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Hydrogeological impact appraisal for dewatering abstractions

Science Report – SC040020/SR1

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Executive summary

This report provides practical guidance on how to assess the hydrogeological impact of groundwater abstractions in connection with dewatering operations at quarries, mines and engineering works. It applies to those who are preparing applications to the Environment Agency for transfer and full licences. The methodology for hydrogeological impact appraisal (HIA) is designed to fit into the Environment Agency's abstraction licensing process, including the changes brought about by the Water Act 2003. It is also designed to operate within the Environment Agency's approach to environmental risk assessment, so that the effort involved in undertaking HIA in a given situation can be matched to the risk of environmental impact associated with the dewatering. The HIA methodology can be summarised in terms of the following 14 steps:

Step 1: Establish the regional water resource status.

Step 2: Develop a conceptual model for the abstraction and the surrounding area.

Step 3: Identify all potential water features that are susceptible to flow impacts.

Step 4: Apportion the likely flow impacts to the water features.

Step 5: Allow for the mitigating effects of any discharges, to arrive at net flow impacts.

Step 6: Assess the significance of the net flow impacts.

Step 7: Define the search area for drawdown impacts.

Step 8: Identify all features in the search area that could be impacted by drawdown.

Step 9: For all these features, predict the likely drawdown impacts.

Step 10: Allow for the effects of measures taken to mitigate the drawdown impacts.

Step 11: Assess the significance of the net drawdown impacts.

Step 12: Assess the water quality impacts.

Step 13: If necessary, redesign the mitigation measures to minimise the impacts.

Step 14: Develop a monitoring strategy.

The steps are not intended to be prescriptive, and the level of effort expended on each step can be matched to the situation. Some steps will be a formality for many applications, but it is important that the same thought-process occurs every time, to ensure consistency. The methodology depends heavily on the development of a good conceptual model of the dewatering operation and the surrounding aquifer. The steps of the methodology are followed iteratively, within a structure with three tiers, and the procedure continues until the required level of confidence is achieved. Advice is also given on how to undertake HIA in karstic aquifers and fractured crystalline rocks.

Contents

1 Introduction

1.1 Purpose of this report

This report describes a methodology and suite of tools for assessing the hydrogeological impact of the abstraction of groundwater for dewatering purposes. Such dewatering abstractions were previously exempt from licensing control, but the Water Act 2003 has removed that exemption. There are now three types of abstraction licence:

- ***Temporary licences:*** for water abstraction for any purpose over a period of less than 28 days.
- ***Transfer licences:*** for water abstraction to transfer water from one source to another without intervening use, or to transfer water within the same source for dewatering activities in connection with mining, quarrying, engineering works etc, again without intervening use.
- ***Full licences:*** for water abstraction for any other licensable use.

The main purpose of this report is to provide practical guidance on how to assess the hydrogeological impacts of dewatering abstractions, for those who are undertaking investigations and preparing technical material to support applications to the Environment Agency for groundwater abstraction licences (transfer and/or full licences) in connection with dewatering operations at quarries, mines and engineering works. The guidance is relevant both to existing dewatering operations and to proposed new operations. Similar guidance for those preparing applications for other groundwater abstractions (for public water supply, irrigation or industrial use, for example) can be found in a separate report (Boak and Johnson 2006). Both reports build on work carried out under an earlier R&D Project (W6-071) on risk-based decision making for water resources licensing (Faulkner *et al* 2003).

A proportion of the water derived from dewatering at many quarries and mines is used for other activities, such as dust suppression, mineral washing, and the manufacture of products like concrete blocks. These quarries and mines are likely to require both transfer and full licences, as will be explained in greater detail in Section 1.4. The Environment Agency intends there to be a common approach to the assessment of the impacts on water resources of any type of groundwater abstraction. The methodology described in this report is applicable to applications for both transfer and full licences.

There are several ways in which the water environment can be affected by surface mineral extraction: by the initial ground investigation works; by the physical presence of the excavation; by the dewatering of workings that operate below the water table; and by contamination of groundwater and/or surface water (Thompson *et al* 1998). This report concentrates on the third aspect, dewatering. A summary of the other aspects, which in general are controlled by different legislation, can be found in Thompson *et al* (1998) or at http://www.goodquarry.com.

1.2 The regulatory context

The Water Act 2003 has introduced significant changes to the abstraction licensing system in England and Wales. Among other provisions, the dewatering of mines, quarries and engineering works, water transfers into canals and internal drainage districts, use of water for trickle irrigation, and abstractions in some areas that were previously exempt, will now need a licence, to make sure that they are managed appropriately and that any impact on the environment can be dealt with (Environment Agency 2003a). In addition, all abstractors now have a responsibility not to let their abstraction cause damage to others, and from 2012, the Environment Agency will be able to amend or revoke permanent abstraction licences without compensation if the abstractions are causing serious damage to the environment.

It is estimated that the ending of the exemption for dewatering will bring about 1,000 existing dewatering abstractions into the licensing system, most of which are likely to require transfer licences. Environment Agency licensing officers are faced with the task of processing the applications for these transfer licences, and to do this, the hydrogeological impact of each of the abstractions needs to be assessed. To make this task manageable, the Environment Agency has set out a framework (for existing dewatering abstractions, and subject to confirmation by the Government) under which it will:

- give at least two years for applicants to submit applications, to enable pre-application discussions to take place;
- have up to five years to determine all the applications, and attach priority to the most significant abstractions; and
- wherever possible, provide the maximum possible formal notice to applicants of any significant material alteration in abstraction regime considered necessary in the interests of overall water resources management.

Before the Water Act 2003 came into force, dewatering of quarries and mines was largely controlled by conditions attached to mineral planning permissions, conservation notices and discharge consents. The discharge consent regime remains in place, and will not be displaced by transfer licences. Even with the introduction of transfer licences for dewatering, the planning system will still control many other potential impacts of quarrying on the water environment, and it is therefore necessary that the two regimes are able to run alongside each other. The interface between the minerals planning system and the new abstraction licensing regime has been the subject of a parallel research project undertaken by Capita Symonds (MIRO Project SAMP2.23), the outputs from which will be available through the Mineral Industry Research Organisation (MIRO).

Even though transfer licences represent a new type of abstraction licence, several other regulatory regimes are still highly relevant to the assessment of the impacts of dewatering on water resources and the water-related environment. These include the Habitats Directive, the Water Framework Directive, and Catchment Abstraction Management Strategies (CAMS). Further information on these and their relevance to dewatering can be found in Appendix 1.

1.3 Applications for transfer licences

It is useful at this point to explain where hydrogeological impact appraisal (HIA) of dewatering abstraction fits into the overall process of applying for a transfer licence. A broad outline of the process for an applicant is as follows:

i. ***Establish whether or not a transfer licence is required.*** Under the Water Act 2003, small abstractions (under 20 m^3/d), whatever the purpose of the abstraction, will not need a licence. Quarries or mines that have not yet commenced dewatering, or where the dewatering quantity is under this threshold, need go no further in the process. Also, there are some circumstances (passive gravity drainage from a hillside quarry, for example) where a licence is not required. If in doubt, contact your local Environment Agency office for clarification.

ii. ***Pre-application discussions with the Environment Agency.*** Assuming the dewatering is licensable, then it is recommended that the applicant engages in pre-application discussions with the Environment Agency. The main purpose of these discussions would be for the applicant to get advice on the following:

   - The main environmental issues in the area, which may need to be specifically addressed by the HIA (for example, low flows in a certain river, or declining water levels in a certain wetland).
   - The level of effort likely to be required during the HIA (see also Section 3.3).
   - Specific information such as the water resources availability status of the area in which the dewatering operation is located, and where to find other relevant information.
   - Whether or not a full licence will be required for other water uses, in addition to the transfer licence (see Section 1.4).
   - Potential impacts on protected rights and other lawful uses of water.
   - Where to look for hydrogeological expertise if the applicant has no such expertise in-house.

iii. ***Undertake the hydrogeological impact appraisal.*** It will be seen below that the HIA methodology takes a tiered risk assessment approach. The level of effort that turns out to be necessary depends on many factors, but primarily on reducing the uncertainty to an acceptable level. It will also be seen later that there may very well be more discussions with the Environment Agency during the course of the HIA. The results of the HIA are written up and form the bulk of the Environmental Statement that will accompany the licence application.

iv. ***Submit the transfer licence application,*** accompanied by the Environmental Statement that summarises the results of the HIA. Note that some applicants may choose to submit the application before undertaking the HIA, but it will have to be done in any case.

Given that the Water Act 2003 has introduced licensing to a previously exempt activity, there will be two types of applicant for transfer licences: those who are already dewatering, and those who have not yet commenced dewatering. For existing dewatering abstractors, the application and HIA processes may be streamlined significantly. In due course, the Government will publish regulations describing the transitional arrangements that apply to existing dewatering abstractors.

1.4 Full licences

The wording of the Water Act 2003 makes it clear that transfer licences apply to the transfer of water from one supply source to another (for example, for navigation and drainage), or the transfer of water within the same source for dewatering activities, in either case without intervening use. The three words *without intervening use* are very important. There are some operations where water is abstracted for dewatering purposes, and pumped straight to the discharge point(s) without intervening use, where it is returned to the ground or discharged into surface water. These operations will only require a transfer licence (assuming the quantity pumped is > 20 m^3/d). However, there are many operations where a proportion of the water abstracted for dewatering is used for other purposes, such as dust suppression, mineral washing, and making cement/concrete products. The quantity of water used for these other purposes is likely to require a full licence, in addition to the transfer licence for the dewatering itself.

Note that the Water Act 2003 refers to the *use* of water, and does not depend on the water being *consumed*. This introduces the subject of consumptiveness, which can be loosely defined as the proportion (usually expressed as a percentage) of the quantity abstracted that is not returned to the environment. For the minerals sector, there has been some research (Mathieson *et al* 2000) on the amount of water typically required for various purposes (0.3 m^3/tonne for washing of aggregates for example). However, this is not necessarily the consumptiveness as defined above, because it does not take into account the proportion returned to the environment (see Box 4.3 for more on consumptiveness). It is essential that quarry and mine operators have a good understanding of what happens to the water that is abstracted, and good estimates of the quantities involved, including how much is consumed. In other words, a good water balance for the quarry or mine should be developed. This will be discussed in more detail later. Measuring or estimating abstraction and discharge quantities, and the quantities of water used for other purposes, is essential for the following reasons:

- Full licences specify quantities of water as part of the licence conditions (typically a total annual quantity and a maximum daily rate). Operators applying for full licences will therefore need to know how much water they are using for the various purposes mentioned above.

- Even though transfer licences do not usually specify quantities, it will be seen later that among other things, knowledge of abstraction and discharge quantities is required for the assessment of the impacts of the dewatering abstraction and the efficacy of the mitigation measures.

One of the functions of this report is to highlight the information that the operators of dewatering systems will need to assemble in order to undertake HIA, so that they can commence the collection of such information as soon as possible, if they have not already done so. Note that the procedure for HIA required for full licences is virtually the same as for transfer licences for dewatering. The same data collection, conceptual modelling, and impact assessment will serve for both purposes with very little adaptation, thus minimising duplication of effort.

While on the subject of types of licence, there is an important difference between full licences and transfer licences. Holders of transfer licences are not protected from derogation, whereas holders of full licences are. Dewatering abstractors can opt to apply for a full licence for the dewatering itself (in addition to the intervening uses of water), if they feel the need to protect the entire abstraction quantity from derogation, but this will attract annual volumetric charges. However, in a recent consultation document, the Environment Agency has proposed that a very low loss factor of 0.003 be applied to the annual charge where dewatering is carried out under a full licence (Environment Agency 2005a).

1.5 Case studies

During this project, the HIA methodology was tested on six case studies of real-life dewatering operations in different hydrogeological settings, as follows:

i. Slate quarry in North Wales.

ii. Opencast coal mine in South Wales.

iii. Karstic limestone quarry in Derbyshire.

iv. China clay pits in Cornwall.

v. Floodplain sand and gravel quarry in Berkshire.

vi. Construction dewatering in Cumbria.

These are referred to at the appropriate points in this report, where they are used to illustrate certain aspects of the methodology and its application in practice, but for convenience, all six case studies are grouped together in Appendix 9.

2 Basic concepts

2.1 Introduction

Before delving into the detail of the HIA methodology, it is useful to discuss some basic concepts that are fundamental to HIA, namely uncertainty, risk, and conceptual modelling. This section also deals with some common misconceptions about the way in which groundwater abstractions behave. Unfortunately there is no magic tool for assessing the hydrogeological impacts of dewatering; the emphasis is still on developing good conceptual models, taking uncertainty and risk into account, and using appropriate tools and techniques to answer specific questions.

2.2 Uncertainty and risk

The Environment Agency's approach to environmental risk assessment is based on the guidelines published by the Government (DETR 2000). In these guidelines, the following definitions are given:

> ***Hazard:*** *a property or situation that in particular circumstances could lead to harm.*
>
> ***Risk:*** *a combination of the probability (or frequency) of occurrence of a defined hazard and the magnitude of the consequences of the occurrence.*

In groundwater abstraction licensing (including dewatering), the hazard is the act of abstracting water, and the risk relates to potential impacts on the environment, or other impacts such as derogation of the rights of existing abstractors. In order to evaluate and use risk assessments effectively as a credible basis for decision-making, it is important to understand how different sources of uncertainty contribute to the final risk estimates. Uncertainty can affect all stages of risk assessment, and environmental scientists are increasingly being required to provide information on how certain their decisions are. Analysing the sources and magnitudes of uncertainties can help to focus discussion, identify knowledge gaps, and feed into decisions about risk management. Uncertainties generally fall into the following categories (DETR 2000):

- ***Model uncertainty:*** where models provide only an approximation of the real environment. Model uncertainty may have two components: (i) conceptual modelling uncertainty due to insufficient knowledge of the system; and (ii) mathematical model uncertainty arising from the limitations of the model selected in accurately representing reality.
- ***Sample uncertainty:*** where uncertainties arise from the accuracy of measurements or the validity of the sample (number and location of sampling points).
- ***Data uncertainty:*** where data are interpolated or extrapolated from other sources.
- ***Knowledge uncertainty:*** where there is inadequate scientific understanding of the processes involved.
- ***Environmental uncertainty:*** where the inherent variability of the natural environment leads to errors in our approximations. For groundwater

systems, this could be the variations in groundwater level and flow that occur due to natural variations in rainfall and evaporation.

Environmental uncertainty cannot be reduced, and knowledge uncertainty can only be reduced by scientific investigation. However, model, sample and data uncertainty can be reduced by the conceptual modelling process. All these types of uncertainty apply to HIA and making decisions about abstraction licences. Consider the example of deriving aquifer hydraulic parameters from pumping test results:

Model uncertainty: there may be very limited knowledge of the real configuration of the aquifer, for example, whether or not it is layered, or whether a confining layer should be regarded as leaky. In addition, the test results may be analysed using an analytical equation that is based on a very idealised model of the real situation. Sweeping assumptions (that the aquifer is of infinite extent, for example) are inherent in all analytical solutions. An aquifer that in practice contains many layers with different hydraulic properties will often be simplified into one or two layers with averaged properties.

Sample uncertainty: results from test pumping usually only represent a small sample in time and space of the overall behaviour of an aquifer. Depending on its length, the test is only sampling a limited volume of aquifer around the borehole, and there may only be results from one or two tests to work with. In addition, there may be inaccuracies in the calibration of the flow meter used to measure discharge during the test, for example.

Data uncertainty: test pumping results from a 7-day or 14-day test are often extrapolated to make judgements about the long-term impacts of an abstraction. Aquifer parameters derived at specific points (boreholes) have to be interpolated to give spatially-distributed parameter values for the whole aquifer, or even one average value.

Knowledge uncertainty: there are many key scientific areas where there is still only superficial knowledge and understanding of how real systems behave. Examples from hydrogeology include river-aquifer interaction, the influence of groundwater on wetlands, the behaviour of saline-fresh water interfaces, and the behaviour of highly-layered aquifers.

Environmental uncertainty: it is recognised that aquifers are heterogeneous in practice, and that aquifer parameters such as transmissivity and storage coefficient vary spatially. In addition, hydraulic conductivity can vary in different directions (a condition known as anisotropy).

It can be seen therefore that uncertainty is involved in many ways even in a routine situation. The most important thing to realise here is that uncertainties combine to produce greater uncertainty. If a single value of transmissivity is assigned to an aquifer or groundwater management unit, then the uncertainty associated with that value is a combination of the types of uncertainty just described. This is not necessarily a problem, as long as the situation is recognised, and decisions are made taking into account the overall uncertainty.

Care should also be exercised when using average parameter values. To continue the test pumping example: imagine that two pumping tests have been conducted on different boreholes in the same aquifer; and different transmissivity values have been derived, say 200 and 600 m^2/d. In many test pumping reports this would lead to the statement that transmissivity varies from 200 to 600 m^2/d, and that an average value of 400 m^2/d has been used from then on. However, this is making several assumptions: that 200 and 600 represent the extremes of the true range of transmissivities; that transmissivities can be arithmetically averaged; that the results can reasonably be applied to other parts of the aquifer; that the assumptions inherent in the analysis are appropriate, and so on. When assessing potential impacts of abstraction, all assumptions must be recognised and taken into account. It is often useful to undertake some form of sensitivity analysis in order to understand the effects of ranges in parameter values on derived quantities (see Box 2.1).

Box 2.1: Sensitivity analysis

Example of simple sensitivity analysis to illustrate the effects of ranges in parameter values on derived quantities, using the Theis equation for unsteady-state flow in confined aquifers (Kruseman and de Ridder 1990):

$$s = (Q/4\pi T).W(u)$$

where $u = r^2S/4Tt$, and $W(u)$ is a function of u (commonly known as the well function), with s being the drawdown at a radius r from the pumping well at time t, in an aquifer of transmissivity T and storativity S, and abstraction taking place at a rate Q. Suppose that the equation is being used to predict the drawdown at a sensitive wetland, using aquifer parameters estimated from previous tests. The quantities Q, r and t are known with reasonable accuracy, and we are using estimated values of T and S to predict s. Let's say $Q = 1{,}000\ m^3/d$, r = 500 m, t = 100 days, $T = 400\ m^2/d$ and $S = 1x10^{-4}$. This gives a prediction for drawdown (s) of 1.63 m. The measured range for T might be 200 to 600 m^2/d (even ignoring the fact that the true range may be much greater), and let's say the range for S is from $5x10^{-5}$ to $5x10^{-4}$. Keeping S at the original value, using the extremes for T gives a range for s of 1.14 to 2.98 m. Keeping T at the original value, using the extremes for S gives a range for s of 1.31 to 1.77 m. However, combining the uncertainties (varying both T and S in the combinations that give the greatest extremes) results in a possible range for s of 0.93 to 3.26 m. Which drawdown turns out to be the 'true' value could have dramatic implications for the wetland.

Some types of uncertainty are easier to reduce than others. For example, drilling more observation boreholes for a pumping test, or conducting tests in several boreholes, will help to reduce the data and sample uncertainty; using a radial flow model with layers (as opposed to an analytical equation) to analyse the results will reduce the model uncertainty. However, reducing knowledge uncertainty may require extended scientific study; and environmental or natural uncertainty is impossible to reduce, and must just be recognised.

2.3 Conceptual modelling

Conceptual modelling is at the heart of both CAMS and the Water Framework Directive (see Appendix 1), and its importance to HIA cannot be overemphasised. In the water resources context, a conceptual model can be defined as a synthesis of the current understanding of how the real system behaves, based on both qualitative *and* quantitative analysis of the field data. Some people take the view that conceptual models are based upon a purely qualitative understanding, with quantitative assessment only coming in during subsequent analytical or numerical modelling. However, in this report, the term conceptual modelling definitely includes quantitative analysis.

A real hydrogeological system is so complex that it will never be possible to study everything in detail; a conceptual model is therefore bound to be a simplification of reality. The important question is to determine what needs to be included in the study and what can be safely ignored. In other words, what observed behaviour do we want the conceptual model to get right, and what don't we mind the model getting wrong? For example, if we are investigating the mechanisms that operate during periods of low flow in a Chalk stream, we may not mind being wrong about the mechanisms that operate during groundwater flooding events (Environment Agency 2002a). Or, when developing a regional groundwater resources model of a coastal aquifer we may choose to ignore the difference in density between fresh and saline water in order to simplify the mathematical representation. We may not mind being wrong about the exact behaviour of the fresh/saline water interface, because we are focussing on larger water resources issues.

Experience has shown that for most aquifer systems there is a small number of crucial factors that *must* be examined in detail, and if any one of these is ignored the conclusions may be seriously in error (Rushton 1998). The focus of the conceptual model should be on the identification of these crucial factors. Continuing the second example above, if the coastal model is of a small Caribbean island, then it probably *will* be important to get the relative positions of the fresh water lens and the underlying saline water right, in order to know how the fresh water can be abstracted without causing problems such as saline upconing. For this purpose, relative density should definitely *not* be ignored. It is helpful to write down the purpose and specific objectives of the conceptual model, as this is invaluable for focussing effort on the right factors. With these comments in mind, the important characteristics of a conceptual model can be described as follows:

- It should concentrate on the crucial factors, that is, the features of the system that are important in relation to the purpose of the project.
- It is based on evidence; even though it is inevitably an approximation or simplification of reality, it must not contradict the observed evidence.
- It is a set of observations, explanations, working hypotheses and assumptions, bearing in mind that there may be more than one explanation for observed behaviour.
- It must be written down; this is a discipline that forces vague ideas to be formalised, and helps to identify weaknesses in reasoning or unjustified assumptions.
- It must be tested; this is an essential part of conceptual model development, as it forces hypotheses to be evaluated and alternatives found if necessary.

It is the last point, testing the model, where the numbers come in and the conceptual model becomes quantitative rather than just qualitative. If there is no quantitative testing, the degree to which the model represents the real system cannot be assessed. Testing with numbers also enables uncertainty to be explicitly addressed, which links conceptual modelling to risk assessment. Conceptual modelling is an iterative or cyclical process (Figure 2.1).

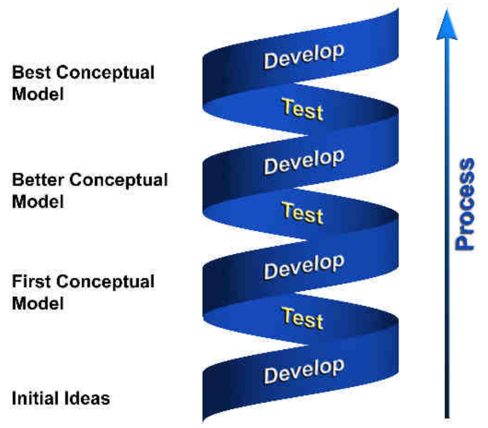


Figure 2.1 The development process for a conceptual model
(adapted from Environment Agency presentations)

The process of developing a conceptual model is as follows:

- Start with initial ideas, such as observations, hypotheses and areas of uncertainty, and write them down.
- Test the model, by for example doing some crude water balance calculations with long-term average values for the various water balance components.
- Based on the results of the testing, re-evaluate the model, rejecting some hypotheses, keeping some and developing some new ones, as necessary.
- Test the improved model, and then continue the cycle of re-evaluation and testing until the initial ideas become the best available conceptual model, as appropriate for the problem being addressed.

It is worth repeating the point that conceptual modelling is continuous and cyclical; it is a process, not a finished product. It is also important to realise that the degree of development of a conceptual model is determined by the availability of data, and the sophistication of the tools that have been used to test the model. Bredehoeft (2005) introduces the phrase 'hydrogeologic surprise', which he defines as the collection of new information that renders one's original conceptual model invalid. From limited empirical data, he estimates that such surprises occur in 20-30 per cent of model analyses. Rushton (2003) goes further, saying that it is his experience that in each modelling study at least one fundamental feature was not identified in the initial conceptual model (but became clear in subsequent modelling cycles).

Superimposed on the continuous cycle of model development and testing is a hierarchical or tiered approach, with basic, intermediate and detailed levels of model. These tiers can be described as follows:

Tier 1 (Basic): Tested using lumped long-term average water balances and simple analytical equations, to arrive at a ‘best basic’ conceptual model.

Tier 2 (Intermediate): Tested using more detailed data, such as time-variant heads and flows, and more sophisticated tools, such as seasonal or sub-catchment water balances (semi-distributed), analytical solutions (to investigate the impact of abstraction on river flows, for example), or two-dimensional steady-state groundwater models.

Tier 3 (Detailed): Likely to be tested using a spatially-distributed and time-variant numerical groundwater model, calibrated and validated against historical data.

The tiered approach to conceptual modelling is illustrated in Figure 2.2, from which it can be seen that the conceptual model is refined within each tier from an initial understanding to the best available model. The diagram also illustrates that associated with each tier is an assessment of the risk involved in the decision being made.

As the investigation progresses through the tiers, the cost increases, but so does the confidence in the model. As confidence increases, so the uncertainty decreases, and the investigation should continue up the tiers until the uncertainty (and therefore the risk) has been reduced to an acceptable level. The level that is considered acceptable depends of course on what the conceptual model is being used for. Common sense must be used, and in general, decisions should be made with the simplest model possible, with refinement of the model required only if a decision cannot be made because the uncertainty is too great.

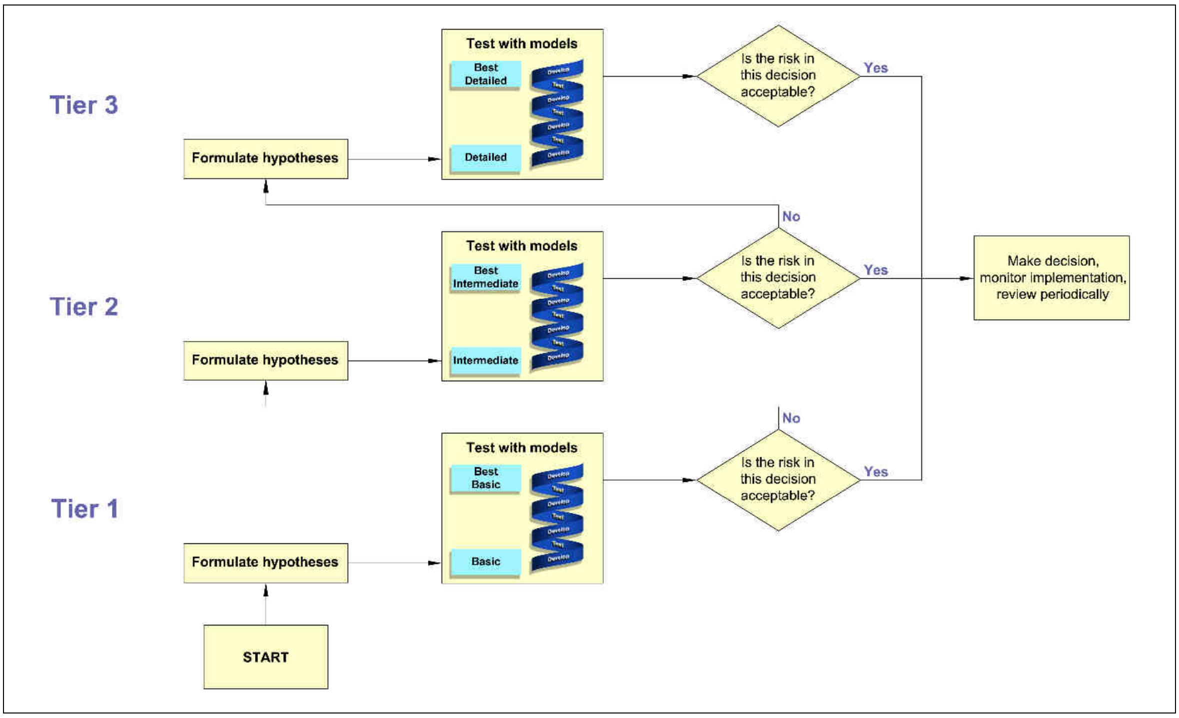


Figure 2.2 Tiered approach to conceptual modelling
(adapted from Environment Agency presentations)

2.4 Common misconceptions

It is assumed that the people undertaking HIAs have some basic hydrogeological knowledge or experience, even though they may not be specialist hydrogeologists. Even so, there are some common misconceptions about groundwater abstractions and the way in which they behave. Examples of such misconceptions are as follows:

- A groundwater abstraction will only have an impact, or the impact will be greater, on water features that are downstream (down the regional hydraulic gradient).
- Impacts of groundwater abstractions will be reduced or limited by recharge, or even that recharge will 'fill up' the cone of depression around a borehole.
- There will only be an impact if there is increased drawdown.
- Results from short pumping tests can safely be extrapolated in space and time.
- Faults are impermeable barriers to flow.
- The drawdown predicted by the Theis equation after 200 days represents drought conditions.

In order to counter these misconceptions, and prepare the ground for explaining the methodology for HIA, some basic principles of groundwater behaviour will now be described briefly. These are not intended to be exhaustive technical explanations, but primarily to provoke thought. Some of the ideas are admittedly counter-intuitive, but they are so important that the reader is directed to two excellent papers, Theis (1940) and Bredehoeft *et al* (1982), if they wish to know more.

Principle 1: The impact of a groundwater abstraction spreads until it has stopped an equal amount of water leaving the aquifer. Imagine an aquifer system that is in long-term equilibrium and where there are no abstractions. Inputs to the system may include recharge from rainfall, regional groundwater flow, and infiltration from surface water. Discharges from the aquifer may include springs, seepages, baseflow to rivers, and regional groundwater flow. A simplified system of an island in a fresh water lake, receiving rainfall recharge, is illustrated in Figure 2.3 (upper diagram). Now imagine that a borehole is drilled, abstraction commences, and a cone of depression starts spreading out from the borehole. Initially, the abstraction takes water from storage, and the flow of groundwater into the lake is unaffected (Figure 2.3, middle diagram). However, as the cone of depression spreads, it eventually reduces the gradient of the water table where it intersects the lake, and groundwater flow into the lake will be reduced (Figure 2.3, lower diagram).

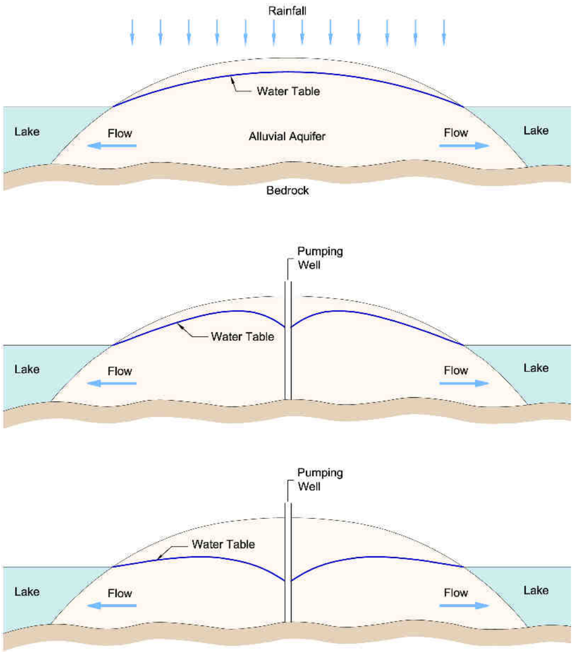


Figure 2.3 Impact of groundwater abstraction on aquifer discharges
(after Bredehoeft *et al* 1982; note that the rainfall continues in the middle and lower diagrams)

The first principle to emphasise therefore, is that the effect of the abstraction will spread until it has stopped an equal amount of water from leaving the aquifer (in both confined and unconfined aquifers). This will usually be in the form of reduced discharges (reduced springflow, reduced baseflow, or reduced seepage, for example). In other words, all groundwater abstractions eventually have an impact; it is only a question of where the impact will appear and how long it will take. This subject was addressed over 65 years ago by Theis (1940), who said:

> *"Under natural conditions…previous to development by wells, aquifers are in a state of approximate dynamic equilibrium. Discharge by wells is thus a new discharge superimposed upon a previously stable system, and it must be balanced by an increase in the recharge of the aquifer, or by a decrease in the old natural discharge, or by loss of storage in the aquifer, or by a combination of these."*

Principle 2: Impacts can be changes in flow as well as water level. It will be seen later that the procedure for HIA differentiates between impacts due to changes in flow and impacts due to changes in groundwater level. Impacts from changes in level are easy to visualise, and include for example, lower water levels in neighbouring boreholes or in groundwater-supported wetlands. Impacts from changes in flow are harder to visualise, but include interception of water that would otherwise have reached a river, spring or estuary, for example. This type of impact is also harder to observe, as it may represent a small proportion of the flow in a river, it may take some time for the impact to reach the river, and river flows are naturally variable. In Figure 2.3 (lower diagram), there are obvious changes in water level caused by the abstraction, but there is also a change in flow from the groundwater into the lake. On the coast of the imaginary island, at the point where the water table meets the lake, the water level has not perceptibly changed, but the gradient of the water table has been reduced slightly, and so the flow into the lake has been reduced.

Principle 3: Rainfall recharge makes no difference to magnitude of drawdown, nor to the size/depth of the cone of depression. From Principle 1, it follows that recharge from rainfall (as opposed to induced recharge from surface water) makes no difference to the eventual magnitude of the impacts of the abstraction (for a linear system[1]), because the recharge was there before the abstraction started, and the water was already going somewhere. Rainfall recharge does not therefore 'fill up' cones of depression, because the cone of depression is superimposed on the pre-existing groundwater profile (a point that is discussed thoroughly in Bredehoeft *et al* 1982, and more recent papers such as Bredehoeft 2002, and Devlin and Sophocleous 2005). It should be said that in some circumstances, drawdown due to abstraction may increase the amount of infiltration (by reducing run-off), which Theis (1940) describes in terms of reducing the amount of 'rejected recharge'. However, this can be viewed as merely altering the timing and/or location of the impact, as rejected recharge can still be regarded as a discharge from the aquifer system. It should also be said that there *is* a sense in which rainfall affects impacts: if rainfall is consistently high, water tables will generally be higher and there may be a higher density of surface water features, and a groundwater abstraction may not have to look so far afield before 'capturing' sufficient water. However, the cone of depression will still be superimposed on a pre-existing groundwater profile.

Principle 4: Impacts are the same upstream and downstream. If an abstraction takes place from a uniform aquifer with an initially horizontal water table, then it is easy to understand that the impacts of the abstraction will be radially symmetrical. If an abstraction takes place from groundwater where there is regional flow (where there is a regional hydraulic gradient), the impacts of the abstraction will still be the same upstream as downstream, *at the same radial distance* (again for a linear system). This follows from the principle of superposition, in that a symmetrical drawdown pattern is superimposed on the pre-existing groundwater gradient.

Principle 5: Impacts can be felt beyond the catchment zone of a groundwater abstraction. Over the past few years, a lot of work has been undertaken on defining catchment zones for groundwater abstractions, related to establishing source protection zones in response to rising nitrate levels in groundwater. By definition, all groundwater within a particular catchment zone will eventually end up at the abstraction borehole in question. However, groundwater abstractions can have impacts beyond their catchment zones (that is, beyond their source protection zone), for example by intercepting water that would otherwise have contributed baseflow to a river. This is illustrated in Figure 2.4. Note that catchment zones are usually based on recharge conditions that occur largely in the winter period, so there can be seasonal differences.

[1] In this context, a linear system means a system where the transmissivity does not vary with time. In other words, a confined aquifer, or an unconfined aquifer as long as the drawdown is not significant compared to the saturated thickness. Unconfined chalk can be non-linear, with transmissivity varying (sometimes dramatically) depending on the seasonal water level.

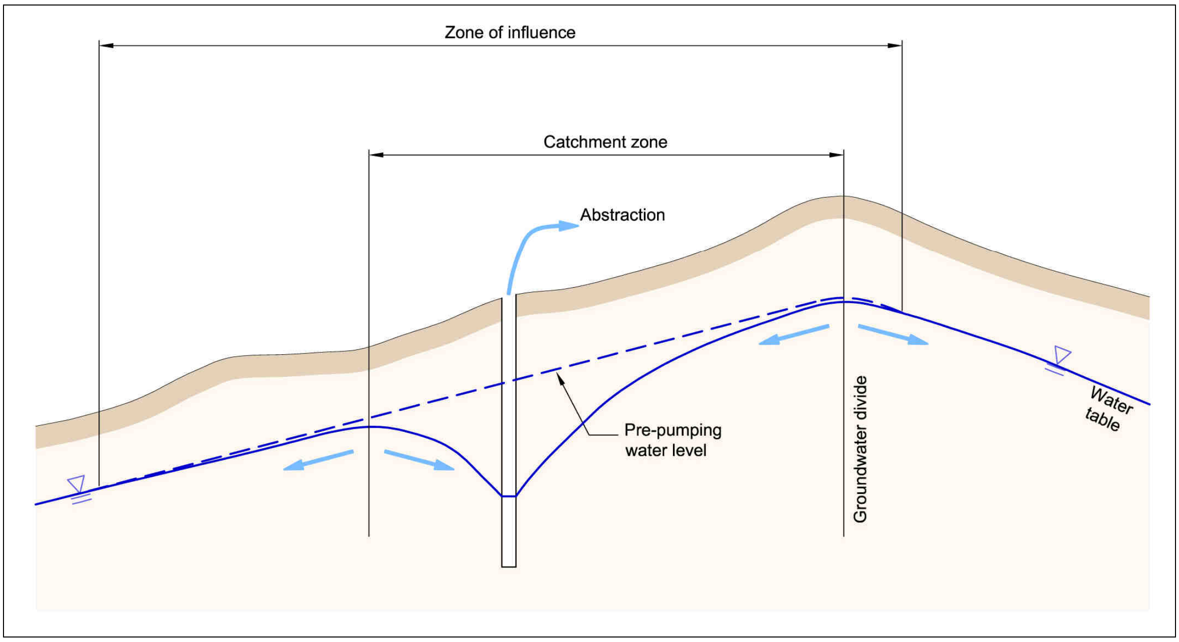


Figure 2.4 Catchment zones and zones of influence
(after Environment Agency 1996)

Principle 6: Groundwater flow divides make no difference to the spread of impacts. From the previous points it follows that groundwater flow divides make no difference to the distribution of the impacts of abstraction, again due to the principle of superposition (see Figure 2.4). The cone of depression spreads out radially and will stop spreading only if it reaches a genuinely impermeable boundary, or if it has prevented an equal amount of water from leaving the aquifer. Abstraction can even change the position of the groundwater flow divide.

Principle 7: Pumping tests are only limited soundings of an aquifer. Data from pumping tests are used to gain information about the hydraulic properties of the aquifer from which the borehole is abstracting. On commencement of pumping, a cone of depression starts moving out radially, and carries on spreading until it has captured sufficient water to balance the abstraction. When interpreting pumping test results, it is essential to realise that they are only limited soundings out into the aquifer for the duration of the test. For example, for a 7-day pumping test, the perimeter of the cone of depression will spread out for 7 days and reach a certain radial distance. The test results *only* represent the volume of aquifer within that perimeter. If the perimeter would have reached a significant hydrogeological feature, such as a barrier, on Day 8, we will know nothing about it from the test results. Extrapolation of results from pumping tests must therefore be undertaken extremely cautiously, with an awareness of the limitations of the data. Incidentally, the speed of propagation of the cone of depression does not depend on the pumping rate, but is related to a parameter known as the aquifer hydraulic diffusivity (transmissivity divided by storage coefficient).

Principle 8: Faults are not necessarily impermeable. For a geological fault to act as a significant barrier to groundwater flow, it needs to be laterally extensive, with hydraulic conductivity orders of magnitude lower than the main aquifer. For example, sections of the major Birmingham Fault have thrown down Mercia Mudstone against the Sherwood Sandstone Group, forming the boundaries of several groundwater management units. However, this is not always the case, and in fact some faults can act as zones of enhanced permeability. In low permeability hard-rock terrain, major fault zones and fracture zones are often assumed to have significantly higher transmissivities than the surrounding rock mass. Banks *et al* (1992) demonstrated that even this assumption cannot be taken for granted; major fracture zones can suffer reduced transmissivity due to the presence of fine-grained fault gouge or clay minerals resulting from weathering or hydrothermal activity. In general, faults within the same aquifer can make a significant difference to local groundwater flow patterns (and thus influence the distribution of local impacts), but they rarely make a significant difference to regional flow patterns (and therefore regional impacts).

Principle 9: Drawdown after 200 days does not necessarily represent drought conditions. In the Stage 2 assessments under the Habitats Directive (see Appendix 1), when assessing the potential impact of a groundwater abstraction on a wetland, for example, the drawdown (at the wetland) after 200 days is often taken as being representative of drought conditions. This is on the basis that 200 days without recharge represents a drought. However, as we have already seen, recharge from rainfall makes no difference to the impacts of a groundwater abstraction, in a linear system. The drawdown predicted by the Theis equation at a given radius after 200 days of abstraction does not necessarily represent the maximum additional drawdown at that radius, with or without a drought. In reality, the maximum additional drawdown at a given radius from an abstraction borehole will occur when the cone of depression has spread until it has stopped an equal amount of water leaving the system; this may be before or after 200 days. Having said that, the 200-day drought could be taken as the starting point for a worst-case scenario. That is, the natural groundwater levels after a 200-day drought could be used as the baseline upon which the additional drawdown due to the abstraction is superimposed.

