded 300 mg/l shortly after completion of flooding of the main body of backfill, but with steady pumping have since dropped to less than 40 mg/l

In deep mine systems, the time taken for contaminant concentrations to fall from the initial post-rebound peak to some long-term asymptotic value is approximately four times as long as the mine took to flood in the first place (Younger 2000). In some recent cases in France, shorter durations have been observed, which are attributed to much of the mine void space becoming isolated from the bulk of the post-rebound groundwater circulation (A Blachere, CESAME, personal communication 2003). Insufficient data exist to characterise the duration of the first flush in recently-flooded backfill; however it is likely that assuming a duration equal to four times the flooding period will err on the side of caution.

One key difference between flooded deep mines and restored backfill is that the latter is often far more accessible to the atmosphere in the long-term. This in turn means that there may well be more opportunity for ongoing pyrite oxidation in opencast sites than in flooded deep mines in the long-term. On the other hand, it is far easier to retrofit pollution prevention measures on former opencast sites than in abandoned deep mines (section 5).

AII.3. Characterisation of pollutant release potential

AII.3.1. Evolution of characterisation methods

Prediction of water quality associated with opencast mines is difficult and challenging, reflecting the complex interplay of hydrological and biogeochemical processes which govern the overall pollutant release mechanisms (Younger *et al.* 2002). Notwithstanding the considerable advances which have been made in relation to characterising the pollution potential of coal mine waste rocks over the last three decades (Kleinmann 2000), it is still fair to say that the science of drainage quality prediction remains in its infancy. This is in part due to the inherent problems of developing reproducible lab testing methods suited to polymineralic rock samples (see Younger *et al.* 2002; Banwart *et al.* 2002). Perhaps even more important is the general neglect of the crucial task of relating the results of geochemical tests on individual samples to the overall pollutant release and attenuation of an entire site and the wider hydrological system within which it sits (e.g. Younger and Banwart 2002). Further problems relate to the 'quality control' of many of the most widely-used prediction techniques. These techniques have primarily been developed by engineers working in the coal mining industry, who's work in general has not been subject to the scientific peer review process which is the *sine qua non* of publication in international refereed journals. Consequently the science and art of opencast drainage quality prediction is largely documented in the "grey literature", i.e. non-refereed conference proceedings, informal reports and various regulatory guidelines. Furthermore, nearly all of the material relevant to the coal sector reflects eastern USA experiences and perspectives, which are not necessarily appropriate for many other coalfields around the world.

Notwithstanding these *caveats*, it is worthwhile briefly reviewing the evolution of the approaches most commonly used today for mine drainage quality prediction. Improving awareness of the environmental implications of acidic mine drainage in the 1960s-70s prompted an increase in the degree of regulation to which mining companies are subjected by public authorities. Assessments of potential environmental liabilities began to be required in support of mining permit applications (e.g. in the US Federal Surface Mining Conservation and Reclamation Act (SMCRA), 1977), and this effectively meant that mining companies had a legal obligation to demonstrate that they had identified potential sources of polluted drainage before they could be granted a mining permit. It fell to the regulators to specify the criteria against which mining applications could be assessed, and this quickly led to calls for standardisation of such criteria. In relation to mine drainage quality prediction in particular, regulatory bodies recognised the lack of applicable tests in the literature and therefore commissioned what is now considered to have been the "first draft" of mine water quality prediction guidelines (Sobek *et al.* 1978), which included the first attempt to standardise two distinct categories of test:

- **static tests**, in which the focus is on determination of the composition and likely geochemical reactivity of samples of rock (usually drill core) from proposed opencast sites, and

- **kinetic tests**, which are effectively 'simulated weathering experiments' (*sensu* Morrison *et al.* 1990), in which samples of rock are reacted with water under laboratory conditions, and the quality of the resultant leachates noted.

Because these two types of test were included in early guidelines such as those of Sobek *et al.* (1978), they have acquired a certain status which has ensured that subsequent research has tended to focus on their further development and refinement, arguably to the neglect of other experimental approaches which might well have proven fruitful had they been pursued. The only serious addition to this two-legged methodological approach since the late 1970s has been the slow advent of computer-based modelling approaches, which offer the unique capability of integrating geochemical and hydrological concepts, and testing their consistency with all of the available field data (e.g. EPA 1994; MEND 2000; Price 1997; Morin & Hutt 1997; Banwart and Malmström 2001; Banwart *et al.* 2002; Younger *et al.* 2002).

AII.3.2. Static tests: acid-base accounting

Static test procedures are so-called because they are generally one-off measurements of a particular set of properties of earth materials expected to be disturbed by mining. There are a number of widely used static test procedures in practical use, including whole rock analysis (chemical, mineralogical and granulometric), 'paste' pH tests acid-base accounting (ABA), (see for example, Sobek et al, 1978 and Price, 1997).

ABA is by far the most widely used of these various methods (Kwong, 2000). Although ABA was first described in 1973, it came to be widely adopted in the mining industry only after its use was advocated in the highly influential US EPA report of Sobek *et al.* (1978). ABA involves the following steps:

(i) The 'maximum potential acidity' (MPA) of the sample in question is determined. MPA is the total acidity which a given rock would produce were all of the pyrite which it contains to be oxidatively dissolved. Although it is possible to determine MPA directly by one of several titrimetric approaches, most previous determinations have been achieved by calculation from the pyritic sulphur content of the rock, using a stoichiometric relationship based on the net proton release resulting from the sum of equations (A1), (A3) and (A4) above. Working through the moles of acidity produced per mole of pyritic sulphur, this leads to the conclusion that for every weight-percent of pyritic sulphur present in 1000 tonnes of the rock, neutralisation of the maximum potential acidity released would consume 31.25 tonnes of calcium carbonate. (On this basis, MPA is expressed in parts of $CaCO_3$ per 1000 parts of rock (ppt as $CaCO_3$), e.g. grammes of $CaCO_3$ equivalent per kilogramme of rock, or tonnes of $CaCO_3$ equivalent per thousand tonnes of rock).

(ii) Determine the neutralisation potential (NP) of the rock. This is achieved by reacting a fixed mass (normally 2 grammes) of crushed rock with a known excess of concentrated hydrochloric acid while heating. The acidity remaining in the cooled, reacted fluid is titrated with a strong base back to pH 7, and compared with the initial acidity in order to determine the amount of protons consumed. As for MPA, NP is conventionally expressed in ppt as $CaCO_3$.

(iii) Next subtract the MPA from the NP: this yields a value known as the 'net-neutralisation potential' (NNP). The consensus view is that values of NNP < 0 ppt indicate that the rock in question can be confidently expected to yield acidic leachate, whereas for values of NNP > 12 ppt, only circum-neutral leachates would be anticipated. Between these two limits (i.e. for 0 ppt < NNP < 12 ppt) the rock in question might produce either acidic or circum-neutral leachates, notwithstanding the positive value of NNP. This is because the actual availability of pyrite versus the potentially neutralising minerals varies significantly from one rock to another, depending on petrological fabrics.

ABA as described above is appealingly simple and logical. However, it is important to recognise that it is based on an array of simplifying assumptions, many of which will not hold in reality. In attempts to account for the consequent shortcomings of simple ABA, a number of modifications to the procedure have been proposed (see Li,1997, and Kleinmann 2000). For example, although many labs determine MPA from total sulphur content, other forms of sulphur present in coals (i.e. organic sulphur and "sulphate sulphur" (sulphur present in sulphate minerals such as gypsum and barite)) do not produce acidity during weathering (Casagrande 1987; Lawrence, 1990; Price 1997). False neutralizing capacity can also be deduced is siderite is present in the rock (Lappako 1994). The use of powdered samples will tend to test the acidifying / neutralising potential of mineral grains which under natural conditions would be located in the interior of clasts, occluded from reaction with oxygen and/or water. Further problems arise from the assumption that the chain of reactions commencing with oxidation of pyrite proceeds as far as equation (A4), so that iron hydroxide solids are produced (see Sobek *et al*, 1978); this may unrealistically define the acidity production in situations where the oxygen supply is limited. Reacting samples with concentrated HCl at high temperature tends to overestimate NP, since minerals that dissolve under these laboratory conditions may not be so readily soluble under field conditions. For a robust interpretation of NNP values mineralogical information is ideally needed, because neutralization can originate from a variety of minerals that react at very different rates under field conditions. This means that it is eminently possible for an ABA result to indicate a positive value of NNP, whereas true kinetic constraints mean that buffering reactions are far less effective in the field. This is not a particular problem in relation to calcite, which naturally dissolves more than three orders of magnitude more rapidly than pyrite oxidation, but it can give the misleading impression that potential buffering by silicate mineral dissolution (see Lappako 1994) is more active than would be the case in the field (where reaction rates are typically two orders of magnitude slower than pyrite oxidation; Younger *et al.* 2002, p. 104).

