ACID DRAINAGE PREVENTION GUIDELINES FOR SCOTTISH OPENCAST COAL MINING: THE PRIMACY OF THE CONCEPTUAL MODEL [1]

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Abstract. 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 terms of 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. Accordingly, the Scottish Environment Protection Agency (SEPA) commissioned an assessment framework for pollution prevention in opencast coal mining. The development of the framework was founded upon a comprehensive critical review of the literature on acidic drainage prediction, from which it emerged that there has been an excessive concentration on simple pollution potential assay tests (acid-base accounting, humidity cell tests) at the expense of rational assessments of contaminant transport pathways. The new Framework described here rectifies this imbalance, emphasising the over-arching importance of developing a robust site conceptual model, which is progressively refined as relevant data (mineralogical, geochemical, hydrological) become available. The conceptual model then provides the basis for risk assessment and impact mitigation planning.

Additional Key Words: acid-base accounting, acidity, coal, conceptual model, guidelines, groundwater, humidity cell test, opencast, prediction, Scotland.

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Introduction

The European Directive on Groundwater (80/68/EEC) was fully transposed into Scottish law by the introduction of the Groundwater Regulations 1998, which 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. Table 1 lists the List I and II substances of relevance to coal mining environments in Scotland. As the comments in Table 1 make clear, given what is known of the mineralogy and geochemistry of coal-bearing strata in Scotland, opencasting poses little risk of introducing List I substances into groundwater; however, it does have substantial potential to lead to the migration of several 'List II' substances. For this reason the Scottish Environment Protection Agency (SEPA) commissioned the development of an assessment framework for pollution prevention in opencast coal mining. 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.
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 not 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.

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. (After Younger and Sapsford 2004, to which reference should be made for the rationale behind the comments given here).

List I substances	**Comments …**
Mercury (Hg) and its compounds	Highly unlikely to be released during weathering of soils and rocks found in Scottish opencast sites
Cadmium (Cd) and its compounds	Unlikely to be released 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	May occasionally be released 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	May occasionally be released 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	May occasionally be released 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	Highly unlikely to be released 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 commonly released unless steps are taken to minimise their mobility.
Fluorides	May occasionally be released though mineral sources in coal-bearing strata are few; dissolved concentrations normally maintained below 5 mg/l by equilibrium with CaF_2 (fluorite).
Ammonia	May occasionally be released 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.

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

Why not just use ABA and humidity-cell tests?

The full literature review upon which the development of the framework was based is presented as an Annex to the published Assessment Framework (Younger and Sapsford 2004; available on-line at: www.sepa.org.uk/pdf/groundwater/opencast_assessment.pdf). The review work ranged very widely over the international literature on prediction of mine site pollution potential (the bulk of which remains non-refereed 'grey' literature, published in-house by various organizations). A very large number of references relate to two categories of test, typically applied to samples of mine wastes (including opencast backfill):

(i) Static tests, which are thus named because they are generally one-off measurements of a particular set of properties of mine waste rocks / tailings. There are a number of widely used static test procedures in practical use, including whole rock analysis (chemical, mineralogical and granulometric), 'paste' pH tests, and acid-base accounting (ABA) (Sobek *et al*. 1978; Price, 1997); of these, ABA is by far the most widely used (Kwong, 2000).

(ii) Dynamic tests, which are essentially dissolution tests conducted on bulk samples of rock to aid prediction of drainage quality from mine wastes (Lapakko, 2003; Sapsford and Williams 2005). (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 scientific terms).

From the perspective of assuring compliance of Scottish opencast coal sites with the Groundwater Regulations 1998, static tests suffer from the drawback 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 1).

Frustration with static test results has spawned extensive endeavours in relation to dynamic testing. Dynamic test methods include various types of laboratory columns (including "humidity cells") and field-based test pads (Morin and Hutt 1997). Of the various options available, humidity cells are the most widely used in the coal sector. They are commonly used to estimate rates of weathering of both pyrite and the various buffering minerals in real rock samples (e.g. Morin and Hutt 1997; Price 1997; Frostad *et al*. 2002; Sapsford and Williams 2005). According to Price (1997), such dynamic testing procedures are able to yield predictions of:

a. the relative rates of acid generation and neutralisation, which can be important in determining if a given body of waste 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 possibly become acid-generating if it is wholly characterised by strongly positive netneutralisation potential (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 , 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^{o}C in lowland Scotland, compared with typical laboratory temperatures of around 25^{o}C). 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 is well known (Younger *et al.* 2002). 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 maximum potential acidity (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). 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. If overestimation of risks became normal this would also likely to lead over time to the gradual discrediting of the regulatory regime, which would in turn mean that the maintenance of overly-stringent controls will be vulnerable to political challenge in the long term. Furthermore, it is often environmentally responsible 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.

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; Banwart *et al.* 2002). The way is now open for the development of more robust predictions of likely site behaviour, during and after opencasting, taking field conditions fully into account. The means for doing so is the development of site conceptual models.

Conceptual modelling

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 definitions in terms of physical hydrogeology, 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 SEPA assessment framework was developed.

Summary of Assessment Framework

Figure 1 summarises the Assessment Framework for opencast coal mining in Scotland. As shown in the Figure, the Framework proceeds in six steps, the fourth of which is a potentially ceaseless feedback loop of conceptual model refinement, depending on the degree of confidence in the assessment and mitigation of quantified risks.