2.5 Hydrogeology of dewatering

There are many similarities between groundwater abstraction for dewatering and abstraction for say public water supply. There are also important differences, including the following:

- The focus of dewatering abstraction is on maintaining a given groundwater level in an excavation, rather than maintaining a certain abstraction quantity.
- The environmental impact of the dewatering abstraction can be significantly reduced by returning the water to the environment, in other words, by implementing mitigation measures.
- Even though active dewatering operations are often relatively short term, there can be permanent impacts on the local hydrogeological system.

For those not already familiar with the subject, further information on the hydrogeology of dewatering can be found in Appendix 2, including discussions of typical hydrogeological settings for quarries or mines, dewatering methods, the application of HIA to construction dewatering, and long-term impacts. In certain hydrogeological settings that are frequently encountered by mines and quarries, there may be special considerations that have to be taken into account when undertaking HIA. These are as follows:

Karst: dissolutional features such as conduits, caves, sinkholes, and closed depressions can develop in any soluble rock type, including carbonate rocks such as limestones and dolomites, and evaporites such as gypsum, anhydrite and halite. Such dissolutional features give an aquifer karstic properties, and the assumptions built into many models and analytical equations (that the aquifer is homogeneous and isotropic, for example) break down. There is far greater uncertainty when predicting impacts or interpreting monitoring data in karstic aquifers, and a slightly different approach to HIA may be required. The problems of karst, and the revised HIA approach, are described in Appendix 3, but it is recommended that you read the remaining sections of this report first. Figure 2.5 shows the main areas in England and Wales where there is potential for the development of dissolutional features. See the GeoSure service provided by the British Geological Survey for details of obtaining this information at larger scales.

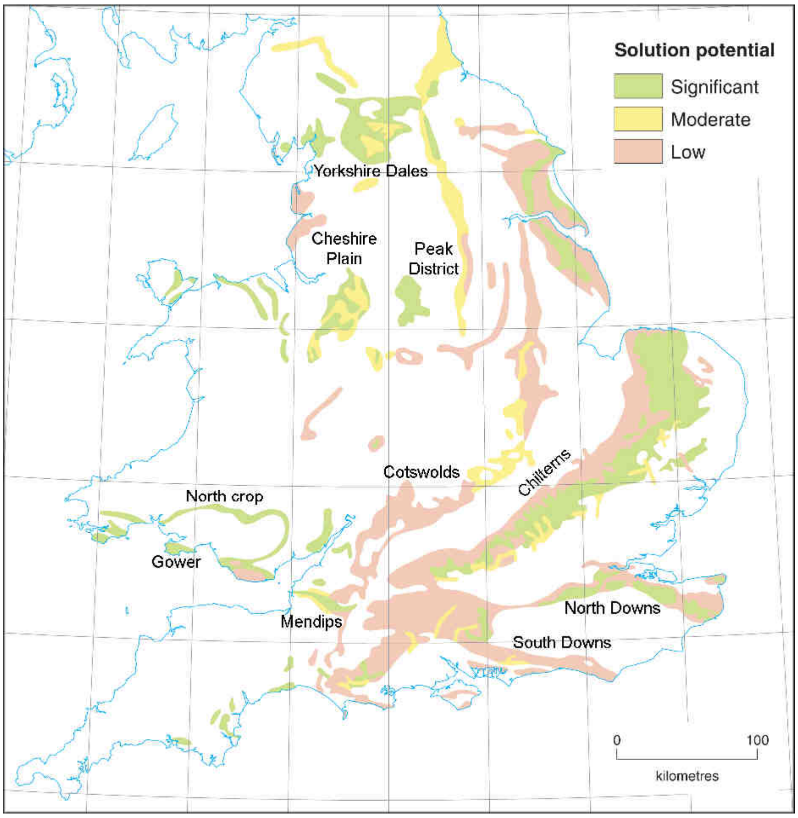


Figure 2.5 Main locations of karst-prone rocks in England and Wales
(adapted from GeoSure information, British Geological Survey)

Fractured crystalline rocks: such as slate, granite, marble, basalt and dolerite are typical of upland terrain in Wales and northern England. The porosity between mineral grains is typically low or negligible, such that the majority of groundwater storage and flow takes place through networks of fractures. Again, assumptions of homogeneity and isotropy are questionable, and these rocks may have to be dealt with in a similar way to karst. See Appendix 4 for further information.

Coal Measures: Appendix 5 discusses the special considerations when undertaking HIA in Coal Measures, including the influence of anthropogenic features such as old underground workings now being encountered by modern opencast workings. Much of the discussion applies equally well to underground mining and to quarries in other settings, such as slate, that also encounter old underground workings.

3 Background to the methodology

3.1 Development criteria

In developing the HIA methodology for dewatering abstractions, certain general criteria were applied. These were that the HIA methodology must:

- be risk-based; that is, the effort and resources used to assess the impacts should be matched to the level of risk of environmental damage.
- emphasise the importance of developing a robust conceptual model of the site that is continually reviewed and updated as new information is collected.
- be able to distinguish between impacts caused by changes in flow, and those caused by changes in water level, and deal with them appropriately.
- take into account the mitigation of impacts by the return of water to the groundwater or surface water system.
- be applicable to any consideration of the long-term impacts after cessation of active dewatering, not just the impacts during the operational phase.
- result in an appropriate level of on-going monitoring, targeted at the issues of real concern.
- identify opportunities to reduce environmental impact by creative use of the abstracted water.
- be compatible with the HIA methodology for conventional groundwater abstractions, while allowing for the special characteristics of abstraction for dewatering.
- be able to cope with a variety of spatial scales (regional and local, for example).

In addition, the HIA methodology is designed to be compatible with the Government's principles of modern regulation. Five principles to be applied to any modern regulatory regime have been set out by the Better Regulation Taskforce (Environment Agency, undated). The regime must be:

Transparent, with clear rules and processes;

Accountable, leading to decisions that can be justified;

Consistent, with the same approach being applied across sectors;

Proportionate, according to the risks involved;

Targeted, with a clear environmental outcome.

Many environmental impacts arising from a groundwater abstraction for dewatering will occur close to the abstraction, especially those caused by changes in the water levels in the surrounding aquifer. However, some impacts caused by changes in flow may occur many kilometres from the abstraction, months or even years after the abstraction has commenced. Most groundwater abstractions are ultimately at the expense of surface water flows, whether they induce additional leakage from rivers or intercept water that would otherwise have discharged to them. Hydrogeological investigations are often undertaken at two scales, regional and local:

Regional scale: typically at the level of groundwater management units, such as those used in the CAMS process, or groundwater bodies as defined by the Water Framework Directive. At this scale, the impact of an individual abstraction may be of little significance, but the cumulative impact of all the abstractions may well be significant. Impacts at a regional scale are often due to changes in flow, and the focus of regional investigations is usually on overall water resources availability.

Local scale: sometimes referred to as the zone of influence of the abstraction, and much harder to define. This zone depends on many factors, such as the size of the abstraction, and the nature of the local hydrogeology. It will be seen later that defining the local zone of influence of the abstraction is an integral part of the HIA process. At the local scale, close to the abstraction, direct impacts of abstraction (on both flows and levels) are much more likely to be significant.

The focus of investigations when undertaking HIA is at the local scale, but the regional picture also has to be taken into account. Under certain hydrogeological conditions, for large abstractions or for confined aquifers, for example, the regional and local scales will sometimes merge. Suffice it to say at this point that the HIA methodology concentrates on the local scale, but moves out as far as is necessary to examine the most distant impacts.

3.2 Tiered approach

The HIA methodology is designed to operate within a tiered approach, which was introduced in Section 2.3, and will now be discussed further. The Environment Agency has chosen a tiered approach for the following reasons:

- It is in line with the Government's recommendations on environmental risk assessment (DETR 2000), which address the issue of having to make robust and defensible decisions on environmental matters in the face of significant uncertainty.
- It enables the level of effort to be matched to the risks associated with the decision being made. For example, when undertaking HIA, much greater effort is likely to be required for a quarry or mine in a major aquifer, close to some Ramsar sites, pumping large quantities of water, compared to a small quarry or mine in an unproductive aquifer, with no sensitive conservation sites in the area, with some seasonal dewatering.
- It minimises unnecessary expenditure on investigations to back up the HIA, because it allows regular assessments to be made of whether the uncertainty has been reduced to an acceptable level.

A rough guide to the level of effort associated with each of the three tiers is as follows:

Tier 1 (Basic): Conceptual model developed from information and data that are fairly easily available from published sources, bodies such as the Environment Agency, the British Geological Survey, and the Centre for Ecology and Hydrology, or the operator's own historical monitoring data. The conceptual model is typically tested using lumped long-term average water balances and simple analytical equations, to arrive at a ‘best basic’ conceptual model. A Tier-1 assessment is likely to be required in virtually all cases (depending on the contents of the forthcoming transitional regulations).

Tier 2 (Intermediate): The sophistication of the conceptual model is increased by testing it using more detailed data, such as time-variant heads and flows, and seasonal or sub-catchment water balances (semi-distributed). More detailed analytical solutions may be used (to investigate the impact of abstraction on river flows, for example), or two-dimensional steady-state groundwater models. Limited field investigations may be required to fill important gaps in the data. Tier-2 assessments are likely to focus on (and be limited to) specific areas of uncertainty that have been highlighted during Tier 1.

Tier 3 (Detailed): The conceptual model represents a high degree of understanding of the hydrogeological and hydrological system, and is likely to be tested using a spatially-distributed and time-variant numerical groundwater model, calibrated and validated against historical data. This is likely to require the collection of data from a wide range of sources, including more field investigations. It is likely that Tier-3 assessments will only be required in a relatively small number of cases.

It is not possible to be prescriptive when describing the tiers, and indeed it is preferable that as much flexibility as possible is retained throughout the process (the information and data requirements will become clearer when the HIA methodology itself is described in Section 4). Having said that, it is entirely reasonable for the dewatering operator to want to know in advance what level of effort is likely to be required for their particular operation, for budgeting purposes as much as any other reason, and a simple scoring exercise will now be described.

3.3 Prediction of likely level of effort

A simple scoring exercise has been developed, and is outlined in this section. It must be emphasised that this (arbitrary) scoring exercise is solely for the purpose of giving a rough idea of the level of effort likely to be required for undertaking HIA at a particular quarry or mine. In no way does it commit the Environment Agency, or indeed the licence applicant, to a particular tier. A set of factors or criteria were first identified, that focus on the most important hydrogeological issues, and that make use of easily-available information. The criteria are as follows:

i. ***Aquifer characteristics:*** based on the general hydrogeology of the aquifer in which the dewatering operation is set, and using classifications that are easily obtainable from groundwater vulnerability maps and geological maps.

ii. ***Water-dependent conservation sites:*** to take into account the presence of environmentally-sensitive sites in the vicinity of the dewatering operation, particularly Special Protection Areas (SPAs), Special Areas of Conservation (SACs), Ramsar sites, Sites of Special Scientific Interest (SSSIs), etc.

iii. ***Water resource availability status:*** based on the CAMS classification for the groundwater management unit in which the dewatering operation is located. Although transfer licences are supposed to imply no consumptive use of water, there can still be significant impacts on other abstractors, and the overall resource status of the unit is a good indicator of the level of stress.

iv. ***Dewatering quantity:*** to take into account the size of the dewatering operation (though not necessarily the size of the excavation itself), and therefore the magnitude of the potential water-related impacts.

Now, within each criterion, a hierarchy of possible scenarios or classes is defined, so that dewatering operations can be systematically assessed against the criteria. The hierarchy is shown in Table 3.1, with further information in Box 3.1.

Table 3.1 Scoring criteria, classes and weights

Criteria and classes	Score	Weight
Aquifer characteristics		
Karst	4	
Principal (major) aquifer	3	2
Secondary (minor) aquifer	2	
Unproductive strata	1	
Water-dependent conservation sites		
Habitats Directive (Natura 2000) sites	4	
Sites of Special Scientific Interest	3	4
Other designations (including National Parks and AONB)	2	
None	1	
Water resource availability status		
Over-abstracted	4	
Over-licensed	3	1
No water available	2	
Water available	1	
Dewatering quantity		
Very large (> 5,000 m^3/d)	4	
Large (2,500 to 5,000 m^3/d)	3	3
Medium (1,000 to 2,500 m^3/d)	2	
Low (< 1,000 m^3/d)	1	

Box 3.1: Further information about the criteria and classes

Aquifer characteristics: simple classification of the aquifer in which the dewatering operation is located, obtained from the groundwater vulnerability maps, published by the Environment Agency, which cover England and Wales at a scale of 1:100,000. These maps can be purchased, or your local Environment Agency office should be able to tell you what your aquifer classification is. Note that karst is not specifically identified on these maps, and some judgement needs to be used. For the purposes of this exercise, if the dewatering operation is in carbonate rock (primarily limestone or chalk), and there is significant fissure flow or the presence of dissolution features, assume the aquifer is karstic (see Appendix 3 for more information, and also Figure 2.5).

Water-dependent conservation sites: look for such sites within a radius of say 3 to 5 km of the dewatering operation, especially if the sites are likely to be in hydraulic continuity with the operation. Information on Ramsar and Natura 2000 sites can be found on the Joint Nature Conservation Committee website (http://www.jncc.gov.uk), and information on SSSIs can be found on the websites of Natural England (http://www.naturalengland.org.uk) and the Countryside Council for Wales (http://www.ccw.gov.uk). Score for the 'highest' site designation present in the search radius. The search radius is admittedly arbitrary, and can be changed if there is already good reason to believe that it should be larger or smaller. If in doubt, talk to your local Environment Agency office.

Water resource availability status: simple CAMS classification for the groundwater management unit in which the dewatering operation is located. Look on the Environment Agency's website (http://www.environment-agency.gov.uk) to find your local CAMS area and to discover whether or not the strategy is complete for that area. In the absence of a CAMS classification, your local Environment Agency office will tell you what the licensing policy is for the location of the operation (if there is none, assume 'Water available').

Dewatering quantity: based on estimated annual average dewatering quantities, expressed as a daily average. This information should already be available to, or can be estimated by, the operator.

To operate the scoring system:

i. For the dewatering operation in question, find out the relevant information and allocate a score for each criterion, depending on what class the operation falls into in each case.

ii. Multiply each score by the relevant weight (last column in Table 3.1), and add up the weighted scores. For example, an operation in a major aquifer, with a Natura 2000 site nearby, in a 'No water available' CAMS unit, pumping 600 m^3/d would score: (3 x 2) + (4 x 4) + (2 x 1) + (1 x 3) = 27.

iii. The highest and lowest possible total scores are 40 and 10 respectively, so now use the guide shown in Table 3.2.

Table 3.2 Relationship between total weighted score and likely level of effort

Total weighted score	Level of effort likely to be required
31 to 40	Tier 3
21 to 30	Tier 2
10 to 20	Tier 1

It is worth repeating that this scoring exercise is presented here solely for the purpose of giving a rough idea of the level of effort likely to be required. In practice, common sense applies at every stage of the HIA methodology, and there will be many exceptions to the anticipated level of effort, usually on the side of minimising the effort. For example, imagine a dewatering operation that scores the maximum under each criterion. In other words, it is in a karst aquifer, with Natura 2000 sites nearby, in an 'Over-abstracted' CAMS unit, and pumping very large quantities of water. Imagine now that the applicant and the Environment Agency are in complete agreement, early on in the process, that the dewatering is impacting a particular SAC, but that there are restrictions or mitigation measures that would remove the risk of impact to the SAC. It would be to nobody's advantage to pursue the HIA relentlessly to the end of Tier 3 if the problem can be identified and solved early on, even at Tier 1 (and it may already be covered by Section 106 agreements under the planning system).

Readers already familiar with this type of scoring exercise will recognise that it is nothing new, but is really a form of multi-criteria decision analysis (MCDA), specifically an adaptation of the linear-additive model. There is a large body of literature on the subject of MCDA, and this report is not the place to discuss it in detail. Suffice it to say that Government guidance on appraisal and evaluation says this about the linear-additive model of MCDA:

> *“Models of this type have a well-established record of providing robust and effective support to decision-makers working on a range of problems and in various circumstances” (Defra 2001)*

The linear-additive model of MCDA goes on to form the foundation for the more detailed recommendations later in the same reference. It is likely that the Environment Agency will continue to refine this simple scoring system, and a spreadsheet will eventually be made available on its website, accompanied by guidance on filling it in.

3.4 Tools and techniques

There are many tools and techniques available that can be of great help when undertaking HIA. Unfortunately, there is no single tool or technique that covers everything, so it is a question of using technical judgement on when to use which tool or technique. It is also a question of being realistic about the limitations and built-in assumptions of each tool or technique. Let us now look briefly at some possible tools and techniques.

3.4.1 Tier 1 tools

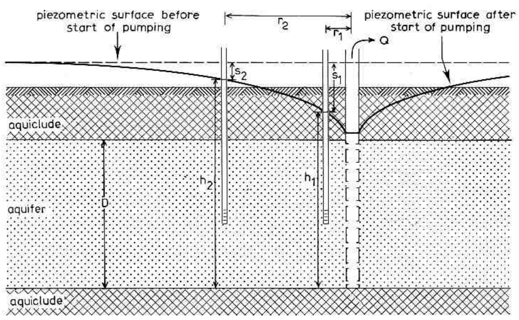
The main tools likely to be used at the level of Tier 1 are simple analytical equations and lumped long-term average water balances. Two good examples of useful analytical equations are the Thiem and Thiem-Dupuit equations, shown in Box 3.2. The equations in Box 3.2 are given in their most general form, but they are commonly used in other ways:

- If only one piezometer is available (at distance r_2 in the diagrams in Box 3.2), the water level in, and radius of, the pumping well (h_w and r_w) can be used instead of the 'inner' piezometer. However, care must be taken to allow for the effects of well losses and the breakdown close to the well of some of the assumptions built into the equations.
- When applied to excavations instead of wells, the equivalent radius (r_e) is often used instead of r_w. For a circular excavation (or circle of wellpoints), r_e would simply be the radius of the circle. Expressions for r_e are available for other shapes such as rectangles (see Preene *et al* 2000).
- The radius of influence (R_o) of a groundwater abstraction, defined as the radius at which drawdown is zero, is sometimes estimated by setting h_2 to the original water table or piezometric surface, if all other parameters are known.
- If R_o can be estimated independently, the equations are often used to estimate the inflow (Q) to a planned excavation, by setting h_1 to the desired level at the edge of the excavation (at r_e).

Box 3.2: Thiem and Thiem-Dupuit equations

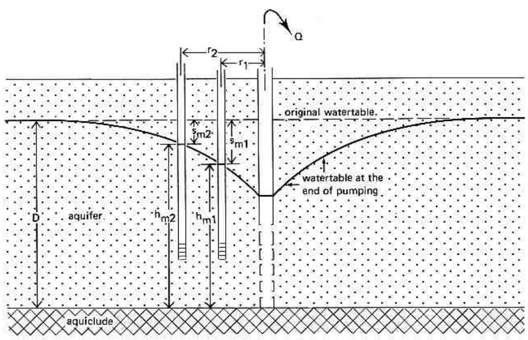
Thiem equation (steady-state confined flow)

$$Q = \frac{2\pi KD(h_2 - h_1)}{2.30\log(r_2/r_1)} = \frac{2\pi KD(s_1 - s_2)}{2.30\log(r_2/r_1)}$$



Thiem-Dupuit equation (steady-state unconfined flow)

$$Q = \frac{\pi K(h_2^2 - h_1^2)}{2.30\log(r_2/r_1)}$$



Both diagrams from Kruseman and de Ridder (1990).
Q = pumping rate [L^3/T]; K = hydraulic conductivity [L/T];
D = saturated aquifer thickness [L];
h_i = elevation of water table or piezometric surface [L];
s_i = drawdown [L]; r_i = radius [L];
where L = length and T = time.

While on this subject, a few words need to be said about the so-called Sichardt formula. This is an empirical formula, $R_o = Cs\sqrt{K}$, where s is the target drawdown in the excavation, R_o and K are as defined above, and C is an empirical calibration factor. If s and K are in m and m/s respectively, to obtain R_o in m, C is usually taken as 3,000 for radial flow (Preene *et al* 2000). The origins of the Sichardt formula are obscure, and the assumptions built into C are unclear, but it is widely used to estimate R_o, with the value obtained then being used as described above to estimate Q. However, its use is not encouraged in this report, partly because of the uncertainty surrounding C, but mainly because it is not consistent with the principle of the impact of an abstraction spreading until it has 'captured' sufficient water (see Section 2.4). The Sichardt formula would give exactly the same answer whether an abstraction was immediately adjacent to a hydraulically-connected river, or was remote from any such features.

Such equations must always be used with care, bearing in mind all the assumptions on which the equations are based. Several different equations may be used when looking at the hydrogeology of a particular dewatering operation. As part of this project, over 20 analytical equations have been assembled from textbooks and other publications, and put into an MS Excel spreadsheet for convenience, for use when assessing the impacts of dewatering. The equations are summarised in Table 3.3. This spreadsheet will be made available in due course by the Environment Agency.

Table 3.3 Summary of analytical equations in Tier 1 tools spreadsheet

Analytical equation	**Comments**
Equations for abstraction	
Thiem Thiem–Dupuit	Steady-state flow to a well in confined and unconfined aquifers
Theis	Time-variant flow to a well in a confined aquifer
Cooper–Jacob	Approximation to Theis equation for time-variant flow in confined and unconfined aquifers
Dupuit–Forcheimer	Steady-state flow (per unit width) into the straight side of a pit, in an unconfined aquifer
De Glee	Steady-state flow to a well in a leaky aquifer
Construction dewatering equations	
Trench with flow from one side Trench with flow from two sides Wellpoints, double row, partial penetration Single well with image well	Variety of equations for confined and unconfined conditions taken from the CIRIA guide to groundwater control (Preene *et al* 2000)
Recharge equations	
Injection well recharge	Recharge into an unconfined aquifer through an injection well
River–aquifer leakage	Vertical flow through low-permeability river-bed sediments
River–aquifer leakage	Flow between river and aquifer through the sides of a fully-penetrating river channel
Miscellaneous equations	
Radius of influence (Niccoli *et al* 1998) Flow to a pit (Marinelli and Niccoli 2000) Radius of influence (Sichardt) Radius of influence (Bear)	Variety of equations for estimating radius of influence and flow to a pit. Included for comparison, but not recommended for use during HIA because R_o is dependent on rainfall or arbitrary constants.
Effective radius of wellfield	For conversion of rectangular wellfield to circular
Darcy–Weisbach	For head loss in a pipe due to friction (can be applicable to flow in conduits, for example in karst)
Operational water balances	
Water balance spreadsheet	Simple tool for calculating operational water balances for dewatering. Can be adapted to suit individual dewatering operations.

Note that some of the equations in Table 3.3, in particular the ones for estimating radius of influence, are not necessarily recommended for use during HIA, but they have been included to enable comparisons to be made between different methods. The reservations on the use of these equations for HIA are based, for example, on the inclusion of rainfall as a parameter, which contradicts the key principle of impacts not being dependent on rainfall recharge. The Sichardt equation has already been discussed.

Apart from analytical equations, the other main type of Tier 1 tool is the water balance. Water balances are an integral part of conceptual modelling. Mathematically, they involve nothing more complicated than basic arithmetic; the hardest part is identifying and estimating all the inflows to and outflows from the water balance. Further discussion of water balances can be found in Appendix 6.

3.4.2 Tier 2 tools

In Tier 2, the conceptual model is tested in more detail, typically using time-variant data instead of steady-state. Tools and techniques likely to be used at the level of Tier 2 include the following:

- *Water balances:* more sophisticated water balances using time-variant (or at least seasonal) data, perhaps examining sub-catchments, and looking at scenarios such as wet, dry and average years.
- *Analytical models:* typically two-dimensional steady-state models based on analytical equations, available as proprietary software, and often used to evaluate the effects of multiple abstractions in a uniform regional groundwater flow field. Examples of such models include WinFlow and TWODAN.
- *IGARF:* a spreadsheet-based tool for examining the impacts of groundwater abstraction on up to two rivers, including estimating depletion of river flow in time and space.
- *Radial flow models:* analytical or numerical models (such as RADFLOW) that consider radial flow to a borehole or well in a variety of aquifer configurations, using radial instead of Cartesian co-ordinates.
- *Recharge spreadsheets:* spreadsheet-based tools (see Environment Agency 2006, for example) are available for estimating recharge, using a water-balance approach coupled with daily soil moisture calculations. The results can usually be fed into a numerical groundwater model.

As mentioned already, water balances are discussed in Appendix 6. Further details on the other Tier 2 tools can be found in Appendix 7.

3.4.3 Tier 3 tools

The main tools likely to be used at the level of Tier 3 are spatially-distributed, time-variant and usually three-dimensional numerical groundwater models, calibrated and validated against historical data. Such models should only be used by experienced hydrogeologists and groundwater modellers, as their use is by no means intuitive. The main modelling codes likely to be used for simulating the impacts of dewatering are:

- *MODFLOW:* a freely available code developed by the United States Geological Survey, which has become the industry standard. Many pre- and post-processors and other useful software modules have been developed for MODFLOW.
- *ZOOMQ3D:* a relatively new code being developed jointly by the University of Birmingham, the Environment Agency, and the British Geological Survey.
- *MIKE-SHE:* a package of models and graphical user interface developed and marketed by DHI Water & Environment.
- *FEFLOW:* a finite-element model (the other three are finite-difference models) developed by WASY GmBH.

Further details on these models, and a general discussion of techniques for modelling dewatering, can be found in Appendix 8. The Environment Agency's current preferred numerical groundwater modelling software is MODFLOW, with Groundwater Vistas as the user interface.

4 The HIA methodology

4.1 Overall HIA structure

The HIA methodology is presented as a sequence of steps (Box 4.1), which should be followed for all groundwater licence applications. This may at first seem onerous, but the process has a logical progression, and the steps impose some discipline on each appraisal. At the same time, the steps in the process are not prescriptive, and the level of effort expended on each step can be matched to the situation. In other words, some steps will be a formality for many applications, but it is very important that the same thought-process occurs every time, to ensure consistency.

Box 4.1: The HIA methodology

Step 1: Establish the regional water resource status.

Step 2: Develop a conceptual model for the dewatering operation and the surrounding area.

Step 3: Based on the conceptual model, identify all potential water features which are susceptible to flow impacts.

Step 4: Apportion the likely flow impacts to the water features, again based on the conceptual model.

Step 5: For the relevant water features, allow for the mitigating effects of any discharges associated with the abstraction, to arrive at net flow impacts.

Step 6: Assess the significance of the net flow impacts.

Step 7: Define the search area for drawdown impacts.

Step 8: Identify all the features within the search area which could potentially be impacted by drawdown.

Step 9: For all these features, predict the likely drawdown impacts.

Step 10: For the relevant water features, allow for the effects of any measures being taken to mitigate the drawdown impacts.

Step 11: Assess the significance of the net drawdown impacts.

Step 12: Assess the water quality impacts.

Step 13: If necessary, redesign the mitigation measures to minimise the flow and drawdown impacts.

Step 14: Develop a monitoring strategy, focussing on the features likely to experience flow or drawdown impacts.

It is fully recognised that the impacts of many dewatering abstractions are mitigated by water being discharged back into the environment. However, the HIA methodology always assesses the impacts as if there were no mitigation, then adds back in the beneficial effects of mitigation. This is done for the following reasons:

- The locations where the impacts are felt may well be different from the locations where the water is being discharged.
- The timing of the abstraction impacts may be different from the timing of the beneficial effects of the discharge (especially if abstraction is from groundwater and the discharge is direct to surface water, for example).
- The combined effect of these first two reasons is that impacts may still be felt even if the consumptiveness of the dewatering operation is zero.
- It cannot always be assumed that dewatering is non-consumptive, and the quantity discharged is often significantly less than

the quantity abstracted (for reasons such as evaporation).

- For existing operations, it provides opportunities to redesign the current discharge arrangements to target the most serious impacts.
- It allows for the fact that some impacts may be permanent, even after the cessation of active dewatering (at which point active mitigation may also cease).

The steps will now be considered in more detail. When following the procedure, the tiered approach described earlier should always be kept in mind, and the procedure repeated as many times as necessary (iterations within the tiers and moving through the tiers) until the required level of confidence has been achieved. Also, the basic principles established earlier (recharge makes no difference to impacts; impacts are the same upstream and downstream; and the abstraction spreads until it has stopped an equal amount of water leaving the aquifer) should be kept very much to the fore.

It is recognised that many quarries and mines have already undertaken detailed hydrogeological investigations for other purposes (in relation to the Habitats Directive, for example, or as part of the Review of Old Mineral Permissions). There is no reason why the results of these investigations cannot be used in support of the HIA, and the applicant's responses under some of the steps of the HIA methodology may simply be to refer to, or summarise, previous work.

4.2 The HIA methodology

4.2.1 Step 1: Establish the regional water resource status

The starting point for the HIA is to establish the CAMS status for the area in which the dewatering operation is located. As described in Appendix 1, each CAMS area will eventually be assigned a resource availability status, from four possible categories:

i. *Water available:* Water likely to be available at all flows including low flows. Restrictions may apply.

v. *No water available:* No water available for further licensing at low flows, although water may be available at higher flows with appropriate restrictions.

ii. *Over-licensed:* Current actual abstraction is resulting in no water available at low flows. If existing licences were used to their full allocation they would have the potential to cause unacceptable environmental impact at low flows. Water may be available at high flows with appropriate restrictions.

iii. *Over-abstracted:* Existing abstraction is causing unacceptable environmental impact at low flows. Water may still be available at high flows with appropriate restrictions.

Some people may already be questioning the relevance of resource availability status to HIA for dewatering. Its main relevance is that groundwater is held in storage in aquifers, and represents a useful resource. In groundwater management units where this resource is over-committed, abstracting large quantities of groundwater just for dewatering purposes may make the resource unavailable for other uses, because some of it is lost to evaporation, for example, or it is discharged to surface water and flows out of the system much more quickly than it would have done as groundwater. This is especially a problem in shallow gravel deposits, where dewatering may represent a substantial proportion of the saturated aquifer thickness (Huxley *et al* 2004). Having said that, in the case of applications for transfer licences, the CAMS resource availability status must be interpreted with care, for the following reasons:

- Transfer licences are not supposed to involve intervening consumptive use of the water (between the abstraction and discharge points). In this case, dewatering abstractions may effectively be 'transparent' as far as CAMS is concerned.
- On a regional scale, the net effect on resource availability of a non-consumptive abstraction will often be negligible, unless the water is being transferred out of the unit.
- The resource availability status is usually generalised for the whole CAMS area, and the hydrogeological picture may look different when viewed at a local scale.

Full assessments have not yet been completed for all CAMS areas, as they are being completed on a rolling programme up to 2008. You can find out which CAMS area the dewatering operation is in, and information on which CAMS areas have assessments available, on the CAMS homepage on the Environment Agency's website. If a CAMS assessment has been completed for your area, then a summary document on CD-ROM can be obtained from the Environment Agency. This document contains plenty of information that is of great help when developing the conceptual model, including the results of the Resource Assessment Methodology (RAM) Framework tests (see Appendix 1). If a CAMS assessment is not yet available, then the Environment Agency will provide the Water Framework Directive (WFD) risk category for the relevant groundwater body. The WFD initial characterisation (see Appendix 1) has placed groundwater bodies into one of four risk categories (risk of failing to meet the WFD objectives in time):

- At risk
- Probably at risk
- Probably not at risk
- Not at risk

Any groundwater body falling within either of the two 'Probably' categories will be the subject of further characterisation, to establish whether it should really be in the 'At risk' or the 'Not at risk' category. If neither a CAMS status nor a WFD risk status is available, then it is likely that the dewatering operation is located in unproductive strata (formerly known as a non-aquifer), and the focus of the HIA is likely to be entirely on specific impacts at the local scale. The focus of the HIA will differ, depending on the CAMS or WFD status, along the lines shown in Table 4.1.

Table 4.1 HIA in relation to CAMS and WFD

CAMS status	WFD status	Comments on HIA
No status defined	No status defined	It is likely that the abstraction is located in unproductive strata (formerly known as a non-aquifer) and the focus of the HIA is likely to be entirely on specific impacts at the local scale.
Water available	Not at risk	HIA is likely to concentrate on specific impacts at the local scale.
No water available	Probably not at risk	The onus is on abstractors to demonstrate that their abstraction is not part of the regional water resources problem. Abstractors may have to accept seasonal restrictions.
Over-licensed	Probably at risk	
Over-abstracted	At risk	HIA needs to demonstrate that the proposed abstraction will not exacerbate the regional water resources problem. There will almost certainly be seasonal restrictions.

4.2.2 Step 2: Develop a conceptual model for the dewatering operation

The key to HIA is a good conceptual model. As has already been emphasised several times, the process of developing a conceptual model is continuous and cyclical, with the development continuing until the level of confidence in the model is sufficiently high to enable a decision to be made. The types of information likely to be needed for developing the conceptual model are as follows:

i. The geology of the quarry (or mine or other excavation) and the surrounding area, focussing on structure, faulting, fracturing etc, and the way in which these influence the movement of groundwater. See Appendices 3 to 5 for guidance on dealing with karst, fractured crystalline rocks and Coal Measures respectively.

ii. The hydrogeology of the area, including known or estimated groundwater levels in different places, likely groundwater flow paths or flow patterns (horizontally and vertically), for example as revealed by places in the excavation where there is ingress or seepage of water.

iii. Surface water features in the area, such as lakes, rivers, springs, ponds, storage reservoirs, lagoons etc, concentrating on their hydrological behaviour, their elevations in relation to groundwater levels, and their potential interaction with groundwater.

iv. The physical configuration of the excavated void, particularly the way in which this may have altered the flow of surface water and groundwater.

v. The dewatering arrangements themselves, including data on typical pumping quantities and the way these vary seasonally.

vi. Details on what subsequently happens to the pumped water, including where and when it is discharged, and the influence of the discharge on the receiving water body.

vii. Other water uses within the quarry or mine operation, such as dust suppression, vehicle washing, mineral washing, concrete batching etc, and how these relate to the dewatering arrangements, if at all.

viii. Any other useful information, such as previous studies on potential impacts on sensitive conservation sites, or mitigation measures already in place (those connected with water, that is).

Conceptual models should be illustrated wherever possible with appropriate maps, sketches, diagrams, graphs and cross-sections, bearing in mind that the level of detail should match the required level of confidence. An integral part of the conceptual model will be a water balance for the dewatering operation, focussing on answering the questions: How much water is actually pumped for dewatering purposes? How much of that water really comes from groundwater, as opposed to pumping surface run-off and direct rainfall out of the excavation? What subsequently happens to that water? How much water is consumed, and is not therefore available for return to the environment? Further information on conceptual modelling, especially developing water balances, can be found in Appendix 6.

At this point, readers may be asking the question, for what area should the conceptual model be developed, as the zone of influence of the abstraction has not yet been defined? Unfortunately, this is a chicken-and-egg situation, in that the zone of influence cannot be defined without a conceptual model, but the area for the conceptual model is not yet clear. This is where the cyclical approach to conceptual modelling comes into its own. An educated guess should be made of the area over which to collect information for the first attempt at the conceptual model, with the area being revised at the start of each cycle. Remember the additional uncertainty involved in karst and hard-rock situations (see Appendices 3 and 4).

4.2.3 Step 3: Identify water features susceptible to flow impacts

The starting point for this step is the question: Which water features are likely to be deprived of water by the abstraction? In other words, at which places will quantities of water (totalling the abstraction quantity) eventually be stopped from leaving the aquifer (or additional flows induced), when the system has eventually achieved a new equilibrium? This step is basically trying to identify the recharge boundary conditions for the conceptual model.

Ignoring any mitigation measures for now, use the conceptual model to identify all the potential water features that are susceptible to flow impacts. These may include springs, rivers, and some lakes or wetlands, but note that it does not yet include other groundwater abstractions (as they will be covered by drawdown impacts later in the procedure). Deciding how far afield to look for potential water features that could be deprived of water by the abstraction is a matter of professional judgement, but the following factors should be borne in mind:

- Known geological features such as the edge of the aquifer, or anisotropy in the hydraulic characteristics of the aquifer, may indicate that the features will not be equally distributed (spatially) around the proposed abstraction. This should be clear from the conceptual model.
- Abstractions from confined aquifers ultimately have to obtain their water from outcrop areas (unless it can be shown that there is vertical leakage into the aquifer through a leaky semi-confining layer). You will probably have to look further afield than in unconfined aquifers.
- Care should be exercised if it is suspected that a river is hydraulically isolated, that is, disconnected from the groundwater. In this case, the abstraction may not increase the amount of leakage from the river, and the perched river may not be impacted by the abstraction. However, it may be necessary to look for water features beyond (on the opposite side of) the river, which may be impacted by the abstraction.
- For high-transmissivity systems, the distance from the abstraction to the water feature will be of less importance than the hydraulic resistance of the deposits between the surface water and the aquifer (see Box 4.2).

Then, using the conceptual model, decide which of these water features may be deprived of water by the abstraction. Note that it is essential to identify at least *some* such features, because they will be needed in subsequent steps. It is usually helpful to prepare a sketch map showing these features, such as the one shown in Figure 4.1.

4.2.4 Step 4: Apportion the flow impacts

Now apportion the flow impacts to these water features, either as portions of the total abstraction quantity, or as percentages (making sure that the percentages add up to 100 per cent). This is illustrated in Figure 4.1. In this example, it has been decided from the conceptual model that the water features likely to be deprived of water by the abstraction are Spring A, a stretch of River A, Wetland B and a stretch of River B, in the proportions 20, 50, 10 and 20 per cent respectively. In other words, if the total abstraction quantity is 12 Ml/d, then we are implying that Spring A will be deprived of 2.4 Ml/d of flow, River A will be deprived of 6 Ml/d, and so on.

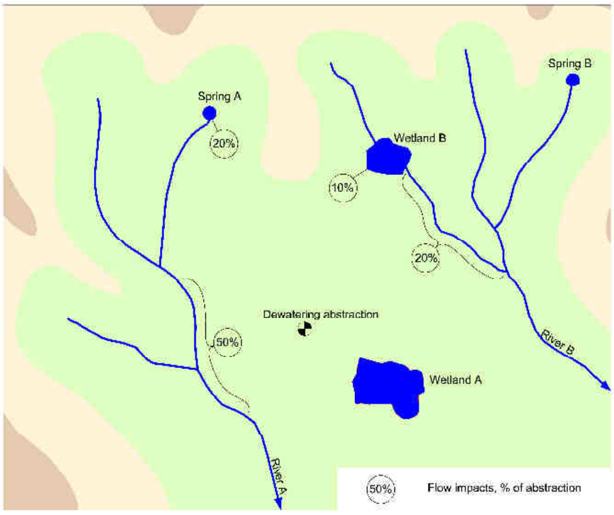


Figure 4.1 Sketch map of water features

Having said that, there is a very important issue here, which is what figure to use for the total abstraction quantity. This is critical to the assessment of flow and drawdown impacts. The most obvious problem is how to allow for surface run-off and direct rainfall that enter the excavation and have to be pumped out by the dewatering system. If the full quantity actually pumped were used, this would give a false impression of the impact of the dewatering, as a proportion of the water pumped would have run off anyway, without reaching the groundwater, before the excavation existed. This issue is discussed in detail in Appendix 6, where it is suggested that the total quantity pumped is factored down according to typical baseflow indices from the rivers in the area.

Box 4.2: Hydraulic resistances

The concept of hydraulic resistance is most commonly applied to layered aquifers. Consider an aquifer with three horizontal layers, each with a vertical thickness (b_i) and hydraulic conductivity (K_i). The vertical hydraulic resistance of Layer 1 is b_1/K_1 and so on. The total hydraulic resistance of all three layers (in a vertical direction) is $b_1/K_1 + b_2/K_2 + b_3/K_3$. The inverse of hydraulic resistance, K/b, is also commonly used, and is usually referred to as a leakage factor or leakance.

The idea of adding hydraulic resistances can be applied to manual apportionment of flow impacts in the following way: instead of a vertical pathway through the horizontal aquifer layers just described, imagine a flow pathway from a river or other water feature to an abstraction, and instead of layer thickness, use the length of the flow pathway. This pathway may consist of several components, for example, low-conductivity river bed sediments, followed by a section of aquifer with certain hydraulic properties, followed in turn by a section of aquifer with different hydraulic properties. Estimate the total hydraulic resistance for the whole pathway by adding the hydraulic resistances of each component. Do this for each water feature, compare the resistances, and apportion the flow impacts to each water feature in proportion to their relative resistances. This is admittedly a crude approach, which ignores factors such as the head difference between each water feature and the abstraction, and the fact that in practice groundwater follows many pathways to get from one place to another. However, it is a quick way of getting a feel for the relative ease with which an abstraction could 'capture' water from various places.