AII.3.3. Dynamic tests: leaching columns and humidity cells

Dynamic tests are so called because they involve experiments where the geochemical processes of rock weathering are measured experimentally over extended periods of time. (Although widely referred to as "kinetic tests" in the literature, the use of the word 'kinetic' in this context is somewhat inaccurate, as the tests rarely yield true kinetic rate constants in the accepted geochemical meaning of the phrase)[18]. Rather, dynamic tests are dissolution tests conducted on bulk samples of rock to aid prediction of drainage quality from mine wastes (Lapakko, 2003). According to Price (1997), such dynamic testing procedures have predictive power in relation to:

1. the relative rates of acid generation and neutralisation, which can be important in determining if a given rock will "go acid".
2. the time before acidity release can be expected, and
3. the drainage chemistry and the resulting downstream loading for each of the probable geochemical conditions.

The merit of these claims is discussed in section AII.3.4 below.

Dynamic test methods include various types of laboratory columns (including "humidity cells") and field-based test pads (Morin and Hutt, 1998). Of the various options available, humidity cells are the most widely used in the coal sector. They are commonly used to generate estimated rates of weathering of both pyrite and the various buffering minerals in real rock samples (e.g. Frostad et al, 2002; Morin and Hutt, 1997; Price, 1997). The history of humidity cell test work dates back to 1949 (see Hornberger and Brady 1994). The early focus of dynamic test work in the 1950s and 1960s was on the laboratory simulation of acidic leachate release at coal mine sites. Early examples of the types of apparatus now known as humidity cells were described from the coalfields of the eastern USA (Hanna and Brant 1962) and from the UK (Glover and Kenyon 1962),

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[18] Just to add to the confusion, the term 'dynamic test' is sometimes applied by Canadian workers to a specific variation of column leaching tests; see Kleinmann 2000, pp. 104 -105)

where the former National Coal Board performed outdoor leaching experiments using 30 cm diameter (91 cm tall) columns filled with coal, which were cemented to tile bases with drainage holes leading to collecting jars. By the late 1960s, the practice of flushing the weathering products out of laboratory columns every seven days was established by Caruccio and Parizek (1967). This seven-day leach interval has been since become standard practice in humidity cell test work, apparently out of convenience rather from any specifically scientific rationale.

The introduction of the SMCRA regulatory regime in the USA in 1977 ushered in a new era of development and implementation of dynamic testing, which was first codified in the US EPA manual written by Sobek et al. (1978). The simulated weathering tests described by Sobek et al (1978) used humidity cells comprising plastic boxes containing 200 g of rock chips and 'inoculated' with acid mine water from a deep mine. Dry air was passed through the plastic boxes for 3 ½ days, followed by moist air for 3 days, and a ½ day to leach the sample (with 200 ml of distilled water) and analyse the leachates (7 day cycle). The use and investigation of various configuration of leaching apparatus continued throughout the 1970s, 80s and into the early 90s. The various developments culminated in the adoption of two specific methodologies (Lapakko, 2003):
- (i) The "modified humidity cell" of Lawrence (1990), which are subjected to 7-day cycles of ventilation and leaching, and
- (ii) Leaching column tests, in which unsaturated samples of rock are continuously irrigated from above.

These two methods are the focus of many recent works (e.g. ASTM 1996; Price 1997; Morin and Hutt 1997; Mills 1998; MEND 2000; Kleinmann 2000; Geidel et al 2001; and Gandy and Evans 2002).   A typical contemporary humidity cell design is shown in Figure AII-1. Most humidity cells these days are *Plexiglas* cylinders, 20cm tall with a diameter of 10cm (ASTM 1996; Price 1997; Morin & Hutt 1997; Kleinmann 2000). In coal mining applications, humidity cells are typically filled with 500 g of rock which has been crushed so that all of it will pass a 6.3mm diameter sieve (Kleinmann 2000, p. 100). An initial leach of 750 ml distilled-deionised water is performed after initial packing of the column, followed by 500 ml leaches at the end of every 7-day aeration cycle.
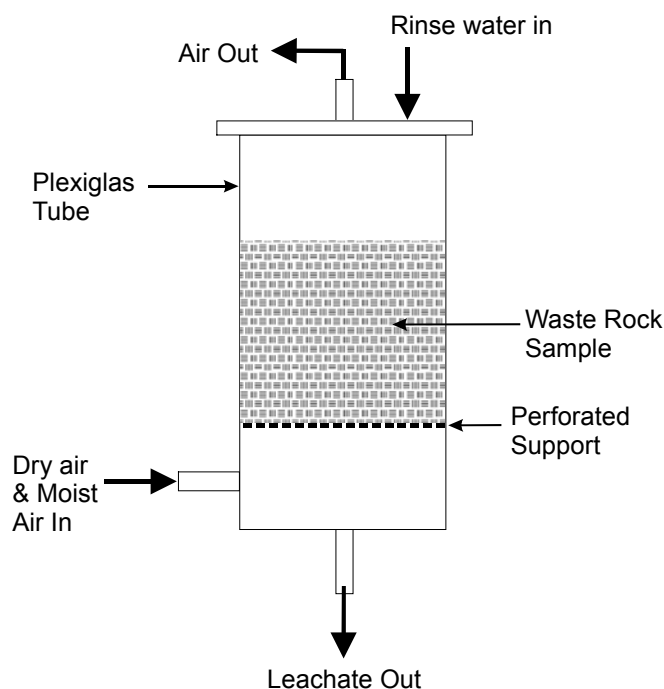

Figure AII-1. Humidity cell for testing the acidic leachate production potential of waste rock samples (after Price 1997).

A number of different techniques are used for introducing the leach water, including 'drip trickle' from above (as in the irrigation of a leaching column), 'gentle' pouring down the sides of the column or (perhaps best of all) filling from the base to avoid trapping air bubbles. Following the practice established by ASTM (1996) aeration is normally implemented by first blowing dry air through the cell for 3 days, followed by 3 days of humid air (the humid air being produced by bubbling air through a vessel filled with water at 30°C, which normally achieves a relative humidity in excess of 95%), after which leaching is performed on the 7th day.