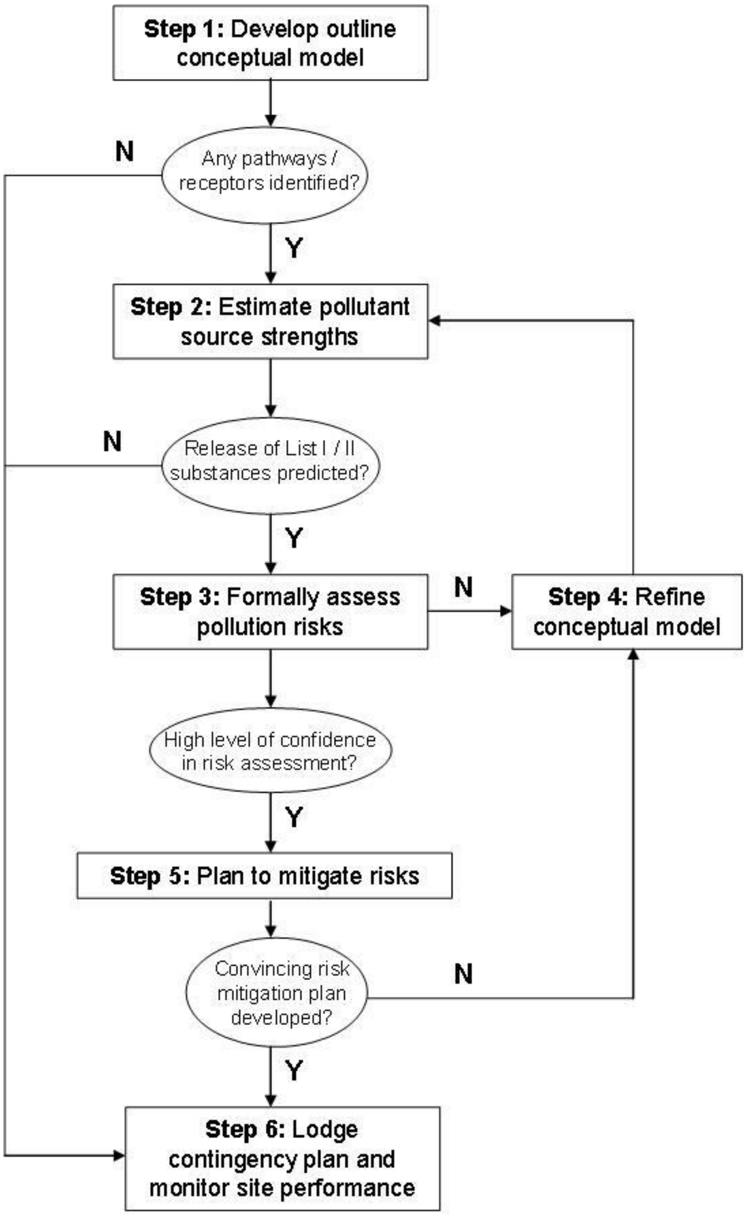

Figure 1: Flow-chart summarising the decision logic of the risk assessment framework for compliance of proposed opencast developments in Scotland with the Groundwater Regulations 1998 and allied regulations.

The constituent steps of the framework are outlined below.

Step 1 - Outline conceptual model: An initial conceptual hydrogeological model for the site must be developed, with explicit coverage of at least 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 no post-closure migration pathways or potential receptors are identified, permission may be granted for the evaluation to proceed directly to Step 6.

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 Fig. 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).

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, albeit that significant modifications of US practice are advocated (Younger and Sapsford 2004). Level 4 includes an element of 'dynamic testing' as used in North America, 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 Younger and Sapsford (2004) 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 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 in the Framework. 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, if 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. In the light of these analyses, the following activities will typically 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

- 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 at least 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 Younger and Robins 2002)
- where the site intersects the pre-mining water table, the design of dewatering operations, specifically addressing:
    - the mode of dewatering, with advance dewatering always being preferred ahead of sump-dewatering, wherever possible (cf. Younger *et al.* 2002)
    - the likely quality of waters to be pumped and their treatment requirements (where appropriate) in order to comply with COPA (Control of Pollution Act, 1974) provisions, and
    - 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
- design of restoration measures (e.g. compaction, clay capping of backfill etc) to minimise long-term infiltration
- specific planning for compliance monitoring

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 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.

Conclusions

In developing an assessment framework for pollution prevention due to opencast coal mining in Scotland to ensure compliance with the Groundwater Regulations 1998 (Younger and Sapsford 2004), a comprehensive critical review of the literature on acidic drainage prediction was undertaken. From this review, it emerged that there has been an excessive concentration on simple pollution potential assay tests (acid-base accounting, humidity cell tests) at the expense of rational assessments of contaminant transport pathways. The new SEPA Framework rectifies this imbalance, emphasising the over-arching importance of developing a robust site conceptual model, which is progressively refined as relevant data (mineralogical, geochemical, hydrological) become available. The conceptual model then provides the basis for risk assessment and impact mitigation planning, implemented in a six-step, iterative process (Fig. 1) aimed at ensuring compliance with the Groundwater Regulations 1998 throughout the full life-cycle.

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