This can lead to some surprising conclusions. Take for example, an aquifer with high hydraulic conductivity (let us say 200 m/d), crossed by two parallel rivers 1 km apart, each with very low conductivity river bed sediments (say a layer 1 m thick with hydraulic conductivity of 0.002 m/d). Now introduce a groundwater abstraction between the two rivers, 100 m from one and 900 m from the other. The hydraulic resistance between the first river and the abstraction would be (1/0.002) + (100/200) = 500.5 days, and between the second river and the abstraction would be (1/0.002) + (900/200) = 504.5 days. This implies that the abstraction will impact both rivers virtually 50:50, even though it is very much closer to one of the rivers. In other words, in this particular situation (high contrast between river bed and aquifer K), the calculation is relatively insensitive to the distance from the abstraction to the river. See also Appendix 4 for another example of the use of hydraulic resistances.

Step 4 is based on the key principles described in Section 2.4, especially Principle 1, that the impact of a groundwater abstraction spreads until it has stopped an equal amount of water leaving the aquifer. In apportioning the flow impacts, we are making a judgement, based on the conceptual model, on where we think the abstraction is stopping water from leaving the aquifer (or 'capturing' discharge from the aquifer). There is of course the important question of timing. The impacts of a relatively new abstraction may not be felt by a water feature for some considerable time, especially in high-storage aquifers. However, this step is also firmly based on the idea that we must assess the long-term impacts of the abstraction, when the system has eventually reached a new equilibrium.

Note that for seasonal or intermittent abstractions, it may be necessary to consider the instantaneous maximum pumping rate in addition to the annual total abstraction. For a steady-state system, the annual total abstraction will have the same overall impact as a short-term abstraction at a high rate followed by a period of zero pumping. However, the seasonal distribution of the impacts may vary. Note also that for a wellfield with widely-spaced wells, the location of the impacts may depend on which wells are currently pumping, even though there is constant abstraction from the wellfield as a whole. The approach to both these problems is to use hydrogeological imagination, asking the question: What timing or location of pumping will cause the worst impacts?

The method used to apportion the flow impacts depends largely on the data available and the acceptable degree of uncertainty. The main methods are as follows:

- *Manual allocation:* if no suitable data are available, or a high degree of uncertainty is acceptable, then the flow impacts can be allocated manually, using professional judgement. For example, for an abstraction close to a spring, it might be decided to allocate 100 per cent of the flow impact to the spring. This is effectively a worst-case scenario for the spring. Alternatively, some simple calculations can be performed, based on the estimated hydraulic resistance between the abstraction and each feature in turn, and then apportioning the impact in proportion to these resistances (see Box 4.2).

- *Analytical tools:* at an intermediate level, the IGARF spreadsheet model may also be useful for investigating possible scenarios using different parameter estimates. Parameters likely to be required include: radial distance of the river from the abstraction; aquifer hydraulic conductivity and storage coefficient; and river bed thickness and conductivity. Choosing a value for the hydraulic conductivity of river bed sediments is particularly difficult, but some guidance on typical values is given in Calver (2001).

- *Regional numerical groundwater model:* if a suitable model is already available, or if a high degree of confidence is essential, then a numerical groundwater model can be used to apportion the flow impacts. This is achieved by performing prediction runs with the proposed abstraction included in the model with all the existing abstractions, and will probably require help from a specialist groundwater modeller.

In the tiered approach being used in this report, these methods correspond roughly to Tiers 1 to 3 respectively, but it is sensible to use the best tool available. That is, if a Tier 1 assessment is being undertaken and a good regional numerical groundwater model is already available, then it would be sensible to use it (although it should be recognised that there might be significant costs associated with this).

4.2.5 Step 5: Allow for mitigation of flow impacts

Having estimated the flow impacts on relevant water features, the mitigating effects of any discharges associated with the abstraction are now added back in, to arrive at an estimate of the net flow impacts. If it is not required for other purposes, the water from many dewatering abstractions is discharged into a convenient surface watercourse, or into a recharge trench or lagoon, where it soaks back into the ground. If done carefully, this can be very beneficial to the receiving water body, by restoring flows or levels, and mitigating the impacts caused by the abstraction. However, assessing the mitigating effects of the discharges is not necessarily straightforward, for the following reasons:

- The quantity of water reaching the receiving water bodies may be significantly less than the quantity originally abstracted. This is known as the 'consumptiveness' of the abstraction (see Box 4.3).
- The receiving water body may not be the one (or may not be the *only* one) being impacted by the abstraction. In other words, the dewatering abstraction may be affecting a certain stream, but the water is being discharged into a different stream. The same can apply to recharge trenches, as the abstraction and recharge may be from and into different groundwater bodies (especially in layered aquifers).
- The timing of the impacts from the abstraction may be completely different from the timing of the benefits of the discharge, especially if the abstracted water is stored before being discharged, or if the abstraction is from groundwater (diffuse and delayed impact on surface water) and the discharge is to surface water (immediate effect on flows in a specific place).
- If the discharge is into a stream that is already being impacted by the abstraction, then the overall benefit to the stream flows will be less than 100 per cent of the discharge quantity. The amount by which the flow is increased above what it would naturally have been is known as the 'net gain' (see Box 4.3).
- Discharges to surface watercourses, especially small streams, may significantly alter the natural flow regime, either by evening out natural flow variability, or by introducing sudden large changes in flow when pumps cut in and out. The discharges may also have water quality impacts, particularly for parameters such as temperature and suspended solids.

Successfully allowing for the mitigation of flow impacts under this step depends on a good conceptual model, a sound understanding of the hydrological and hydrogeological mechanisms at work, and taking care to add discharges back into the correct surface water and groundwater bodies, after taking into account issues such as consumptiveness and net gain.

4.2.6 Step 6: Assess the significance of the net flow impacts

If the full RAM Framework calculations are available as part of a completed CAMS assessment, then that contains well-defined methods for determining the environmental acceptability of flow impacts, particularly for rivers. This involves using information on four ecological indicators (fish, macro-invertebrates, macrophytes and physical characteristics) to define Environmental Weightings, which in turn are used with long-term flow duration curves to derive appropriate Ecological River Flow Objectives (see Environment Agency 2002b). The CAMS summary document also contains the resource status, the locations of river assessment points, and details of any adverse regional trends.

Box 4.3: Consumptiveness and net gain

Consumptiveness: Usually expressed as a percentage, the consumptiveness of an abstraction is the proportion of the quantity abstracted which is *not* returned to the environment. Traditionally, standard percentages have been used for various water uses. For example, abstractions for spray irrigation were normally regarded as 100 per cent consumptive, whereas abstractions for the flow through fish farms were often classified as 0 per cent consumptive. Abstractions for mineral washing were typically regarded as about 5 per cent consumptive. However, consumptiveness was rarely studied in any detail, and the percentages were effectively rules of thumb. In the context of quarry or mine dewatering, the quantity of water reaching the receiving water bodies may be significantly less than the quantity originally abstracted, even if there has been no specific 'intervening use' of the water. Possible reasons for this include the following:

- Open-water evaporation, especially if the water is first discharged into settling ponds before being pumped on to the final discharge point.
- Water leaving the site in the form of wet gravel or rock (when transporting the product).
- Water used for dust-suppression or wheel-washing etc, much of which would subsequently evaporate.
- Water discharged into recharge trenches that does not find its way to the water table, for example because it is retained as storage in the unsaturated zone, it is lost as interflow within the unsaturated zone, or it is lost through subsequent evapotranspiration.

Establishing the consumptiveness of the dewatering abstraction is critical to assessing the net impacts of the abstraction, and may also determine whether or not a full licence is required in addition to a transfer licence. Consumptiveness is best estimated using a water balance, focussing on accounting for all the abstracted water (see Appendix 6). Interestingly, a recent Environment Agency consultation document has proposed the following loss factors (equivalent to consumptiveness): 100 per cent for dust suppression; 60 per cent for certain industrial purposes and when conveying material; and 3 per cent for mineral washing (Environment Agency 2005a). Another Environment Agency consultation document (Environment Agency 2005b) strongly encourages the use of water audits (which include establishing consumptiveness) to improve the efficiency of use of abstracted water.

Net gain: For a river being augmented by groundwater, the net gain is the amount by which the river flow is increased above what it would naturally have been without the augmentation. Let us suppose that under natural conditions, the flows in a certain river are being supported by baseflow from groundwater. Then, the groundwater is abstracted for dewatering purposes and the water discharged into the same river. At first, the abstraction will draw entirely from groundwater storage, and the river flows will benefit from 100 per cent of the discharge. However, the abstraction will start to intercept natural baseflow to the river, and the net gain in river flows will progressively fall, as a proportion of the discharge merely makes up for the reduction in natural baseflow. If the abstraction is continuous, the net gain may eventually fall to zero, as the system reaches a new equilibrium. In the context of quarry or mine dewatering, this is not necessarily a problem, as it can be argued that a net gain of zero is also a net impact of zero. Further information on net gain and the augmentation of river flows from groundwater can be found in Rippon *et al* (2003).

The suggested procedure under Step 6 is as follows:

i. Confirm whether or not the dewatering operation is consumptive. If the consumptiveness is minimal, then the rest of the procedure will be a formality, unless the timing and locations of the discharges with respect to the flow impacts are a problem.

ii. If the dewatering abstraction is found to be consumptive, and the output from a CAMS assessment is available, then identify the Groundwater Management Unit (GWMU) and Assessment Points (APs) (see Appendix 1) that will be impacted by the abstraction. If the status of both the GWMU and APs is 'Water available', the presumption by the Environment Agency will be to grant the licence subject to local considerations (on which the HIA should therefore concentrate). If the status is 'No water available' or worse, the applicant will need to demonstrate that their abstraction will not cause a deterioration in status.

iii. If CAMS output is not available, then use Tests 2 to 4 from the five CAMS groundwater tests. That is, compare summer baseflow to the predicted flow impact (Test 2), look for trends in groundwater levels or quality (Test 3), and look for other evidence of unacceptable groundwater abstraction impacts (Test 4).

iv. If neither a CAMS status nor a WFD status is available, and the dewatering operation is in unproductive strata, then as mentioned earlier (Table 4.1), the focus of the HIA will be on looking for specific local impacts.

Other comments on assessing the environmental acceptability of flow impacts on various features are as follows:

Rivers: In addition to the magnitude of flow impacts, the timing of the flow impacts is especially important for rivers. Abstractions at a constant rate can be assumed to change river flows at a constant rate. Seasonally variable abstractions may also change flows by the long-term average rate of abstraction, due to the diffuse nature of groundwater flow, but they may have a seasonal impact. In this case, a summer reduction should be assumed as a worst-case scenario, unless a time-of-impact analysis clearly shows that the flow reduction will occur in the winter. In the absence of a suitable numerical groundwater model, IGARF is a useful tool for assessing the impact of groundwater abstractions on rivers. Using basic parameters and an appropriate analytical solution, IGARF can give estimates of the magnitude, timing and distribution of the flow impact upstream and downstream. When using IGARF, it is important to understand the assumptions built into the model, and therefore its limitations. For example, it assumes that all the abstracted water will eventually be derived from the river in question, so the proportion of the total abstracted water allocated to a particular river (under Step 4) should probably be used as the modelled abstraction rate. If necessary, the opinion of a hydroecologist or similar specialist should be sought on whether the magnitude, timing and distribution of the potential flow impacts are environmentally acceptable.

Wetlands: Impacts on wetlands are notoriously difficult to assess, mainly because they are often very complicated hydroecological systems that are not well understood, but also because they can be very sensitive to water level changes as well as flow impacts. Some wetlands have exacting water quality requirements that are dependent on factors such as: receiving water of different qualities from a variety of sources, including upward groundwater flow; a flushing flow through the wetland being maintained; or a water quality gradient across the wetland being maintained. Wetlands should therefore be given special attention, and a conceptual model of the local system is necessary. Flow impacts should then be treated in the same way as for rivers, with specialist advice being sought on the acceptability of the impacts, not forgetting cumulative impacts. Impacts on wetland water levels will be considered later (Step 11).

Estuaries: In the past, the environmental impact of abstractions on estuaries has often been considered not to be significant, except perhaps for some very large abstractions. However, the importance of estuaries is increasingly being recognised, for birds in particular, and many are now designated under the Habitats Regulations. If necessary, technical specialists should be consulted on the importance of fresh water flows into an estuary. Unless further research reveals otherwise, estuaries should be treated in the same way as rivers, with a proportion of the flow impact allocated to the point closest to the abstraction, the impact investigated using a numerical model or IGARF, and a judgement made on the acceptability of the impact.

If there is the potential for impact on SSSIs or Natura 2000 sites, there are specific legal obligations associated with assessments for these sites. If you think you may be impacting such a site, contact your local Environment Agency office at the earliest opportunity for advice. In many cases, it will be difficult to determine precisely whether or not a net flow impact is ecologically significant. In these cases, you should present your conclusions based on the information you have available, together with the supporting information on how you reached that conclusion.

4.2.7 Step 7: Define the search area for drawdown impacts

We turn in this step to identifying and assessing drawdown impacts. First we must define the search area within which we are going to look for water features susceptible to drawdown impacts. In effect, we are defining the radius of influence (usually denoted as R_o) of the abstraction, which is the radius at which drawdown is zero. Various methods are in common use for estimating R_o, such as Sichardt's empirical formula, which was discussed in Section 3.4. However, the HIA methodology takes a different approach, which is to base the definition of R_o on a good conceptual understanding of the dewatering abstraction and the surrounding area, bearing in mind the key principles outlined in Section 2.4.

Take the sketch map from Steps 3 and 4 (Figure 4.1), and identify the flow impact feature that is furthest away (in radial distance) from the dewatering abstraction. An assumption is now made that drawdown is zero at that radius. This may not be strictly true in practice, but it is a pragmatic assumption, which enables calculations to be made in the steps that follow, and should not lead to significant errors. Conceptually, it fits with the principle that the impact of the abstraction (the cone of depression) keeps spreading until it has stopped an equal amount of water leaving the aquifer. This is in effect setting R_o manually, based on the conceptual model. Using the example from Figure 4.1, the furthest flow impact feature is Spring A, so draw a circle, centred on the dewatering abstraction, passing through Spring A (the solid red line on Figure 4.2).

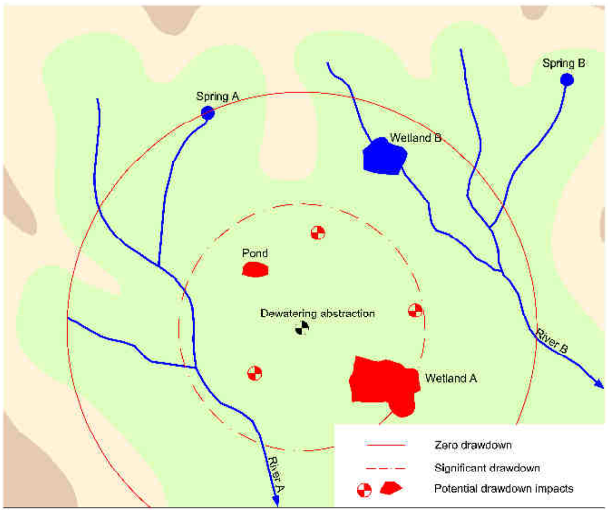


Figure 4.2 Sketch map showing potential drawdown impacts

It may be possible to rule out sections of the resulting circular search area on the following hydrogeological grounds:

- The presence of an impermeable boundary, such that it is not considered worth looking on the other side of it from the dewatering abstraction.
- The presence of impermeable layers between aquifers, such that water features that are clearly hydraulically isolated from the dewatering abstraction can be ruled out.

- The presence of a major 'recharge boundary', for example a river in complete hydraulic connection with the dewatering abstraction, such that it is considered unlikely that there will be drawdown impacts beyond it.

It may also be possible to reduce the radius of the circular search area by defining a 'significant' drawdown. This is discussed further under Step 9.

4.2.8 Step 8: Identify features susceptible to drawdown impacts

Having defined the search area, look for all the features that could potentially be impacted by drawdown, such as abstractions, protected rights, ponds, wetlands and some springs. This should involve a search through readily available information, such as:

- The results of any water features surveys that have been carried out in the past.
- Abstraction licence and protected rights maps held by the Environment Agency. Bear in mind that there may be small abstractions exempt from licensing (which may nevertheless be protected rights), and springs used for stock watering.
- Records of domestic supplies, available from the Environmental Health Officer in the local authority. These records are far from complete however, and another approach is to examine records of public water supply pipes to locate properties that are not connected to the public water supply.
- Records of boreholes and wells held by the British Geological Survey.
- Ordnance Survey maps.
- Databases of conservation sites (see Box 3.1 and Appendix 1).

Ideally, certain key pieces of information should be collected on the boreholes and wells that are the licensed abstractions and protected rights, namely, total depth, depth to pump intake, current typical pumping water level, and which aquifer is exploited. This information is important for the assessment of potential derogation (see Step 11).

Depending on how far away the furthest flow impact feature is, the potential search area could be very large, and a great many potential drawdown impact features might be present within the search area. How thoroughly Step 8 should be undertaken is a matter of judgement. A door-to-door and field-by-field survey is the only sure way of identifying all the relevant water features. This is obviously too onerous at Tier 1, but may be necessary at Tier 3. The results of this step should be added to the sketch map, as shown in Figure 4.2 (the water features coloured red).

4.2.9 Step 9: Predict maximum drawdown impacts

The water features identified in Step 8 should now be examined in turn, to predict the magnitude and timing of the maximum drawdown impact. For predicting the magnitude of the steady-state drawdown, it is suggested that the Thiem or Thiem-Dupuit equation should be used as appropriate. These equations were introduced in Box 3.2, and a full explanation of their derivation and the assumptions on which they are based can be found in Kruseman and de Ridder (1990). For an excavation in an unconfined aquifer, the simplified situation is as shown in Figure 4.3.

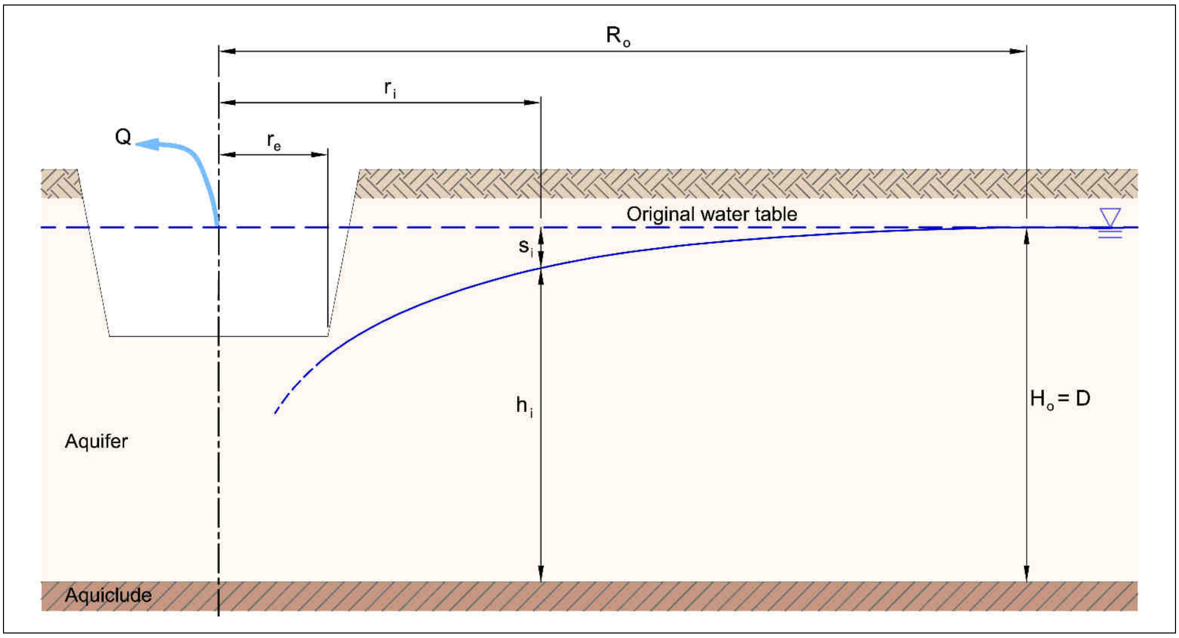


Figure 4.3 Parameters for the Thiem-Dupuit equation

For existing dewatering operations, where the abstraction rate (Q) is known, we have identified R_o in Step 7, where we assume drawdown is zero (H_o = D), so for each water feature, insert the appropriate radius from the abstraction (r_i) and calculate the predicted water table elevation (h_i), and hence the drawdown ($s_i = H_o - h_i$). For convenience, the Thiem-Dupuit equation rearranged to solve for s_i is as follows:

$$s_i = H_0 - \sqrt{H_0^2 - \frac{2.30Q\log(R_0 / r_i)}{\pi K}}$$

For proposed dewatering operations, where Q is not yet known, a slightly different approach is needed. We still specify R_o from the conceptual model. Knowing the target drawdown at the edge of the excavation (at r_e in Figure 4.3), we can calculate Q. Then we use this value of Q to estimate the drawdowns at other radii (r_i).

The Thiem-Dupuit and Thiem equations are usually applied to abstractions from boreholes, as illustrated in the diagrams in Box 3.2. In the case of quarries or other open excavations, especially those with very wide open pits, the question arises of where to measure the radius from, the centre of the excavation, the actual abstraction point (the sump itself, for example), or the edge of the excavation? This could have a significant influence on the drawdown calculations, especially for wide, shallow excavations, such as those in floodplain gravels. Guidance from CIRIA (Preene *et al* 2000) is to take the centre of the excavation as the origin for the equation (for measuring the radii, as shown in Figure 4.3), but professional judgement should be used, with the most appropriate centre-point chosen, especially in irregularly-shaped excavations. Note that in thin gravel aquifers, the assumptions built into the equations may break down anyway.

The Thiem-Dupuit and Thiem equations are of course for steady-state conditions only. As such, they represent the worst-case scenario, because they give the drawdown if abstraction continues until steady-state conditions have been achieved. If necessary, to get a fuller picture, the Theis equation can be used to estimate the time to a certain impact (after commencement of pumping). It may be that the impact will not be significant for many years, beyond the projected life of the quarry or mine, for example. The Theis equation introduces the time factor and aquifer storativity for unsteady-state flow conditions. Various options are available, including the conventional Theis equation for confined conditions, the Cooper-Jacob approximation to the Theis equation, or other versions for unconfined and leaky aquifers. These are all described in Kruseman and de Ridder (1990), and need to be used by an experienced hydrogeologist, as it is very important to take into account the assumptions inherent in each analytical solution.

The uncertainty of the drawdown estimates should be explicitly defined as far as possible. This will depend largely on the level of detail in the conceptual model. Uncertainty in the aquifer parameters can be explored by sensitivity analysis using a realistic range of values for transmissivity (or hydraulic conductivity) and storativity in the relevant equations (Box 2.1).

When looking at certain water features, particularly wetlands, other factors need to be taken into account. Wetlands often have complex internal structures, with low permeability layers of accumulated silt and organic matter. The drawdown predictions made with the Thiem and Theis equations apply to the aquifer beneath the wetland, and will not necessarily be manifested in the wetland water level itself. This will be discussed further under Step 11. Three problem cases are worth explaining:

- ***Seasonal abstractions:*** For seasonal or periodic abstractions, two scenarios should be considered. Firstly, use the annual average abstraction, in m^3/d, for Q in the steady-state Thiem or Thiem-Dupuit equations. Secondly, if necessary, use the Theis or another suitable unsteady-state equation to calculate drawdown at the end of the actual pumping period at the (higher) actual pumping rate.

- ***Non-linear systems:*** Aquifers where the transmissivity varies with time cannot be regarded as linear systems. The classic example is unconfined chalk, where the transmissivity may vary, sometimes dramatically, depending on the seasonal water level. As far as drawdown impacts are concerned, the problem is the speed of the impacts and how this changes seasonally. Again, professional judgement from a hydrogeologist is needed, using the approach of performing two calculations, one ‘slow’ calculation using aquifer properties when water levels are low, and one ‘fast’ calculation for when water levels are high. The ‘slow’ case is probably more important in terms of assessing drawdown impacts, as when water levels are seasonally high, drawdown impacts are probably less likely to be significant.

- ***Karst and fractured crystalline rock:*** Care needs to be taken when dealing with groundwater abstraction from karstic aquifers and fractured crystalline rock. The assumptions inherent in analytical equations such as those of Thiem and Theis usually break down, and it is no longer reasonable to pretend the aquifer is homogeneous and isotropic (see Appendices 3 and 4).

Before moving to the next step, another issue needs to be discussed briefly, related to the shape of the drawdown profile shown in Figure 4.3 (in reality a cross-section through the cone of depression). It can be seen that assuming the drawdown is zero at radius R_o, the radius r_i can be dramatically less than R_o before the drawdown s_i becomes significant, because of the shape of the curve. This introduces the possibility of defining a more realistic radius for the search area, within which the drawdown is likely to be significant, to reduce the number of water features that have to be individually assessed for drawdown impact (the dashed red circle on Figure 4.2). With a little rearrangement, the Thiem and Thiem-Dupuit equations can be used to define that radius, having chosen an arbitrary value for significant drawdown (s_i).

This immediately raises the question of what is a significant drawdown, or what justification is there for any particular value of drawdown? The answer is that it depends on the nature of the water features identified in Step 8. If the water features include sensitive wetlands, for example, then a small drawdown would be significant (perhaps of the order of 0.05 m). Specialist advice should be sought, if necessary, on what is a reasonable figure to use, (see for example Brooks *et al* 2004). Bear in mind that we are only setting a search radius, so err on the side of caution if there is any doubt. Also, there is no reason why different search radii should not be used for features with different sensitivities.

4.2.10 Step 10: Allow for mitigation of drawdown impacts

This step has been included because it is possible that discharges from the dewatering can be used to mitigate drawdown impacts. It is much more common for discharges to be used to mitigate flow impacts (considered under Step 5 above), but it is worth considering drawdowns separately, to maintain the logic of the sequence of steps. Examples of mitigation of drawdown impacts include the following:

- Discharges to wetlands (those that are unconnected to surface water) to maintain water levels.
- Discharges to other surface water bodies that are expressions of the local groundwater level, such as certain lakes and ponds, to maintain water levels.
- Indirect support of such water levels by discharging to a suitably-placed recharge trench or injection wells, to maintain local groundwater levels.
- Direct supply of water to a third-party user, to replace or compensate for a groundwater source that has been derogated, or provision of an alternative water supply such as connection to the mains.

In the same way as for mitigation of flow impacts, the measures for mitigation of drawdown impacts need to be considered carefully, to arrive at a net drawdown impact, while bearing in mind any peculiarities of the local conceptual model. In particular, it is worth remembering that the timing and location of the drawdown impacts from the dewatering abstraction may be different from the timing and location of the benefits from the mitigation. The appropriateness of particular mitigation measures will depend on the type and complexity of the site impacted. If the site is designated, some mitigation measures may not be appropriate, and will need to be agreed with Natural England or CCW.

4.2.11 Step 11: Assess the significance of the drawdown impacts

The significance of the potential drawdown impacts should now be assessed. These can be described in terms of three categories, derogation of existing abstractors, environmental impacts on water bodies and wetlands, and subsidence/desiccation. Taking each of these in turn:

Derogation of existing abstractors: The Environment Agency defines derogation as preventing a person entitled to a protected right from abstracting water to the extent authorised on their licence. This obviously covers cases of pumping water levels being lowered below the current pump intake, but increased pumping costs (which inevitably result from lower pumping water levels, because the pump is working against a greater head) do not qualify as derogation. Key things to consider when assessing derogation are as follows:

- In practice, the pumping water level (PWL) cannot be allowed to fall too close to the level of the pump intake, otherwise there are problems with drawing in air, leading to cavitation damage in the pump. The size of the buffer zone between the deepest PWL and the pump intake varies depending on operational practice and pumping rate. For a large public water supply abstraction, pumping several million litres per day, the preferred buffer zone may be as much as 10 or 15 m. A small domestic supply on the other hand, only pumping several hundred litres per day, may only require 1 m or so. The basic details of typical PWLs, and existing pump intake level are essential for assessment of derogation.

- Derogation impacts should be judged by considering both typical conditions and dry or drought conditions. For dry conditions, considering the water levels that would occur in a drought of 1-in-10-year return period would be reasonable, rather than the worst case conditions. This frequency of drought is classified as a moderate drought under the British Hydrological Society classification system (Mawdsley *et al* 1990). Also, the annual groundwater recharge during the 1-in-10-year drought was recommended for use as part of an indicator of hydrological severity, developed during an R&D project for the National Rivers Authority (NRA 1995). Using a moderate drought assumes that all abstractors will accept some restrictions on their ability to abstract in serious or severe droughts, and so derogation does not apply to these conditions.

- The effects of the uncertainties in drawdown estimates should also be considered before making a decision about derogation (see Box 2.1 for an example). If the drawdown at which derogation will occur is greater than the predicted drawdown but falls within the range of possible drawdowns established by the sensitivity analysis, then the uncertainty may be too great to make a clear decision at this stage. Further monitoring may be required, during a pumping test or during the period of a short-term licence.

- Note that for transitional applications (existing dewatering abstractions, see Section 1.3), abstractions that derogate may still be allowed to have licences that derogate, as the derogation aspects of the legislation are disapplied.

If derogation is predicted or occurs, then there may have to be negotiation between the applicant and the potentially affected abstractors, before a licence can be granted. Options include lowering the pumps, deepening the borehole, providing an alternative supply, or even paying compensation.

Environmental impacts on water bodies and wetlands: The local conceptual model, and information gained during water features surveys and any other investigations are used to estimate the potential environmental impacts of the abstraction on the water levels within any ponds, wetlands, meres, fens and springs. There is currently limited capability for predicting the ecological impacts of water level reductions on wetlands, and more research is urgently needed. The drawdown at water bodies and wetlands can be estimated in a similar fashion to the estimation of the potential for derogation, and the following factors should be considered:

- Some wetlands may be perched on a low-permeability substrate, and may not be in hydraulic continuity with the aquifer beneath. For example, this is thought to be the case for many upland blanket bogs on low-permeability hard-rock terrain.

- The Thiem and Theis equations actually predict the drawdown in the aquifer under the wetland. This is not necessarily the same as water level changes in the wetland itself, because of the equivalent effect to there being lower-permeability river-bed sediments in a river. For most wetlands, there is some resistance to flow between the wetland and the underlying aquifer, due to the build up of sediment and organic material. A wetland leakage factor 'C' is sometimes used (C = K/b, where K is the vertical hydraulic conductivity and b is the thickness, of a semi-confining layer beneath the wetland). Flux of water vertically between the wetland and the groundwater depends on C and on the hydraulic head difference, but C is very rarely known with any certainty. This subject is discussed at length by Williams *et al* (1995).

- The impact may have to be judged by considering typical conditions, moderate drought conditions, and perhaps the predicted or historical worst-case conditions for some particularly sensitive sites. This is difficult, and relies largely on the judgement of ecologists. The impacts depend on the predicted changes in water level, the time of year, the duration, and how often they may occur. If the changes are within the normal level fluctuations experienced at the site, it will be the increased duration of the lower levels that may be significant. If the level may be reduced beyond the range of normal level fluctuations, it will be how often and for how long this occurs that may be significant.
- Many wetland species require variations in water level through the year, and may be adversely affected if water levels are kept too constant artificially.
- For SSSIs and Habitats Directive sites, as mentioned under Step 6, a parallel investigation of the potential impacts may be necessary. Under the relevant legislation, it must be demonstrated that there is no adverse effect on the designated site, which could mean that *any* additional drawdown at the site is regarded as unacceptable.
- Drawdowns may result in changes in water chemistry, particularly if a wetland is transformed from a discharge area to a recharge area, which may also affect the wetland flora and consequently the fauna (Harding 1993). Impacts of drawdowns will depend on whether the wetland is surface- or groundwater-dependent or a combination of both. The timing of an impact can also be important.
- Some impacts on water features may be easier to predict. For example, if the predicted drawdown is sufficient to dry up a spring for substantial periods, then the impact on any species depending on the spring flow should be fairly easy to judge as unacceptable.

The known water level and flood regime requirements of wetland species and communities are summarised by Brooks *et al* (2004) and Whiteman *et al* (2004). These give upper and lower limits of tolerance to either soil water tables or depths of water or a mixture of the two, and in most cases a preferred range of levels (as illustrated in Figure 4.4).

In applying these levels to communities of species, a precautionary approach should be taken based on the most sensitive species in the system and specialist species with narrow tolerance to fluctuations in water level. Some vegetation is very sensitive to the depth of the unsaturated zone, and even changes of less than 0.1 m can have a severe impact on some ecosystems in marshy areas and brooklands (RIVM and RIZA 1991). Acreman (2004) provides further guidance on impact assessment of wetlands. Guidance on the management of groundwater to maintain wetland ecological character can be found on the Ramsar website (http://www.ramsar.org).

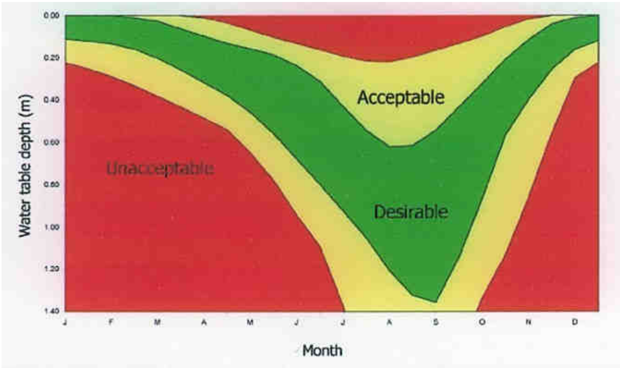


Figure 4.4 Example of water table threshold requirements for particular plant community
(from Brooks *et al* 2004)

In the absence of any better information, Williams *et al* (1995) recommend the use of a '10 per cent rule', namely that the effects of groundwater abstraction (that is, the additional drawdown) should be limited to 10 per cent of the difference between the mean and minimum summer water levels in the aquifer underlying the wetland. This rule-of-thumb is based on the premise that for wetlands to survive, they must be able to cope with the natural fluctuations in the water table, and the rule compares the additional change in head in the vicinity of the wetland with the relative magnitude of the natural variations. This rule is based on natural (pre-impacted) wetland conditions, and cannot always be applied to current water level variations, as the wetland may already be badly affected by abstraction.

Subsidence and desiccation: Subsidence and desiccation are closely related, and often occur together. In the context of groundwater abstraction, they can be defined as follows (Kirk *et al* 2000):

> ***Subsidence****: the settlement of the land surface or buildings as a direct consequence of a decline in groundwater levels produced by groundwater abstraction.*
>
> ***Desiccation:*** *the drying out of the soil or exposed strata beyond natural levels of variability as a consequence of groundwater abstraction.*

In saturated aquifers, the water in the pores between the grains of material helps to support the weight of the rocks and soils above. If the porewater pressure is reduced, by lowering the piezometric level in a confined aquifer for example, then a greater proportion of the weight is borne by the grains of material, as opposed to the porewater. This increases the effective stress in the formation, and under certain circumstances this can lead to significant compaction, which is manifested at the surface as subsidence. Subsidence induced by groundwater abstraction tends to occur in thick sequences of unconsolidated or poorly consolidated sediments, especially when a large percentage of the sequence consists of high-compressibility clay (Freeze and Cherry 1979). Internationally, there have been notable examples of such subsidence in the San Joaquin valley of California, around Shanghai, in southern Taiwan, and around Mexico City (Price 1996), with the ground surface subsiding by several metres. In the UK, very few cases have been reported that can be directly attributed to groundwater abstraction. The best-known example is central London, where large abstractions over a long period of time caused subsidence of about 0.2 m by the 1930s (Kirk *et al* 2000). Most subsidence is caused by other factors, such as dissolution and collapse of soluble strata (as is the case around Ripon, North Yorkshire), or the collapse of old mine workings.

Desiccation usually occurs in unconfined aquifers, or in certain circumstances such as confining clay beds that dry out when the groundwater level falls below the base of the confining layer. Common problems associated with desiccation are as follows:

- *Change in plant communities:* Different species of plant have different abilities to take up water from the soil zone, both in terms of the depth to which their roots extend, and the ability of the roots to absorb water against the capillary forces in the soil. Over time, a plant community develops that is in equilibrium with the water availability regime. If the water table is lowered by groundwater abstraction, then the availability of water in the unsaturated zone may change as well, and may affect the plant communities.

- *Cracking of clay soils:* Soils with high clay content tend to crack when they dry out, resulting in the classic image of polygons separated by deep cracks. This makes working the soil difficult, and can lead to a breakdown in the soil structure.

- *Shrinkage of soils with high organic matter:* Soils with high organic matter content, such as peat, tend to shrink when they dry out, often causing the ground surface to lower significantly (Figure 4.5). This can lead to structural damage to property if there is differential settlement caused by uneven drying out of near-surface strata (the same applies to clay soils).

- *Damage to archaeological remains:* Many buried archaeological remains depend for their preservation on being kept submerged in water, that is, in being below the water table. This is usually because of the protection from oxidation, but can also be due to the chemical properties of the groundwater. Lowering of the water table by groundwater exploitation can expose the archaeological remains to air, leading to decomposition and degradation. English Heritage has published a strategy for the conservation and management of monuments at risk in England's wetlands (English Heritage, undated).

Figure 4.5 Peat shrinkage causing ground surface to lower, exposing tree roots

Again, very few cases of desiccation have been reported that can be directly attributed to groundwater abstraction. Most cases of desiccation are caused by other factors, primarily land drainage, but also changes in land use or climate. Predicting when subsidence and/or desiccation are likely to occur as a result of groundwater abstraction, and calculating the magnitude of their effects, are very difficult. Formulas are available from the world of geotechnical engineering (the Terzaghi and Koppejan formulas, for example), but detailed knowledge of soil/rock parameters such as compaction coefficients is required. Useful information on estimating subsidence can be found in Domenico and Schwartz (1998) and in Preene *et al* (2000). Work in the Netherlands has shown that ground surface levels can react to seasonal changes in groundwater level, sometimes by as much as 15 cm (TNO 2004). Models are available, for example as a subsidence module for MODFLOW, but they really need to be used and interpreted by an experienced geotechnical engineer.

Finally, the reversibility (or otherwise) of the potential drawdown impact should be taken into account. Whereas it may be relatively easy to reverse the effects of derogation, the damage caused to a wetland or an archaeological site by a significant or prolonged drop in water level may be irreversible. This will affect the degree of confidence that it is necessary to achieve.

4.2.12 Step 12: Assess the water quality impacts

Groundwater quality is given great prominence in the WFD, with the achievement of 'good' status being just as dependent on quality as on quantity, and it is therefore treated separately here. However, methods for determining regional water quality impacts are not nearly as well-developed as for quantitative impacts. For example, in the RAM Framework, the approach is just to capture and record comments on existing water quality problems where they affect the ecology, so that they can inform the next step in the overall CAMS process, which is the sustainability appraisal. In addition, the uncertainty associated with impacts on water quality is inherently greater than with quantity, as the impacts are much harder to identify, measure, and prove. The Environment Agency is developing regional groundwater quality monitoring strategies, and will soon be in a position to define the background groundwater quality of all the principal aquifers at least, if not for all secondary aquifers. This is an essential requirement of the WFD, and data on all groundwater bodies are being collected so that the water quality status of each body can be defined. In fact, part of the WFD procedure includes identifying specific water quality pressures on the aquifer. In the context of quarry or mine dewatering, the main threats to water quality include the following:

- Discharge of poor quality water into receiving water bodies, with the main problem being sediment-laden water being discharged into surface watercourses.
- Poor-quality run-off from unvegetated surfaces within the excavation.
- Pollution from accidental spills of fuels, oils, solvents etc within the excavation, often exacerbated by the fact that groundwater vulnerability is increased due to the soil and unsaturated zones having been largely removed.

These threats however, are not really related to the dewatering abstraction itself, and are managed by other means, such as discharge consents, minerals planning guidance, and the Integrated Pollution Prevention and Control (IPPC) regime. Potential water quality impacts of the dewatering abstraction itself (and therefore covered by the transfer licence) are largely related to changes in the groundwater flow pattern in the aquifer. In some cases a numerical groundwater model will be available which is also suitable for modelling contaminant transport. This should be used to examine the impact of the proposed abstraction on the water quality. However, in the vast majority of cases there will not be this luxury. For these cases, the approach should be to ask the question, How is the flow pattern in the aquifer likely to be altered by the proposed abstraction? A basic picture of the flow patterns in the aquifer should be available from the conceptual model, and the usual combination of tools, professional judgement and expert opinion from technical specialists should be used to assess how these flow patterns will be altered. A judgement can then be made on whether or not the water quality impacts are likely to be significant. Issues to watch out for include the following:

- Pollutant plumes from point sources (such as old landfills) accelerating or changing direction. Briefly, plumes in groundwater move by advection, that is with the water flow, and by dispersion. Dispersion is largely independent of the flow velocity so that a new abstraction, unless very large, is unlikely to change the rate of dispersion significantly. Advection takes place with the flow of water, so that increased rates of flow will increase the plume movement in a similar way. Dewatering can therefore draw pollution into previously unpolluted parts of the aquifer, as reported by Morgan-Jones *et al* (1984), who also noted that pulling in poor quality groundwater in this way can have knock-on effects of having to dispose of poor quality discharge water.