The duration of humidity cell tests is a matter of continued debate. In general, most authorities on the subject argue that test work should continue until 'geochemically stable rates' of acidity production are attained. There is little agreement on what the criteria of recognition of "stable rates" should be. Morin & Hutt (1997) suggest that "steady state is arbitrarily defined as the point at which the last five weeks fluctuate within a definable and constant range". The ASTM (1996) suggested a minimum 20-week leaching programme, while acknowledging that some cells may require a longer programme (though the criteria by which this is decided are not stated). Recent laboratory studies by Sapsford (2003) show that 'steady' rates of leachate production are often not attained for 5 months or more, and in some cases it may be necessary for tests to continue as long as 20 months before a convincingly long period of 'steady rate' acidity release will have been observed. The reasons for these long durations lie in the relatively slow establishment of mature colonies of sulphide oxidising bacteria under laboratory conditions, probably reflecting the lack of access to natural waters carrying viable spores of appropriate genera.
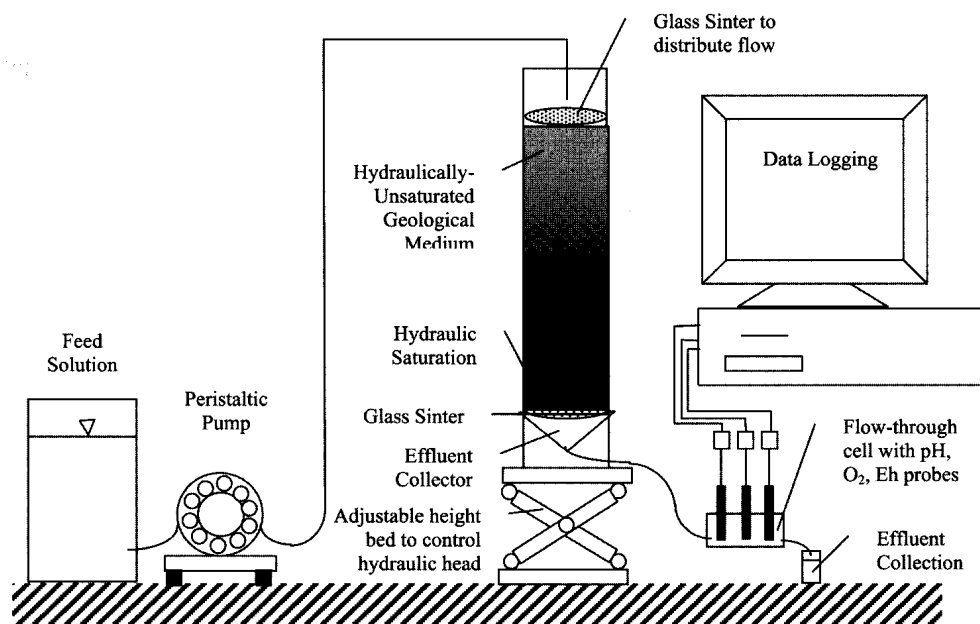
Figure AII-2. Schematic diagram of a typical leaching column used to investigate the pollutant-release potential of mining wastes, as applied recently in the UK to the study of coal mine spoil leachates (after Banwart *et al.* 2002).

Virtually no standardisation has yet developed in relation to leaching column tests. In essence, most leaching columns closely resemble humidity cells, in that they are conducted in vertical columns of similar dimensions. Figure AII-2 illustrates a contemporary leaching column design. However, leaching columns differ from humidity cells in their operation. As the lack of any air vents on Figure AII-2 indicates, no aeration is enforced. Also water is added continuously to the top of the column by 'drip trickle', rather than being restricted to weekly leach cycles. This means that the column hosts an unsaturated subsurface flow system at all times, albeit saturated conditions may develop near the base of the column (see Figure AII-2) depending on the hydraulic conductivity of

53

the material in the column and the rate of water application. Recent examples of the application of leaching column tests to coal mining wastes in the UK are described by Gandy and Evans (2002) and Banwart *et al.* (2002). Significantly, although these leaching column studies yielded interesting results as good as any others reported in the literature, none of the scientists who undertook the work argued that the leaching column results were directly applicable as predictors of field conditions. Rather, the leaching column tests were regarded as contributing towards the parameterisation of site models which take hydrological processes into account (Gandy and Evans 2002; Banwart *et al.* 2002).

AII.3.4. Limitations of static and dynamic tests

One of the key drawbacks of static tests, at least from the perspective of assuring compliance of Scottish opencast coal sites with the Groundwater Regulations 1998, is that they provide information only on the acidity/alkalinity balance of mine drainage waters. Given the specificity of the Groundwater Regulations 1998 (which actually do not mention acidity at all) this is a significant limitation. However, it is fair to say that the most elevated concentrations of most polluting metals occurs in acidic mine waters, so that knowledge that a given water is likely to be acidic at least flags up the likely need to analyses for specific metals of concern (as listed in Table AII-2).

In section AII.3.3 it was noted that Price (1997) argues that dynamic tests have predictive power in relation to:
(a) The relative rates of acid generation and neutralisation, which can be important in determining if a given rock will "go acid".
(b) the time before acidity release can be expected, and
(c) the drainage chemistry and the resulting downstream loading for each of the probable geochemical conditions.
Before commenting on these claims in relation to opencast coal mining in Scotland, it is worth noting that Price (1997) made these claims in the particular context of hard-rock metal mines in Canada, in which the acid-base balance is often far more precarious than in coal-bearing strata, and in which the low permeability of much of the enclosing bedrock can lead to rather more circumscribed hydrogeological systems than obtain in the complex, previously deep mined coalfields of Scotland. Hence the comments which follow are not a critique of Price (1997) *per se*, but an evaluation of the validity of these remarks in the context of Scottish opencast coal mining.

In relation to point (a), while it is certainly true that a given body of backfill cannot possible become acid-generating if it is wholly characterised by strongly positive NNP values, the fact that permanent submergence can completely arrest pyrite oxidation means that the obverse cannot be claimed: in other words, a negative NNP in even the major part of the backfill does not necessarily mean that a site will "go acid", as long as the hydrogeological conditions do not favour wholesale pyrite oxidation.

Similarly in relation to (b), the time required before exhaustion of alkalinity depends critically on the hydrogeological configuration of the site after restoration (see Younger and Banwart 2002 for an extended discussion of time-scale issues affecting the long-term management of abandoned mine sites).

As regards point (c), the results of dynamic tests cannot be directly used to predict downstream loadings without taking into account the issues of scale-up which affect the transition from lab- to field-scale. Of course dynamic tests produce artificial leachates which can in principle be analysed (just like real leachates collected in the field) for any analytes of interest. However, it is inadvisable to read too much into the detection of relatively "exotic" List I and II metals (i.e. Hg, Cd, Cr, Pb, Sn, Ba, Be, B, U, Ti, Mo, Sb, Ag, Te, Th, Co and V) plus the List II metalloids (As and Se) in such artificial leachates. This is because it has been repeatedly found that **field rates of pollutant release from mine wastes are typically two to three orders of magnitude less than**

laboratory-determined rates for the same rocks (Banwart *et al*. 2002). At least five major causes have been identified for this systematic lab-field discrepancy (Banwart *et al.* 2002), namely:

- Particle size effects: Large clasts contribute much to the mass of a body of backfill, but very little reactive surface area; in contrast, laboratory tests focus on the smaller-sized particles[19], which have much higher specific surface areas than the large clasts, and thus tend to react much more vigorously than them in the presence of O_2 and H_2O.

- Temperature effects: In many countries, and certainly in Scotland, field temperatures average less than half the values typically maintained in laboratories (e.g. typical groundwater temperatures of around 10^oC in lowland Scotland, compared with typical laboratory temperatures of around 25^oC). Mineral weathering occurs more rapidly at higher temperature, so that lab tests tend to over-estimate reaction rates.

- Spatial variations in mineralogy: Given the limited resources typically available for lab testing, there is a tendency to selectively test material which is suspected to be acid-generating (e.g. black shales, which are often pyritic). On real field sites such material may be of limited extent, and much leachate will actually originate from less polluting materials, providing dilution to the more acidic waters.

- Hydrogeological complexity and preferential flowpaths: The fact that backfill tends to be hydraulically heterogeneous and to contain highly preferential subsurface flowpaths was explained in Section AII.2.2 above. These preferential flowpaths, which correspond to clusters of very large clasts within the backfill, simply cannot be reproduced within laboratory columns. In practice, a vigorous exchange between more- and less-mobile waters occurs in real backfill, with concomitant mixing and dilution of the more concentrated pollutant sources. By contrast, lab columns tend to more closely mimic the granulometric and pollutant generating properties of the finer-grained zones within the spoil, utterly failing to represent the cobbly zones.

- Oxygen availability: Oxygen diffuses far more slowly through water than through air. Consequently, perched zones of saturation within or above backfill tend to greatly hinder the ingress of oxygen to pyritic zones within the spoil. If no oxygen reaches a pyritic zone, then no acidity will be generated, irrespective of the lab-determined MPA of that zone.