- Dilution of poor quality surface water being adversely affected. It is well-established that groundwater contributes baseflow to rivers, and supports other surface water features. The effect on surface water quality of changes in groundwater quality must not therefore be ignored. This could be a direct effect, such as polluted groundwater entering surface water and causing its quality to deteriorate. It could be an indirect effect, such as a reduction in clean groundwater baseflow on which the river depended to dilute an effluent discharge. It could also be a more subtle effect, such as altering the water quality gradient across a wetland.

- Increased risk of saline intrusion. For saline intrusion there is a divide between fresh water and the saline water with a brackish mixing zone in between. Abstractions can disturb the equilibrium between the fresh and saline water, and cause the boundary to move. It is not necessary to reverse the groundwater gradient, but just to disturb the equilibrium. There are simple methods of analysis to model this process (see Todd 1980, for example).

4.2.13 Step 13: Redesign the mitigation measures

Assuming that the uncertainty has been reduced to an acceptable level, it may be that the environmental impacts of the dewatering abstraction are still unacceptable. However, with dewatering, as opposed to highly-consumptive abstractions such as those for public water supply and irrigation, there is the opportunity to influence how and where the water is discharged back into the environment to maximise the mitigatory effects. Designing or redesigning effective mitigation measures may be the only way for an abstraction licence to be approved in a stressed catchment, so this step could be very important.

From the work done so far, it should be known whether the timing and location of the impacts from the dewatering abstraction are different from the timing and location of the benefits from the discharge. If they *are* different, then it should be possible to improve the situation so that the abstraction can still be permitted. Practical measures that could be taken include the following:

- Shifting the point(s) at which water is discharged into surface watercourses.

- Splitting the discharge into several separate places, into different watercourses or into several points down the length of the same watercourse.

- Controlling the discharge rate and timing, usually by means of intermediate storage facilities, to restore some flow variability in the receiving watercourse, or to augment the flows at the most critical times.

- Careful placing of recharge trenches in relation to the impacted feature (and usually further away from the dewatering abstraction) to maximise benefit and minimise recycling of water.

- Installing low-permeability barriers to reduce the dewatering quantity, and to maintain groundwater levels on the side of the barrier away from the abstraction.

These techniques are not new, and are already in widespread use; the point is that the conceptual model can be used to optimise the mitigation. Wardrop *et al* (2001) provide several examples of designing simple low-cost mitigation measures based on a good conceptual model. A longer discussion of mitigation measures can be found in Section 5.5.

4.2.14 Step 14: Develop a monitoring and reporting plan

The final step in the HIA involves developing a plan for monitoring and reporting. The subject of monitoring in general, and also monitoring plans specifically, are covered in Section 5. Suffice it to say the following here:

- Monitoring associated with HIA is deliberately not prescriptive, because like the HIA itself, it should be based on risk, and should be appropriate to each case.

- The monitoring should be focussed on the water features that have been identified during the HIA as being susceptible to flow and drawdown impacts (see Section 5). This is a somewhat similar approach to the source-pathway-receptor concept often used in water quality, contaminated land and landfill studies. The abstraction is equivalent to the source, the sensitive water feature is the receptor, and the pathway is the aquifer in between.

- Monitoring the long-term groundwater level changes in a borehole that is not too close to be directly affected by individual abstractions provides a very useful indication of the 'health' of the water balance in the aquifer. In other words, if the outflows are consistently greater than the inflows, water will be taken from storage, and this manifests itself as an overall decline in groundwater levels.

4.3 After each tier

4.3.1 Consultation

Assuming that you have reached the end of a Tier 1 investigation (which may have involved several iterations), it is recommended that you consult the Environment Agency before proceeding any further, for the following reasons:

- If any of the water features that have been identified as being impacted (or potentially impacted) is a SSSI, SAC, SPA, or Ramsar site, then the Environment Agency will need to trigger the relevant consultation process with Natural England or CCW.
- It may be that agreement can be reached with the Environment Agency after a Tier 1 investigation, without having to proceed any further (even if it was initially thought that a Tier 2 or 3 investigation might be necessary). This is likely to be because there is no significant impact, or because the impact is obvious and can be easily mitigated, or because the impact cannot be mitigated and is unacceptable.
- It may also be that the impacts and associated mitigation measures are already covered by Section 106 agreements under the planning system.
- It is the Environment Agency's role to assess the cumulative impacts of all abstractions in a certain groundwater management unit or groundwater body, and should it be necessary to continue with the HIA process beyond Tier 1, the Environment Agency may be able to supply missing data, or advise you on aspects that need a closer look.
- The HIA methodology is intended to be flexible and based on risk, so following on from the previous point, it may be that the Environment Agency just asks for a certain aspect to be looked at more closely, without having to go through a full Tier 2 investigation.

The same comments apply at the end of Tier 2, perhaps even more so, as the costs associated with a Tier 3 numerical groundwater model can be significant.

4.3.2 Reducing the uncertainty

At the end of each tier, and indeed each iteration within a tier, it is necessary to judge whether there is sufficient confidence in the conceptual model developed so far to enable a decision to be made. In other words, has the uncertainty been reduced to an acceptable level? The decision itself concerns whether or not the impacts of the abstraction are acceptable, and making this judgement is not straightforward. The Environment Agency has to balance the impacts caused by the abstraction against the benefits of the abstraction to the applicant. Impacts will undoubtedly occur and some or all may be deemed acceptable. The difficulty is deciding when the impacts become unacceptable either individually or in combination. Impacts that may be unacceptable include:

- Derogation that is not agreeable to existing abstractors.
- Any flow reduction that might lead to failure to achieve statutory flow obligations.
- Any significant reduction in water level leading to environmental damage at an environmentally sensitive wetland site, such as a SAC or SPA.

It may be helpful to identify those impacts that are clearly acceptable and to rank the remainder with the worst impact, or expected to be worst, at the top. If the uncertainty is still considered to be too great, the procedure should be as follows:

- Bearing in mind the discussion on uncertainty and risk earlier, try to determine where the areas of greatest uncertainty lie, and where efforts to reduce uncertainty would best be focussed. Is the uncertainty in the conceptual model, in the data and the way in which it was sampled, in scientific knowledge, or in the inherent variability of the environment? Or in more than one of these? Another useful approach is to establish what would need to go wrong – what would have to occur – for the impacts to be unacceptable. This sometimes helps to clarify whether or not the uncertainty is indeed too great.

- Reduce the uncertainty, by improving the conceptual model. Assuming that the starting point is the first 'dry run' through the impact appraisal, this will normally be achieved by further data collection, and possibly field investigations such as pumping tests. The cycle of conceptual modelling, from basic to best basic, from intermediate to best intermediate, etc, should be continued until the uncertainty has been reduced to an acceptable level.

4.3.3 Recording your findings

Recording your findings under each step of the HIA is highly recommended, as this acts as an audit trail. The written record needs to be detailed enough to enable someone else to understand how the conclusions were reached. This means recording the conceptual model, its assumptions, and how it was tested, verified, developed and used to make decisions on the potential impacts of the abstraction. It is useful to record models, mechanisms or hypotheses that were considered but then rejected, as it is important to know that they were at least considered. Uncertainties should be explicitly identified and documented. It is also important to record the sources of the data used in the appraisal. It is helpful if the record of the conceptual model includes sketches, maps and cross-sections. One of the reasons audit trails are important is that if an assessment is made in good faith based on the best evidence available at the time, then it is defensible, even if new information subsequently shows it to be wrong.

5 Monitoring and mitigation

5.1 Purpose of monitoring

The subject of monitoring has been mentioned briefly several times already in this report, but it is so important to HIA that it deserves a section of its own. This section is, however, by no means an exhaustive treatment of the subject, but serves to highlight the main issues of relevance to HIA. Although the focus of this section is on monitoring water levels and flows, monitoring can also include water quality sampling and ecological surveys. In the context of dewatering, the main purposes of monitoring are as follows:

- To establish the baseline environmental conditions before the commencement of dewatering (for new excavations, or for extensions to, or deepening of, existing excavations).
- To fill in gaps in the knowledge of the hydrogeology and hydrology of a dewatering operation and its surrounding area. In other words, to improve the conceptual model by reducing the uncertainty.
- To provide the data necessary to undertake a water balance for the quarry or mine, for example by measuring abstraction and discharge quantities, and water usage within the operation.
- To demonstrate compliance with conditions attached to relevant abstraction licences or discharge consents.
- To trigger mitigation measures or temporary cessation of dewatering, if the water level in a receptor (such as a wetland) falls below an agreed threshold, for example.
- To provide early warning of adverse impacts on receptors such as sensitive water-dependent ecosystems, or other abstractions.
- To accumulate data during the lifetime of a time-limited licence that can be used when the time comes to review the licence.
- To provide information for the day-to-day operation of the dewatering system (and where appropriate, the mitigation system), thus enabling the dewatering to be optimised.
- To enable the post-closure period (on permanent cessation of dewatering) to be managed successfully.

The penultimate point, on optimising the dewatering (and mitigation) system, is sometimes forgotten when concentrating on environmental issues and compliance with licences and consents. Operating costs can be reduced significantly by adjusting the dewatering system in response to good quality monitoring data.

5.2 Principles of monitoring

In some cases, monitoring data are collected haphazardly, without any clear idea of why certain types of data are being collected. Again in the context of dewatering, the main principles of monitoring are as follows:

- The overall objectives of the monitoring system should be clearly defined. There will probably be more than one objective, and these are likely to be related to the main purposes described above. For example, the main objective of a monitoring system might be: to quantify accurately the dewatering abstraction, the water used for other purposes within the quarry or mine (dust suppression, mineral washing, etc), and the amount discharged to the environment, to enable an accurate water balance to be drawn up, and consumptiveness to be established. It is a good discipline to write these objectives down, if only to focus the mind on what the monitoring system is trying to achieve.

- The design of the monitoring system should be based on a good conceptual model. Having said that, it may of course be that there is significant uncertainty about the conceptual model, which the monitoring is trying to address. The design should therefore be based on the best current understanding of the conceptual model, with the design reviewed and the system adapted as understanding increases.

- The design of the monitoring system should be risk-based, that is, the burden of monitoring effort should be reasonable and appropriate for the environmental risks associated with the dewatering in question. One way to achieve this is to focus the monitoring on the water-dependent features that have been identified during the HIA as being potentially impacted by the dewatering. Again, it is useful to write down an explicit description of why the monitoring system is designed in a certain way.

- There should be a clear idea of what the function of each individual monitoring point is, so that all parties are clear why a specific type of data is being collected, and what are the specific issues of concern. This can easily be forgotten over time if not written down, especially as personnel often change within the lifetime of a monitoring system.

- The construction of each monitoring point should be appropriate for its function. For example, for monitoring water levels in a wetland, it would be no good using a deep borehole that is not hydraulically connected to the wetland itself.

- The same thought should go into deciding at what frequency the data should be collected, and the system reviewed periodically. It may be that money can be saved by reducing monitoring frequency once an understanding has been gained of the variability of the parameter being measured. In other words, there may be little point in collecting daily readings from a borehole in a Triassic sandstone aquifer (where water levels tend to vary slowly, due to the high storage). Conversely, once-daily measurements of flow in a ‘flashy’ stream might be far too few, as flow peaks due to quick run-off from storm events might be missed completely. In karst aquifers, the monitoring frequency should be high because of the rapid response.

- Stringent quality control procedures are necessary to ensure that monitoring data are of satisfactory quality. Such procedures should include the routine calibration of measuring instrumentation, the routine manual checking of automated instrumentation, the routine screening of data for both instrumental malfunction and operator errors, and the documentation of all of the above procedures. It is also essential to safeguard the continuity of monitoring, and to retain and archive all monitoring data (including the quality control procedures themselves) systematically.

The value of writing down the objectives and the thinking behind the design of the monitoring system has already been emphasised. It is good practice to include these in a written Monitoring Plan, which should cover the following subjects (as a minimum):

i. The overall objectives of the monitoring system.

ii. The reasoning behind the design of the monitoring system, relating the design to specific water-dependent features that are at risk.

iii. The function of each individual monitoring point.

iv. Construction details of each monitoring point, including drawing and photographs.

v. Health and safety risk assessment for each monitoring point.

vi. Justification for the frequency of data collection at each monitoring point.

vii. Records of, and justifications for, changes made to the monitoring system in response to periodic reviews.

Not the least of the reasons for preparing a written Monitoring Plan is the fact that agreement will need to be reached with the Environment Agency on the monitoring system.

5.3 Practical design considerations

The main physical parameters being directly measured by the monitoring system are likely to be groundwater levels, surface water flows and levels, water quality, abstraction quantities, discharge quantities, and water usage within the quarry or mine. Parameters such as regional rainfall and evapotranspiration are more likely to be obtained from sources such as the Environment Agency and the Met Office. For more detail on the design and installation of monitoring facilities, see, for example, Brassington (1998), and Environment Agency (2003b). However, it is worth making some points here about practical design considerations:

- Measurements of groundwater levels from at least three different points, ideally arranged in a triangle, are necessary to determine groundwater gradient and therefore the likely flow direction. However, flow in fissures and conduits is not always in the same direction as the regional groundwater gradient. Also, in areas of pronounced topography, especially in low-permeability terrain, groundwater flow and distribution of heads may be strongly three dimensional, and nested piezometers or multi-level observation boreholes may be required to characterise such a head distribution adequately.

- Measurement techniques do not have to be sophisticated. For example, dewatering abstraction may be estimated by knowing the pump capacity and multiplying by pumping hours, or by timing with a stopwatch the filling of a container of known volume. Practical advice on undertaking field work, installing monitoring equipment and collecting monitoring data can be found in Brassington (1998), Environment Agency (2003b), and Younger *et al* (2002).

- Many types of monitoring equipment require regular calibration, including pressure transducers, flow meters and flow gauges, etc. Ignoring the need for such recalibration may risk the integrity of the collected data, and their acceptability to the Environment Agency.

- Each monitoring point should be labelled clearly with a unique identifier (for example, a number painted on the borehole cover), so that there is no chance of confusion, especially if the monitoring is undertaken by several people. The unique identifiers should be part of a clear, unambiguous numbering scheme for all the monitoring points, marked on a master plan.

- All monitoring points for water levels should be surveyed so that the water levels can be related to each other. Water levels are usually reported as metres above Ordnance Datum (maOD), having been converted from metres below a fixed mark (such as the lip of the borehole casing) at each monitoring point. It is important that this fixed mark at each monitoring point is clear, so that there is consistency between readings, especially if the monitoring is being carried out by different people. Ideally, the Monitoring Plan should include a picture or diagram of each monitoring point, clearly indicating the mark from which water levels should be measured.

- The locations of monitoring points may not necessarily be ideal, as there may be constraints associated with land ownership or access. If this is significantly weakening the value of the monitoring system, then it should be discussed with the Environment Agency at an early stage.

- Permission is likely to be required for the installation of monitoring points such as boreholes and flow gauges, even if they are on land owned by the quarry or mine operator. Permission may be required from other bodies in addition to the Environment Agency, such as Internal Drainage Boards, Natural England or CCW (for any work affecting a SSSI, for example), or the Coal Authority (to drill a borehole through a coal seam or working).

- Consideration may need to be given to measures to prevent vandalism of, or interference with, the monitoring points.

- One specific problem associated with large open excavations is that some monitoring sites may be lost as the void expands and deepens. It is therefore necessary to ensure that critical monitoring sites will not be lost during expansion, or at least that this is anticipated and planned for in the design of the monitoring system.

- Health and safety must of course be taken into account, so that the monitoring points can be visited safely and the data collected safely. Common hazards associated with monitoring include lone-working, working in or adjacent to: deep or flowing water; open excavations with unstable sides; haul roads used by heavy machinery; unprotected large-diameter wells; and confined spaces (below-ground chambers or basements are often classified as confined spaces, and access may be required to dip a borehole). The relevant searches for buried services and pipelines etc must be conducted before drilling boreholes.

5.4 Interpretation of monitoring data

Monitoring data are not collected for their own sake, and are almost worthless unless they are interpreted and reported. It is assumed that the reader is familiar with the basic principles of recording and presenting data, in an MS Excel spreadsheet for example, and with simple statistical analyses such as averages, maxima, minima and trends. In the context of HIA, there are several things to watch out for when interpreting monitoring data:

Firstly, great care needs to be taken when using existing boreholes or wells as monitoring points for groundwater levels. Unless you have a good idea what the borehole/well construction is, the water level data may be difficult to interpret. This is a particular problem in layered aquifers separated by aquitards (or a wetland separated from the aquifer by a low-permeability layer), especially if in reality there is a different hydraulic head in each layer. The monitoring point may be a deep borehole with grouted casing through the shallow aquifer layers, with the water level actually representing the piezometric head in a deep aquifer layer (as opposed to the shallow water table). Or it may be a borehole left open through several different aquifer layers, with the water level representing an amalgam of the hydraulic heads in the different layers. In terrain with significant topographic gradients, especially in low-permeability rocks, groundwater heads can vary by several metres over tens of metres' depth. To determine water-table level in such situations, the borehole response zone must straddle or be immediately below the zone of water table fluctuation, and *not* at some depth below it.

Secondly, spot measurements of groundwater levels taken from monitoring wells or boreholes can be affected by many different factors, which need to be considered when interpreting the water level reading. These can include:

- *Barometric pressure:* water levels in boreholes penetrating confined aquifers can be significantly affected by high or low atmospheric pressure, the effect being most marked in rigid, consolidated rock (Price 1996).
- *Tides:* water levels in boreholes penetrating confined coastal aquifers can respond to the tidal cycle. In fact, the nature of the response can be used to estimate aquifer hydraulic characteristics. Similarly, water levels in boreholes can respond to regulation of river levels or locking events on navigable rivers.
- *Abstraction:* some monitoring takes place in boreholes that are themselves used for abstraction. It is obviously essential to know whether or not the pumps are operating when the water level reading is taken, or whether the water levels are still recovering from a recent pumping period.
- *Other abstractions:* similarly, if water levels in the monitoring borehole are likely to be affected by abstraction from nearby boreholes, it is essential to know the pumping patterns in those boreholes.
- *Natural seasonal changes:* ideally, a reasonably long data record from a monitoring borehole, prior to the commencement of dewatering, will be available, to reveal the natural seasonal behaviour of the groundwater levels. This is especially important when dealing with periods of drought. Wardrop *et al* (2001) present a case study where suspected impacts of dewatering were, on closer inspection, identified as the effects of a period of extremely low precipitation and high evaporation.

Thirdly, monitored groundwater levels may be in a perched aquifer, with completely different water levels or piezometric heads in deeper aquifer layers. It is important to know which aquifer layers the dewatering abstraction is likely to affect, and this should become apparent during conceptual model development. On a similar point, water levels in flooded excavations can be affected by sealing or 'blinding' of the pit floor by low-permeability silt, which can have the effect of perching the water level above what it would otherwise be if the flooded pit were in complete hydraulic continuity with the main water table. Having said that, Gandy *et al* (2004) point out that documented instances of pit blinding are very rare (possibly because of a lack of research on the phenomenon).

Fourthly, it must be recognised that most monitoring data represent measurements taken at particular points, which are then used to draw conclusions about aquifer properties or behaviour over a wide area. This is a reasonable approach in homogeneous aquifers with inter-granular groundwater flow, but can be fraught with difficulties in heterogeneous aquifers, with flow occurring in fissures, fractures, or conduits. Karst conditions are the most extreme example of this, and it is perfectly possible for monitoring boreholes to completely miss important impacts caused by dewatering, if a certain conduit does not happen to be penetrated by the borehole, for example (see Appendix 3).

5.5 Mitigation measures

With dewatering abstraction, as opposed to consumptive groundwater abstraction, there are options of reducing groundwater flow to the abstraction point, or returning the abstracted water to the environment. These options come under the general title of mitigation measures. There are many possible mitigation measures that can be used to minimise or eliminate the hydrogeological impacts of dewatering. These include (Huxley *et al* 2004):

- Limiting the depth and/or surface extent of the excavation or dewatering system so that the zone of dewatering influence does not affect sensitive features.
- Dewatering in small cells, one at a time, to reduce the pumping rates required and the surface area that has to be dewatered.
- Using 'closed-circuit' dewatering systems that involve recharging the abstracted water to ground within or close to the site, rather than pumping off-site.
- In the case of floodplain sites, recharging abstracted water back into the aquifer to reduce the impact on surface water flows.
- Recharging the abstracted water directly into surface watercourses, lakes or other water features that could potentially be affected.
- Installation of a low-permeability cut-off barrier around all or part of the site, or between the excavation and sensitive features.
- Continuous monitoring of water levels in nearby abstraction wells, or flows in watercourses, so that additional mitigation measures can be provided as required.

Such mitigation measures are practical and effective, and some are already common within the quarrying industry. This point was strongly emphasised by Wardrop *et al* (2001), who presented a series of case histories covering all stages of mineral operation from ground investigation to after-use. They also made the point that while mitigation measures should ideally be put in place in advance, to prevent predicted impacts, prediction is an imperfect science, and in practice some mitigation will inevitably be reactive. Having said that, when a protected site such as a SAC is potentially under threat, waiting for an impact before reacting with mitigation would probably not be acceptable, and it may be necessary to demonstrate that effective mitigation measures are already in place before an impact is observed.

It is important to understand the hydrogeological and hydrological implications of existing or planned mitigation measures, and their effects on water levels and flows. The design of the mitigation measures should be based on the conceptual model of the dewatering operation and surrounding area, and the following factors also need to be borne in mind:

- The mitigation measure that is ideal from a hydrogeological or hydrological point of view may not always be possible in practice, with land ownership and access being the main constraints. For example, the ideal mitigation measure may be to construct a recharge trench immediately adjacent to a sensitive wetland, but the land owned or leased by the operator may be too far away from the wetland to make a recharge trench effective.

- The effectiveness of recharge trenches is often limited by infiltration capacity, and it is difficult to match the infiltration rate with the pumping rate. In other words, unless the infiltration capacity is sufficient to cope with the rate at which water is being discharged into the trenches, then temporary storage and/or overflow facilities will also be required. Infiltration capacity also tends to decrease with time, as the infiltration surfaces of the recharge trenches become clogged with fine sediment. Recharge trenches usually work best in shallow high-permeability aquifers, with shallow water tables, when positioned close to the impacted feature, and constructed in combination with cut-off barriers to minimise recycling (Huxley *et al* 2004).

- For mitigation measures involving discharging to surface water, it is important to take into account the natural flow variations that would normally be expected in the receiving watercourse. Hydroecologists are increasingly recognising the role played by periods of high and low flows, and natural flow recessions, in the life-cycles of water-dependent flora and fauna. Discharging large quantities at a fixed rate into a watercourse can mask the natural flow variation, and adversely affect the hydroecology. Solving this problem usually involves temporary storage of the abstracted groundwater, so that it can be discharged at varying rates and times. Differences in water quality between the water being discharged and the receiving water also need to be taken into account.

- One of the potential impacts of dewatering is the derogation of existing nearby abstractions through lowering of groundwater levels. The simplest mitigation measure, which has been widely adopted to counter the effect of such derogation, is the replacement of existing groundwater supplies with mains water, provided at the dewatering operator's expense. Although suitable for small individual domestic abstractions, such schemes may not be able to deal with larger abstractions. In these cases, direct replacement of lost resource from pumped dewatering abstraction can be a satisfactory alternative. However, maintenance of appropriate water quality is required if this approach is adopted. It may be necessary to separate poor quality surface run-off from good quality groundwater inflow, and to monitor water quality more carefully.

- Some mitigation measures in themselves may have adverse impacts. For example, for the derogation of another groundwater abstraction, an alternative mitigation measure is to drill a replacement borehole for the affected party. However, unless designed carefully, that borehole might itself affect a third party.

As mentioned in Section 1.1, Thompson *et al* (1998) identify four distinct aspects of surface mineral extraction by which the water environment can be affected: by the initial ground investigation works; by the physical presence of the excavation; by the dewatering of workings that operate below the water table; and by contamination of groundwater and/or surface water. This report, and the mitigations measures described above, concentrate on the third aspect, dewatering. The other aspects are in general controlled by different legislation, and each aspect has its own risks and its own recommended mitigation measures. Further information on mitigation measures in general can be found in Thompson *et al* (1998), and on recharge features in particular in Huxley *et al* (2004).

5.6 Links between monitoring and mitigation

Assuming that monitoring data have been collected, and correctly interpreted, they are still of only academic interest unless there is a mechanism in place for something to happen in response, if necessary. In other words, there should be a direct link between monitoring and mitigation, which is usually achieved by the use of agreed trigger levels. For example, Huxley *et al* (2004) present a case study at Condover quarry in Shropshire, where extraction of sand and gravel is taking place in close proximity to Bomere Pool, a Ramsar wetland site. Two trigger levels (based on water levels in Bomere Pool) were agreed between the Mineral Planning Authority (MPA) and the quarry operator. If the pool water level falls below the first trigger level, monitoring frequency is doubled, the cause of the drop in water level investigated, and a report submitted to the MPA. If the pool water level continues to fall, and reaches the second trigger level, dewatering ceases and monitoring frequency doubles again, until it has been established whether or not the dewatering is the cause. If dewatering *is* found to be the cause, then an active remediation scheme would be proposed. This system (of two trigger levels) is analogous to the system recommended for landfill sites, where the first trigger level would be described as a 'control level' (Leeson *et al* 2003). Note that in landfill legislation, the term 'trigger level' usually has a specific meaning, namely, a compliance level or regulatory standard. In this report, the term is used in a more general sense.

Choosing appropriate trigger (or control) levels, and agreeing what should happen when those levels are reached, are by no means straightforward, even with the cooperation of all parties. The following factors need to be borne in mind:

- Trigger levels usually err on the side of caution, but it is important that they are based on risk, and that the first trigger level reached should not necessarily result in immediate operating restrictions. In the example from Condover, just described, the first trigger level actually triggers further investigation, rather than the blame being immediately laid at the door of the dewatering abstraction. This is good practice, because the observed impacts may turn out not to be due to the dewatering abstraction. However, there will of course be cases where it is immediately clear that the dewatering is to blame, and it may be important to implement mitigation measures as soon as possible, to minimise the damage to a receptor.

- The feasibility and effectiveness of the proposed mitigation measures should be established and agreed as early as possible, so that they can be implemented quickly and efficiently if and when required. If there is a significant delay between a trigger level being reached and the mitigation measures being implemented, this could either result in unacceptable damage to a receptor, or the enforced cessation of dewatering for an extended period (leading to loss of production etc), especially if the mitigation measure originally proposed is found not to be effective in practice.

- The trigger levels need not necessarily be the same throughout the year. It has already been pointed out that the water requirements of many wetland species vary through the year (Section 4.2, Step 11), and there is no reason why the relevant trigger levels should not also vary. A diagram similar to that shown in Figure 4.4 could be developed for the trigger levels, which would be operated in the same way as the control curves used for surface water impoundment reservoirs. As an alternative method of depicting natural variations in water level, and acceptable deviations from the natural levels, Mawdsley *et al* (2002) suggest the use of water level duration curves, showing the percentage of time that a certain water level is exceeded (Figure 5.1). This concept should be familiar to all hydrologists, by analogy with flow duration curves.

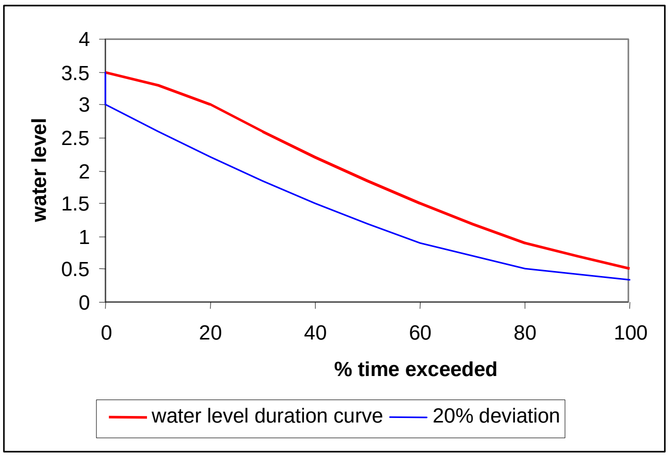


Figure 5.1 Example of water level duration curve, showing possible deviation
(from Mawdsley et al 2002)

- The ecosystem of a sensitive water feature, which may potentially be impacted by dewatering, may not be well understood. In other words, it may be difficult to define the water level and flow requirements of protected species of flora and fauna, and to judge their ability to recover from water-related stress. It is also recognised that in areas that have been subject to development (anthropogenic influences) for decades, if not centuries, it is not necessarily obvious any more what the 'natural' conditions are. In those cases, the focus is usually on preventing deterioration, and maintaining favourable conditions for the protected species. If in doubt, consult the Environment Agency.

- The statistical behaviour of historical monitoring data needs to be understood, so that realistic decisions can be made when setting trigger levels. For example, if the measured values of a certain parameter vary about a mean, and the mean is gradually increasing (in other words, there is a trend), the trigger level may not be invoked until the mean has reached it, as opposed to the first time a spot value exceeds the trigger level. There are recognised statistical techniques for dealing with situations like this, and the monitoring frequency should be influenced by the variability of the parameter.

Another example of setting trigger levels, this time for augmentation of surface water flows in the Mendips, is presented in Box 5.1. There is also further discussion of trigger levels in relation to karst aquifers, in Appendix 3.

If sufficient historical monitoring data are available, and there is a good understanding of the conceptual model, it may be possible to develop a empirical relationship between parameters such as rainfall and evapotranspiration that can be seen as driving the system, and parameters such as groundwater levels and river flows that can be regarded as responding to the drivers. At its simplest, this relationship would be derived by multiple correlation, with no physical basis to the correlation coefficients, and is sometimes referred to as a climate response model. This technique needs to be used with great care (see Appendix 3, under Step K6, where such models are described as predictive models), but in the right circumstances there is potential for it to complement or even replace the use of trigger levels. Used correctly, such a model can highlight the effects of natural fluctuations in climate, so that dewatering operators do not get unfairly blamed for adverse impacts on sensitive receptors.

Box 5.1: Example of setting trigger levels for surface water flows

Torr Quarry is located in the Mendip Hills in Somerset, and extracts Carboniferous Limestone. A system of stream augmentation was put in place by the quarry operator, in response to concerns that the long-term quarry dewatering may cause springs to run dry and stream flows to be reduced. Torr Quarry is often quoted in discussions of quarry dewatering, and the background to the environmental and planning issues is well summarised by Thompson *et al* (1998), as Case Study 1. Agreement on the system of stream augmentation was reached between the operator and the Mineral Planning Authority in 2000, that is, before the Water Act 2003. The legal mechanism for the agreement was Section 106 of the Town and Country Planning Act 1990. Among other things, the relevant Section 106 Deed of Agreement contains details of the river augmentation measures, the technical aspects of which are as broadly as follows:

- A facility for the storage of augmentation water had to be constructed, with a capacity of at least 425 million litres. From this storage facility, water can be pumped to augment the flows in three different watercourses, Whatley Brook, Alham Stream and Nunney Brook.
- In Whatley Brook, augmentation occurs when flow declines to less than a trigger level (5.8 l/s) measured at a reference gauge just upstream of the augmentation point.
- In Alham Stream, augmentation occurs at various rates, depending on the flow in a reference stream (Midford Brook, assumed to be unaffected by the quarry dewatering), with the aim of maintaining at least 20 l/s flow, but with a maximum augmentation rate of 10 l/s.
- In Nunney Brook the system is similar to Alham Stream, that is, governed by flows in Midford Brook, with the aim of maintaining 35 l/s, but with a maximum augmentation rate of 15 l/s.
- Stream flows and quarry dewatering rates are measured at 60-minute intervals.
- In addition, groundwater level surveys are carried out at monthly intervals, water quality data are collected, and surveys of the ecology of the target streams are undertaken.

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List of abbreviations

AONB	Area of Outstanding Natural Beauty
AP	Assessment Point (on rivers as part of the RAM Framework)
BFI	Baseflow Index
BGS	British Geological Survey
CAMS	Catchment Abstraction Management Strategy (or Strategies)
CEH	Centre for Ecology and Hydrology
CCW	Countryside Council for Wales
CIRIA	Construction Industry Research and Information Association
CWS	County Wildlife Site
DCPE	Double Continuum Porous Equivalent
Defra	Department for the Environment, Food and Rural Affairs
DETR	Department of the Environment, Transport and the Regions
DFPC	Discrete Fracture Porous Continuum
FAO	Food and Agriculture Organization (of the United Nations)
GIS	Geographical information system
GWMU	Groundwater Management Unit
HIA	Hydrogeological impact appraisal
HOST	Hydrology of Soil Types
IGARF	Impacts of Groundwater Abstraction on River Flows (spreadsheet tool)
IPPC	Integrated Pollution Prevention and Control
JNCC	Joint Nature Conservation Committee
maOD	Metres above Ordnance Datum
mbgl	Metres below ground level
MCDA	Multi criteria decision analysis
MIRO	Mineral Industry Research Organisation
MORECS	Met Office Rainfall and Evaporation Calculation System
MOSES	Met Office Surface Exchange System
MPA	Mineral Planning Authority
NRA	National Rivers Authority
PoM	Programme of measures
PWL	Pumping water level
R&D	Research and development

RAM	Resource Assessment Methodology (part of the CAMS process)
Ramsar	Not actually an abbreviation, but a reference to the international Convention on Wetlands, signed at Ramsar, Iran, in 1971
RBD	River Basin District
RBMP	River Basin Management Plan
REV	Representative elementary volume
SAC	Special Area of Conservation (under the EC Habitats Directive)
SP	Self potential
SPA	Special Protection Area (under the EC Birds Directive)
SSSI	Site of Special Scientific Interest
UKTAG	United Kingdom Technical Advisory Group (on the WFD)
VLF	Very low frequency
WFD	EC Water Framework Directive
WRMU	Water resource management unit

Glossary

Abstraction: Removal of water from groundwater or surface water, usually by pumping

Anisotropic: Condition of an aquifer in which the physical properties vary with direction

Aquiclude: Geological formation through which virtually no water moves

Aquifer: Subsurface layer or layers of rock or other geological strata of sufficient porosity and permeability to allow either a significant flow of groundwater or the abstraction of significant quantities of groundwater (from WFD)

Aquitard: Poorly-permeable geological formation that does not yield water freely, but may still transmit significant quantities of water to or from adjacent aquifers

Baseflow: The proportion of flow in a river that is contributed by groundwater

Conceptual model: A synthesis of the current understanding of how a real system behaves, based on both qualitative and quantitative analysis of field data

Cone of depression: Depression in the water table or piezometric surface around a groundwater abstraction

Confined (aquifer): Saturated aquifer that is isolated from the atmosphere by an overlying impermeable formation

Consumptiveness: The proportion of the total quantity abstracted that is consumed, and not available for return to the environment

Derogation: Abstraction of water that prevents a person entitled to a protected right from abstracting water to the extent authorised on their licence

Desiccation: The drying out of the soil or exposed strata beyond natural levels of variability as a consequence of groundwater abstraction

Dewatering: The control of groundwater levels, usually by abstraction, to enable activities such as construction and mineral extraction to continue below the natural water table

Drawdown: The vertical distance between the static water table or piezometric surface and the surface of the cone of depression

Groundwater: All water below the surface of the ground in the saturation zone and in direct contact with the ground or subsoil (from WFD)

Hazard: A property or situation that in particular circumstances could lead to harm

Heterogeneous: Non-uniform in structure, composition or properties

Homogeneous: Uniform in structure, composition and properties

Hydraulic conductivity: A measure of the rate at which water can flow through a medium (the constant of proportionality in Darcy's Law)

Isotropic: Condition of an aquifer in which the physical properties are the same in all directions

Karst: Terrain composed of or underlain by carbonate rocks that have been significantly altered by dissolution

Net gain: The amount by which river flow is increased above what it would naturally have been (when augmenting with groundwater)

Recharge: The process by which water is added to groundwater, or the amount of water added to groundwater in a given period

Risk: A combination of the probability (or frequency) of occurrence of a defined hazard and the magnitude of the consequences of the occurrence

Subsidence: The settlement of the land surface or buildings as a direct consequence of a decline in groundwater levels produced by groundwater abstraction

Storativity: A dimensionless measure (also known as **storage coefficient**) of the amount of water released from or taken into storage in an aquifer per unit surface area for a unit change in hydraulic head

Transmissivity: A measure of the ease with which water can flow through a saturated aquifer (the product of the hydraulic conductivity and the saturated thickness)

Unconfined (aquifer): Aquifer where the water table is exposed to the atmosphere through unsaturated overlying material

Water balance: Quantification of all the inputs to, outputs from, and storage changes within, a given water system

Water table: The surface of a body of unconfined groundwater where the pressure is at atmospheric pressure

Appendix 1: Regulatory context

Introduction

This appendix, first referred to in Section 1.2 of the main report, provides more information on the regulatory context for the licensing of groundwater abstractions related to dewatering. Even with the coming into force of the Water Act 2003, and the introduction of transfer licences as a new type of abstraction licence, several other regulatory regimes are still highly relevant to the assessment of the impacts of dewatering on water resources and the water-related environment. These include the Habitats Directive, Catchment Abstraction Management Strategies (CAMS), and the Water Framework Directive, which will now be described in more detail.

Habitats Directive

The EC Habitats Directive (92/43/EEC) is transposed into UK legislation as the Conservation (Natural Habitats etc) Regulations 1994, commonly referred to as the Habitats Regulations. The Directive requires Member States to designate sites based on species and habitats listed in its annexes, combined with existing designations from the Birds Directive (79/409/EEC). Once agreed by the European Commission, these sites become part of a European network, called Natura 2000. Member States must then take measures to maintain or restore sites to a favourable conservation status, depending on the habitats and species for which the sites have been selected. Natura 2000 sites are Special Protection Areas (SPAs), classified under the Birds Directive, or Special Areas of Conservation (SACs), designated under the Habitats Directive. Although not specifically required by the Habitats Regulations, similar protection is afforded to sites designated under the Ramsar Convention on Wetlands, as a result of a Government policy statement in November 2000.

The Habitats Regulations require the Environment Agency, as a Competent Authority, to ensure that no Environment Agency activity, permission, plan or project results in an adverse effect on the integrity of a European site, unless there are imperative reasons or overriding public interest, *and* there are no alternative solutions, *and* compensatory measures are provided. This applies to direct and indirect effects of activities and permissions. The assessments of applications for proposed licences, including transfer licences, are carried out in close consultation with Natural England or the Countryside Council for Wales (CCW). The Regulations specify that this should be done in four stages, as outlined below:

Stage 1 Identifying relevant applications: Any application for an abstraction licence for groundwater or surface water from a hydrological system, part of which is a Natura 2000 site, must be considered. For groundwater, this means any abstraction which is in hydraulic continuity with the Natura 2000 site. It may not be known whether or not there is hydraulic continuity at this early stage, so it should be assumed that there is, unless it can be demonstrated that the abstraction is *not* in hydraulic continuity with the site.

Stage 2 Assessing likely significant effects: This stage is basically a risk assessment exercise, aimed at answering the following questions:

- Is there a potential impact (see below for examples of impacts) that the abstraction might have on the interest features of the site, either directly or indirectly?
- Are the interest features sensitive to this impact?
- Is each potential impact likely to affect the interest features of the site?
- What is the significance of the scale or magnitude of the impact?

The judgement of significance is made on a case-by-case basis, taking account of local circumstances and the site-specific combination of interest features. It is important not to consider the abstraction in isolation, but to consider possible cumulative effects of all the abstractions that are in hydraulic continuity with the site. After consultation with Natural England or CCW, applications having no effect, or effects that are trivial, can be progressed without further consideration under the Habitats Regulations. Otherwise, a more detailed assessment needs to be undertaken (Stage 3).

Stage 3 Appropriate assessment: The aim of the appropriate assessment is to decide whether it can be ascertained that the integrity of the site will not be adversely affected by the proposal. The starting point for a water resources appropriate assessment is usually an understanding of the hydrological and hydrogeological functioning and water budget of the Natura 2000 site (in other words, a good conceptual model). The understanding of the hydrogeological impacts is then linked to an assessment of the potential ecological impacts. It is recognised that other factors may contribute to an apparent effect on the site, which is not attributable to a water resources authorisation. For example, an increase in scrub cover can result in drying of a wetland, and maintenance works on a watercourse can reduce the extent and frequency of surface water flooding. Test pumping and even detailed numerical modelling may be necessary during Stage 3, but the word 'appropriate' indicates that the scope and content of the assessment will depend on the location, size and significance of the proposal.