The marked discrepancy between field- and laboratory-determined weathering rates for pyrite and the various buffering minerals is thus explicable and anticipatable (Malmström *et al.* 2000; Younger *et al.* 2002). Nevertheless, many practitioners simply neglect the existence of any such discrepancy, and uncritically use lab-determined values of NNP (or other measures of pollutant source strength) to infer the likely quality of drainage associated with a future opencast mine (see Kleinmann 2000). From a regulatory perspective, the fact that such 'blind' use of lab-derived NNP values errs heavily on the side of caution could be interpreted as a good thing; however, there is a serious risk that over-estimating pollution potential by as much as three orders of magnitude would lead to the unnecessary sterilisation of coal reserves which might well prove to be of strategic economic importance in coming decades. Incessant over-estimation of risks is also likely to lead over time to the gradual discrediting of the regulatory regime, which will in turn mean that the maintenance of overly-stringent controls will be vulnerable to political challenge in the long term. Furthermore the Memorandum of Understanding (MoU) between SEPA and the Coal Authority which was signed in 2001 seeks (*inter alia*) to encourage opencast coal operations in locations where environmental problems from previously abandoned mines can be addressed by extraction and restoration techniques. There are therefore a number of reasons why it is unwise to yield to the pessimism inherent in uncritically equating the results of dynamic leaching tests with predictions of contaminant loadings from field sites.

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[19] Typically less than 6.3mm diameter in humidity cells based on the recommendations of the ASTM (1996), and never more than 10cm diameter, given the diameters of the *Plexiglas* columns used in such cells. It has been found by experiment that particles any larger than about 4mm do not contribute measurably to leaching (Banwart, S.A., *personal communication* 2004).

Fortunately, there is no technical need to yield to such a pessimistic approach: the fruits of recent peer-reviewed research have shown that it is possible to resolve the discrepancy between lab-measured mineral weathering rates and their application in simulations of field-scale pollutant generation / attenuation processes (Malmström *et al.* 2000; Younger *et al.* 2002; Banwart *et al.* 2002). The way is now open for a re-appraisal of the possible role of modelling techniques in developing more robust predictions of likely site behaviour during and after opencasting.

AII.3.5. Modelling approaches: accommodating geochemical kinetics and hydrogeological complexity

A recent thorough review of the application of mathematical models to the simulation of pollutant release and attenuation in mined systems has been presented by Salmon (1999). Recent publications document the practical application of such models to abandoned coal mines in Scotland (e.g. Chen *et al.* 1999) and northern England (Banwart and Malmström 2001). A wide range of mathematical approaches have been applied to such problems, ranging from empirically-derived 'performance models' (e.g. Younger 2000), through flow-path geochemical mass balance models (e.g. Chen *et al.* 1999), to fully 3-D numerical reactive transport models (e.g. Eriksson and Destouni 1997; Bain *et al.* 2000; Gandy and Evans 2002). However, all of these models share the following trait in common: they are all predicated on the notion that, provided the hydrology of a mined system can be specified at least in broad outline (e.g. positions of system boundaries and average flow rates), then it is possible to quantitatively account for observed water qualities. It is also increasingly the case that <u>future</u> water quality can be predicted for such systems within order-of-magnitude correspondence (e.g. Banwart and Malmström 2001; Gandy and Evans 2002). This is clearly an improvement on simply applying lab-derived ABA estimates to full field-scale without attempting to take the complexities of hydrology into account. Although applications of mathematical modelling in mine water quality prediction is still a relatively recent development, it is finding increased regulatory acceptance in North America, Australia and certain European countries (most notably Sweden; e.g. Eriksson and Destouni 1997). The time is probably ripe for appraising its suitability for application to Scottish conditions. However, such a development would only make sense if it were adopted as part of an overall risk assessment methodology. The following section considers the current state-of-the-art relating to the application of formal risk assessment approaches to opencast mines.

AII.4. Formal risk assessment techniques

AII.4.1. The "source-pathway-receptor" evaluation framework

A formal framework for pollution risk assessment has now been adopted throughout much of the modern groundwater management industry. In the UK, legislation introduced in 1990[20] in relation to the investigation and remediation of contaminated land required the application of a risk-based approach to site assessment. This development (and analogous developments in other countries) prompted a more general development of risk-assessment tools appropriate for groundwater pollution investigations. The assessment framework which is currently finding widest application worldwide originated in the USA as the 'risk-based corrective action' (RBCA) guidelines for the assessment of subsurface hydrocarbon pollution (ASTM 1995). The RBCA guidelines introduced the discipline of risk assessment founded on *a priori* identification of "sources, pathways and receptors[21]". A hazard (i.e. a potential risk) only becomes a real risk where all three of these elements exist: a hazardous

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[20] In Scotland, the relevant legislation is Part IIA of the Environmental Protection Act 1990 (which was inserted by Section 57 of the Environment Act 1995)
[21] originally referred to as 'targets' in ASTM (1995)

Site Investigation Cost and Effort	Data Quality Objectives	Uncertainty	Tier 1.

Tier 1.
- Identify sources, pathways, targets
- Evaluate exposure routes/scenarios
- Decisions
  - No further action,
  - Use site in its present state,
  - Design remediation or
  - Tier upgrade

Tier 2.
- Additional site investigation
- Determine model parameters
- Reactive transport modelling
- Decisions
  - No further action,
  - Use site in its present state,
  - Design remediation or
  - Tier upgrade

Tier 3.
- Spatial resolution of site properties
- Numerical modelling
- Stochastic representation
- Decisions
  - No further action,
  - Use site in its present state or
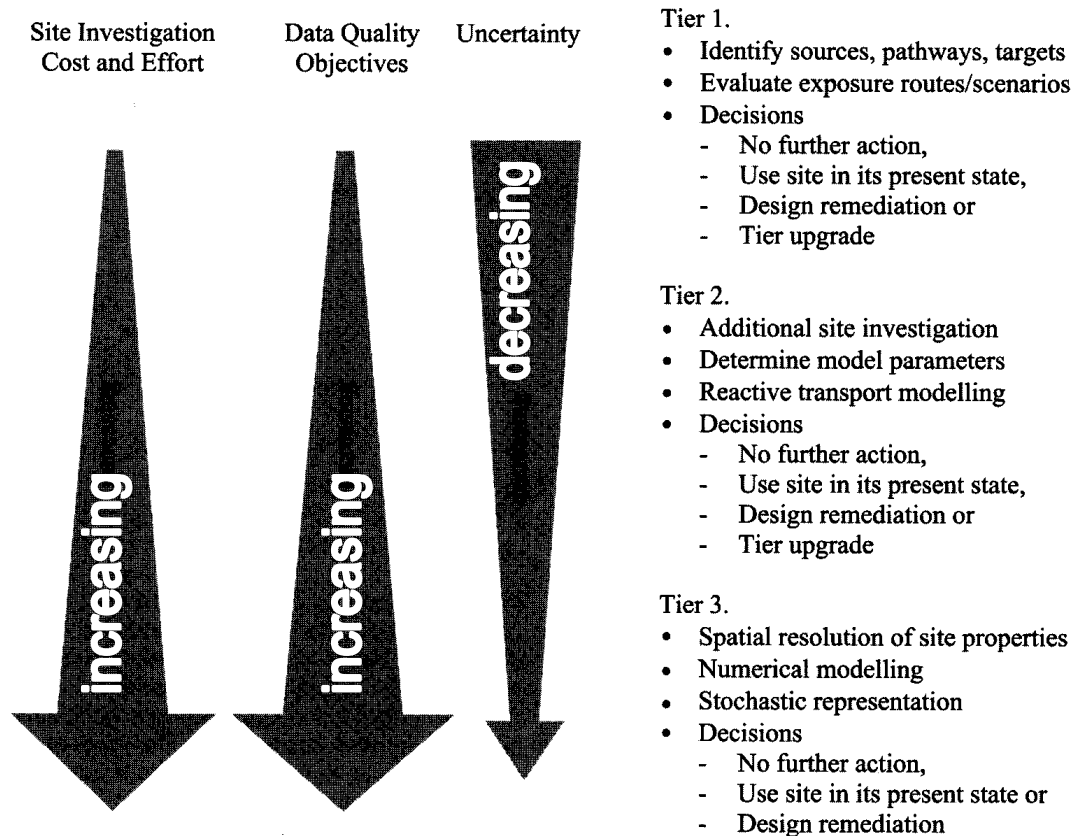 - Design remediation

Figure AII-3: The tiered approach to risk assessment originated by ASTM (1995), and recently adapted for application to mine site evaluations (after Banwart *et al.* 2002).

source which has no pathway to a receptor poses no risk. In many cases all three elements will indeed be present, and after these have been identified, a tiered approach to further risk assessment is established (Figure AII-3), in which further refinement of system understanding is pursued, by means of collection of further data which in turn supports conceptual and (where appropriate) mathematical modelling of sources, pathways and receptors.

AII.4.2. Application of formal risk assessment to mine sites

Banwart *et al.* (2002) have extended the tiered risk assessment approach outlined in Figure AII-3 to the evaluation of pollution risks associated with mine sites. The aim of the approach represented in Figure AII-3 is to guide decision-making in relation to the use of site investigation resources to achieve an appropriate level of risk assessment. Tier 1 represents the preliminary assessment of a site: If the site is relatively easy to understand using existing information alone, then a reliable decision may be taken without further ado as to the degree of 'corrective action' which the site may require in order to ensure it achieves specific objectives (such as compliance with the Groundwater Regulations 1998). "Existing information" in this context would typically include sufficient hydrogeological data to permit the development of a robust conceptual model of groundwater flow system boundaries, flow directions and magnitudes of fluxes, and sufficient water quality and / or mineralogical data to allow confident evaluation of the potential for pollutant generation. If the currently available data are insufficient (in either type or quantity) to support making such a decision (in other words the conceptual model for the site remains too uncertain) then a decision should either be made to abandon the effort or else upgrade to a higher tier. Upgrading to a higher tier demands additional site investigation and more intense data analysis, with the aim of improving the basis upon which management decision can be made.

Table AII-3: Summary of data quality objectives and assessment methodology for the application of 'Tier 1' of the formal risk assessment approach (Figure AII-3) to mine site evaluation (after Banwart and Malmström 2001). Note that this Table relates primarily to <u>existing</u> sites, and as such is not directly applicable to the risk assessment of <u>proposed</u> opencast sites (see Section 2 of main text).

Assessment step	Data required	Corresponding data sources
1. Identify • Possible pollutants • Source areas • Transport pathways • Sensitive targets	1. Discharge composition 2. Mine petrology and mineralogy 3. Stratigraphy/structural geology 4. Groundwater flow paths 5. Location of site boundaries, discharge points, aquifers, streams, boreholes, etc.	1. Water quality analyses 2. Mine records and geological memoirs 3. Mine records and geological memoirs 4. Mine records and geological maps 5. Site maps and plans, regulatory authority records
2. Site water balance	1. Recharge to subsurface 2. Evapotranspiration 3. Number of site discharges 4. Discharge flow rates 5. Groundwater flows	1. Hydrological atlas 2. Hydrological atlas and knowledge of mine ventilation 3. Site maps and plans 4. Pumping records and environmental monitoring records 5. Borehole monitoring records, geological maps, hydrogeology handbooks
3. Discharge solute mass flows	1. Discharge flow rates 2. Discharge solute concentrations 3. Temporal variability in same	1. Environmental monitoring records 2. Environmental monitoring records 3. Environmental monitoring records
4. Source mineral weathering rates	1. Identify source minerals at site 2. Discharge solute mass flows 3. Weathering reaction stoichiometry	1. Geology reports and mine records 2. Step 2. Results 3. Mineralogy handbook
5. Source mineral abundance	1. Spatial extent and porosity of workings and deposits 2. Volume and mass of reactive rock 3. Mineral composition of rock	1. Site maps and mine plans 2. Site maps and mine plans 3. Mine records, geological memoirs, mineralogy handbooks
6. Source mineral longevity	1. Source mineral weathering rates 2. Total reactive source mineral abundance	1. Results from Step 4 2. Results from Step 5
7. Exposure scenario evaluation	1. Longevity of acidity generating minerals 2. Longevity of metals generating minerals 3. Longevity of alkalinity generating minerals	1. Results from Step 6 2. Results from Step 6 3. Results from Step 6

In implementing this approach, each tier is assigned appropriate 'data quality objectives', i.e. the type and quality of information required to carry out the particular level of assessment and reach a decision. Banwart and Malmström (2001) recently proposed 'Tier 1 data quality objectives' and assessment methodology for risk-based assessment of water problems associated with mine sites. As noted by Banwart *et al.* (2002), the proposed methodology is particularly suitable for "shallow, relatively well-characterized mine environments" such as bodies of opencast backfill. Table AII-3 summarises the proposed methodology and the site data which its implementation requires. Even for proposed opencast sites which have been extensively characterised, it will often be difficult for owners and operators to meet data quality objectives beyond those for Tier I assessment as defined in Table AII-3. This is because of the specialized knowledge and data that are required for quantifying water flow, geochemical reaction rates and contaminant transport pathways in complex hydrogeological systems. Taking a cue from previous work relating to deep mine discharges (Younger 2000a), Banwart *et al.* (2002) proposed that the cost and effort involved in Tier I risk assessment could be reduced by developing a series of generalised conceptual models for the

relatively complex hydrogeological systems which many opencast sites represent. The development of generalised conceptual models has two immediate benefits. Firstly it reduces the amount of effort spent in developing a site-specific conceptual site model. Secondly it allows the collation and analysis of site-specific data to focus on those factors most likely to yield significant reductions in overall uncertainty.

AII.5. Pollution prevention and treatment technologies

AII.5.1. Principal options

Having assessed risks, it is necessary to select the most appropriate technological response. The following summary of the available options is based upon a recent keynote review of the available technologies (Younger 2002a). There are three principal options (none of which are mutually exclusive):

1. Compliance monitoring, in which natural processes or previous engineering measures (2 and 3 below) are believed to be preventing the migration of List I and II substances, but monitoring is undertaken to ensure that this belief is (and remains) valid.
2. Prevention / minimisation of pollutant release processes.
3. Mine water treatment, by either active (chemical dosing etc) or passive (wetlands etc) means.

Each of these technologies is considered in further detail in the following sections. Before discussing their relative merits and disadvantages, it is important to briefly consider the decision-logic which ought to be deployed when choosing between the principal options. Figure AII-4 summarises the coarse-scale logic, by means of which a choice is made between complianc monitoring on the one hand, and various treatment strategies on the other. Consideration of Figure AII-4 at once reveals that reasonable steps to minimise long-term pollutant release should always be taken. It will also be observed that that the flow-chart given in Figure AII-4 is essentially never-ending, in that a residual duty to retain precautionary monitoring will always remain (albeit this may eventually amount to no more than occasional visual inspection).

Figure AII-5 attempts to place the lower two-thirds of Figure AII-4 into a technical context, by relating the decision logic (for all activities except implementation of feasible preventative measures) to the flow magnitude and pollution intensity of the mine water in question. The axes of Figure AII-5 have been deliberately labelled in a qualitative manner to allow for the fact that the precise definitions of the terms 'high flow' and 'high contaminant concentrations' in a particular case will crucially depend on the characteristics of the catchment in question (in terms of climate and runoff patterns etc) such that these categories may well be reached at lower thresholds in, say, the relatively dry eastern areas of the Fife Coalfield than in the rain-soaked Macrihanish or Ayrshire coalfields of the west coast.

The key lesson of Figure AII-5 is that choosing between active and passive remediation options is often not a matter of simple preference: we simply do not have passive technologies capable of coping with the most voluminous and most contaminated of mine water discharges, so that nothing less than intense active treatment is likely to suffice for such cases.

We now turn to further discussion of the various options listed above.