Stage 4 Determination of the application: Authorisations under the Habitats Regulations may include conditions designed to avoid adverse effects on the integrity of the Natura 2000 sites. If it has been determined that there is no adverse effect from the proposal on the site's integrity, then the Environment Agency can authorise the permission. The original proposal may be modified to include mitigation or licence conditions to ensure that there is no adverse effect. If there are no mitigation measures or licence conditions that can ensure that the site's integrity will not be adversely affected, then the proposal will be refused.

Examples of impacts that are relevant to water resources (see Stage 2 above) are:

- Changes in wetland water levels and surface flooding regime.
- Changes in river flow or velocity regime.
- Modifications to surface water catchments.
- Reduced dilution capacity or increased residence times.
- Changes in water chemistry or salinity regime.
- Changes in fresh water flows to estuaries.
- Habitat loss.
- Entrapment (fish kill associated with water intake structures).

These impacts are just as relevant to transfer licences as to full abstraction licences, although it must be said that with transfer licences for dewatering, there is by definition no intervening use of the water. This presents opportunities to eliminate or mitigate the impacts by returning the water to the environment, although precisely how this is done will have to be considered carefully in order to ensure that the integrity of the European site is protected.

Sites of Special Scientific Interest

The main legislation governing Sites of Special Scientific Interest (SSSIs) is the Wildlife and Countryside Act 1981, as amended by the Countryside and Rights of Way Act (2000). Natural England and CCW are the bodies responsible for identifying, notifying and protecting SSSIs (in England and Wales respectively). They investigate activities that are damaging SSSIs and can take appropriate action, including securing restoration. The Government has set a Public Service Agreement target that 95 per cent of all nationally-important wildlife sites should be in favourable condition by 2010. The Environment Agency is obliged to notify Natural England or CCW before issuing a permission that may cause potential damage to a SSSI. A review was undertaken jointly by the Environment Agency and English Nature (now Natural England), to identify SSSIs in England that are potentially affected by abstraction (English Nature and Environment Agency 1999). Of 358 sites reviewed, six sites were identified as facing potential impacts from mineral or gravel extraction, but all six sites were categorised as not significantly affected by abstraction.

Catchment Abstraction Management Strategies

In 1999, the Government published an important document, *Taking Water Responsibly* (DETR 1999) outlining its decisions, following consultation, on changes to the abstraction licensing system. Many of the proposed changes required new legislation, now embodied in the Water Act 2003. However, some changes were achievable within the powers already held by the Environment Agency, and the most important of these was the development of Catchment Abstraction Management Strategies (CAMS). CAMS make more information on water resources allocation publicly available, and allow the balance between the needs of abstractors and those of the aquatic environment to be determined in consultation with the local community and interested parties. The CAMS process is described in detail in Environment Agency (2002a), and in very simple terms it can be summarised as follows:

i. *Definition of CAMS areas:* England and Wales have already been divided up into 126 CAMS areas plus three ‘corridor’ CAMS (for the Rivers Severn, Trent and Thames). The rest of the process is being applied to each area, on a rolling programme to cover the country by 2008.

ii. *Pre-consultation:* for each CAMS area, stakeholder groups are set up, in order to raise awareness and to request information and comments.

iii. *Resource assessment and resource availability status:* this is achieved by using the Resource Assessment and Management (RAM) Framework, that has been developed by the Environment Agency (see below).

iv. *Sustainability appraisal:* this uses the Government’s approach to sustainable development to consider the wider implications of options for water resources development, such as the environmental impacts, social implications, economic impacts, and impacts on natural resources.

v. *Consultation:* this provides an opportunity for all stakeholders to comment on the proposed strategy.

vi. *Final CAMS document*: the final document is published and implementation begins, with the strategy being updated annually and reviewed every six years.

As mentioned above, decisions about resource assessment and resource availability status are made using the RAM Framework, which takes an integrated approach to assessing the groundwater and surface water resources available within the catchment (Environment Agency 2002b). The RAM Framework consists of the following stages:

i. Define CAMS area, collect and integrate existing data, develop conceptual understanding.

ii. Highlight CAMS rivers, tributaries, aquifers, groundwater outflows and local issues.

iii. Assess the ecological sensitivity of rivers to abstraction, to arrive at a hydroecological Environmental Weighting.

iv. Identify CAMS river Assessment Points (APs) and groundwater management units (GWMUs).

v. Conduct a preliminary river AP resource assessment.

vi. Assess the GWMU resources (see the five tests below).

vii. Map and integrate river AP and GWMU assessment results.

viii. Review and iterate to refine and prepare standard output for illustrative years.

ix. Finalise maps of resource availability status.

As far as groundwater is concerned, the RAM Framework uses five tests to determine whether there are resources available within the aquifer unit for further licensing, or whether it is fully-licensed, over-licensed, or over-abstracted. The tests explicitly consider the links between the aquifer and hydraulically-connected rivers, and they have been developed largely for aquifer units where the link to rivers is the limiting factor. The five tests for groundwater resource assessment are as follows:

Test 1 (natural recharge and inflow resource compared to abstraction): considers only the annual mean recharge and for this reason can be carried out using limited data. The test gives an upper bound to the possible sustainable yield as it assumes the aquifer unit has infinite storage (generally only approximately true for sandstone aquifers), and that the unit is watertight with no losses to rivers, springs or other features.

Test 2 (summer baseflow or groundwater outflow compared to abstraction impacts): is the major test and considers both the flows to rivers required in summer and the importance of these flows to the river environment. It may also consider outflows that may be needed to prevent saline intrusion into the unit or to support adjacent, hydraulically-linked units. Scenario groundwater outflows are compared with the flow needs of the river to assess the resource availability in the aquifer.

Test 3 (observed trends in groundwater levels or quality): uses long-term trends in groundwater level or quality to identify whether the unit is being over-abstracted, without explicit modelling. This test does not help to identify in advance where such problems may occur but is useful in identifying units that are clearly over-abstracted. It is useful where there are problems in defining acceptable summer outflows, for example in confined conditions.

Test 4 (other evidence of unacceptable groundwater abstraction impacts): uses anecdotal evidence to help to identify units that may already be over-abstracted. It assumes that those reporting damage to rivers or wetlands are correct to associate this with groundwater abstractions. In practice, this test is used to highlight issues for further study or monitoring to try to gain harder evidence, rather than to limit the resource directly.

Test 5 (optional local tests): allows for local knowledge and experience to influence the decision on resource availability. There may be local details that may be important but not covered in the general procedures, such as links to wetlands, storage or drought recharge. This test cannot override the results of Tests 1 to 4.

Water Framework Directive

The Water Framework Directive (2000/60/EC) was approved by the European Union in December 2000, and is often described as the most significant piece of European water legislation for over 20 years. The Water Framework Directive (WFD) had to be transposed into Member States' legislation by 2003, and will be implemented in stages up to 2015. It enshrines in law a holistic approach to water management, and it rationalises and updates previous piecemeal legislation by setting common EU-wide objectives for water. The purpose of the WFD is to establish a framework for the protection of inland surface waters, transitional waters, coastal waters, and groundwater, which, among other things:

- prevents further deterioration and protects and enhances the status of aquatic ecosystems and associated wetlands;
- promotes sustainable water use based on a long-term protection of available water resources; and
- ensures the progressive reduction of pollution of groundwater and prevents its further pollution.

EU Member States are required to achieve "good surface water status" and "good groundwater status", and also to prevent deterioration in the quality of those waters that are already good. Ecological quality, in addition to chemical quality, is taken into account in the assessment of status, for surface waters in particular. For groundwater, the assessment of status must take into account quantity as well as quality. The emphasis of the WFD is on anthropogenic activities and pollution, as opposed to naturally-occurring substances. Under the WFD, all groundwater has to be protected from new or on-going pollution. However, not all groundwater has to be managed in relation to the specific objective of good groundwater status. The concept of groundwater bodies has been introduced, which embraces:

- the groundwater that is in continuity with ecosystems and can place them at risk, either through the transmission of pollution or by unsustainable abstraction that reduces baseflow;
- the groundwater that can provide for the abstraction of significant quantities of water for human use (with the definition of "significant" in the WFD being anything over 10 m^3/d).

A 'groundwater body' is therefore the management unit under the WFD that is necessary for the subdivision of large geographical areas of aquifer in order for them to be effectively managed. The concept also provides a convenient way of grouping, monitoring, managing and reporting on adjacent small blocks of aquifer of differing hydrogeological nature but with similar hydrogeological properties. Groundwater bodies have been delineated by the Environment Agency jointly with the British Geological Survey (BGS), based on conceptual hydrogeological models.

One of the underpinning principles of the WFD is that of integrated river basin management. Groundwater bodies (and surface water bodies) are assigned to River Basin Districts (RBDs), based on hydrological catchments, with coastal waters and groundwater being assigned to the most appropriate RBD. For each of the RBDs, a River Basin Management Plan (RBMP) must be produced, followed by a detailed Programme of Measures (PoM). This is the main mechanism for achieving the objectives of the Directive. The WFD recognises that there are costs associated with achieving the objectives, as well as benefits. Cost-benefit analysis of the proposed PoM, and indeed of existing water use, forms an integral part of the process.

Initial characterisation of all groundwater bodies has been undertaken, involving an assessment of the status of each groundwater body (quantitative and chemical), identification of the pressures to which the groundwater body is subject, determination of the potential impacts of the pressures, and finally an assessment of whether the groundwater body is at risk of failing to achieve good status by 2015. Further information on the initial characterisation, including the draft pressures and impacts maps, can be found on the Environment Agency's website.

All groundwater bodies identified as being at risk during the initial characterisation will be the subject of more detailed investigations, known as further characterisation, the aim being to design a PoM that ensures that the groundwater body achieves good status by 2015. During the initial characterisation of abstraction pressures on groundwater, the conservative assumption was made that all groundwater use is consumptive (UKTAG 2003). Where the use turns out to be non-consumptive (and the specific example is given of water used for sand and gravel washing being returned to the aquifer), this will be picked up by the further characterisation.

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Appendix 2: Dewatering

Introduction

This appendix, first referred to in Section 2.5 of the main report text, provides more information on the hydrogeology of dewatering for those not already familiar with the subject, highlighting both the similarities and the differences between dewatering and 'conventional' groundwater abstraction.

Background to dewatering

In the context of mineral extraction, the main reason why quarries or mines are dewatered is to enable extraction of the mineral to continue below the water table. This increases mineral recovery, and allows more flexibility in exploiting different grades of mineral to achieve the desired product blends. In certain circumstances, quarries can of course be worked wet, that is, extraction of the mineral continues below the water table without dewatering. However, this is generally only in unconsolidated or relatively soft formations such as sand, gravel and Chalk, where drag-lines or dredgers can be used. Dewatering takes place in quarries, mines and construction sites as part of a wider water management system that can have other objectives, including the following:

- To remove excess water that enters the excavation from rainfall or surface run-off.
- To reduce or prevent floor heave or piping in the base of an excavation if there is upward groundwater pressure.
- To improve slope stability by reducing pore pressures, to avoid sloughing or slope failures on the walls of the excavated void. In addition to the obvious safety implications, this can significantly affect the economics of a quarry or mine. Careful design of a dewatering system can allow the walls to be steeper, thus maximising mineral reserves for a given quarry or mine footprint, or reducing overburden stripping.
- To reduce in advance the water content of *in situ* minerals, to facilitate their extraction (making the use of explosives easier, for example) and immediate processing (by dry-screening, for example).
- To improve the trafficability of the benches and other horizontal surfaces for the movement of heavy machinery.

With these objectives in mind, let us now compare abstraction for dewatering with abstraction of groundwater for say public water supply. The focus of groundwater abstraction for public water supply is to abstract a given quantity of water in the most efficient manner, within certain environmental, water quality and licensing constraints. It is in the interests of the abstractor to minimise the drawdown for a given abstraction rate, because this directly affects pumping costs, and reduces environmental impacts. The approach taken for groundwater abstraction for dewatering is completely different. The main differences are as follows:

- The focus of the design of the dewatering system is to maintain a given groundwater level (as opposed to quantity), and it is usually in the interests of the abstractor to minimise the quantity of groundwater pumped out to maintain a given level. Depending on the hydrogeological setting, the quantities abstracted may vary seasonally, if not from day to day. Dewatering may not be necessary at all at certain times of the year.

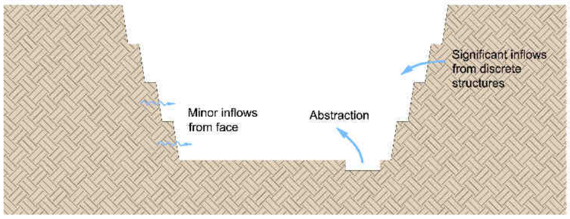
- Dewatering abstraction is often short term, with regular changes in the dewatering system. For example, a sand/gravel operation may only dewater the cell actually being worked, for periods of the order of a year, before moving on to a new cell and progressively restoring the worked-out cells.
- The system operator sometimes has the option of reducing the quantity of water reaching the dewatering pumps by constructing flow barriers (using cut-off walls, slurry trenches, grouting or ground freezing, for example), particularly in the construction dewatering context. Natural flow barriers can also be exploited by careful system design.
- The environmental impact of the abstraction can be significantly reduced by returning the water to the environment, by augmentation of surface water flows or by recharge of groundwater. A significant factor that affects the degree to which the impacts can be reduced is the amount of water consumed by the dewatering operation.
- Depending on the restoration plan on cessation of dewatering operations, there can be permanent impacts on the hydrogeological system. For example, open water bodies can act as ongoing groundwater abstractions due to evaporation.

Despite these differences, the hydrogeological principles of groundwater abstraction for whatever purpose are the same, even if the objectives of the abstractor and the design of the abstraction system are different. Artificial flow barriers just affect the way in which groundwater flows towards the abstraction point, and excavated voids can often be regarded as very large diameter wells. If someone decided to operate an ordinary abstraction borehole with the objective of maintaining a given pumped water level rather than a given abstraction quantity, the amount of water pumped out would vary in exactly the same way as for dewatering.

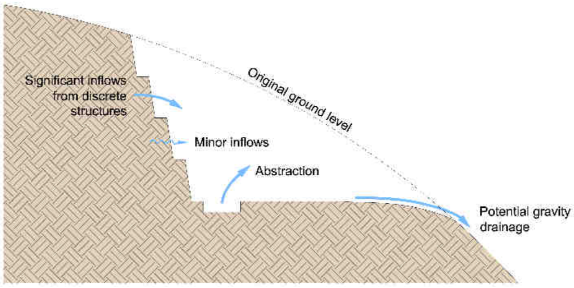
There are of course many quarries or mines that do not require dewatering, if working above the water table, or some that only need to be dewatered seasonally. 'Groundwater control' is a common phrase used instead of 'dewatering'. It is a good description of what is trying to be achieved, and is the title of the main CIRIA report on the subject (Preene *et al* 2000).

Hydrogeological settings

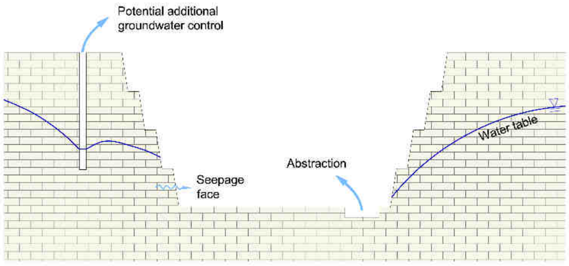
Rates of groundwater inflow to an excavation will depend on a wide range of factors such as the dimensions of the excavation, the local water resources balance, and the hydraulic properties of the soils and rocks being excavated. With so many variables, no two dewatering operations are exactly the same in terms of their hydrogeological setting, and it is important not to jump to conclusions when conceptualising a particular operation. Nevertheless, it can be useful to identify generic hydrogeological settings. There are many different ways to categorise hydrogeological settings (see for example Stuart and Davies 2002, and Geoffrey Walton Practice1988), but for the purposes of this report, seven different settings can be described and illustrated, as follows (Figure A2.1):



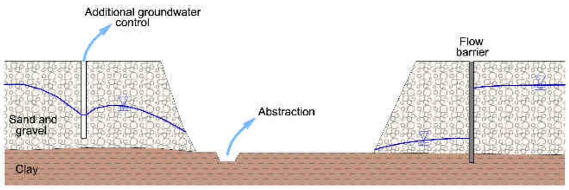
1) Variable-permeability hard-rock formations: Generally occur in igneous or metamorphic formations, with very variable hydraulic properties, and also in karstic limestone. Local groundwater flows can be significant if transmissive structures are intersected. Excavations are often deep.



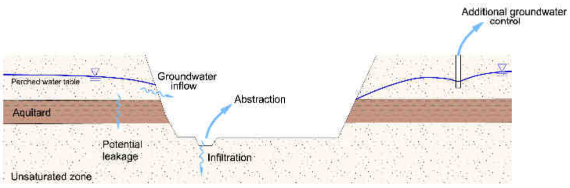
2) Variable-permeability hard-rock formations on slope: As for Setting 1, but dewatering may be possible by gravity drainage. Hillside excavations are common in Scotland, Wales and northern England.



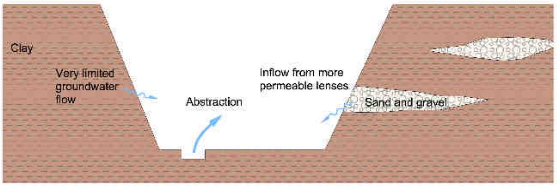
3) High-permeability consolidated formations: Generally in sandstone or limestone formations, with varying proportions of intergranular and fracture/fissure flow. Karst development in limestone can mean significant groundwater impacts. Excavations are often deep.



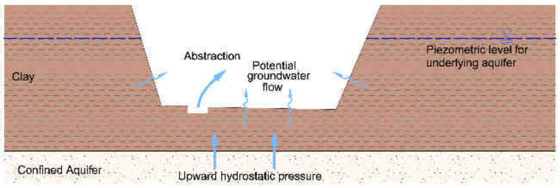
4) High-permeability unconsolidated formations: Commonly in minor aquifers, such as shallow fluvio-glacial sands and gravels, which may or may not be hydraulically connected with the underlying unit. Excavations are usually relatively shallow.



5) Excavation through a perched unconfined formation: Can occur in interbedded clay/sand units overlying a major aquifer, for example, the Lower London Tertiaries over chalk.



6) Low-permeability unconsolidated formations: Generally in unproductive formations such as clay, with few issues of groundwater control. However, significant inflows may occur from sand/gravel lenses, or where there is over-consolidation and fracturing.



7) Low-permeability formations with underlying confined aquifer: As for Setting 6, but with potential issues of ground heave or piping in the base of the excavation from upward groundwater pressure.

Figure A2.1 Generic hydrogeological settings for quarries or mines

These seven settings are by no means exhaustive, and it will always be possible to identify exceptions or variations. However, they serve to illustrate the wide range of hydrogeological conditions that can be encountered when excavating minerals or dewatering for construction, and they may be useful pointers when commencing a conceptual model from scratch. This appendix is written in general terms so that it can apply to mining, quarrying and construction dewatering. Some of the features illustrated in Figure A2.1, such as the additional groundwater control and flow barriers, are commonly encountered in construction dewatering, even though they may be rare in quarrying and mining.

The subject of karst is introduced under Settings 1 and 3 in Figure A2.1. The development of karstic features in carbonate rocks can have dramatic effects on their hydrogeological behaviour. When trying to predict the hydrogeological impacts of dewatering, the level of uncertainty encountered when dealing with karstic aquifers may be an order of magnitude greater than for most non-karstic aquifers. It will be seen later (in Appendix 3) that great care needs to be taken when developing conceptual models for quarries or mines in karstic rocks.

Dewatering methods and drawdown profiles

There are many different methods for dewatering excavations. Pumping from a sump at the lowest point of the excavation is the most common method, and the one most frequently encountered in the quarrying industry. If the sump pumps can keep up with the rate of groundwater inflow, and the drawdown around the excavation is sufficient to ensure slope stability (where relevant), then this method may be all that is required. However, the sump method may not be suitable for aquifers with large storage capacities and/or high hydraulic conductivity, or shallow wide workings. Other dewatering methods include the following:

- Wellpoint systems, or shallow wells with suction pumps.
- Deep vertical wells with submersible pumps.
- Horizontal drains drilled into the slope face.
- Drainage galleries behind the slope face.
- Eductor (or ejector) systems.
- Electro-osmosis.
- Collector wells (horizontal borings leading into a central well).
- Drainage trenches down, along or at the foot of the slope face.
- Passive relief wells and sand drains.

Many of these methods are rarely used, or tend only to be used in construction dewatering, and this report is not the place to describe them in any more detail. The subject is covered thoroughly elsewhere, particularly in Preene *et al* (2000). The main point is that the actual method used to get the water out of the ground does not usually matter for the purposes of hydrogeological impact appraisal. Many of the methods are hydrogeologically equivalent, and will probably result in similar drawdown profiles around the excavation (as illustrated in Figure A2.2), and indeed in similar total abstraction quantities.

From a hydrogeological point of view, dewatering for construction or engineering works is no different to dewatering for quarrying or mining, and the general principles of HIA

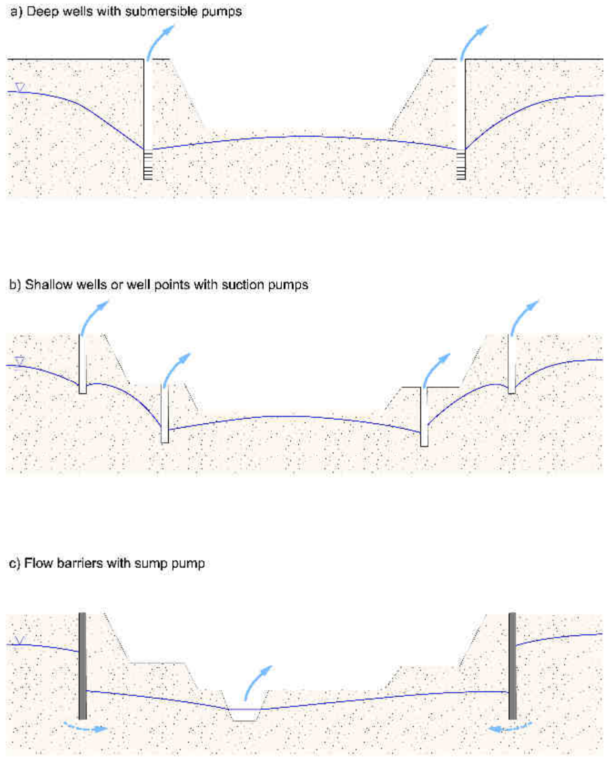


Figure A2.2 Hydrogeological equivalence of dewatering methods

can still be applied. However, note that the term 'dewatering' is used in a general sense in this report. Preene *et al* (2000) point out that when lowering the water table level in unconfined fine-grained soils that do not drain freely, the term 'pore water pressure control' is more appropriate, because the soil may remain saturated (with negative pore water pressure above the new water table).

The application of the HIA methodology to construction dewatering is illustrated by Case Study 6 in Appendix 9.

When undertaking HIA for a dewatering operation, the important thing is not to become sidetracked too much on the dewatering methods, but to concentrate on their implications for groundwater flow and drawdown profile around the excavation. While on this subject, it is worth pointing out that the drawdown profile in the immediate vicinity of deep excavations, especially in fractured rock, does not necessarily follow the shape predicted by analytical equations. It is often observed that the slope of the cone of depression is very steep, with the drawdown being far less than expected at a given radius from the excavation. There are several factors that can contribute to this phenomenon (not all of which may apply in every case), as follows:

- Additional hydraulic head losses as the groundwater flow towards the open excavation becomes turbulent (or non-Darcian) instead of laminar, especially when there is a seepage face on the wall of the excavation (see for example, Dudgeon 1985).
- Groundwater levels being controlled by geological structure, and changing in a series of steps rather than a smooth curve. Depending on where the edge of the excavation is in relation to the structure, there can be sudden jumps in water level away from the excavation.
- Localised increases in the permeability of the rock close to the excavation, either because of the effects of blasting, or because of lithostatic unloading (removing the weight of rock when excavating the void, sometimes causing fractures to open up).

Attempting to draw groundwater level contours around an excavation in fractured rock can be highly misleading, and much more attention should be given to geological structure and the presence of features such as seepage faces.

Long-term implications

With conventional groundwater abstractions from boreholes, the borehole itself usually represents a very small physical disturbance to the aquifer, and when abstraction ceases, given enough time, the hydrogeological system would eventually return to its pre-abstraction state. There are of course potential problems in layered aquifers if the borehole acts as a new connection between previously-unconnected layers, and unprotected boreholes may represent pollution pathways. Clear guidance is available (Environment Agency 1999) on best practice for closing down or abandoning groundwater sources, so that they do not have long-term impacts. With quarries and mines however, there can be significant and permanent impacts on the hydrogeological system as a result of the excavation, once operations have ceased. For quarries or opencast mines that required dewatering when in operation, the most common closure plans are as follows:

- Restoration to open water, usually achieved by allowing the groundwater to find its own level in the abandoned excavation, often accompanied by landscaping and creation of interesting habitats within and around the open water body.
- Backfilling the void with inert material or previously-excavated overburden, to restore the original ground surface profile, or at least to ensure that the finished ground surface is above the new water table.

The long-term implications of these closure plans can include the following:

- *Evaporation from open water:* This represents the equivalent of an ongoing abstraction. The amount of water 'lost' through evaporation from a body of open water depends on many factors, such as wind speed, solar radiation, water temperature, and air humidity.
- *Changes in groundwater level due to open water:* If the water table was inclined before mineral extraction started, then the post-closure groundwater levels are likely to be different, because the surface of an open water body is effectively horizontal (see Figure A2.3c).
- *Changes in groundwater flow paths due to different permeability fill material:* If closure involves backfilling, it is rarely possible to recreate exactly the hydraulic properties of the original *in situ* material when compacting the fill material. This can lead to different post-closure groundwater flow paths and levels, as illustrated in Figure A2.3d.
- *Acid mine drainage:* aeration of previously anaerobic zones can lead to water quality problems in the future, with oxides being dissolved when groundwater levels rise again on the cessation of dewatering, for example.

Assuming that active dewatering ceases on closure of the quarry or mine, then a transfer licence would no longer be required, and the post-closure impacts just described would need to be anticipated and controlled by the planning system instead of the abstraction licensing system. They are mentioned here for completeness, and because the conceptual model developed to support the transfer licence application will help in the prediction of post-closure impacts.

There are of course other uses to which voids are put on cessation of mineral extraction, the most obvious being conversion to landfill. Landfills are controlled by a different body of legislation, and separate guidance is available on hydrogeological risk assessments for landfills (Leeson *et al* 2003).

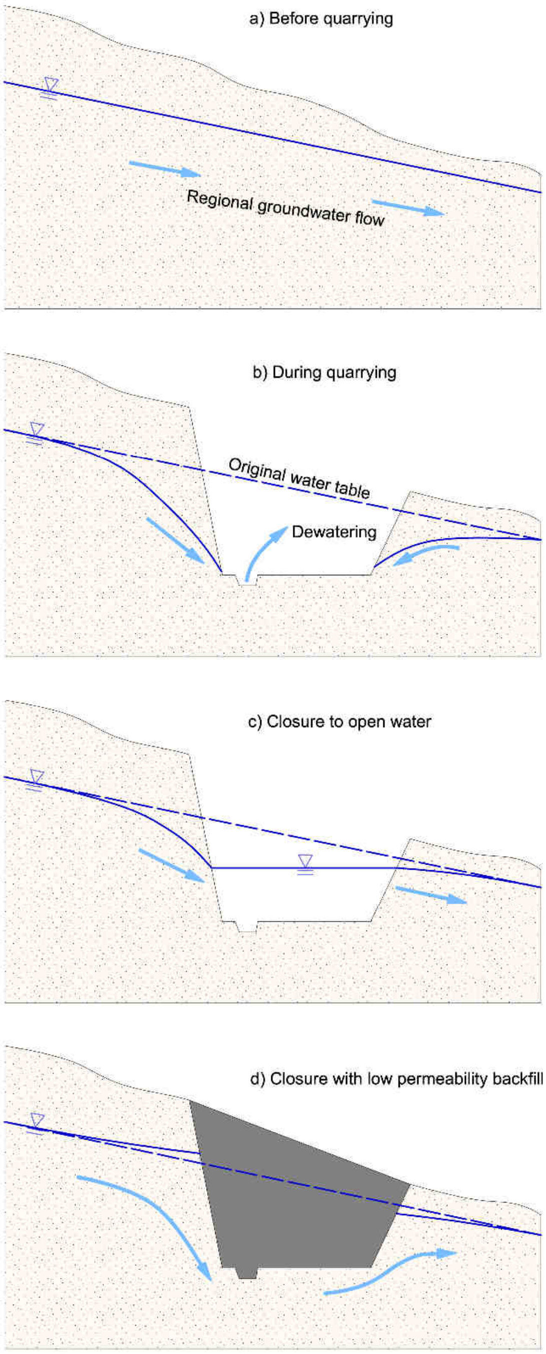


Figure A2.3 Long-term impacts of quarry or mine closure plans

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Appendix 3: Karst

Introduction

This appendix, first referred to in Section 2.5 of the main report, introduces the subject of karst hydrogeology, and discusses the particular characteristics of karst aquifers that need to be taken into account when developing conceptual models and undertaking HIA for dewatering operations within soluble (karst-prone) rocks. Also covered are the various modelling approaches that can be applied in karst aquifers, and the development of a 'monitor-and-mitigate' approach to the management of hydrogeological impacts.

Acknowledgement on authorship: the text for this appendix was largely contributed by Professor Peter Smart, School of Geographical Sciences, University of Bristol, Bristol, BS8 1SS, UK.

What is karst?

Karst terrains are the product of enhanced groundwater circulation that has developed preferentially due to the solubility of the terrain. They can develop in any soluble rock type including carbonate rocks such as limestones and dolomites, and evaporites such as gypsum, anhydrite and rock salt (halite). Where any of these rocks are present, the underlying groundwater system may be karstic in nature. Given the high vulnerability of karstified aquifers and the considerable difficulties in predicting the effects of groundwater abstractions in them, the precautionary principle indicates that groundwater systems developed in these rock types should be considered as karstified until this is proven not to be the case.

Karst terrains can often be recognised by the presence of a distinctive suite of landforms including: limestone pavements and other small-scale surficial and sub-soil dissolution forms (termed *karren*), sinking streams, blind and dry valleys, closed depressions of a variety of sizes and origins, caves and springs (Quinlan *et al* 1991). Of these, the closed depression and dry valley are perhaps the most useful general indicators of karst.

From the point of view of HIA, it is important to distinguish between the morphological and functional recognition of karst. Many landscapes continue to display karst landforms developed in earlier phases of landscape development, even though the groundwater systems underlying them no longer function in a karstic manner. For instance, in the Carboniferous Limestone of south-west England and Wales, caves developed when the limestones were first exposed sub-aerially in the Triassic are often intercepted in quarries and other excavations. It is often found that the caves have been occluded by sediment fill, and in some cases mineralisation, and that they are no longer conduits for groundwater flow. Such fossil karst terrains are termed *paleokarst*. They are the result of major changes in the boundary conditions for karst development, caused by changes in climate, sea level, and patterns of sediment supply (Osborne 2000). Thus, although the presence of a distinctive karst morphology may indicate that the associated aquifer is actively karstic, this need not necessarily be the case. However, the precautionary principle should again be applied, with the aquifer assumed to be actively karstic unless it can be shown that it is not.

Karst groundwater systems

Karst groundwater systems are unusual because they develop channel or conduit flow, that can give rise to very rapid and highly localised movement of groundwater. In carbonate rocks, there is a strong non-linearity in the rate of dissolution as chemical equilibrium is approached, so that some under-saturation persists if there is significant flow, allowing continuous dissolutional enlargement of the openings through which groundwater flows. Thus, any initially open pathways such as joints or bedding planes through which groundwater flows may be subject to dissolutional widening (Worthington 1999). Such enlarged channels are frequently organised into a dendritic, hierarchical, tributary network that feeds to major springs (Figure A3.1)(Bakalowicz *et al* 1995). Such hierarchical channel networks result from the strong positive feedback between the circulation of fluid and the rate of dissolution, which is primarily dependent on groundwater flux. Thus flow routes with large discharges tend to develop most rapidly, and capture flow from adjacent smaller openings that have higher heads, building a dendritic tributary network very similar to that of surface water drainage. Indeed, this analogy can be extended because, like surface rivers, springs fed by karst conduits can often have well-defined underground catchments, although these do not necessarily conform with the surface topography.

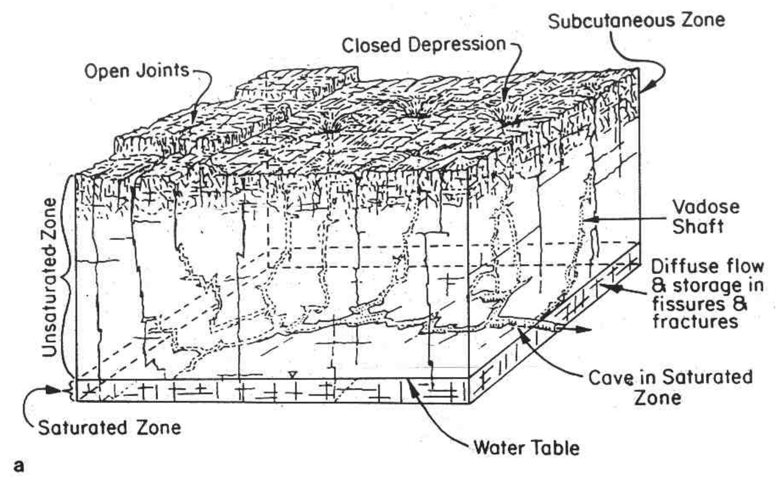

Figure A3.1 Features of a karst groundwater system
(from Bakalowicz *et al* 1995)

The majority of dissolution in soluble rocks occurs where the solvent phase first contacts the mineral phase, as this is when it has a high degree of under-saturation. This may be at the bare bedrock surface where soil is absent, but more generally is at the base of the soil. There is thus a tendency to develop a zone of elevated dissolutionally-enhanced porosity within the shallow subsurface. In carbonate aquifers, where dissolution is driven primarily by carbonic acid derived from the elevated carbon dioxide (CO_2) concentrations present in the soil atmosphere (resulting from root respiration and bacterial decomposition of organic matter), this zone may be particularly pronounced. It is termed the *epikarst aquifer* or subcutaneous zone (Figure A3.1). In contrast to conventional aquifers, in karst there is substantial storage and redistribution of recharge within the epikarst aquifer (Williams 1983; Smart and Friederich 1986). Failure to recognise the significant contribution of this zone to the hydrological behaviour of the karst groundwater system can lead to substantial errors in forward predictions. There are however considerable difficulties in developing techniques to evaluate the importance of the epikarst aquifer at any individual site, and in the incorporation of its behaviour in predictive models.

Karst aquifers are best considered as triple-porosity aquifers, although in some aquifers the smallest scale openings may not be hydrologically significant (Quinlan *et al* 1996; Worthington 1999). At the smallest scale is matrix porosity, comprising inter-crystalline and inter-granular pores of small diameter (50-500 µm). At the intermediate scale are fractures that have experienced little or no dissolutional enlargement and have typical widths of <1 mm. Because of their small apertures, flow is laminar in both these types of opening. However, at the largest scale of dissolutional channels, apertures range from several millimetres in dissolutional fissures to metres in cave conduits, and under most head conditions flow is turbulent. The development of turbulent flow in karstic channels is important because it allows sediment transport by groundwater flow, which may impact upon water quality. More significantly, flow can no longer be described using Darcy's Law (which applies only to laminar flow) and conventional approaches to groundwater flow modelling are inappropriate.

Hydrogeological impacts in karst groundwater systems

The nature and type of the impacts of groundwater abstraction in karst aquifers (which exhibit groundwater flow in conduits) differ from those in aquifers where groundwater flow is predominantly intergranular in a number of ways:

- *Impacts on groundwater levels and flows:* these are often of a much greater magnitude, because of the very high transmissivities of the conduits. The impacts also tend to be irregularly distributed, because of the highly heterogeneous distribution of transmissivity in karst aquifers. Larger impacts occur along the line of (and in the vicinity of) conduits, including at springs where the conduits discharge – potentially a long way from the dewatered excavation. Smaller impacts occur in areas more distant from conduits where intergranular and small fracture flow dominate – potentially quite close to the excavation being dewatered.

- *Ground subsidence and collapse:* lowering of groundwater levels can cause ground subsidence and collapse in karst terrain. Reduction of pore (or larger void) water pressures causes an increase in the effective stress borne by the aquifer or overlying materials (solid phase), and if the increased effective stress exceeds the strength of these materials, subsidence or ground collapse will occur. The collapse feature usually takes the form of a closed depression, called a sinkhole or doline. Subsidence and formation of sinkholes in karst terrain can occur naturally or it can be human-induced through groundwater abstraction. However, Newton (1976) showed that, of an estimated 4,000 sinkholes formed in Alabama between 1900 and 1976, only 50 (about 1 per cent) were natural collapses. The most widely-reported subsidence problems in the UK are those in the region of Ripon, North Yorkshire, which lies on the outcrop of the very soluble Permian gypsum deposits.

- *'Within aquifer' impacts:* in contrast to aquifers where intergranular flow dominates, karst aquifers can *contain* features of geoecological value. These include rock-forms (such as speleothems) and hypogean fauna. Groundwater abstraction can endanger the favourable hydrological conditions for the formation and maintenance of these features.

In the saturated zone of many mature karst aquifers, water storage is predominantly within the matrix and fracture porosity, which is often termed the diffuse flow component of the aquifer. However, groundwater movement is almost wholly via the channel porosity of the conduit system (Atkinson 1977; Worthington *et al* 2000). Thus, any attempt to predict the impacts of groundwater abstraction or dewatering that does not adequately characterise the behaviour of these two different components of the karst groundwater system is likely to be inadequate. The major difficulty here arises because, whilst the general characteristics of both the conduit network and the diffuse flow system can be determined using appropriate techniques, the actual distribution, location and topology of the conduit network are generally neither known, nor amenable to reliable and precise prediction. Without detailed information on the system responsible for transmission of the majority of the groundwater circulation, prediction of the impacts of abstraction can never be considered reliable in karst groundwater systems. The emphasis must therefore be on a *monitor and mitigate* approach, which will be explained later.

Types of carbonate aquifers

Early attempts to describe the behaviour of carbonate aquifers and the springs that drain them were focussed on the extent to which conduit flow in channels was developed, compared with more diffuse flow in a ramifying fracture network (Schuster and White 1971). Alternative models considered the nature of recharge, which could be as autogenic percolation from the surface of the unconfined aquifer, or as a concentrated allogenic input from streams derived by surface run-off from adjacent clastic terrains that entered the limestone at stream sinks or swallets (Newson 1971). However, further studies demonstrated that the extent of storage was also of considerable significance in controlling aquifer and spring behaviour (Atkinson 1977).

In fact, the predominant type of recharge, the extent of storage and the mode of transmission of groundwater within the aquifer are all important and essentially independent characteristics of any particular carbonate aquifer. Thus, an individual spring catchment in a specific carbonate (or evaporite) aquifer can be considered in terms of its position in a three-dimensional space (Figure A3.2; Smart and Hobbs 1986) defined in terms of:

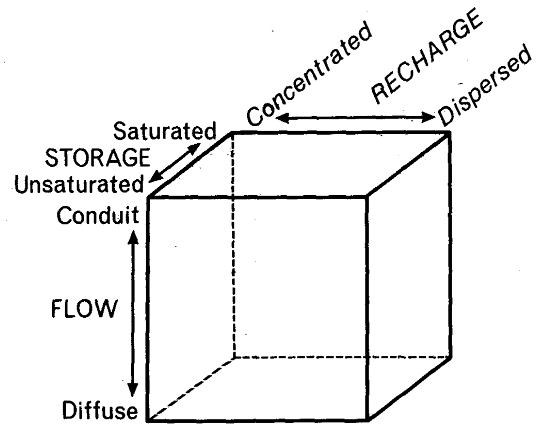

Figure A3.2 Carbonate aquifer model
(from Smart and Hobbs 1986)

- *Recharge*: a continuum between concentrated (swallet) and diffuse (distributed percolation) end-members;
- *Flow*: a continuum between conduit (cave) and diffuse (matrix) end-members; and
- *Storage*: a continuum between high and low storage end-members.

The position of an aquifer in this three-dimensional space can be determined by its configuration. For instance, whether there are stream sinks, losing streams and shafts draining closed depressions, that is concentrated recharge, or whether these features are absent and recharge is diffuse (Figure A3.3). Alternatively, quantitative indicators can be used, such as the recession coefficient (which is an indicator of storage).