**Figure AII-4: Decision logic for the selection of remedial options for polluted drainage
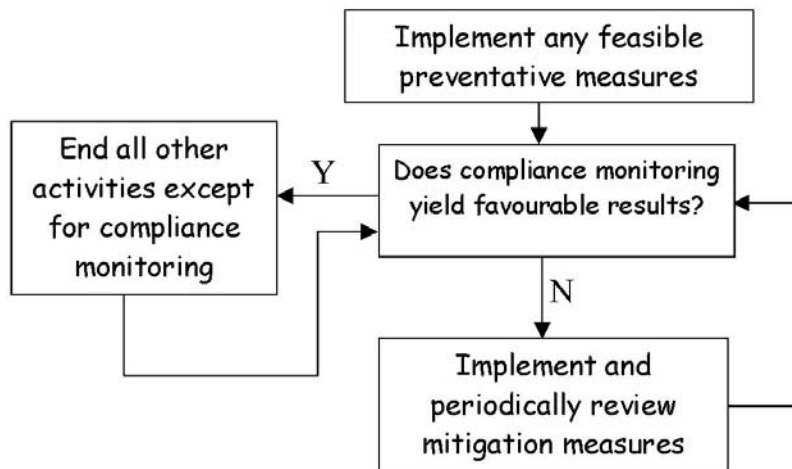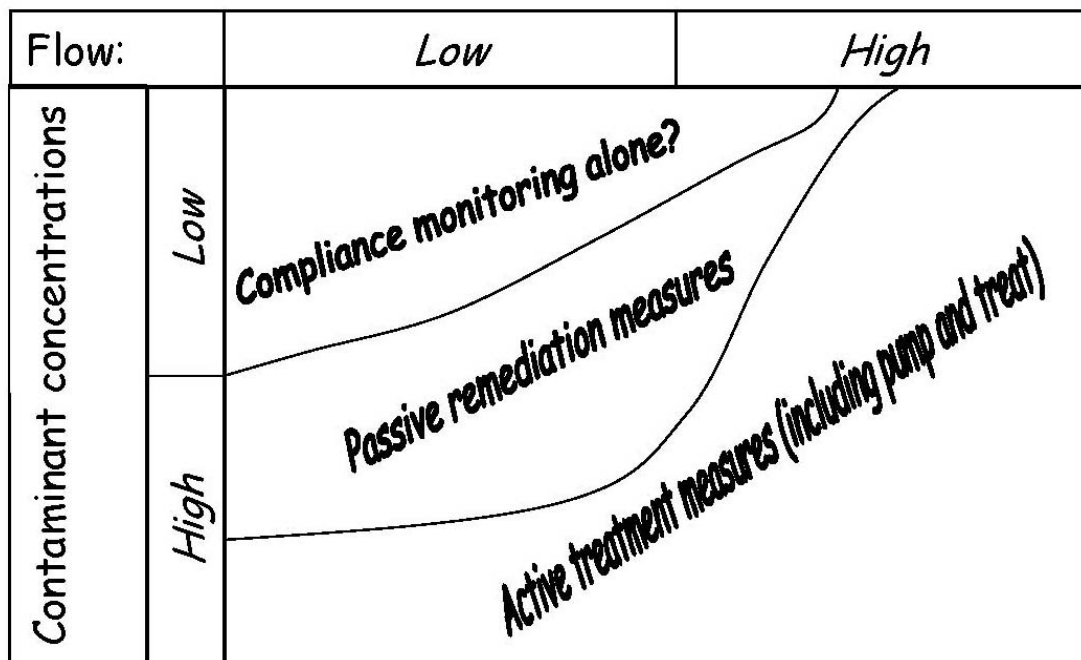associated with opencast backfill (after Younger 2002a).**

**Figure AII-5: The applicability of the three principal options for mine water remediation as
functions of contaminant concentrations and leachate flow rates (after Younger 2002a)**

AII.5.2. Compliance monitoring

This will often be the "sustainable" long-term outcome for most sites. However, before it can be implemented it is essential all necessary steps have been implemented to prevent release and transport of List I and II substances. Assessment of the circumstances under which compliance monitoring alone can be relied upon as a long-term management strategy can only be made on a site by site basis. Such an assessment demands an evaluation of the degree to which natural and engineered pollution prevention processes are likely to operate (and to be sustained over time) in the ambient hydrogeological setting. This in turn demands conceptual modelling, drawing together hydrological, mineralogical and geochemical information into an overall synthesis.

AII.5.3. Prevention / minimisation of pollutant release processes

From the discussion presented in Sections 2.1 and 2.2, it follows that strategies intended to minimise the release of contaminants to backfill leachates must focus on eliminating one or other of the following pre-conditions for the oxidation of sulphide minerals: (i) the presence of oxygen (ii) the availability of moisture (iii) the presence and activity of iron- and sulphur-oxidising bacteria.

The access of both atmospheric oxygen and moisture to backfill can be severely limited by the installation of low-permeability covers. Adequate covers will generally include a coarse grained "capillary break" layer (to prevent upward migration of pore-waters in response to surface layer desiccation), overlain by a low permeability cap (typically 0.5 m or more of clay, compacted such that it retains a hydraulic conductivity no greater than 10^{-5} m·d^{-1} usually with a final veneer of vegetated topsoil. Specific guidelines on the installation and after-care of clay caps have recently been published by the European Commission (PIRAMID Consortium 2003). Selective diversion of surface waters away from known zones of infiltration to backfill may also be possible in some cases (see Younger 2000b). Where backfill adjoins open deep mine voids (for instance, remaining in the final highwall) these might provide leachate migration pathways to groundwater systems. Where the prevention of such migration seems desirable, it may be technically feasible to plug the entrances to these voids so as to limit their ability to transmit leachate. Details of this type of approach and examples of its application are provided by Younger and Wolkersdorfer (2004). However, as this approach entails substantial costs and possibly also significant long-term risks, it is not to be advocated lightly.

Besides denying bacteria moisture and oxygen using covers, direct inhibition of the activities of iron- and sulphur-oxidising bacteria using bactericides has also been attempted on a number of occasions (for a brief review, see Younger *et al.* 2002). Although such efforts have met with a certain degree of success in controlling the leaching of acidity from coal stockpiles and overburden stores of modest areal extent, the beneficial effects rarely last any longer than 6 months, making this a very expensive option in practice. It is also not feasible to apply this approach to very large bodies of opencast backfill.

AII.5.4. Active treatment technologies

"Active treatment" denotes the use of conventional wastewater treatment unit processes, which typically require ongoing inputs of electrical power and/or chemical reagents in a closely-controlled process (which usually demands frequent operator attention). Although active treatment is most commonly used to improve the quality of surface discharges of mine water, it can also be applied to polluted groundwaters in "pump-and-treat" mode. The treated groundwaters can then either be discharged back into the subsurface, or discharged to a surface watercourse.

The classic approach to active treatment of acidic and/or ferruginous mine drainage involves three steps (Younger *et al.* 2002):
(i)   oxidation (usually by means of a simple cascade), which helps to convert soluble ferrous iron (Fe^{2+}) to far less soluble ferric iron (Fe^{3+}), as well as allowing pH to rise by venting excess CO_2 (where present) until equilibrium with the atmospheric CO_2 content is attained.

(ii)   dosing with alkali (usually hydrated lime (Ca(OH)$_2$), and less frequently caustic soda (NaOH)), both to raise the pH (thus lowering the solubility of most problematic metals) and to supply hydroxyl ions for the rapid precipitation of metal hydroxide solids.

(iii)   accelerated sedimentation, usually by use of a clarifier or lamellar plate thickener, often aided by the addition of flocculants and/or coagulants. Current practice in the industry favours the re-circulation of an aliquot of iron hydroxide sludge into the influent of the sedimentation unit, which has been shown to favour the densification of the sludge overall. This practice is called the "high density sludge" process and it typically yields sludges with 25 – 30% solids by volume, as opposed to the 5% solids contents typically obtained without re-circulation.

For most purposes this time-honoured approach will suffice. However, where it is important that the treatment process yields a net reduction in mine water salinity, then alternative approaches will most likely be necessary. In some cases, it may prove worthwhile applying conventional desalination technology (i.e. flash distillation and reverse osmosis) to mine waters. However, these processes are extremely costly. In South Africa, innovative research is yielding new approaches to the desalination of mine waters which result in far cheaper processes. In one approach, termed 'biodesalination', co-treatment of acidic mine waters with sewage from the adjoining cities is effectively "using one waste stream to cancel out the other", in a process which is extremely efficient at removing sulphates from the mine water (by means of bacterial sulphate reduction). Other technologies employing bacterial sulphate reduction are available as commercially-proven turn-key operations, and are particularly suitable where very low metals concentrations must be attained in active treatment plant effluents, and where one or more of the metals recovered from the water is valuable (e.g. zinc/copper). Another very promising desalination process of South African lineage is the SAVMIN^{TM} process (Smit 1999), which is essentially a variant of conventional alkali dosing and sedimentation in which a series of cyclical precipitation and sedimentation steps eventually lead to recovery of virtually all of the previously dissolved sulphate as potentially marketable gypsum. The key to achieving this is a step in which sulphate is removed down to residual concentrations of only a few mg·l^{-1} by equilibration of the water with respect to the aluminium sulphate mineral ettringite ($3CaO.3CaSO_4.Al_2O_3.31H_2O$), which is stable only in a narrow range of high pH (11.6 - 12.0).

Beyond these neutralisation and desalination technologies, there is a considerable range of alternative approaches to the active treatment of mine waters, many of them borrowed from the field of metallurgical processing, which all have potential as niche applications in cases in which recovery and re-use of metals is an economic possibility. These approaches include (Younger *et al.*, 2002): (1) sorption and ion exchange processes (2) solvent extraction (3) electrochemical extraction (4) biochemical extraction techniques (5) the barium sulphide process (in which sulphate is removed from mine water by precipitation of $BaSO_4$) (6) biological trickle filters. Few of these techniques are ever likely to enjoy widespread uptake in practice, on account of their costs and limited track record, and are therefore not considered further here.

AII.5.5. Passive treatment technologies

The term "passive treatment system" has been formally defined with by the European Commission's PIRAMID project as follows: "A water treatment system that utilises naturally available energy sources such as topographical gradient, microbial metabolic energy, photosynthesis and chemical energy and requires regular but infrequent maintenance to operate successfully over its design life". The working definition of "infrequent" in this context is currently around six-monthly (PIRAMID Consortium 2003).

The development to date of passive treatment technologies applicable to mine waters is documented in detail by Younger *et al.* (2002). Types of passive system currently in use include:
      (i)   Aerobic, surface flow wetlands (reed-beds)
      (ii)   Compost wetlands with significant surface flow

(iii)   Mixed compost / limestone systems, with predominantly subsurface flow (so-called <u>R</u>educing and <u>A</u>lkalinity <u>P</u>roducing <u>S</u>ystems (RAPS))

(iv)   Subsurface reactive barriers treating acidic, metalliferous ground waters

(v)   Closed-system limestone dissolution systems for zinc removal from alkaline waters

(vi)   Roughing filters for the aerobic treatment of ferruginous mine waters where there is no room for a surface wetland.

Each of the above technologies is appropriate for a different kind of mine water (i.e. acidic versus neutral), or for specific hydraulic circumstances. The degree to which each type of system can currently be considered to be "proven technology" corresponds to the order in which they are listed above. This ranking of confidence is reflected in uptake rates to date (Younger, 2000b).

One of the principal attractions of wetlands as treatment systems is the possibility of integrating them into the surrounding landscape, and achieving healthy connections with the existing eco-systems in the area. Integration of wetlands into a landscape at the level of aesthetics is readily attainable, as a number of recent projects illustrate (see the case studies later in this paper). Ecological integration is rather harder to achieve in practice, however, due to a number of factors including (i) physical limitations on the areas available for treatment (ii) the frequent requirement that treatment wetlands in riparian areas be surrounded by flood defence bunds, which preclude two way exchanges of water, solutes, sediments and plankton with adjoining rivers; (iii) engineering limitations, such as the need to allow freeboard at the perimeters of systems and the need to prevent erosion by extreme flows.

AII.6. Environmental regulations and Scottish opencast coal mining

AII.6.1. Water management in Scotland and opencast coal mining

The brief summary which follows is drawn from a recent European Commission-sponsored research report into the institutional issues surrounding mine water management in the UK (Amezaga and Younger 2002).

The Scottish Parliament has the power to issue laws, regulations and statutory guidance in environmental issues. Although Scottish law is written specifically for Scotland it is frequently modelled on similar legislation introduced previously or simultaneously to cover the same issues in England and Wales. In current Scottish law the Scottish Executive bears the primary responsibility for the regulation of minerals planning and water management in Scotland. SEPA is charged with implementing those aspects of environmental legislation which are not managed by local authorities (most of the planning functions) or the Executive itself (relating mainly to a few items of regulation in which SEPA itself may be subject to permitting).

To date the control of pollution from active mines has effectively been managed by two procedures:

(i)   Broad preventative measures are implemented as part of the planning process. The developer of a proposed opencast mine has to apply for planning permission in accordance with the Town and Country Planning (Scotland) Act 1999. Planning applications must be supported by formal Environmental Impact Assessments (or Environmental Statement in the case of very small projects), following the stipulations of the Environmental Impact Assessment (Scotland) Regulations 1999. In determining opencast applications, the Local Authority consults with the planning liaison team in SEPA, which then refers to the officers in the corresponding Area Offices. SEPA expects to see (a) a risk assessment (including a strategy for managing site drainage) covering the impact of the proposed development on the water environment (b) plans for the operational phase and (c) proposals for the closure phase.

(ii)   Regulation of operating opencast sites is achieved by SEPA by means of the system of consents to discharge originally introduced under the Control of Pollution Act 1974 (recently amended by the Environment Act 1995). Discharge consent controls apply to both groundwaters and surface waters. This system of regulation is one of the most mature of its type in the world, and is generally considered to be working very effectively.   Indeed, so well does this system work that it leaves little room for constructive commentary on improvements to the operational phases of opencast site developments. However, it must be stressed that this regime facilitates controls at the point/s of discharge only, and precludes any real involvement in 'pollution prevention' activities within working sites. The COPA regime therefore limits SEPA's interventions to reactive controls as opposed to pro-active involvement with operators in the development of pollution minimisation strategies.

Most recently, SEPA has acquired new powers (as yet unused) to directly influence how opencast operations are undertaken, by means of issuing notices pursuant to the Groundwater Regulations 1998 (see Section AII.6.2 below).

Although the post-closure management of mines has always in theory been subject to control under (i) above, a number of salutary negative experiences demonstrate that, all too often, insufficient attention was paid to the post-closure arrangements during the permitting of former underground and opencast coal mines in Scotland (e.g. Norton 1983; Robins 1990; Younger 2001). Having learned from these experiences, SEPA now expects closure management strategies to be fully considered at the environmental impact assessment stage of the planning process. Given the long time-delays which necessarily separate the initial planning application from the eventual period of site closure, however, there is plenty of scope amongst both public authorities and private operators for either overlooking the original provisions as the staff who originally dealt with a given site move on or for a failure to adequately track unanticipated developments during the working of a site which effectively invalidate the original closure plans proposed before the commencement of extraction. To avoid these problems, there is a good case for requiring operators to supply SEPA with certain basic information on water quantities, qualities and likely post-closure hydrological changes (including mitigation plans for any predicted post-closure pollution) six months before final site closure. Since 1998 such a requirement has in fact existed for the operators of <u>deep mines</u>, who must comply with the terms of the Mines (Notice of Abandonment) Regulations 1998. However, these regulations <u>do not</u> apply to opencast coal mines (E P Watson, *personal communication* 2003). This is arguably a significant gap in regulatory provision, at least in terms of transfer of information needed to design rational contingency plans for post-restoration water management at former opencast sites.

Beyond the legal requirements policed by SEPA, other institutional arrangements also influence the environmental management of coal mining in Scotland. In particular, SEPA works in close collaboration with the Coal Authority (CA) under the terms of a Memorandum of Understanding (MoU) signed in 2001. The primary focus of this MoU is the prevention and progressive reduction in pollution associated with abandoned coal mines. It also encourages existing operators to plan for abandonment and develop their understanding of the associated environmental issues, and it specifically seeks to encourage opencast coal mining in locations where extraction and restoration techniques can contribute to the resolution of environmental problems associated with previously abandoned mines. All active coal mining in Scotland (entirely opencast nowadays) is concentrated in the central belt and limited areas in the south west.