The scheme put forward by Smart and Hobbs (1986) offers the most practical scheme for evaluation of the extent of karstic behaviour in a carbonate aquifer. Whilst it is unlikely in practice that the position of any individual aquifer can be determined with any precision in the recharge-flow-storage three-dimensional space, the rationale for the scheme is rather that it is indicative of the type of aquifer behaviour to be expected, and the problems that are likely to be of significance in its management. Hobbs and Gunn (1998) identify four types of karst aquifer with respect to assessment of the potential impacts of quarry dewatering (Figure A3.4):

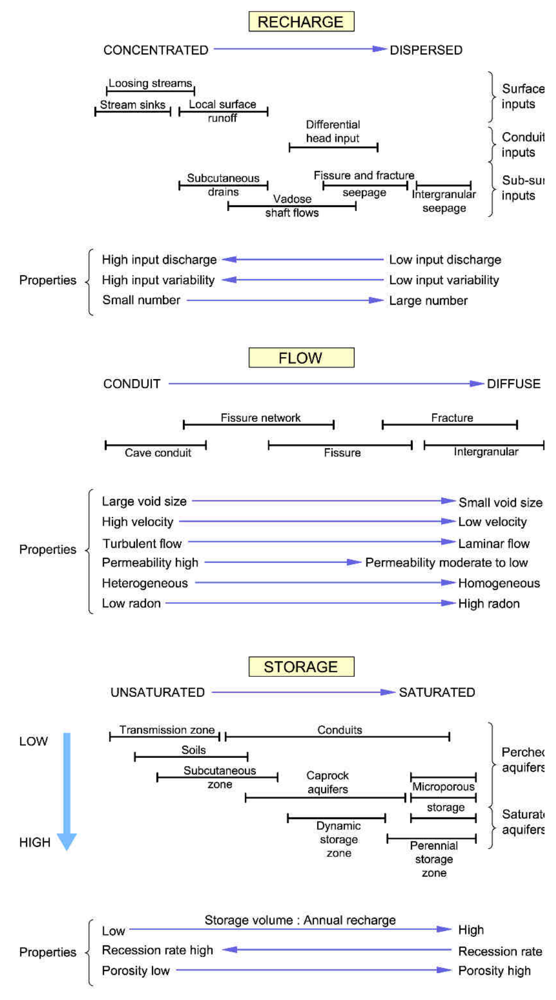

Figure A3.3 Karst aquifer characteristics
(from Smart and Hobbs 1986)

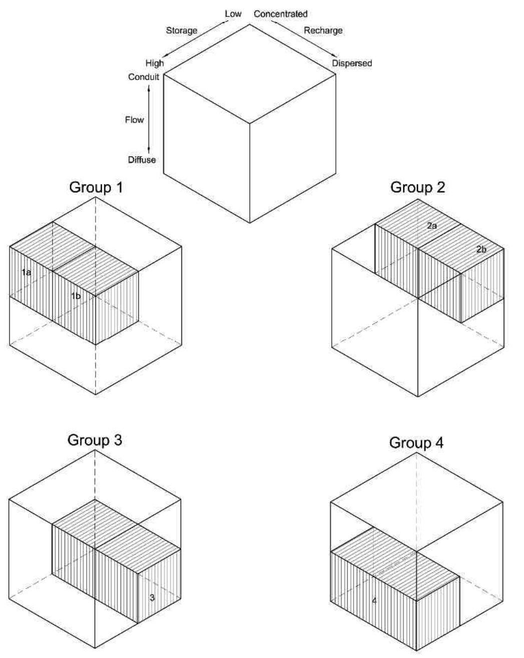

Figure A3.4 Carbonate aquifer classification
(after Hobbs and Gunn 1998)

Group 1 aquifers (high storage, conduit flow, variable recharge) represent the most difficult in terms of prediction of impacts, because conduit flow is well developed. They also have a high risk of spring contamination. Because they have high storage, there is also a substantial groundwater resource that may be impacted by any abstraction. Group 1 is subdivided into aquifers that have a high proportion of concentrated recharge (Group 1a), and those that do not (Group 1b). Group 1a aquifers pose the more difficult situation because there is a tendency to develop higher conduit densities where stream sinks are present in border karst, and the risks of conduit intersection and derogation are thus large. St Dunstan's Well in the Carboniferous Limestone of the East Mendips, Somerset is an good example of a Group 1a spring. It has several proven feeder swallets with a proportion of spring flow fed from allogenic sources, and a well-developed conduit system that has been explored by cave divers over much of its length. The spring was abandoned as a source of supply because of persistent pollution from adjacent limestone quarries, and suspended sediment entering the main swallet from the discharge of other quarries in the non-carbonate catchment area (Stanton 1977). Parts of the Chalk of south-east England where conduit flow is developed, such as the Havant and Bedhampton Springs of Portsmouth (Atkinson and Smith 1974) are also in this category, as are parts of the unconfined Jurassic Limestones, such as the Great Oolite of the southern Cotswolds, which feeds the head of the By Brook and Sherston Avon (Smart 1977). There are however substantial differences in the behaviour and configuration of these three examples, indicating the range of aquifer types that may be included in these initial broad groupings.

Group 2 aquifers (low storage, conduit flow, variable recharge) differ from Group 1 in having much less storage. Thus, whilst they retain the difficulties associated with predicting the impacts of abstraction where conduit flow is present, the lower storage means that the number of water supplies and size of springs supported by the aquifer is likely to be much smaller. Having said that, abstraction impacts can spread much more quickly, and be more intense, in low-storage systems. Also, in such systems, removal of the unsaturated zone by quarrying may have a significant impact on spring baseflow. Ogof Ffynnon Ddu, in the North Crop of the Carboniferous Limestone in South Wales is an example of such a system, having an extensive cave system and significant concentrated recharge via a large stream sink. Many minor springs in perched carbonate aquifers in the inter-bedded limestones and clastics of the Carboniferous Yoredale Series of Yorkshire are also in this category.

Group 3 aquifers have dispersed recharge, diffuse flow and low storage. Such systems are 'secondary or unproductive aquifers', and their development is thus likely to be less contentious than for the other groups. Perched springs in the Great Oolite of the Cotswolds, or in the Silurian limestones of Wenlock Edge may be in this category. Hobbs and Gunn (1998) also suggest that sub-valley limestone aquifers, such as those in the Carboniferous Limestone of the Ribble Valley (Yorkshire) are also of this type, but these may be rather better considered as Group 4 because of the high perennial storage potential below spring level.

Group 4 represents aquifers with diffuse flow, high storage and variable recharge. Such aquifers provide a significant groundwater resource, and impacts of any abstraction for dewatering may therefore be significant. They are often developed by boreholes rather than by spring abstraction. Examples include Pwllwy Spring, in the Carboniferous Limestone of South Wales, many (but not all) parts of the Chalk aquifer such as Great Givendale Springs (Pitman 1978).

Prediction of hydrogeological impacts in karst

For HIA, the critical issue is to determine whether or not conduit flow is occurring in the aquifer. Where conduit flow is present, most analytical equations and conventional groundwater modelling strategies are inappropriate and, if they are used, predictions of impacts will be highly uncertain. If conduit flow is not present, then more conventional techniques may still be applicable. It is worth emphasising again that, in the context of HIA, there should be a high burden of proof on a conclusion that conduit flow is *not* a feature of a groundwater system. The criteria that may be used to recognise aquifers that have conduit flow are as follows:

- Recharge to the aquifer occurs at discreet sink points.
- Hydrologically active caves are known from the area.
- Discharge from the aquifer is limited to a few discreet high-discharge springs.
- The rate of groundwater movement, determined by tracer tests, is high (Figure A3.5).
- Tracer detection in observation wells is focussed at specific sites, rather than forming a

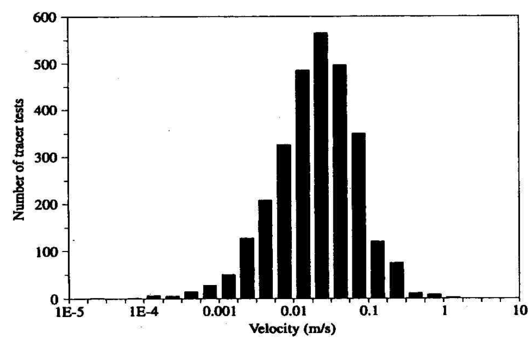

Figure A3.5 Velocities in conduits from 2,877 tracer tests in carbonate rocks between stream sinks and springs
(from Worthington *et al* 2000)

general breakthrough curve.

- Flow in the aquifer is turbulent, as indicated by the calculated Reynolds number, or transport of suspended sediment to the springs.
- Under baseflow conditions, linear troughs are present in the piezometric surface mapped from boreholes.
- Hydraulic gradients tend to decrease in a down-gradient direction in karst groundwater systems, whereas they tend to increase in non-karst (intergranular flow) systems. Such a pattern also implies a down-gradient increase in hydraulic conductivity.
- There is a non-linear relationship between spring discharge and water level observed in boreholes.
- There are abrupt changes in water quality at springs during recharge events.
- There are rapid changes in water levels in boreholes following rainfall (more indicative of concentrated recharge and conduit flow in unsaturated zone than in saturated zone).
- There are very large differences in the hydraulic conductivity determined at different scales within the aquifer.
- There may be marked differences between the isotopic and geochemical characteristics of water sampled from individual boreholes, and between these and springs.
- There is an anisotropic and heterogeneous response of observation boreholes to abstraction.
- There are non-linear relationships between drawdown in observation wells and the rate of abstraction from a pumping well.

It is important to recognise that with the possible exception of the results of tracer tests, none of these criteria provides an unequivocal indication of conduit flow behaviour, but where several of the criteria are met, the balance of interpretation should lie firmly in this direction. To understand further the nature and application of these criteria, the reader is referred to two papers that provide contrasting (karst and non-karstic) interpretations of the hydrology of the Smithville PCB spill site in the Silurian Dolomites of the Niagara Escarpment, Ontario, Canada (Worthington 2002 and Zanini *et al* 2000). There is also an interesting series of papers that debate the contribution of conduit flow in the very important Edwards Aquifer of Texas, USA (Halihan *et al* 2000; Mace and Hovorka 2000; Worthington 2002). The consequences of incorrectly accepting the non-karst model are graphically illustrated by problems associated with the construction of interceptor sewer tunnels in Milwaukee, Illinois (Rovey and Cherkauer 1994, Burke 2002; Day 2004).

Great care is needed when using information from boreholes. Water level and aquifer properties such as transmissivity determined from boreholes are unlikely to reflect conditions in the conduit flow part of the aquifer as the probability of a borehole intersecting a conduit is very low. Worthington (1999) estimates the probability is between 0.0037 and 0.075 (based on maps of ten extensive cave systems), but this probably represents an overestimate, as the surveys include dry passage no longer actively involved in groundwater flow, and the examples are drawn from areas known to be highly cavernous. Thus, data from boreholes are likely to be unrepresentative and unreliable (especially if used to develop and test numerical models). In contrast, springs in carbonate aquifers are the natural output points for the conduit network, and thus provide a sampling point indicative of its behaviour. In terms of aquifer contamination, they also integrate conditions over a large area, and are thus more useful as sampling points than boreholes, the catchments for which are poorly known (and usually exclude the conduit system).

Management of potential hydrogeological impacts in karst

'Monitor and mitigate'

Because of the difficulties in the reliable prediction of the hydrogeological impacts of groundwater abstraction from karst aquifers, it may not be possible to use the 14 steps of the main HIA methodology (introduced in Box 4.1). An alternative approach to the management of such impacts is required, which can be described as 'monitor and mitigate'. This can be summarised in terms of eight steps (Box A3.1), which is effectively a revised HIA methodology for karst. These steps will now be discussed in turn.

Box A3.1: HIA methodology for karst

Step K1: Establish the regional water resource status.

Step K2: Develop a conceptual model for the abstraction and the surrounding area.

Step K3: Identify sensitive sites.

Step K4: Commence preliminary monitoring at those sites.

Step K5: Design and demonstrate effective mitigation measures for the sensitive sites.

Step K6: Specify trigger levels for the mitigation measures.

Step K7: Continue surveillance monitoring at the sensitive sites.

Step K8: If necessary, implement mitigation measures when trigger levels have been passed.

Step K1: Establish the regional water resource status

This step is exactly the same as Step 1 of the main HIA methodology. It is still important to establish the water resource status from CAMS or the WFD, because it provides the context for the remaining steps.

Step K2: Develop a conceptual model for the abstraction and surrounding area

The overall approach to developing a conceptual model in karst is no different to other types of aquifer. Ideally, a conceptual model for a karstic groundwater system should include the same components as expected for a non-karstic groundwater system, as detailed in Step 2 of the main HIA methodology. It should also include the following additions, where possible:

- The location, dimensions and character of karst-related features, for example, closed depressions and dry valleys, in the topography.
- A description of the fracture/fissure/conduit network where it is accessible. For instance, spacing, aperture, orientation, morphology, sediment fill, groundwater flow status.

- The locations, flow rates (and dynamics) and character of discrete recharge features such as swallow holes.
- The location, depth, morphology and hydrological functioning of any epikarst.
- A description of the recharge process(es), including a quantitative estimate of the importance of diffuse and concentrated recharge.
- The locations, flow rates (and dynamics) and character of discrete discharge features such as springs. For example, how do the springs respond to rainfall, and what is the annual baseflow discharge profile of the springs?
- A description of proven hydraulic connections, for example through conduit or fissure systems, between specific locations or features.
- Information on the three-dimensional course, dimensions and hydraulic properties of connecting features.
- A summary description of groundwater flow processes, including a quantitative estimate of the relative importance of higher velocity flow in fissures and conduits, and lower velocity flow in small aperture fissures and the rock matrix.
- An estimate of the porosity and specific yield of the rock matrix.
- An estimate of the porosity contributed to the bulk aquifer by any interconnected fissure or conduit system.
- A description of the annual and longer-term storage dynamics of the groundwater system. For example, volumes of accessible storage during high and low groundwater level conditions.

If karst features are identified, an inventory (database or spreadsheet), ideally linked to a GIS for production of annotated maps, should be developed.

Under this step, the following three-stage process is recommended for identification and characterisation of a karst groundwater system:

i. Consideration of generic information, from the literature, relating to the type and scale of karst features that develop in specific geological formations and situations around the country. Awareness of this information in relation to the geological formation in question at a particular site will provide a good starting point for the development of a conceptual model.

ii. Desk study involving inspection of a range of materials, including maps, literature and databases, which can provide specific information about karst features at, or in the vicinity of, the site in question.

iii. Field investigations, which can be used to confirm and extend the understanding of the groundwater system at a site. Investigation techniques include field surveys, groundwater tracing, downhole geophysical logging and test pumping, continuous groundwater level and/or spring discharge monitoring, continuous water quality monitoring, and geophysics.

These stages of investigation will provide a basic conceptual model of the groundwater system, from which it should be possible to place the system within the carbonate aquifer classification described earlier, and therefore to assess whether groundwater flow in conduits is occurring. In turn, it should be possible to identify the type and nature of hydrogeological impacts that could be caused by the proposed groundwater abstraction. Using this information, it can be decided whether impact prediction using the main HIA methodology is possible, or whether the 'monitor and mitigate' approach should be used. Further field investigations and analysis could be necessary in order to improve the conceptual model of the system at any stage during this process. Further information on this three-stage process is given in a separate section below, after the remaining karst HIA steps have been described.

Step K3: Identify sensitive sites

This step is equivalent to Steps 3 and 8 of the main HIA methodology, where water features susceptible to flow and drawdown impacts are identified. Possible features include springs, rivers, lakes, wetlands, other abstractions and protected rights. Defining the search area for such features in karst is very difficult, for all the reasons of unpredictability already discussed. This step will have to be guided by the conceptual model and previous experience in the area, bearing in mind that in karst, impacts can manifest themselves over relatively long distances without impacts necessarily being seen closer to the abstraction. Potential water quality impacts, as outlined in Step 12 of the main HIA methodology, should also be borne in mind when identifying sensitive sites.

Step K4: Commence preliminary monitoring at those sites

Once sensitive sites have been identified and agreed with the Environment Agency, it is important to commence monitoring at the earliest opportunity. The availability of initial monitoring data is essential for the agreement of trigger levels, and several years of data are usually needed to give an indication of the effects of inter-annual variation in hydrological conditions.

It is important to ensure that the frequency of monitoring is adequate to document the short-term changes in conditions that may occur in karst aquifers (Quinlan *et al* 1991), and 15 to 30-minute intervals may be necessary. In the case of water quality monitoring at karst springs, similar sample intervals will probably prove necessary. Such high sample frequencies can create substantial problems in data display and archiving over the long term, unless this aspect has been anticipated.

It is also important to ensure that the frequency of monitoring is adequate to define critical values, such as minimum water levels, with an acceptable degree of precision. In the case of rest water level data from boreholes in UK carbonate aquifers, biweekly monitoring appears to provide an optimum balance between data requirements and staff costs for manual monitoring (Smart *et al* in prep), although automated monitoring with much higher frequency is of course preferable.

See Section 5 of the main report for a detailed discussion of monitoring in general.

Step K5: Design and demonstrate effective mitigation measures for the sensitive sites

Various aspects of the design and implementation of mitigation measures are presented under Steps 5, 10 and 13 of the main HIA methodology, and in Section 5.5 of the main report, and the same points apply to karst. Unfortunately, there has been relatively little formal evaluation of mitigation measures in karst aquifers. Often, mitigation schemes have been developed and evaluated on an informal *ad hoc* basis, rather than being formally proposed and tested. The practicability and effectiveness of mitigation is perhaps the most significant element of uncertainty in the monitor-and-mitigate scheme, and is the specific rationale for the requirement that the success of any mitigation scheme should be *demonstrable*. The development of theoretical schemes whose practicability and effectiveness have not been demonstrated is thus not acceptable.

In some cases the objective of mitigation may be the maintenance of groundwater levels, for instance beneath a sensitive wetland, through groundwater recharge. However, artificial recharge of karstified limestones is difficult. Whilst high rates of point recharge can be achieved at swallets, providing a direct way of maintaining flow at the associated spring, the injected water may not replenish the diffuse flow zone. Direct recharge to this zone is problematic; injection boreholes frequently have low capacity in massive karstified limestones, and may also suffer from sealing if there is inadequate control of suspended sediment in recharge waters. The use of extended linear features such as French drains or trenches that distribute the applied water, and may also penetrate the subsoil epikarst aquifer linking to transmissive flow paths, may prove more successful and more robust. Similar effects may be achieved by recharge to the course of losing streams, which frequently have good connectivity with the subsurface.

The effectiveness of any recharge scheme is greatly increased if there are hydraulic barriers between the site of recharge and the dewatering abstraction (to reduce recirculation). Given the potential difficulties in selection of recharge sites, it is imperative that the viability of proposed recharge mitigation is demonstrated, both in terms of long-term capacity and effectiveness in maintaining water levels at the site to be protected. This work must be undertaken prior to mitigation becoming necessary (and in some instances before any authorisation is granted by the Environment Agency).

In some cases, measures have been taken to restrict groundwater inflow to mineral workings, and thus retain surface flows and regional groundwater levels. Such measures include the sealing of influent stream beds, for instance by lining with concrete or puddling with clay (practices widely adopted on Coal Measures streams affected by sub-surface dewatering in the Forest of Dean, Aldous *et al* 1986), and the development of grout curtains. The latter are very expensive, and in karstified aquifers their utility is usually uncertain (see for example, Lolcama *et al* 2002). They are therefore unlikely to be generally adopted in mitigation.

The mitigation measure that is ideal from a hydrogeological or hydrological point of view may not always be possible in practice, with land ownership and access being the main constraints. For example, the ideal mitigation measure may be to construct a recharge trench immediately adjacent to a sensitive wetland, but the necessary permission may not be obtainable from the landowner.

One final issue to address is the replacement of storage lost by removal of the unsaturated zone. The commonly accepted specific yield of a Carboniferous Limestone aquifer is 1 per cent. However, some studies suggest that up to 50 per cent of this apparent yield is derived from delayed drainage from the unsaturated zone rather than from head reduction in the saturated zone (Smart and Friederich 1986). A solution that has been adopted in the East Mendip area is the provision within the quarry development of additional storage equivalent to that lost from the unsaturated (and in some cases saturated) zone, by provision of a water table sump pond. There has been no formal evaluation of the performance of such ‘balancing ponds’, but the relative ease of specification of their design volume, and their ready incorporation into development schemes, suggests that the adoption of this scheme is neither problematic nor prejudicial.

Step K6: Specify trigger levels for the mitigation measures

Trigger levels form the link between monitoring and mitigation. Trigger levels can be defined using a number of different hydrological variables, such as rest water levels in observation boreholes or streamflows, and employ a variety of statistical parameters such as minimum and maximum annual water levels or a magnitude-frequency descriptor such as the 95-percentile flow frequency. Previous experience (Dudgeon 1997) suggests that maximum water levels are a better indication of derogation than minimum levels, the latter tending to be more dependent on summer effective rainfall (Smart and Jones, in prep).

It is also normal to take some account of particular conditions. Abnormally dry years can be excluded when assessing requirements for mitigation; for example, dry years being defined as total annual rainfall less than or equal to 95 per cent of the long-term average, or total rainfall less than 50 per cent of the long term average for that month in at least three months of the year, two of which are consecutive. Such exclusions may not however be appropriate if particularly sensitive sites such as wetlands are subject to mitigation, or where short-term (quarterly or semi-annual) reporting is required.

Three different approaches for deciding whether mitigation is required are as follows:

i. *Comparison with a 'control' site:* Real-time monitoring data from an appropriate control site can be compared with that from the monitoring site(s) for assessment of impacts. The advantage of such an approach is that it can permit the effects of inter-annual climatic variability and systematic climate change to be accounted for in the assessment of impacts. The major problem with this approach is that it is usually difficult to find a control site that has similar behaviour to that of the pre-development monitoring site. Figure A3.6 shows a control site (Chantry) responding differently to active recharge and sustained natural baseflow (Smart and Jones, in prep). The scale of this problem is demonstrated by the fact that only 40 per cent of monitoring boreholes in the East Mendips exhibited statistically significant correlations with a nominated unaffected reference borehole. In adopting this approach, it will therefore be necessary to demonstrate that the behaviours at the control site and the monitoring site are reasonably correlated. Uncertainties in such correlations should also be propagated to give confidence intervals (depicted as error bars, for example) for identification of impacts. It is also important that the chosen reference site is not itself affected by the development. Given the considerable uncertainties in prediction of the extent of impacts from dewatering in karstified limestones, this may pose a problem.

ii. *Identification of impacts through statistical analysis:* Change can be detected statistically within a single time series whose nature is defined prior to development. A minimum of three years pre-development monitoring is recommended, although eight years is considered to provide a more robust indication of inter-annual variability. A number of statistical techniques are available to synthetically extend such monitoring data, and more importantly to account for the effects of inter-annual variations in effective precipitation (Knotters and van Walsum 1997). The extended data are then compared with real-time monitoring data to assess impacts. It is also necessary to employ statistical testing to determine the onset of change in hydrological series, that is, impacts (see review by Kundzewicz and Robson 2004, and associated papers in this special volume). To date, such techniques have received limited use in monitoring the impacts of dewatering, but in future could form the core of any decision-making system.

iii. *Comparison with a predictive model:* Impacts can be detected by comparing monitoring results with outputs from predictive models developed using pre-development monitoring data. Statistical techniques such as multiple regression can be used to relate key properties of the monitoring series (such as annual minimum water level) with a range of potential predictive variables such as mean annual effective rainfall or summer rainfall. This approach has, for instance, proved effective in wetlands (De Castro Ochoa and Munoz-Reinoso 1997). Again, in assessing exceedance of trigger levels, the errors in the predictive models should be considered. Some of these errors might be quite large as explained variance is typically between 50 and 80 per cent for simple bivariate linear least squares regressions. A more significant problem is that for many monitoring boreholes, it is difficult to develop simple predictive models. Using monitoring data from the Mendip Hills, simple predictive equations could be developed for only 25 per cent of boreholes for maximum water level, and 35 per cent for minimum water levels (Smart and Jones, in prep).

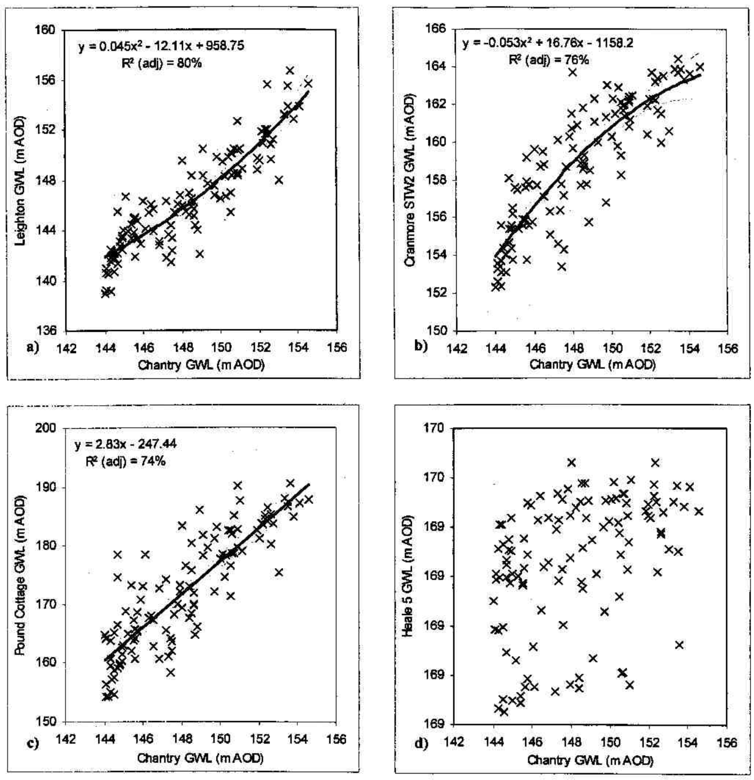

Figure A3.6 Comparing monitoring sites with a control site
(from Smart *et al.* in preparation)

The decision on how to set the trigger levels will need to be taken on a case-by-case basis, in discussion with the Environment Agency. See Section 5.6 of the main report for further discussion of trigger levels.

Step K7: Continue surveillance monitoring at the sensitive sites

Failure to comply with the requirements for suitable high quality monitoring may have substantial implications for operators adopting the monitor-and-mitigate scheme. If trigger levels are apparently exceeded due to failures in monitoring, costly mitigation operations may be started that could be avoided. Conversely, if trigger levels are apparently not exceeded, but adverse effects occur, remedial damages may be awarded. Finally, if there is a demonstrable failure to comply with requirements of the development permission, operations may be compulsorily halted. There are therefore considerable commercial incentives to ensure that monitoring is undertaken to the highest standards.

Step K8: If necessary, implement mitigation measures when trigger levels have been passed

If previous steps have been addressed satisfactorily, this step should be self-explanatory. Note that it will also be necessary to agree with the Environment Agency the procedure for stopping the mitigation, in other words, for recognising when conditions have improved sufficiently for the mitigation to be no longer needed. This will usually be a seasonal 'improvement' in conditions as a result of rainfall and recharge, but may also be a longer term improvement in response to operational changes to the dewatering system. This step also implies continued monitoring, in order to judge the long-term effectiveness of the mitigation measures, bearing in mind issues such as net gain (see Box 4.3 in the main report).

Final comments on HIA steps

It is not intended that the eight steps described here should compete or clash with the 14 steps of the main HIA methodology. Rather, they represent a recognition that in the case of karst aquifers, the predictive elements of the HIA methodology (in particular Steps 4 and 9) may not be feasible, for the reasons laid out in this appendix. The default approach should always be to follow the main HIA methodology, but if the problems posed by the karstic nature of the aquifer are just too great, then these eight steps are an alternative approach. They should be treated in exactly the same way as for the main HIA methodology, that is: iteratively; not prescriptively but with flexibility based on professional judgement; and as part of a tiered risk-based approach.

A summary of the way in which the eight karst steps relate to the 14 steps of the main HIA methodology is given in Box A3.2.

The application of HIA to a quarry in karstic limestone is illustrated by Case Study 3 in Appendix 9.

Box A3.2: Relationship between main HIA methodology and karst HIA

The main HIA methodology

Step 1: Establish the regional water resource status.

Step 2: Develop a conceptual model for the abstraction and the surrounding area.

Step 3: Based on the conceptual model, identify all potential water features which are susceptible to flow impacts.

Step 4: Apportion the likely flow impacts to the water features, again based on the conceptual model.

Step 5: For the relevant water features, allow for the mitigating effects of any discharges associated with the abstraction, to arrive at net flow impacts.

Step 6: Assess the significance of the net flow impacts.

Step 7: Define the search area for drawdown impacts.

Step 8: Identify all the features within the search area which could potentially be impacted by drawdown.

Step 9: For all these features, predict the likely drawdown impacts.

Step 10: For the relevant water features, allow for the effects of any measures being taken to mitigate the drawdown impacts.

Step 11: Assess the significance of the net drawdown impacts.

Step 12: Assess the water quality impacts.

Step 13: If necessary, redesign the mitigation measures to minimise the flow and drawdown impacts.

Step 14: Develop a monitoring strategy, focussing on the features likely to experience flow or drawdown impacts.

The karst HIA methodology

Step K1: Establish the regional water resource status.

Step K2: Develop a conceptual model for the abstraction and the surrounding area.

Step K3: Identify sensitive sites.

Step K4: Commence preliminary monitoring at those sites.

Step K5: Design and demonstrate effective mitigation measures for the sensitive sites.

Step K6: Specify trigger levels for the mitigation measures.

Step K7: Continue surveillance monitoring at the sensitive sites.

Step K8: If necessary, implement mitigation measures when trigger levels have been passed.

Characterisation of karst groundwater systems

It was recommended above (under Step K2) that a three-stage process be used for the identification and characterisation of karst groundwater systems during conceptual model development. Some flesh will now be put on the bones of the three stages.

Stage 1: Assess generic karst potential and form as a function of geological formation

There now follows a brief review of the type and nature of karst development that is found in the more important karst-prone geological formations in the UK. In alerting the investigator to the types of feature and function that could exist in a particular geological formation, it provides a useful starting point for development of a conceptual model of a karst groundwater system. Within the UK, karst may be developed in any geological formation that includes carbonate and evaporite minerals. The main locations of karst-prone rocks in England and Wales are illustrated in Figure 2.5 in the main report, repeated here as Figure A3.7 For convenience.

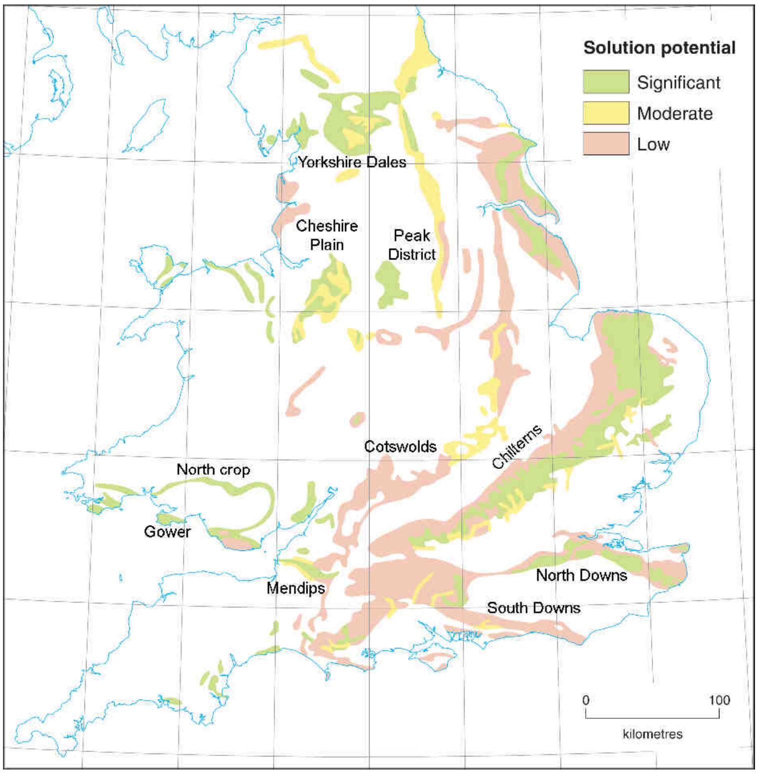

Figure A3.7 Main locations of karst-prone rocks in England and Wales
(adapted from GeoSure information, British Geological Survey)

The most extensive mature karst in the UK is developed in the *Dinantian Carboniferous Limestone* that is present in (Gunn 1994; Waltham *et al* 1997): the Mendip Hills and the Bristol area in south-west England; the north and south crops of the South Wales Coalfield, including the Forest of Dean; the Clwyd Hills of North Wales; the Peak District of Derbyshire; and the Yorkshire Dales. Similarly, well-developed karst is also found in the scattered outcrops of *Devonian Limestone* in south-west England, and in the *Cambro-Ordovician Durness Carbonates* of north-west Scotland. Both the Carboniferous and Devonian Limestones are extensively quarried, and provide a major source of aggregate within the UK (23 per cent). The Carboniferous Limestone is also a locally important aquifer (about 2 per cent of UK public water supply), being extensively developed for supply in the Mendip Hills of Somerset, in South Wales and in Derbyshire.

Karstification has also occurred within the extensive *Jurassic Limestones*, which crop out in a broad band from east Yorkshire in the north through the Midlands to Dorset in the south, and in the Chalk of south-east England. The latter is the most extensive carbonate formation within the UK, supplying about 50 per cent of UK groundwater abstraction. Small areas of *Silurian limestones* crop out in the western Midlands and Welsh Borders, but karst development is limited. Finally, in north-east England the *Permian Magnesian Limestone* is a locally significant aquifer that has well-developed surficial karst features and some caves (Cooper 1986). The Permian formations also include evaporites, predominantly *anhydrite* and *gypsum*, which are present as far south as Nottingham, and well-developed karst is present in the vicinity of Ripon. *Halite* (rock salt) is also extensively present in the Triassic of the Cheshire basin, and a subdued salt karst is present (Waltham, 1989).

There are important regional differences in the style and extent of karst development within each of these formations:

Carboniferous and Devonian Limestones

Flow in the Carboniferous and Devonian Limestones tends to be predominantly in conduits because of: 1) their massive, well-bedded nature; 2) their frequent structural association with adjacent or overlying clastic beds that feed allogenic streams (termed border karst); and 3) their very low matrix porosity. Thus, for instance, Atkinson (1977) reported for the Cheddar catchment that whilst 99 per cent of the storage supplying base-flow recession was derived from the diffuse flow part of the aquifer (predominantly in fissures), 60-80 per cent of the flow was focussed in conduits.

There are, however, substantial differences in conduit density between different areas, and even between different parts of the same aquifer. As already indicated, where topographically elevated clastic terrain adjacent to the limestones is absent, and where they are not overlain by clastic rocks or impermeable regolith such as till, the size and density of conduits is typically much reduced. A good example of this is Broadfield Down, which is a Carboniferous Limestone anticline immediately south of Bristol. Unlike the nearby karst of the Mendip Hills, it has no adjacent exposure of the Old Red Sandstone to provide an allogenic catchment. Such aquifers are predominant in much of the Peak District away from the Edale Shale and Millstone Grit margins. Conduit development is also less prominent when the limestones have only recently been exposed, and the period for karst development is therefore limited. For instance, there is a marked contrast in the maturity of conduit development between the west and east Mendips, the latter having only recently been stripped of its Mesozoic cover.

In many areas the Carboniferous Limestone extends to depth below the main rivers and is often also confined. There is substantial groundwater circulation in such situations, but this is generally through more diffuse openings, rather than conduits. Such deep circulation is often important as it is thermal in character, and feeds springs used for therapeutic spas and tourism such as at Bath Hot Springs and Buxton Spa.

The geometry of conduit networks in the Carboniferous Limestone shows a strong dependence on the geological structure. Cave development occurs preferentially along bedding planes rather than joints and faults, because the former are often laterally continuous and therefore offer the potential to host transmissive routes. Bedding planes may also represent breaks in the original deposition of carbonate sediments that offer favourable geochemical conditions for subsequent dissolution. For example, bedding planes may represent: 1) more porous paleokarst surfaces; 2) soils that host pyrite capable of oxidising to sulphuric acids; or 3) readily soluble evaporites such as gypsum. Such surfaces are termed inception horizons by Lowe (2000), and are often of prime importance in mapping out the initial layout of more transmissive openings that will subsequently develop into conduits.

In the Yorkshire Dales, the limestones are predominantly flat lying. Recharge enters near the top of the limestone via joints that lead downward via vertical shafts into near horizontal dendritic conduits, which in turn discharge to springs at or near the base of the limestone (Figure A3.8). In contrast, in dipping limestones such as those of the Mendip Hills, recharge occurs via bedding planes exposed at the truncated surface of the dipping beds. The points of discharge are, however, typically at the upper surface of the limestones, and therefore water that flows downwards in conduits developed along bedding planes must flow upwards through the stratigraphy (and across bedding planes) to discharge. Under phreatic conditions, pressure is sufficient to drive water upwards via 'riser' tubes developed on faults and joints (Figure A3.9). With increasing maturity, the amplitude of the resulting groundwater flow loops declines as the conduit can develop more efficient routes via the increasingly dense network of enlarged fractures in the diffuse flow zone (Ford 1999).

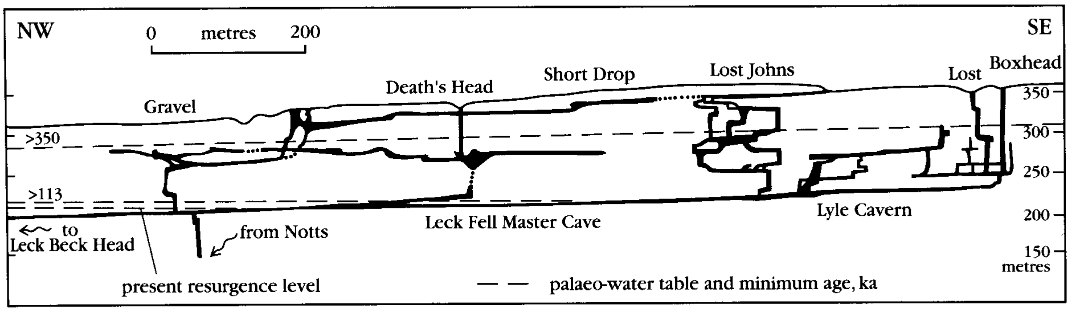

Figure A3.8 Cross-section through karst in the Yorkshire Dales
(from Waltham *et al* 1997)

A particular feature of many limestone aquifers that is well exemplified by the Carboniferous Limestone is the development of significant storage in the unsaturated zone in the epikarst aquifer. The importance of such storage can be readily appreciated in caves, where many perennial drips are observed, even at quite shallow depth, despite strong seasonal variations in the availability of soil water and recharge. Estimates for the Cheddar catchment in the Mendip Hills indicate that up to 50 per cent of the baseflow storage in the limestone may be within the unsaturated zone (Smart and Friederich 1986).

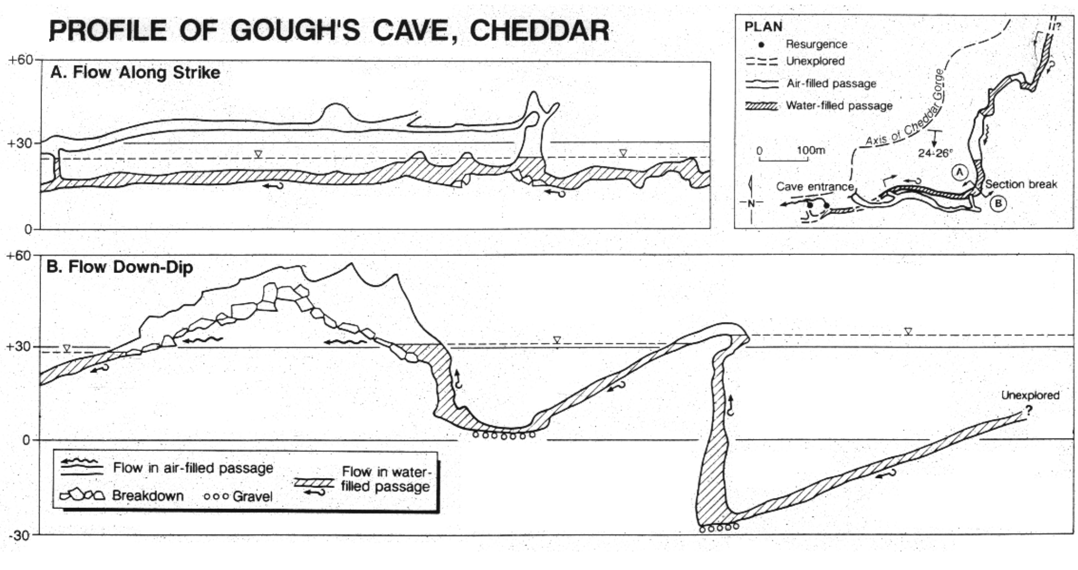

Figure A3.9 Riser tubes developed on faults and joints

The second important aspect of the epikarst aquifer is its role in the focussing of shallow recharge into transmissive conduits (Gunn 1981). Below the soil within the unsaturated zone, there is a general decline of permeability with depth as fewer fractures are opened by unloading, and there is a general decline in the amount of dissolution with depth as the acidity generated in the soil is consumed by reaction with the limestone (Williams 1983). At low rates of recharge, the limited capacity fractures that penetrate to greater depth drain residual shallow storage. As rates of recharge increase, the capacity of these routes is exceeded, and a shallow perched saturated zone develops. Where slopes are steep this may conduct water downhill, and sometimes ephemeral springs develop at the slope base. In the flatter terrain typical of many UK Carboniferous Limestone areas, lateral water movement will be towards higher capacity routes through the unsaturated zone. These routes, which may form initially at joint intersections or where joints have been subject to dilation, are preferentially enlarged by the focussed discharge of aggressive soil water to form vertical conduits or shafts. A local centripetal drainage system centred on these shafts then develops in the epikarst, with more dissolution in the centre and less around the margins. This results in the formation of a closed depression, which in turn serves to concentrate recharge (Williams 1985).