Unlike in England, Scottish Local Authorities retain a statutory duty to promote the recreational use of the water environment, which appears to result in them being more involved in water issues than their counterparts south of the Border. The Local Authorities in coal mining areas also own large areas of "brownfield" sites including many derelict bings, which they are usually seeking to re-develop within the planning system, thereby avoiding their designation as contaminated land under the Part IIA of the Environmental Protection Act 1990 (albeit a requirement still exists (under PAN

33) to secure an equivalency of remedial action to that which would be expected under full implementation of Part IIA). SEPA has an important role in the decision-making in relation to these schemes. In many cases, opencasting operations will form an integral part of the redevelopment process (e.g. Harrison 2000), further underscoring the importance of developing a rational assessment framework for the evaluation of proposed opencast developments.

AII.6.2. The Groundwater Regulations 1998

One of the most recent developments in the regulation of the water environment in Scotland has been the enactment of the Groundwater Regulations 1998 which complete the transposition into Scottish law of the requirements of the European Directive on Groundwater (80/68/EEC). The Groundwater Regulations 1998 effectively complement the existing regime of water pollution control in Scotland (introduced by the Control of Pollution Act 1974, as outlined above) by specifically forbidding the introduction of certain substances (List I) into groundwater, and by placing strict limitations on the extent to which other substances (List II) may be permitted to enter groundwater. Table AII-2 summarises the various List I and List II substances, and comments on possible sources of each of these in Scottish opencast coal sites. Opencast mining is considered to be an activity under the Groundwater Regulations 1998. As such SEPA has powers to control opencast activities by issuing a notice under the Groundwater Regulations 1998.

A code of practice aimed at helping owners and operators of mineral extraction sites (including opencast coal sites) to comply with the requirements of the Groundwater Regulations 1998 has recently been published by the Scottish Executive (2003). This provides sound advice on the control of pollution from non-mineral pollution sources during opencasting operations. Given its very wide scope (it includes all types of mining and quarrying, not just opencast, and is only 29 pages in total length) the coverage given to risk assessment for proposed new opencast sites, and for the post-closure management of such sites, is necessarily limited. Nevertheless, it clearly identifies the need for the "assessment of the risk of release of List I and List II substances from storage in stockpiles or backfilling, prior to the activity being undertaken". It briefly mentions the European Commission's current draft Best Available Technology Reference (BREF) document on the management of tailings and waste rock (IPPC Bureau 2003) (which is discussed in the following section), as well as noting the existence of laboratory leaching tests as described by Kleinmann (2000). As has already been explained in Section AII.3.4 above, both the protocols for existing laboratory tests, and the use of the results which they yield, have significant shortcomings in relation to assessing risks relating to compliance with the Groundwater Regulations 1998. That discussion clearly indicates the need for substantial expansion upon the brief remarks on this subject given in the Code of Practice (Scottish Executive 2003).

AII.6.3. Emerging waste management regulations

At the time of writing, the Scottish Executive (2004) are consulting on proposed regulations to tighten the regulation of the management of non-mineral wastes from mines and quarries. The significance of this initiative in relation to assuring compliance with the Groundwater Regulations 1998 is that it will further limit any risk of post-restoration groundwater pollution by List I and List II substances arising from anthropogenic sources (as listed in the right-hand column of Table AII-2). This further underlines the importance of the present document, focused on the minimisation of pollution risks due to the leaching of back-filled rock materials.

In June 2003 the European Commission formally published a proposal for a daughter directive to the over-arching Waste Framework Directive[22], with the particular purpose of improving the management of waste from the onshore extractive industry in Europe. The proposed Directive is formally entitled "Directive on the management of waste from the extractive industries" (COM (2003) 319 final), though it is informally referred to as the "mining waste directive" (MWD) in most

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[22] i.e. Directive 75/442/EEC, as amended by later Directives 91/156/EEC, 91/692/EEC and by Commission Decision 96/350/EEC.

industry circles. The development of this proposed Directive is one of a number of policy initiatives which were prompted by the spectacular failures of tailings dams at Aznalcóllar (Spain) in 1998 and at Baia Mare (Romania) (2000). These events led to widespread questioning of existing provisions for mine waste management in Europe. As the proposed MWD recognises, tailings dams are not the only source of environmental problems associated with mine waste management, and it also dedicates significant coverage to waste rock piles and other types of mine waste depository. Many of the provisions of the MWD echo established best-practice in these pre-existing national-level management regimes

In parallel with the development of the proposed MWD, the European Commission's IPPC Bureau in Seville has developed a non-binding Best Available Technology Reference (BREF) document on mineral processing operations (IPPC Bureau 2003). Because the European IPPC Directive explicitly exempts the mining sector from its control, none of the recommendations of this BREF can be taken as prescriptive, nor as a defence against prosecution in the event of pollution problems arising despite implementation of its recommendations. The document nevertheless comprises a compendious collection of useful advice on the management of wastes arising from most forms of mining still active in Europe. Its very comprehensiveness is in some ways its downfall, however, as in 490 pages it covers so many different types of mineral processing operations that the details given on any one type are necessarily restricted.

 Although Section 4.3 of the BREF considers management of 'ARD' (so-called "acid rock drainage"), it devotes little more than half a page (sub-section 4.3.1.1) to the entire topic of "Prediction of ARD Potential". What is more this half-page is simply a presentation of a plot of NP versus MPA[23] for a single gold mine site. There is no coverage at all of the crucial hydrogeological elements of risk assessment in relation to opencast coal mining (or any other sort of mining for that matter). Despite its compendious size, therefore, the long-awaited BREF is sadly of little or no relevance to the risk assessment of opencast coal mine sites. (On the positive side, Sections 4.3.1.2 and 4.3.1.3 of the BREF do contain some useful material on preventative technologies such as capping or submerging pyritic spoil, echoing the detailed presentation on this topic in the guidelines of the PIRAMID Consortium (2003), as well as providing a useful expansion upon the key points made in 2.5.3 of the present document).

AII.6.4. Gaps and challenges

The review of legislative provisions and regulatory practices presented in the preceding sections generally demonstrates that there are relatively few gaps in the administration of opencast coal mining in Scotland. In particular, the control of pollution during the active phase of site operation is usually stringent and effective, especially in relation to discharges to watercourses, over which SEPA exercises control under the COPA legislation. However, in relation to pollution <u>prevention</u> measures, such as those advocated in this document, control is exercised primarily via the planning process, usually with limited direct involvement of SEPA in overseeing actual implementation (even where the measures were originally included in site planning conditions at SEPA's behest). This is arguably a gap in the implementation of otherwise mature regulatory arrangements. The present lack of direct regulation of water abstraction by opencast operators for dewatering purposes is arguably another 'gap' in regulation. However, SEPA have recently developed a policy on abstraction, which seeks to ensure that requirements of the Water Framework Directive are taken into account during the adjudication of planning applications. In addition, the power to specify a maximum water <u>quantity</u> on a discharge consent provides an indirect mechanism for reactive abstraction control, and this mechanism has certainly been used in the past in relation to opencast operations (see Norton 1983).

The only real gaps in regulatory provisions arguably relate to the long-term management of water post-closure, and these gaps relate more to practice than to legislation. The planning system already provides a legal platform for enforcing adequate long-term planning for opencast sites. In

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[23] MPA, as defined in Section AII.3.2 of this document, is referred to as 'AP' in the BREF (Figure 4.4)

practice, however, problems arise due to deviations of site conditions from those which were anticipated during planning. In many cases this may actually be symptomatic of a historic failure to specify sufficient *a priori* hydrogeological and geochemical risk assessment for sites (especially for older sites). The antidote is a formal review procedure, in which initial risk assessments are periodically revisited and evaluated to ensure that they remain valid; if not, they can be updated in the light of new information.   As a minimum precaution, it would be wise to enact some provision which enforces a final evaluation of the plans for site restoration and after-care shortly before coaling ceases. While such provisions have existed since 1998 for deep mines (i.e. the Mines (Notice of Abandonment) Regulations 1998), no equivalent review provision exists for opencast sites. While such a provision could be introduced by the development of specific regulations under the Water Environment and Water Services (Scotland) Act 2003, it could equally be enforced simply by inclusion as a condition on planning approvals.

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