The presence of closed depressions is thus a useful indicator of maturity in karst development. It also infers a high degree of connectivity between cave conduits that feed the springs at depth, and the near surface (Figure A3.1). For this reason, particular care is needed in interpreting the significance of closed depressions in relation to HIA. There is also considerable indirect evidence to suggest that autogenic recharge may be delivered to the saturated zone storage, not by slow percolation as in conventional models, but via shafts and conduits. The epikarst is capable of reacting rapidly to recharge, thus the head in conduits in the saturated zone that are fed from this source may exceed that in the adjacent diffuse flow zone, which will only respond when slow percolation increases. Water will therefore move from the conduit into diffuse storage ('Flood flow' in Figure A3.10). When the storm recharge has been dissipated, and head in the conduit reduces, conditions will return to the more normal pattern (diffuse>conduit head) with release of water from diffuse storage into the conduit.

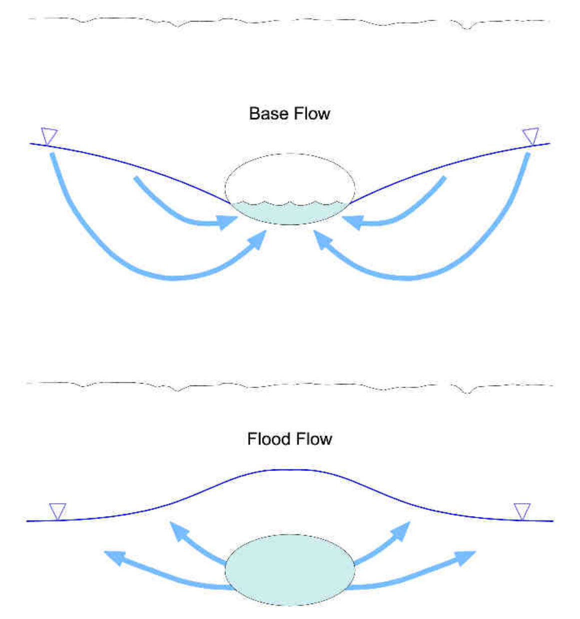

Figure A3.10 Diffuse and conduit heads

Whilst the northern part of the UK was glaciated during the last three glacial periods (the only ones for which there is extensive evidence), southern England was not directly affected by ice. This has resulted in marked contrasts between the karst terrain of northern England (for example, the glacially-scoured limestone pavements and over-deepened U-shaped glacial valleys of the Yorkshire Dales), and that of southern England (for example, the soil and regolith-mantled surface of the Chalk and Mendip Hills). Southern England did however suffer periods of intense periglacial activity, with cryoturbation and mass movement on the hill slopes, and intense run-off at times of snow-melt. These conditions, in association with periods when the ground was frozen and recharge was restricted, gave rise to the development of active surface run-off in valleys that are now largely dry (Smith 1975). Dry valleys are also characteristic of the Peak District, although here the lower parts of the valley networks remain active, as in the case of the River Wye.

Both periglacial and glacial conditions resulted in substantial changes to the pre-glacial karst landscapes, with mantling of the surface, valley infill and the truncation and blockage of earlier caves by sediment (Ford 1983). Thus, for example, much of the outcrop of the Permian limestones is mantled in glacial deposits, and recharge is limited. Lowering of the valley floor by glacial (and fluvial) erosion often resulted in initiation of a new level of cave development, and the abandonment of previous conduits. In many karst areas extensive networks of dry and often sediment-filled fossil passages are generally present.

Jurassic, Permian and Cretaceous (Chalk) Limestones

The Jurassic, Permian and Chalk limestone aquifers tend to be gently dipping and develop a marked scarp-and-dip terrain, which in the case of the Chalk forms elevated downlands. Down-dip they become confined, and can be very productive aquifers.

The major difference between the Carboniferous Limestones and the Jurassic, Permian or Cretaceous limestone aquifers is matrix porosity; the former have little or no matrix porosity but the latter have up to 40 per cent porosity. In the case of the Jurassic limestones, matrix (largely intergranular) porosity ranges up to 15 per cent, but in the Chalk, matrix porosity may reach values as high as 40 per cent. In the case of the Chalk, which is made up of very fine particles, the average size of matrix pores is very small, and they remain water-filled by capillary tension.

In all three aquifers, the bulk aquifer specific yield (1 to 3 per cent) is mainly contributed by fractures and fissures and is much lower than the matrix porosity. Thus, rather than providing useable storage, the matrix porosity is more significant in buffering geochemical changes by diffusive exchange between the mobile fracture water and the essentially static matrix water. However, in the Chalk in particular, some models allocate a large proportion of aquifer recharge to slow vertical piston displacement in the vadose zone towards the water table. Others suggest that recharge is predominantly via fractures, but that exchange between these and the matrix retards the downward progress of chemical changes.

Traditionally, none of these aquifers are considered karstified, but small caves are known in all three. In the case of the Jurassic and Permian limestones, these caves are often developed by mechanical processes associated with cambering and foundering. A few dissolutional caves are known from the Chalk, most notably Beachy Head Cave, but in the neighbouring Chalk of the Paris Basin much more extensive caves are known. Karst landforms such as closed depressions, dry valleys and stream sinks are also numerous and widespread (Farrant 2001).

Flow in all three aquifers is predominantly via dissolutionally-enlarged fractures, which are capable of supporting high abstraction rates at boreholes. There is extensive and unequivocal evidence from tracer tests that groundwater flow in the Chalk and Jurassic Limestones can be very rapid (Smart 1977 and 1995, MacDonald *et al* 1998). Tracer breakthrough curves suggest that the conduits may comprise a complex anastomosing network of dissolutionally enlarged fissures and tubes developed on bedding planes. Using travel time and discharge data, some authors have also modelled the conduits as single tubes whose diameters were of the order of 0.75 m (Atkinson and Smith 1974), comparable to that of Beachy Head Cave.

As for the Carboniferous Limestone, conduit flow is more likely to be significant in the Mesozoic limestone aquifers where concentrated recharge occurs from sinking streams. This is in fact quite common for all three formations:

- In the Great Oolite of the South Cotswolds, streams develop on the underlying Fullers Earth and run down-dip onto the limestones where they lose water and eventually sink completely (Smart 1977).
- In the Lincolnshire Limestone, large streams flow onto the outcrop from the west and significant recharge to the limestone is associated with both diffuse leakage and individual sinkholes in the stream beds. Local stream sinks also develop on the feather edge of the overlying impermeable deposits such as the Forest Marble, and there is significant concentrated recharge to the aquifer (Bradbury and Rushton 1998).
- The Chalk is capped by a variety of impermeable deposits including the Paleocene and the Clay-with-Flints over large areas. The majority of Chalk swallow holes are concentrated on the margins of these deposits (Farrant 2001), and many have been proven to support well-developed conduit flow to springs over distances as large as 20 km (MacDonald *et al* 1998).

Both the Chalk and the Permian limestones have significant numbers of closed depressions, and thus the potential for concentrated recharge, although lateral flow in the Chalk epikarst is likely to be much less important because of the high matrix porosity. Large numbers of sediment-filled pipes are however known from the Chalk suggesting that some focussing of recharge does occur.

Groundwater storage

Whilst both recharge and flow are controlled to a large extent by the nature of the host formation, in the case of storage it is the positional relationship between the carbonate and non-carbonate formations that is critical (Smart and Hobbs 1986). Where the base of the carbonate aquifer lies above the elevation of the main river valleys (the base–level for cave development), a permanent saturated zone is generally absent, and storage is limited to that in the unsaturated zone and any perched aquifers that may overlie the carbonates (right-hand element in Figure A3.11).

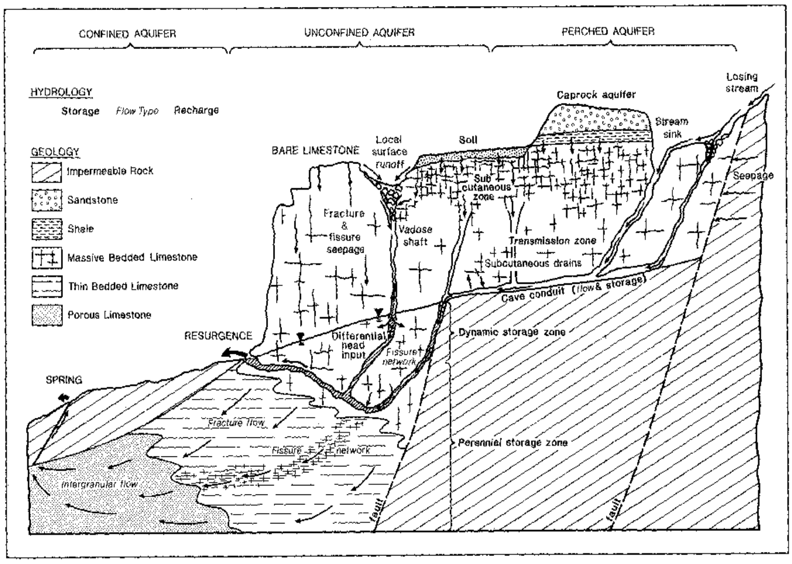

Figure A3.11 Groundwater storage zones in karst aquifers (Smart and Hobbs 1986)

The resurgence fed by White Scar Cave in the Carboniferous Limestone of the Yorkshire Dales is a classic example of this situation, with the irregular basal unconformity frequently revealed at the base of the incised cave passages (Waltham *et al* 1997).

Where the base of the aquifer extends down to the elevation of the lowest possible unconfined outlet, a zone of dynamic saturated storage will be formed, which will vary seasonally in magnitude. The volume of this zone (and any unsaturated storage can be estimated from integration of the volume under the baseflow recession curve. This situation is more characteristic of other springs in the Yorkshire Dales, such as Keld Head in Kingsdale, where much of the underwater conduit has been mapped by cave divers.

Finally, where the carbonate formation extends below the base-level, a zone of perennially-saturated storage may occur, and part of the aquifer may be confined, as is often the case for the Chalk and Jurassic limestone aquifers (left-hand part of Figure A3.11). Many of the Mendip Hills springs are of this kind, as are important abstractions in South Wales such as Schwyll Spring and the Pontnewynydd Springs down-dip of the Ogof Draenen cave system.

In some karst systems where there have been substantial changes in the boundary conditions of the aquifer, the development of this large volume of perennial storage has a major effect on reducing karstic behaviour. Examples of changes in boundary conditions include the infill of glacially over-deepened valleys (such as at Keld Head, Kingsdale in the Carboniferous Limestone of Yorkshire) or postglacial sea level rise in coastal karst aquifers (such as in the Severn Tunnel Great Spring in the Carboniferous Limestone of the Chepstow area). The travel time of tracer dye from the Cas Troggy sinks to the Severn Tunnel Great Spring is much longer (tens of days) than would be anticipated in a clearly conduit flow aquifer.

Stage 2: Desk study

Important resources for a desk study are listed below, with some comments on their advantages and limitations:

Hydrogeological maps: The British Geological Survey (BGS) publishes hydrogeological maps that cover most of the principal (major) and secondary (minor) aquifers of Britain. In combination with the background geological mapping on which these maps are based, the following mapped features are of relevance at a scoping level in deciding whether a groundwater system is karstic: course of intermittent streams; sinkholes; and proven underground hydraulic connections. Geological memoirs are also an obvious source of information on karst features.

Ordnance Survey mapping and other topographic surveys: The characteristic topographic features of karst terrain (closed depressions, dry valleys, sinking streams, etc), can be identified from Ordnance Survey mapping and other topographic surveys. The size of some types of feature, such as closed depressions, means that they can only be identified on a large-scale map or survey. Examples of map and topographic surveys include:

- Ordnance Survey ‘landscape and feature’ mapping at 1:50,000 scale (currently branded ‘Landranger’), 1:25,000 (Explorer) and 1:10,000 scales. All are available in hard-copy or digital format. The smaller-scale maps can be of limited use for detailed interpretation of landform features as the contour intervals are large (up to 10 m) and the maps can be quite ‘busy’ with other features.

- Ordnance Survey Landform PROFILE. This is a digital ground surface elevation dataset, available in grid-point or contour formats. Elevations are reported on a 10-m grid at a precision of +/- 5 m or better. The source of the dataset is contours surveyed at 1:10,000 scale.
- Ordnance Survey Landform PROFILE PLUS. Similar dataset to PROFILE, but elevations are reported on a grid as small as 2 m in urban and floodplain areas, at a precision down to +/- 0.5 m. The source of the dataset is LiDAR remote sensing. This dataset does not have national coverage at the time of writing.

Literature: Extensive research has been carried out on karst landforms and hydrogeology, and there is a significant amount of scientific literature relating to this work. There are a number of options for web-based literature searches. The British Cave Research Association (BCRA) publishes a number of regional karst memoirs and other books dealing with karst and caves. A publications list is available from the BCRA website (http://www.bcra.org.uk). The reference Ward *et al* (1998) contains an appendix summarising the method and results of tracer tests at 52 different sites in the UK (in Chalk and sandstone aquifers only).

Databases and other digital resources: The *Natural Cavities Database* (supporting report: Applied Geology Ltd 1993) was completed in 1994, and is not thought to have been updated or extended. It is held by the BGS, which can supply site-specific (GIS-compatible) information. The dataset has national coverage but the spatial coordinates of features are not very precise. At the time of writing, the BGS is producing a *Karst Geohazards database*. This includes data on stream sinks, springs (including non-karstic springs), caves, dissolution hollows and instances of building damage related to dissolution. Sources of information include field mapping, air photographs, internal reports, expert knowledge, caving club journals and magazines, the Chelsea Speleological Society Records. The database is also being checked against the National Cavities Database. GIS-compatible data from the database are available from the BGS under licence. There are currently 10,000 entries in the database, but it does not yet have national coverage. Coverage is restricted mainly to: Chalk areas of central southern England and the Chilterns, Dorset and part of the South Downs; Carboniferous Limestone - Mendip, Derbyshire and parts of South Wales; and Gypsum and Halite - North East England including Ripon, plus Cheshire, Stafford and Droitwich. Under its GeoSure service, BGS can also provide maps of karst-prone rocks, with solution potential rated as 'Significant', 'Moderate' or 'Low' (from which Figures 2.5 and A3.7 are derived).

Stage 3: Field investigations

Various field investigation methods are described below. They each tend to characterise specific elements of a karst groundwater system. For example, tracer tests yield information about large-scale, rapid fracture flow, whereas testing of boreholes, which are less likely to intercept any large-scale interconnected fracture network, will tend to yield information about flow in the rock matrix or in non-connected smaller-scale fractures. Hence, since a comprehensive understanding of the karst groundwater system in question is required, as wide a range of investigation techniques as possible should be employed. Field investigation techniques include the following:

Field inspection

Investigations under Stages 1 and 2 will provide general information on the nature of the aquifer and the regional disposition of inputs, transmission and output features. Field inspection of all major karst features, such as stream sinks and extensive cave systems, and as many minor features as possible, is necessary to confirm their nature and possible significance. Where mineral workings such as open quarries and underground mines are involved, an opportunity is afforded to see the three-dimensional subsurface structure of the aquifer in a manner that is rarely possible elsewhere. Remarkably, relatively little use has been made previously of this kind of opportunity, and experimental work has focussed on more conventional hydrological instrumentation. The field mapping approach is also advocated by Veni (1999), who provides useful methodological guidance and an example of how such work can inform further field investigation, for example, the use of mechanical excavation at sites to confirm the existence of underlying cave voids. An important consideration in carrying out any site inspections in mines, caves and quarries is that they should conform fully with all statutory safety requirements.

Field inspection often requires careful and systematic site survey. This may involve walking transects on level ground, with the transect spacing being sufficiently low that significant features will not be missed (Veni employed a 15-m interval for the Camp Bullis Military Training Installation site in Texas, USA). In quarries, rather more is learned from inspection of the exposed quarry faces, and a useful procedure is to walk each accessible bench. In some cases, however, features can also be mapped across the quarry floor. If possible, such surveys should document every significant karstic feature present (a tape recorder provides an efficient way of recording this information in the field, but can be somewhat slow to transcribe subsequently). The features should also be located on a map such as a current quarry plan, or fixed by survey or GPS (although the latter may not however provide particularly precise height control).

Details of the orientation, aperture and continuity of any specific karstic fissures or conduits should be obtained. It is important to distinguish between open voids and sediment-filled features as the latter may no longer be functional. Brief details of any fill (colour, texture, cementation, etc), and any distinguishing morphological features such as vadose or phreatic scallops and flow features should also be recorded. A photograph of each feature is also useful. In many cases it may be more useful to map general rather than individual features, for instance recording of the location of a zone with a high density of dissolutionally enlarged fractures and their general nature, rather than individual fractures.

Following the field survey, it is useful to develop a general classification for the features observed, such that a general overview of their nature, number and distribution in space and with depth below the surface can be described. It is also useful at this stage to indicate their possible function within the aquifer, and in particular to separate those features indicative of present-day karstic function and those which are paleokarst. Separation of the latter can often be achieved by consideration of the nature and origin of their sediment fills, for instance many paleokarst neptunian dykes in the Mendip Hills can be identified by their zoned sequence of Mesozoic sediment infills, and the presence of calcite mineralisation.

Sometimes, the results of general field surveys of karst features such as closed depressions, sinking streams and springs is at variance with detailed subsurface observations made in excavated voids, the latter indicating a fissure or fracture flow aquifer, whilst the former indicates a well-developed conduit flow system. There are several reasons this may be the case, for instance the quarry may be within a part of the aquifer not traversed by conduits, or has not yet penetrated to the depth of active conduit development. It is also important to recognise that even in quite large quarries, the scale of the sample may be insufficient to include the relatively infrequent but very transmissive conduits that control groundwater transmission.

Overall, field surveys are useful because they provide information that may allow a conceptual model of the aquifer structure to be developed. In combination with the recharge, flow and storage continua of Smart and Hobbs (1986), this may allow the aquifer to be placed within the overall classification system, and an indication of its likely sensitivity to abstraction to be obtained. In most cases, additional hydrometric characterisation will also be needed.

Groundwater tracers

Groundwater tracing is a widely-used technique for investigation and characterisation of karstic groundwater systems. A review of the theory and practice of groundwater tracing was provided by Ward *et al* (1998), and the following summary information is drawn partly from this document.

Tracing involves injection of a dye or other tracer into groundwater. The fate of the tracer, as demonstrated by the nature of its recovery downgradient from the injection point, can be interpreted to infer aquifer properties. The ideal tracer moves with the groundwater without altering the flow. It must be inexpensive, easily detected at trace concentrations and stable for the length of the experiment. Background levels of the tracer must be low, and the tracer should be easy and safe to handle and non-toxic to humans and the environment (Davis *et al* 1980). The most commonly used tracers in the UK are Rhodamine WT, Photine CU (optical brightener), Sodium Fluorescein and Bacteriophage T7 (host: *E.Coli* B).

Recovery of tracer during a test proves that there is a hydraulic connection between the injection and recovery sites, for example, swallet or borehole to spring. In contrast, non-recovery of a tracer does *not* prove that there is no hydraulic connection between the injection and recovery sites, as the tracer could have been either adsorbed onto aquifer materials or diluted beyond the detection limit. Analysis and interpretation of the tracer recovery concentration profile can provide information about karst aquifer properties, as follows:

- Based on the time after injection at which the peak tracer concentration occurs, the time of travel and apparent velocity of the dye in the aquifer can be calculated. For example, Worthington *et al* (2000) reported a mean velocity of 1,900 m/d for 2,877 sink-to-rising tests in karst aquifers in North America (test results shown in Figure A3.5).
- The amount of tracer recovered can be calculated by integrating under the tracer breakthrough curve and multiplying the result by the discharge at the recovery point. The result can then be expressed as a percentage of the injected tracer mass (that is, the percentage recovery). Interpretation of the percentage tracer recovery is somewhat uncertain as loss of tracer can be through adsorption within the aquifer or division of flowpaths.
- If discrete (conduit) flowpaths are under investigation, the saturated volume of the flowpath can be estimated by multiplying the time-of-travel by the discharge at the recovery point. In turn, and using the straight line distance between the injection and recovery sites, estimates of conduit dimensions can be derived.

Successful planning and execution of a tracer test is a significant challenge, although it is well within the capabilities of a professional scientist or engineer. Ward *et al* (1998) provide a comprehensive planning and execution methodology for tracer testing aimed at this audience, assuming they have no previous experience. There are also a number of organisations, such as consultants and university departments, to which tracing exercises can be sub-contracted. In addition to the obvious choices that must be made (type of tracer, injection and recovery sites, etc), attention must be paid to the following:

- The purpose of the tracer test and the methods to be used for analysing the results.
- Deciding the mass of tracer and the mode of injection. Worthington and Smart (2003) provide a method for determining tracer mass for swallet-to-spring tests. The mode of injection depends on the type of injection site. Techniques are available for injecting tracers evenly through the saturated depth of a borehole, and for injecting tracers into 'dry' closed depressions or sinkholes.
- Observation of injection and sampling protocols to avoid cross-contamination.

Borehole investigation and testing

When considering the use of borehole testing for characterisation of a karst groundwater system, it must be remembered that large-scale, solution-enhanced fissures or conduits always represent a very small proportion of rock volume in a karstic aquifer. For example, Worthington (2002) presents summary data on ten of the best-studied cave systems in the world and shows that the volume of the caves represents between 0.004 and 0.48 per cent of the total volume of the bedrock in which they are developed. Hence, the probability of a borehole intercepting a large-scale fissure or conduit is extremely low. The following techniques and observations involving boreholes can be used during an investigation to help in identification or characterisation of a karstic groundwater system:

- Downhole geophysical logging: 1) direct observation of any fissures intersecting a borehole using CCTV or other optical techniques; 2) calliper logging to identify fissures through their effect on bore geometry; 3) identification of significant groundwater flows up or down a borehole, most probably from and to fissures intersecting the borehole, through downhole flow logging.
- Determination of hydraulic conductivity at varying scales. In most types of aquifer, it is expected that measured values for hydraulic conductivity will increase as the volume of aquifer being tested increases. Hence, hydraulic conductivity measured by laboratory core tests will be lower than those measured by large-scale pumping tests. This is true in karst groundwater systems, but to a much greater extent (Figure A3.12; also, Kiraly 1975). The box plots in Figure A3.12 show median, upper and lower percentiles, range of distribution, and also points defined as statistical outliers (black dots).
- Multiple determinations of hydraulic conductivity over a network of boreholes. Karst aquifers will tend to exhibit a larger range in values for hydraulic conductivity or transmissivity than other aquifers. Figure A3.13 shows results of multiple slug tests for various types of limestone aquifer. The hydraulic conductivity values for the three limestones in 'natural' condition vary over at least two orders of magnitude, with those for the karstified Carboniferous Limestone varying over four orders of magnitude.

Values for a relatively uniform non-karstic aquifer might be expected to vary over one order of magnitude or less.

- Boreholes can be used as either injection or recovery points for tracer tests. The tests can use one, two or numerous boreholes, and can be under natural or imposed (through pumping) hydraulic gradients.
- Price (1994) details a method for assessment of the relative importance of fissure flow and intergranular flow, through pump testing using a downhole packer system to isolate specific vertical intervals of the aquifer.
- The spatial distribution of groundwater levels measured in boreholes can indicate heterogeneity in the distribution of hydraulic conductivity that is a characteristic of karstic aquifers. Typical examples include: 1) markedly non-symmetrical cones of depression during pumping tests (Palmer 1999); and 2) small-scale troughs in the water table resulting from groundwater levels being lower in aquifer conduits than in the surrounding rock.

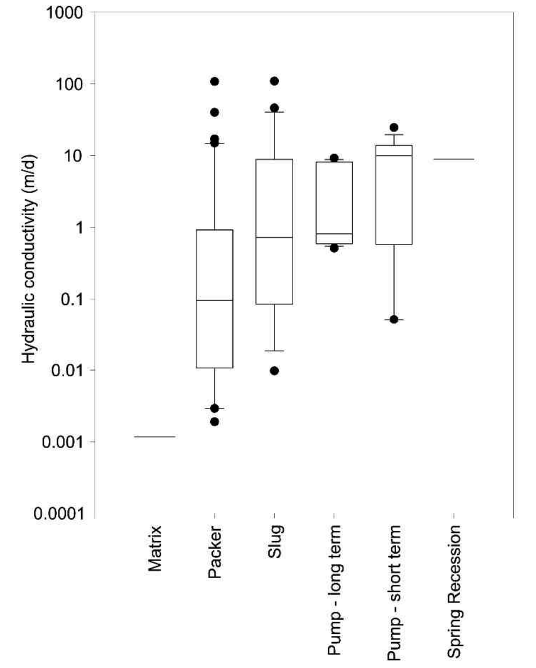

Figure A3.12 Effect of aquifer test scale on estimated hydraulic conductivity for Carboniferous Limestone aquifer of the Mendip Hills (Smart, unpublished)

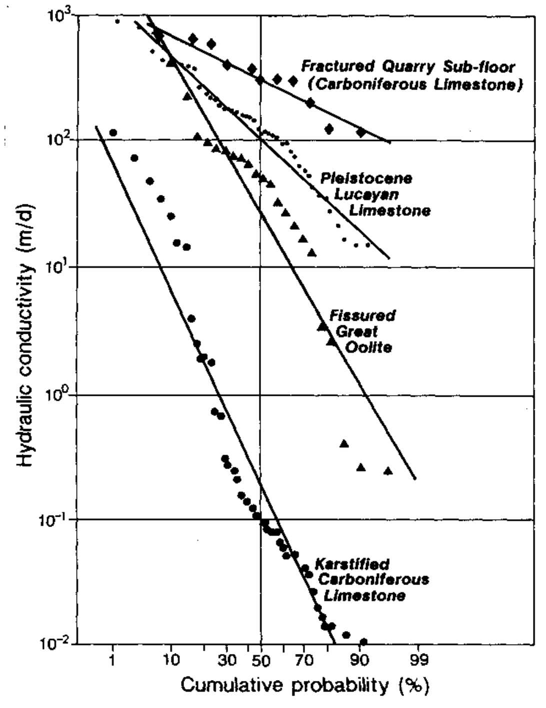

Figure A3.13 Ranges in hydraulic conductivity for different limestones (from Smart *et al* 1991)

Continuous groundwater level and/or spring discharge monitoring

Groundwater levels and spring discharges in karst groundwater systems can respond rapidly to rainfall because large-scale, interconnected fracture networks, in both the unsaturated and saturated zones, can transmit water quickly. In order to collect useful data in this regard, monitoring frequencies must be higher than the typical period of rainfall or recharge events, and continuous monitoring is most useful.

Continuous water quality monitoring

Recharge that moves through a karst aquifer rapidly, through an interconnected fracture network, has a much lower solute concentration than longer residence (matrix) water. Hence, a reduction in solute concentration in either a borehole or at a spring after a recharge event can be assumed to be an indicator of rapid karstic flow through the aquifer.

Geophysics

Microgravity surveying has been used with some success to detect sub-surface cavities in karst terrain, and is probably the most useful geophysical technique for this purpose. For example, McGrath (2003) used this technique extensively in South Wales and Ireland to detect subsurface cavities in karst terrain. In many cases, the existence of cavities detected through microgravity survey was verified by borehole drilling.

Subsurface resistivity surveying has been used to identify air-filled and water-filled cavities (very high and very low resistivity respectively) in karst terrain. Greenfield (1979) suggests, however, that cavities can only be detected directly where the depth:radius ratio is less than 0.5, and that any successful detections where the ratio is greater than this are most probably related to the effect of an underlying cavity on the moisture status of the overlying soil/rock.

Measurement of self-potential (SP), and in the current context streaming potentials induced by concentrated subsurface fluid flow, has been used to detect and locate cavities in karst terrain. Quarto and Schiavone (1996) showed that SP anomalies exist over air-filled cavities in sedimentary rocks, but found that results could be ambiguous and that the use of a suite of geophysical methods (see above) could reduce these ambiguities.

Geophysical survey techniques need to be applied intensively in order to gain useful results, and are therefore generally not cost-effective on a regional scale. They do, however, have the potential to provide information at a high spatial resolution over smaller areas identified by other, less intensive, field investigations.

Water balance calculations

A water balance calculation is an important test of a conceptual model of a karstic groundwater system. Accounting for groundwater inputs to and outputs from a system, either through estimation or measurement, can lead to the identification of sizeable unaccounted for groundwater fluxes. Depending on the geometry of the system, these unaccounted for fluxes could be accounted for by important point recharge or discharge features, or a large conduit or conduits in the aquifer, which have not been included in the conceptual model.

Numerical modelling of karst aquifers

This appendix finishes with a brief introduction to the problems associated with numerical modelling of karst aquifers. See also Appendix 8 for a general discussion of the issues surrounding numerical groundwater modelling of dewatering operations.

Modelling using conventional techniques

The simplest and most common approach adopted for the modelling of groundwater flow in carbonate aquifers has been to assume that the density of fractures is sufficient at the scale of modelling that the aquifer can be treated as an equivalent porous medium. Under this assumption, a system can be modelled using numerical schemes based on Darcy's Law, using a code such as the widely used MODFLOW package. Some success has been claimed for such an approach at the regional scale (Scanlon *et al* 2003), and where flow is predominantly diffuse (Teutsch 1990). In the latter case, the strong scale-dependence of hydraulic conductivity characteristic of carbonate aquifers may not extend to the largest (basin) scale, which would be dominated by the conduit system. An effective representative elementary volume (REV) may therefore be defined, the scale of which is suitable to populate the cells used to discretise the model domain (Teutsch and Sauter 1991). The measurement scale must be greater than heterogeneity length and less than model domain scale if measurements are to be useful for model parameterisation (Figure A3.14).

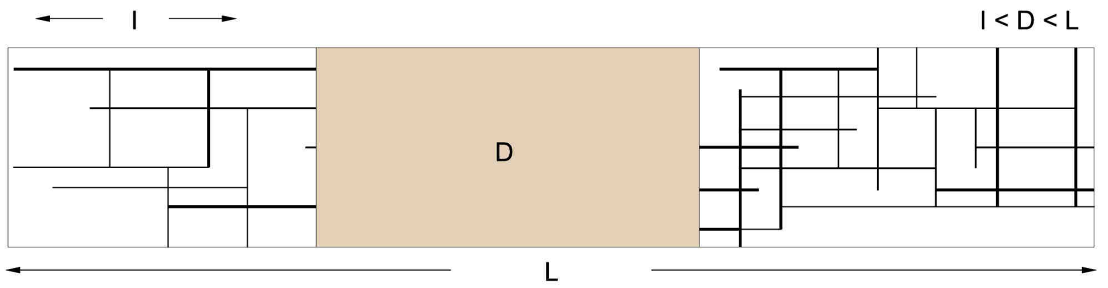

Figure A3.14 Schematic example of length scale hierarchy
(Teutsch and Sauter 1991) (l = heterogeneity; D = measurement scale; L = model domain scale)

Similarly, for regional-scale applications, the relatively large scale of the cells may be sufficient to include the most transmissive elements of the scale permeability hierarchy. In these situations, conventional distributed groundwater flow models may be useful for evaluation of the impacts of abstraction on regional groundwater flows. They cannot however provide more local information such as the detailed flow paths required for definition of well protection zones.

An alternative to a fully-distributed model is to employ a lumped-parameter model. In such models the aquifer is divided into a number of separate and generally sequential units that are treated as tanks (Figure A3.15). These units receive direct recharge in proportion to their area. Flow is permitted between adjacent cells, depending on head difference, aquifer cross-sectional area and hydraulic conductivity. The hydraulic conductivity and specific yield of each cell can also be specified as varying with depth, to introduce some non-linearity into the overall

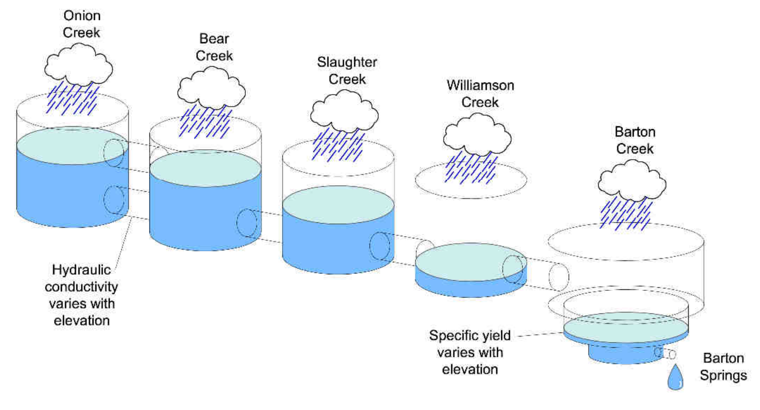

Figure A3.15 Schematic of lumped-parameter model cells and connections (Scanlon *et al* 2003) response of the system. Such models have a number of advantages over distributed models:

- They are intuitively accessible and easy to explain.
- They are simple to use and can be easily run using a spreadsheet.
- Results are obtained very rapidly, allowing easy calibration and testing of alternative model scenarios.
- They are highly parsimonious in terms of both system representation and data requirements.

Their major disadvantage is their inability to fully represent the potentiometric surface, and thus the regional pattern of groundwater flow. Their function is also strongly dependent on the representative borehole selected to characterise heads in each cell. Notwithstanding these limitations, lumped-parameter models can provide a useful initial insight into the possible effects of abstraction, and have been used in several modelling studies of dewatering. However, there are several compelling reasons why conventional groundwater flow models are unsuitable for simulation of the effects of abstraction in karst aquifers with well-developed conduit flow (Huntoon 1994; Quinlan *et al* 1996):

- The range of flow rates between matrix and conduit flow may span as much as 30 orders of magnitude, and even in the diffuse flow zone sampled by slug tests in boreholes, hydraulic conductivities typically span 5-6 orders of magnitude (Figure A3.13). Whilst the spatial averaging of these values required for application of an equivalent-porous-medium model may be possible, the use of this average may greatly underestimate the most rapid movement of groundwater and contaminants through the most transmissive (conduit) routes.
- Water level and transmissivity determined from boreholes are unlikely to reflect conditions in the conduit flow part of the aquifer as the probability of a borehole intersecting a conduit is very low. As mentioned earlier, Worthington (1999) estimates that the probability is between 0.0037 and 0.075, based on maps of ten extensive cave systems, but this probably represents an overestimate, as the surveys include dry passage no longer actively involved in groundwater flow, and the examples are drawn from areas known to be highly cavernous. Thus, both model parameterisation and testing using borehole data are likely to be unrepresentative and unreliable. In contrast, springs in carbonate aquifers are the output points for the conduit network, and thus provide a sampling point indicative of its behaviour. In terms of aquifer contamination, they also integrate conditions over a large area, and thus provide more useful sampling points than boreholes, the catchment for which is poorly known and excludes the conduit system.
- Boundary conditions in karst aquifers are often time dependent, whereas in conventional groundwater flow models they are typically invariant. Many karst aquifers show a complex dependence of head and discharge, with the development of overflow springs at high flow that supplement the underflow springs that provide the major groundwater outlet at low flow. There may also be switching of flow between adjacent catchments when critical water levels are reached and high level conduit segments begin to function (see the modelling of Jeannin 2001).
- There are very large changes in the effective properties of the aquifer over short distances (high heterogeneity), with the transition between conduit

and diffuse zones representing a major discontinuity in aquifer properties. Such discontinuities often result in numerical instabilities in groundwater flow models. Whilst these instabilities may be reduced by increasing the grid resolution and smoothing the transition, this graduated scheme is physically incorrect.

- In addition to the strong heterogeneity of karst aquifers, they are also often strongly anisotropic, with the preferred flow direction dictated by the structure of the dissolutional fissure and conduit network. Under these conditions, flow may not be in the same direction as that of the maximum hydraulic gradient, as is usually assumed in groundwater flow models, but may be sub-parallel to it (Palmer 1999).
- Flow in fissures and conduits is turbulent. Thus, the assumption of a laminar flow regime implicit in numerical groundwater flow models using Darcy's law is inappropriate. Specifically in karst, there will not be a linear change of flow with change in head as in conventional aquifers.

Modelling of turbulent groundwater flow in karst conduits

In the case of lumped-parameter models, turbulent flow between cells can be readily incorporated by replacement of Darcy's Law by a turbulent flow equation such as the Chézy-Manning equation. Barrett and Charbeneau (1997) modelled the Barton Springs segment of the Edwards aquifer using this approach, but reported that the model gave poor agreement with heads observed at representative boreholes in each model cell.

An alternative scheme has been developed by Dudgeon (1985) specifically for modelling inflow to open pits and quarries. The scheme solves the modified Dupuit equation for both laminar and turbulent radial inflow to a fully or partially penetrating sump in an unconfined aquifer. It also incorporates the development of a free seepage face in the wall above the void floor, a feature often observed in sub-water table excavations. The onset of turbulent flow in the region of steepened hydraulic gradient adjacent to the pit, and resulting increase in hydraulic gradient and seepage face height result in a substantial reduction in drawdown compared to that derived from conventional radial schemes with laminar flow and no seepage face. Calculated inflows are also lower.

The Dudgeon model has been successfully applied to the East End limestone quarry in Queensland, Australia. When the assumptions of radial inflow and aquifer isotropy were met, the model performed well, but as the quarry developed and became more rectangular the results required adjustment for the preferential development of the cone of depression along the strike.

In the case of aquifers with well-developed conduit flow, the Dudgeon model is inappropriate. Where the conduit is intersected by the quarry, both pumped abstraction and the extent of the regional drawdown may be grossly underestimated. Even where conduits are not directly intercepted, but rather act as recharge sources providing water by leakage either laterally or at depth in the aquifer, abstraction volumes may be substantially underestimated. In such situations, the calibration of the Dudgeon model using existing drawdown data may provide spurious predictions of the future abstraction and cone of depression, due to the failure of the basic assumptions of the model.

Modelling of both conduit and diffuse flow

An alternative to conventional groundwater flow models that employ a single hydraulic continuum are those that recognise the double-porosity nature of the aquifer. In such models, a network of laterally-continuous fractures provides a more transmissive route embedded in a lower-permeability fracture or matrix porosity component. The simplest approach is the double continuum porous equivalent (DCPE) in which both media are considered to extend throughout the model domain (Figure A3.16). Such a scheme has been implemented as part of the MODFLOW package as DP-MODFLOW (Lang *et al* 1992). The advantage of this approach is that no information is required as to the location and topology of the conductive fractures. However, the disadvantage is that the characteristics of the two continua and the exchanges between them must be obtained by calibration. Flow is also assumed to be Darcian.

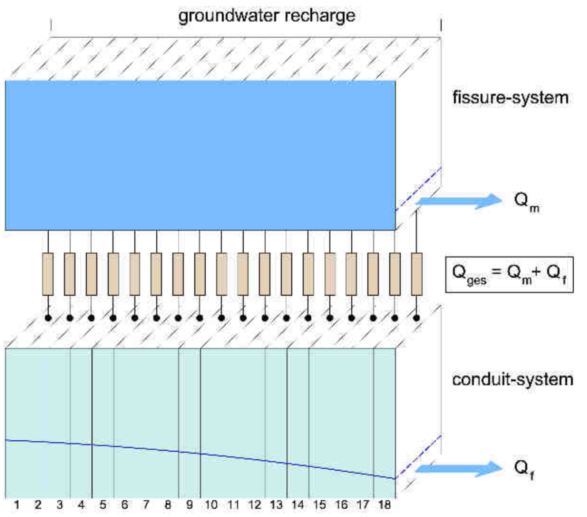

Figure A3.16 Schematic of DCPE model
(Teutsch 1990)

A more sophisticated approach is discrete fracture porous continuum (DFPC) models in which the position and nature of the conduit network is explicitly included in the model. In such models it is also possible to employ non-Darcian behaviour in the conduit network by adoption of the Darcy-Weisbach equation to describe flow (Liedel *et al* 2003). Such models provide the possibility of more realistic simulation of flow in karst aquifers, and unlike DCPE models, do not require calibration. However, the detailed specification of the layout of the conduit system is not generally possible, nor can it be easily predicted, and so these models are not generally applicable. However, Mohrlok and Teutsch (1997) have shown that DCPE models can encapsulate most of the behaviour exhibited in DFPC models, and therefore DCPE models remain the model most applicable to karst aquifers. Nevertheless, there remain significant issues in the modelling of flow in karst aquifers:

- Significant aspects of both conduit behaviour (especially under turbulent flow), and the interaction of conduit and diffuse flow are at present poorly understood, and inadequately described in the DCPE models.
- In the case of DFPC models, the nature of head loss within conduits is poorly understood. Rather than being a continuous property associated with skin friction, it may rather be irregular, with head reduction occurring where short segments of vadose flow occur (Figure A3.9), or where the conduit is restricted by sediment accumulation or breakdown.
- In karstified aquifers that have very low hydraulic gradients, local changes in the nature of the aquifer such as faults with a non-permeable gouge can control hydraulic gradient, rather than this being a response to continuous head loss in the diffuse flow zone. Such local effects may however be difficult to identify and incorporate into the diffuse flow component of either DCPE or DFPC models.

At present there is therefore no reliable numerical model that can be applied routinely to predict the impacts of groundwater abstraction in karstified aquifers with well-developed conduit flow. Whilst both lumped-parameter models and DCPE models may be of some utility, both require calibration, and cannot be relied upon to predict reliably future conditions, which may differ substantially from those used for calibration.

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Appendix 4: Crystalline rock

Introduction

This appendix, first referred to in Section 2.5 of the main report text, discusses the particular characteristics of fractured crystalline rocks that need to be taken into account when developing conceptual models and undertaking HIA of dewatering in such rocks.

Acknowledgement on authorship: the text and diagrams for this appendix were contributed by David Banks of Holymoor Consultancy, Chesterfield, UK.

Characteristics of crystalline rock aquifers

Crystalline rocks, such as slates, granites, marbles, basalt and dolerite are quarried for ornamental, building and roofing stone, for aggregate or for roadstone. These rocks are typically of very low hydraulic conductivity (except where fractures have been enhanced by dissolution or weathering; see Appendix 3). Porosity between mineral grains is typically low or negligible, such that the majority of groundwater storage and flow takes place through networks of fractures. Such rocks are especially typical of upland terrain in Wales, Scotland and northern England. In south-west England, crystalline rocks also occur, although the lack of recent glaciation has often allowed significant thicknesses of weathered material to develop in the near-surface environment. Fractures and fracture zones are also typically more deeply weathered. Thus, southern crystalline rock terrain may enjoy elevated hydraulic conductivities and storage characteristics compared with northern, glaciated terrain.

The following characteristics, peculiar to fractured, crystalline rock aquifers, should be borne in mind in making any assessment of the hydrological impacts of dewatering:

- The generally low hydraulic conductivity (K) means that groundwater inflows to excavations will *usually* (though not always) be low.
- The low K, and the typically elevated topography and rainfall of crystalline rock terrain in the UK mean that there is usually a high density of rejected recharge and surface water features. This implies that cones of depression will *usually* be limited in their development.
- Crystalline rock aquifers are typically *heterogeneous* and of *poor connectivity*. Borehole yields and hydraulic properties vary strongly within relatively short distances. Furthermore, two closely spaced boreholes or excavations may intercept two different fracture systems and thus experience very different yields and groundwater chemistries, and may be in poor hydraulic continuity.
- Crystalline rock aquifers may be *anisotropic.* In other words, they may have greater hydraulic conductivities in one direction (often corresponding with well-developed, open, connected fractures) and low conductivities in another.
- The low K and elevated topography often results in strong three-dimensional hydraulic head gradients and a significant vertical component to groundwater flow. Groundwater heads can increase or decrease by

several metres or tens of metres as the depth of an observation borehole increases. Thus, one must be very careful when designing groundwater observation networks to consider the depth of the response zone of the borehole.

Estimating groundwater inflow

Because fractured crystalline rocks often (though not always) have a low hydraulic conductivity, groundwater inflows to excavations will often be modest in size and will tend to be derived from a limited number of transmissive fractures rather than a diffuse seepage face. Run-in of surface water and rainfall, and losses due to evaporation, may be significant compared to groundwater inflow and will need to be quantified.

Although it may often be possible to visibly identify locations of groundwater inflow and even to measure the approximate groundwater influx, it may sometimes be difficult to distinguish 'true' groundwater from surface water or interflow. The following simple techniques can be used to distinguish these elements:

- *Temperature:* Groundwater will typically have a stable temperature of some 10-12℃ throughout the year. Surficial water tempe ratures will fluctuate daily and seasonally. Groundwater inflows may effectively be identified as inflows of 'warm' water during winter spells where the air temperature is <0℃ and surface water sources are frozen.
- *Chemistry:* Groundwater can often be distinguished from surficial waters on the basis of water chemistry. Crystalline rock groundwaters will often exhibit elevated pH and electrical conductivity, whereas surface waters tend to be relatively acidic and of low mineralisation in much upland, crystalline rock terrain. If analyses are available, groundwater inflows may be characterised by contents of sodium, calcium, alkalinity, fluoride or radon.

Estimating hydraulic conductivity of crystalline rocks

The hydraulic conductivity of crystalline rocks can vary over several orders of magnitude, depending on the fractures that are present in the rock mass. Considering a large enough volume of aquifer (a *representative elementary volume*), however, allows one to apply a value of bulk hydraulic conductivity to conceptual, analytical and numerical modelling approaches. Unfortunately, intensive studies of the properties of crystalline rock aquifers have often only been carried out in specific localities (such as nuclear repositories), and we must estimate hydraulic properties from proxy information such as borehole yield. In fact, it is common practice to take the specific capacity (S_C) of a drilled borehole as being approximately proportional to aquifer transmissivity, allowing hydraulic conductivity to be estimated. Banks (1992) argued that:

$T_a = {}^{S_C}\!/_{\alpha}$, where α is a constant of value about 0.9 and T_a = 'apparent' transmissivity.

Thereafter, $K = {}^{T_a}\!/_{D}$, where D is the saturated borehole depth.

If enough values of K (K_1, K_2, K_3 etc.) can be derived from boreholes in the aquifer, the bulk hydraulic conductivity (K_b) can be estimated as the geometric mean of the individual values:

$$K_b = \sqrt[n]{K_1 \times K_2 \times K_3 \times ..... \times K_n}$$

As hydraulic conductivities and borehole yields are approximately log-normally distributed, K_b can also be approximated by the median value of K.

In some British crystalline rock aquifers, enough borehole yield data may exist to allow meaningful estimations of K_b. Indeed, the Minor Aquifer Properties Manual (Jones *et al* 2000) may provide such values directly, particularly for units such as the Coal Measures or Millstone Grit which, although not true crystalline rock aquifers, share many similar properties. Alternatively, the Norwegian and Swedish Geological Surveys possess large data sets of borehole yields for many of the same Caledonian and Precambrian geological units that crop out in northern Britain and Wales (Banks *et al* 2005). For example, in Norway, the median borehole yield from all crystalline rock lithologies is found to be 600 (± 17) l/hour. In Norway the median borehole depth is 56 (± 0.58) m. Assuming that yields are measured at a near-maximum drawdown of, say, 40 m, the median yield corresponds to an apparent transmissivity of:

$$T_a = \frac{S_C}{0.9} = \frac{0.6}{0.9 \times 40}$$ = 0.017 m^2/hour = 0.4 m^2/d

Assuming the water table is at 5 m below ground level, the median saturated borehole depth is 51 m. This equates to a bulk hydraulic conductivity for Norwegian crystalline rock of:

$$K_b = \frac{T_a}{D} = \frac{0.4}{51}$$ = 0.008 m/d

Lithologically-based subsets of large borehole-yield data sets can be used to calculate specific values for, for example, Precambrian granites, Caledonian shales etc, although variation between lithologies is less than one might intuitively expect (Banks *et al* 2005).

The confidence interval on the median borehole yield will allow the calculation of the confidence interval on the bulk hydraulic conductivity; whereas the interquartile range within the borehole yield data set will provide an indication of the variability of hydraulic conductivity within an aquifer unit. This permits a probabilistic approach to estimating the magnitude of likely impacts from a dewatering operation in a crystalline rock aquifer.

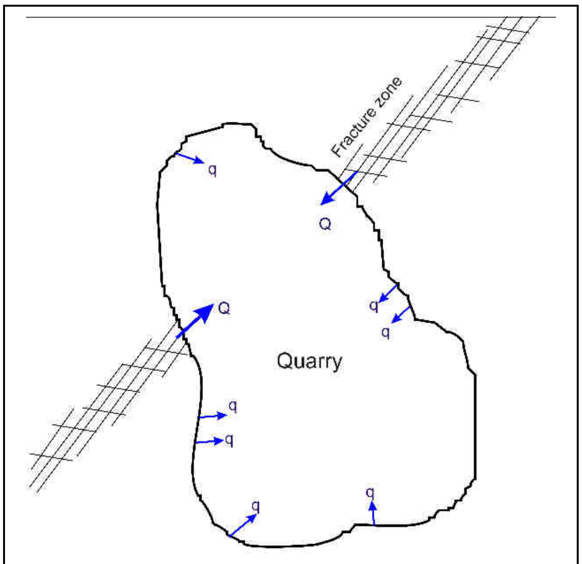

Figure A4.1 Schematic plan view of an excavation in hard rock (This figure illustrates major groundwater inflows (Q) from a fracture zone and several minor inflows (q) from individual transmissive fractures in the quarry face)

Specific transmissive features

Of course, a mine or quarry might intersect features such as faults or fracture zones that might be expected to be more transmissive than a single value of bulk hydraulic conductivity might indicate (Figure A4.1). Such features might provide significant inflows of groundwater to a quarry or may act as conduits, transmitting impacts further afield than might otherwise be expected.

In a completed mine or quarry, such features might be recognised visually in the walls of the excavation as zones of particularly intense and/or open fracturing. They may be purely tectonic in origin or may be chill zones at the margins of a dolerite sill or dyke. Zones of enhanced transmissivity may be sub-vertical or they may be approximately horizontal.

Prior to excavation, sub-vertical fracture zones and dykes might be recognised by the following methods:

- *Topography:* Examination of topographic maps and the terrain for lineaments, that is, linear topographic features, depressions or valleys that may correspond to faults, fracture zones or dykes.
- *Remote sensing:* Examination of aerial or satellite images for such lineaments.
- *Geophysical traverses:* Very low frequency (VLF) electromagnetic induction, magnetometry, resistivity profiling and electromagnetic induction (EM) can be effective at detecting sub-vertical features. Georadar can assist in detecting sub-horizontal features at modest depths.

Remember, however, that not all fracture zones and faults are transmissive to groundwater flow. Sometimes, they can be sealed with fault gouge or clay mineralisation produced by weathering or hydrothermal activity. The techniques mentioned above are *not* good at distinguishing between transmissive and poorly transmissive fracture zones (Banks *et al* 1994).

Wetlands in crystalline rock terrain

The elevation, climate and poorly draining nature of crystalline rock terrain means that most wetland environments associated with such terrain are *upland blanket bogs.* These are typically *ombrogenous* mires, supported by rainfall rather than groundwater flow. They are characterised by *Sphagnum* moss, sometimes ericaceous plants or lamb's wool moss. Such bogs are acidic (pH 3.8 to 4) and often 2 to 4 m thick.

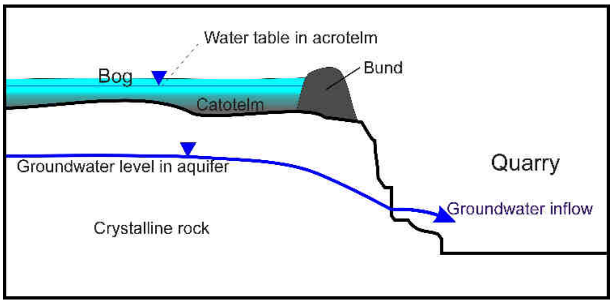

Figure A4.2 Schematic section of an excavation in a crystalline rock aquifer
(This figure shows that the main water table may be decoupled from a wetland perched on its surface. Dewatering of the quarry may in this instance have no impact on the wetland. But the fact that the excavation intersects the wetland may lead to water in the bog acrotelm spilling over the quarry face. This can be hindered by a bund of low permeable material.)

Because they depend on rainfall and poor subsurface drainage, upland bogs *may* not be in continuity with the water table in the underlying crystalline rock. If this is the case, then one would not expect them to be affected by dewatering (Figure A4.2). However, bogs can drain in an uncontrolled manner over the lip of an excavation, resulting in adverse ecological impacts on the bog and unwanted water ingress to the excavation. This can be hindered by simply installing a bund to contain the wetland at the top of the face (Figure A4.2). It would of course be necessary to demonstrate the engineering stability of the bund during dewatering operations, and to consider the long term impacts on cessation of dewatering.

If it is suspected that the bog is in continuity with the water table, the impact of dewatering may still be low, but some assessment is needed of the *hydraulic resistance* between the bog's *acrotelm* (permeable, living upper zone) and the crystalline bedrock aquifer. The *acrotelm* is underlain by the lower part of the bog, the dead, peaty, humified *catotelm* (Ivanov 1981). The hydraulic conductivity of the *catotelm* is usually low, between 10^{-4} and 10^{-9} m/s, depending on the degree of humification and species of bog plants (*Sphagnum* catotelm is less permeable than *Phragmites*-based catotelm, for example). The vertical conductivity K_V is much less than the horizontal K_H. Additionally, there may be a layer of, for example, till, separating the bog from the crystalline rock.

If we consider the sequence in Figure A4.3, we can calculate the downward leakage from the *acrotelm* if the groundwater head in the crystalline rock drops from h_A to h_B (1.5 m) as a result of dewatering. A thickness (b) of 1.6 m of catotelm with $K_V = 10^{-8}$ m/s (say) has a hydraulic resistance of:

$b/K_V = 1.6 \times 10^8$ s

1.5 m of boulder clay with $K_V = 10^{-9}$ m/s (say) has a hydraulic resistance of 1.5×10^9 s. The total hydraulic resistance (R_{tot}) is thus found by addition, and equals 1.66×10^9 s. The vertical leakage is found by:

$(h_A - h_B) / R_{tot} = 9 \times 10^{-10}$ m/s

or 29 mm per year. A judgement can then be made as to whether this is likely to be significant for the overall water balance of the wetland.

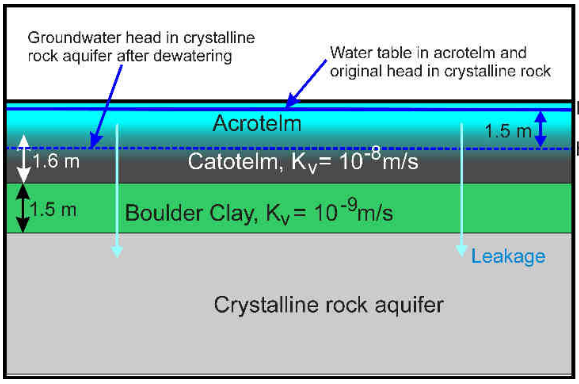

Figure A4.3 Schematic section of a bog developed on a boulder clay till substrate overlying a crystalline rock aquifer
(The bog may be in some degree of hydraulic continuity with the crystalline rock aquifer and a calculation of potential leakage through the catotelm and the till can be performed. It is assumed that a dewatering activity causes the head in the crystalline rock aquifer, which was initially similar to that in the acrotelm, to drop 1.5 m from h_A to h_B.)

HIA in fractured crystalline rocks

The general philosophy and procedures described in this report *can* be used to assess the likely hydrogeological impacts of dewatering in crystalline rock terrain. However, the above observations suggest that any results or prognoses will be associated with a greater degree of uncertainty (for example, regarding the shape and size of any cone of depression and the magnitude of any impact) than with more homogeneous, isotropic aquifers. In some cases, it may be decided that the characteristics and behaviour of the fractured rock is more similar to a karstic aquifer than it is to a homogeneous aquifer. It may be appropriate in those cases to use the 'monitor and mitigate' approach, with the revised methodology of eight steps described in Appendix 3. Professional judgement must be used to decide which approach is most appropriate in a given situation.

The application of HIA to a quarry in fractured crystalline rock (slate) is illustrated by Case Study 1 in Appendix 9.

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Appendix 5: Coal Measures

Introduction

This appendix, first referred to in Section 2.5 of the main report text, discusses the hydrogeology of the Coal Measures, and the particular factors that need to be taken into account when assessing the hydrogeological impact of dewatering opencast coal mines. Many of the comments apply equally well to underground coal mines, and indeed to certain types of quarry, particularly slate, where opencast excavations encounter old underground workings.

Acknowledgement on authorship: the text and diagrams for this appendix were contributed by David Banks of Holymoor Consultancy, Chesterfield, UK.

Hydrogeology of the Coal Measures

The Coal Measures of England and Wales typically comprise repeating sequences of mudstones, siltstones, sandstones, residual soils and coals, deposited in deltaic or shallow marine environments and subsequently lithified. They are typically defined as a secondary (minor) aquifer by the Environment Agency, supporting limited numbers of modest domestic, agricultural and occasional public water supply abstractions. The Coal Measures form a complex aquifer system, with the following characteristics:

- It is a partially discontinuous, multilayered system, where sandstone aquifers are separated by siltstone and mudstone strata that are commonly described as aquitards (even though some evidence suggests that their near-surface hydraulic conductivity is far from negligible; Banks *et al* 1997).
- It is dominated by fracture flow (see Appendix 4), except where surface weathering has enhanced intergranular permeability.
- It is heterogeneous and anisotropic. Sedimentological properties change rapidly with depth and also laterally (sandstone lenses etc.). The strata are often heavily faulted and enjoy preferential fracture directions.
- It is an 'anthropogenically enhanced' aquifer complex, having been intensively mined across large areas, resulting in significant changes to hydrogeological properties.
- It is subject to strongly three-dimensional groundwater flow and head distributions, due both to mining and to natural topography.

Assessing hydrogeological impacts in Coal Measures aquifers

The Coal Measures have historically been worked for fireclay, pot clay, ganister, ironstone and sandstone, but have most importantly been mined for coal. England and Wales have a legacy of several centuries of underground mining in these strata, although today, all but a handful of the deep underground mines are closed. In contrast, there has been an increasing tendency towards winning of coal through opencast mining, sometimes through voids over a kilometre long and over 100 m deep.

The general philosophy and procedures described in this report *can* be used to assess the likely hydrological impacts of dewatering in Coal Measures terrain, and the additional principles outlined in Appendix 4 are also relevant. The known hydrogeological properties of the Coal Measures are documented in the Minor Aquifers Properties Manual of Jones *et al* (2000). Alternatively, aquifer properties can be estimated from the hydraulic testing of exploration holes or the statistical distribution of specific capacities of boreholes, using the procedure of Banks (1992), outlined in Appendix 4, and illustrated in Table A5.1. In Table A5.1, note that median borehole specific capacity can act as an approximate surrogate of median aquifer transmissivity (Banks 1992), and that median transmissivity is very similar to geometric mean (lognormal distribution).

Table A5.1 Specific capacities and transmissivities in the Coal Measures of the Pennines

	East Pennines	West Pennines
Specific capacity ($m^3/d/m$)	N = 27	N = 95
25% specific capacity	9	8
Median specific capacity (N)	21	28
75% specific capacity	112	104
Estimated median transmissivity (method of Banks 1992)	23	31
Transmissivity (m^2/d)	N = 65	
25th percentile	8	
Median	16	
75th percentile	33	
Arithmetic mean	46	
Geometric mean	16	

Data from Jones *et al* (2000)

Studies of hydraulic conductivity on Coal Measures cores from the Pennine area typically yield median (intergranular) hydraulic conductivities of 5×10^{-5} to 1×10^{-3} m/d (Jones *et al* 2000). Borehole tests tend to yield higher values: Banks *et al* (1997) and Cripps *et al* (1993) found values in excess of 10^{-2} m/d in shallow borehole tests, underlining the importance of fracture flow pathways (not measured in core tests) and shallow weathering processes.

An anthropogenically enhanced aquifer

Beneath the Coal Measures outcrop and subcrop, centuries of mining have removed coal from various depths, sometimes in excess of 1 km. Underground mines may have been joined by roadways, drainage soughs or underground canals, creating hydraulically interconnected, potentially transmissive pathways stretching over tens of kilometres. Old pillar-and-stall workings may still be open voids, or they may have collapsed to form zones of goaf (collapsed strata) within the Coal Measures sequence. Modern long-wall mining deliberately allows worked out strata to collapse over large areas behind the working face. The strata above the collapsed seam develop subsidence fracturing, leading to zones of enhanced hydraulic conductivity (Booth 2002).

Furthermore, to allow deep working, underground mines may have been dewatered by systems of free-draining adits or soughs, or by pumping (Younger 2004). Following closure, previously pumped mines will typically accumulate groundwater or recharge water, with water levels rising until the mine water is able to overflow to the surface via an open shaft, a sough or (in some cases) through natural fracture pathways. Various computer codes are available to predict the rate of rise of minewater levels if the geometry of the mine, its interconnectivity with other mines and the sources of inflow, are known, from the relatively simple MIFIM code of Banks (2001), through the GRAM concept of Sherwood and Younger (1997), to the more recent application of SHETRAN-VSSNET (Nuttall *et al* 2002).

It seems that the rate of discharge from relatively shallow, flooded, abandoned mines is approximately in proportion to their areal footprint, such that a mine's 'water make' can be estimated from the product of its 'footprint' and the approximate recharge (Table A5.2). In deep mines, however, the limited hydraulic conductivity of overburden materials will restrict downward recharge (Younger *et al* 2002).

Table A5.2 The rate of overflow discharge from various abandoned mines, compared with their areal footprint and the water make from larger coalfields

	Area (km^2)	Actual discharge (l/s)	Areally-distributed water make (mm/year)
Mines, mostly shallow (Banks, unpublished data)			
West Yorkshire	0.6	4–7	210–368
South Yorkshire	0.79	13	519
Derbyshire	1.8	23	394
North Wales	1	10	316
Derbyshire	2	26	406
West Yorkshire	2	40	631
North Wales	3	30	316
Scotland	4.3	45	330
Scotland	8	>50	>197
Coalfields (Younger et al *2002)*			
Nottinghamshire			73
Durham			110
Scotland			219

Many underground mines are in the process of water-level recovery. Others continue to be dewatered via soughs or by pumping (for example, by the Coal Authority for environmental reasons). Depending on the vertical hydraulic conductivity of the aquifer complex and the quantity of available recharge, the dewatering of mines *may* have led to the drainage of the overlying sequence (that is, development of unsaturated conditions), or it may merely have led to the development of a strong three dimensional downward head gradient (Figure A5.1).

When planning a new opencast coal mine it is important to know:

- Whether the strata are saturated or unsaturated (this will strongly influence the amount of water required to be dewatered). Information from a single deep observation borehole may erroneously suggest that the water table is below the base of the excavation (Borehole B in Figure A5.1). Nested piezometers may be required to clarify the true distribution of heads and position of the water table.

- Whether minewater levels and groundwater heads are rising (rebounding) following deep mine closure. This will have consequences for the evolution of dewatering requirements with time, and for estimation of post-completion water levels in an opencast void. This information is particularly relevant if the planned void physically intersects mine shafts, workings or roadways.

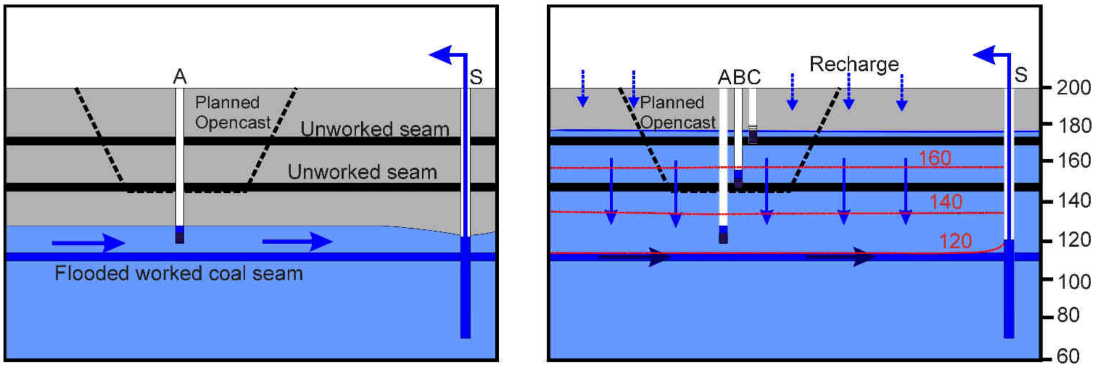

Figure A5.1 Groundwater heads in mined terrain still subject to active dewatering, in this case by pumping from a shaft (S)
(A low groundwater head measured in a deep piezometer (A) may indicate that strata have been dewatered by pumping down to that level (left). Alternatively (right), the low groundwater head may simply be a result of a very strong downward head gradient in largely saturated strata (red lines are schematic groundwater head contours), induced by pumping. Multilevel piezometers (A, B, C) are required to clarify the situation.)

Inflows to opencast coal mines

One can estimate inflows to opencast coal mines or the impacts caused by dewatering them using the techniques described elsewhere in this report. However, it is important to remember that the natural hydraulic conductivity of strata may have been enhanced by fracturing caused by former underground working. Furthermore, opencast excavations may intersect roadways, workings, goaf, or shafts from former underground mines which, if flooded, may contribute significant inflows to the excavation. Goaf, occupying the location of a former coal seam, may provide a high transmissivity stratum, especially if comprised of sandstone fragments. Goaf seams may, however, be almost indistinguishable to the untrained eye in the wall of an excavation, especially if the roof rock is mudstone or if the goaf has been compressed by a significant depth of overburden. In such circumstances its hydraulic conductivity may be correspondingly low.

Roadways act as highly conductive linear pathways for groundwater flow from other parts of a flooded mine complex. Because water flow in them is linear and may be turbulent, many common groundwater flow equations (which assume radial, laminar flow) cannot be applied. It may be more fruitful to apply other approaches (selected carefully according to whether flow through the roadway is 'full-bore' or not, and turbulent or laminar), such as:

- the Chézy or Strickler formulae for flow in an inclined channel
- the Bernoulli or Darcy-Weisbach equations for pipe flow.

Not only can roadways permit large water inflows to excavations from other parts of a flooded mine complex, they will also transmit changes in minewater head rapidly over large distances. For example, if an opencast void intersects a flooded roadway, the decline in head due to void dewatering may result in water level declines in the interconnected mine complex several kilometres away. Alternatively, plugging a flooded roadway or adit could cause mine water to back up in a mine system, leading to a multitude of potential impacts, from uncontrolled breakout of mine water and mine gas to geotechnical instability. Plugging of mine voids and shafts should only be contemplated after a thorough mining engineering and environmental risk assessment, and consultation with the Coal Authority. Shafts pose essentially similar risks to intersected roadways. If the mine workings accessed by the shaft are flooded to an elevation above the base of the pit, the shaft could result in substantial inflows to the base of a deep excavation (Figure A5.2).

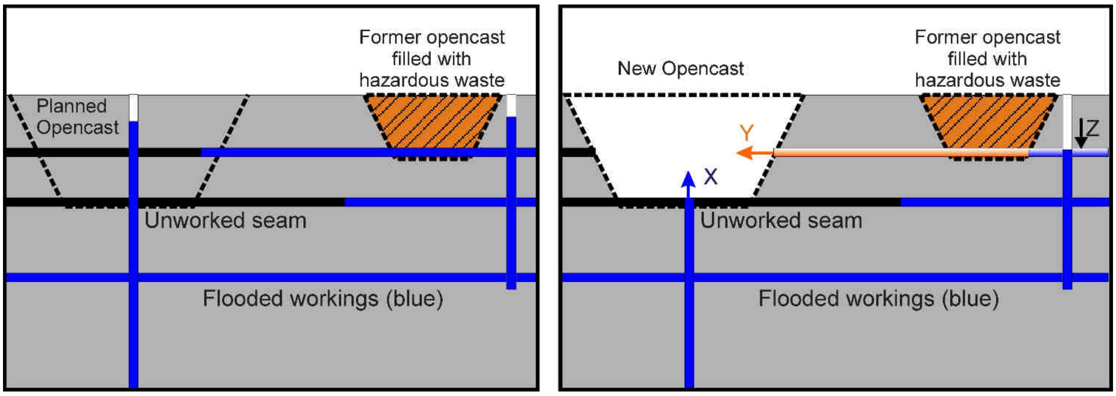

Figure A5.2 Risks associated with dewatering opencast voids
(X = inflow of mine water from former shaft; Y = inflow of minewater from roadway or former workings, contaminated with hazardous waste; Z = decline in mine water levels over significant area. Inflows X and Y may also be contaminated with iron, manganese, sulphate and acid from weathering of sulphide minerals in the coal-bearing strata.)

It is also important to remember that, following abandonment, a mine's shafts may have been sealed or filled in a specific manner with due respect for potential rising mine water levels or gas emissions. If shafts fall within the area of a planned opencast, the Coal Authority should be consulted to identify any need for sealing the shafts below the base of the excavation during the excavation or restoration phases. Furthermore, it should be remembered that mine openings may also provide potential contaminant transport pathways following cessation of dewatering and restoration of a void, especially if any of the proposed backfill is potentially contaminating.

Water quality

Coal Measures strata are associated with sulphide minerals such as pyrite and marcasite (both allomorphs of FeS_2). On exposure to oxygen and moisture (during quarrying and mining operations), these minerals may be oxidised:

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{++} + 4SO_4^{=} + 4H^{+}$$

releasing dissolved iron, sulphate and acid. These are the characteristic components of so-called acid rock drainage (or acid mine drainage). Water rich in these components may be generated as inflows from flooded mine workings intersected by an opencast excavation, by sulphide-rich strata in the walls of the excavation itself or by spoil/overburden removed from the active opencast and re-deposited elsewhere in the void. Such iron-rich or acidic water may require treatment prior to discharge.

Finally, Coal Measures terrain has historically often been heavily industrialised and may have a large proportion of contaminated land. Mine voids and shafts, or former quarries or opencasts may have been used for disposal of hazardous wastes. Mine workings can act as conduits for rapid transport of contaminants. It is not unknown for water flowing from former mine workings into a new opencast void to be heavily contaminated by wastes derived from nearby contaminated sites. This also creates problems for disposal of dewatering water.

Sources of data

The Coal Authority holds the following information:

- An archive of mine plans that are available for public inspection. While these are relatively complete and accurate for recently abandoned mines, this may not be the case for older mines. Furthermore, private mine owners may historically have under-reported the size of their workings for tax reasons.
- A network of minewater observation points (boreholes or shafts) to monitor the rebound of water levels in the larger mine complexes (Figure A5.3).
- A database of known discharges from abandoned mines and current pumped discharges.

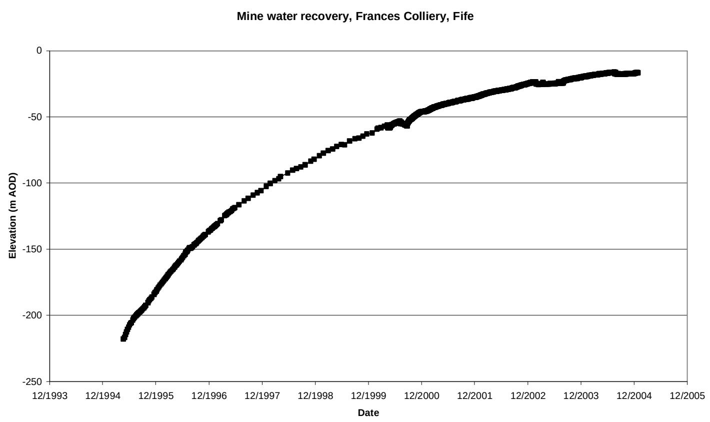

Figure A5.3 An example of rising heads in the Frances Colliery, Fife
(based on data available from the Coal Authority)

The application of HIA to an opencast coal mine is illustrated by Case Study 2 in Appendix 9.

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INWATCO

In addition to the references listed above, further useful information can be found in:

MIRO (2005). *Integrated water management in former coal mining regions (INWATCO) - guidance to support strategic planning.* Mineral Industry Research Organisation, November 2005.

This document describes the results of a demonstration project jointly funded through the EC Life Environment Programme, which aimed to investigate new approaches to mine water monitoring and management, and to reduce the risks and uncertainties associated with strategies for integrated management of water resources at river basin scale in former coal mining regions. The hydrological monitoring programme implemented through the project demonstrated the following factors related to the operation of mine water drainage systems and their response to meteorological variation (MIRO 2005):

- Large-scale mine drainage systems, although hydraulically interconnected, may exhibit a wide range of responses to rainfall recharge.
- Infiltrating rainwater has the potential to drain to deeper workings rapidly, depending on coalfield configuration and interconnectivity.
- Internal storage characteristics will determine the scale and responsiveness of minewater systems to recharge variation.
- Complex systems may not respond as expected to pumping control schemes.

Appendix 6: Water balances

Introduction

This appendix, first referred to in Section 3.4 of the main report, discusses in more detail the subject of water balances, and certain aspects of conceptual modelling. The water balance (sometimes referred to as a water budget) is a simple concept and a powerful tool, that can be applied to a wide range of situations, at any desired spatial or temporal scale, and with different degrees of complexity. For example, a water balance may be conducted for an entire catchment, or a single storage pond. It may use lumped quantities averaged over a long period, or it may include daily calculations. At its simplest, a water balance for a given system can be described as:

[water entering the system] = [water leaving the system] ± [change in storage]

The most important points to bear in mind when developing water balances are as follows:

- The components of the water balance must be based on a good conceptual model of the system in question. This may well involve some iteration, because the water balance may be being used to develop or refine the conceptual model.
- It is essential to define the system clearly, including the boundaries and individual components (inputs and outputs). For example, if the water balance is for a groundwater body, the components may include recharge, vertical leakage through less-permeable overlying strata, flows to or from surface water features, lateral flows to or from adjoining aquifers, abstraction, and change in storage. The boundaries of this system will typically consist of the water table, the base of the aquifer, and some convenient vertical boundaries like the edge of the aquifer. However, if the water balance is for the operational dewatering system itself, the boundaries are much harder to visualise, as they may encompass the interconnected system of pumps, pipes, storage facilities and processes that make up the entire water management system of the mine or quarry. It is usually helpful to draw a sketch of the system and its components.
- When defining components, be careful to avoid double-counting. For example, recharge to groundwater is already a mini-water balance involving rainfall, evapotranspiration, run-off, storage in the soil and unsaturated zone, etc. Also, look out for recirculation within the system, such as discharge to a river finding its way back into the groundwater via seepage through the river bed.
- Some of the components will be relatively easy to quantify (such as abstractions and discharges). Other components cannot be directly measured, but can be estimated using other information (for example, estimating groundwater flow from aquifer properties and hydraulic gradients). It is good practice to attempt to quantify all the components separately, both inputs and outputs, rather than relying on the 'out-of-balance' to quantify the poorly-understood components.
- Estimates should be made of the uncertainty associated with quantifying each component, and the sensitivity of the overall water balance to the possible range of values for particular components.

- It can be helpful to deliberately choose the time period for the water balance so that the change in storage is zero, especially if the change in storage is difficult to quantify. For example, for groundwater, choose a period when the groundwater levels are roughly the same at the beginning and end of the time period.

Water balances and dewatering

The importance of water balances to undertaking HIA for dewatering operations has been emphasised several times in the main report. It is essential that quarry and mine operators have a good understanding of what happens to the water that is abstracted, and good estimates of the quantities involved, including how much is consumed. Measuring or estimating abstraction and discharge quantities, and the quantities of water used for other purposes, is important for the following reasons:

- Full licences specify quantities of water as part of the licence conditions (typically a total annual quantity and a maximum daily rate). Operators applying for full licences will therefore need to know how much water they are using for various purposes.
- Even though transfer licences do not usually specify quantities, knowledge of abstraction and discharge quantities is required for the assessment of the impacts of the dewatering abstraction and the efficacy of the mitigation measures.
- In water resources planning, there is a growing focus on the efficient use of water and the justification of the need for the water. In a recent consultation document (Environment Agency 2005), the Environment Agency advocates the use of water audits as a way of improving water efficiency. Water audits usually involve some form of water balance calculations.

Typically, a water balance can be applied in one of three ways to a dewatering operation:

i. A catchment water balance, looking at the overall groundwater and surface water budget.

ii. A closure water balance that assesses the formation of a pit lake (post-closure).

iii. A water balance on the operational water management within the quarry.

These will now be discussed in turn.

Catchment water balance

Some specific aspects that need to be considered when applying a standard catchment water balance to dewatering operations include the following:

- Standard water balances are usually completed for an entire groundwater or surface water catchment (that is, at a regional scale). The scale of most dewatering operations compared to the size of the regional catchment makes it unlikely that the local effect of the operation will be noticeable. Consequently, such water balances are more likely to be applied at a local level, for the sub-catchment of the operation itself.

- However, to complete a realistic water balance, even for a sub-catchment, good data and information are required on components such as regional and local groundwater flow patterns and quantities, surface water flows, rainfall, evapotranspiration, abstractions and discharges. This level of detail is hard to achieve at many dewatering operations, because of the lack of reliable data.

- In addition, it is not always obvious where to place the boundaries for the water balance, especially when the groundwater and surface water catchments are not coincident. The choice of boundaries has a significant influence on what the inflow and outflow components will be for the water balance.

- Water can be reused within the same sub-catchment, for mitigation, for example. The volumes of water that infiltrate back and change the base-case recharge figure need to be included in the assessment, and great care needs to be taken to avoid double-counting.

In the context of HIA, catchment water balances are only likely to be used when undertaking a detailed numerical groundwater modelling study as part of a Tier 3 investigation, in which case experienced hydrogeologists and groundwater modellers are likely to be involved. Catchment water balances are often aimed at establishing the magnitude and distribution of groundwater recharge. Recharge is of course very important when undertaking investigations of the resources of an aquifer or catchment (as frequently done by the Environment Agency), but the focus of the HIA methodology is on the capture of discharge from an aquifer and predicting local impacts.

Post-extraction (closure) water balance

If planning and physical conditions allow, quarries are often restored to open water. For a large void, the lake can take a significant time to develop. The management of such a closure operation can be helped by construction of a water balance to assess how long it will take for the lake to develop, and what the principal flow elements will be. Inflows comprise:

- Groundwater flow into the pit lake – the rate of this will alter as the groundwater system recovers and head-dependent flows decrease.

- Direct precipitation onto the lake surface – will alter with the elevation and area of the lake.

- Pit wall run-off above the lake surface – will reduce as the lake rises and the pit wall area decreases.

- Run-off from contributing catchments upgradient of the pit (unless diverted).

Outflows comprise:

- Evaporation from the surface of the lake - will change according to the elevation and therefore area of the lake.

- Downgradient groundwater flow – if conditions dictate.

To construct a model, the first step is to use the final pit geometry to develop a numerical relationship between volume and elevation. A spreadsheet algorithm can then be constructed that relates the net volume of water in the pit to the elevation. The algorithm can then also calculate the time taken to fill the volume between any two elevations. The time taken to fill all the elevation stages will determine the time taken to finalise the pit lake. Where outputs are higher than inputs, the lake level will stabilise at some intermediate elevation; where inputs are greater than outputs the lake will fill to the top, or to overflowing. The management of the post-closure period is outside the scope of the HIA (being covered by the planning regime as opposed to transfer licences), and will not be discussed further here.

Water balance on operational water management

This is the most important type of water balance as far as HIA of dewatering operations is concerned. The water balance for the dewatering operation should focus on answering the questions: How much water is actually pumped for dewatering purposes? What subsequently happens to that water? How much water is consumed, and not therefore available for return to the environment?

Such water balances are an integral part of conceptual model development during an HIA for a dewatering operation. They allow water to be tracked from start to finish through the system and are a good indication of whether the conceptual model is valid. If there is a large discrepancy within the water balance the conceptual model should be revised if necessary. It may be that a water use has been overlooked, or simply that a parameter has been entered in the wrong units. For a first pass, the components of the water balance may be lumped into yearly totals. This is useful practice to ensure that a large component is not missing. Once all water uses have been identified, a more refined balance can be constructed, looking at monthly or weekly totals. The refined balance allows the critical times throughout the year to be identified (for example, if more water is required for stream support than is available to discharge once other water uses have been taken into account).

Inputs: The only inputs to the operational water balance are the volumes of water pumped from the sumps or boreholes.

Outputs: The outputs are all the discharges and uses that dispose of this water. Output components commonly include:

- Discharge to surface water: usually controlled by a discharge consent, and can be relatively easily measured or estimated.
- Discharge to ground: either just as a means of disposal, or as part of mitigation measures such as recharge trenches or lagoons, again usually controlled by a discharge consent.
- Other off-site discharges: such as compensation or replacement water supply for a derogated third party, or sometimes discharge to sewers.
- Vehicle and wheel washing: with actual consumption estimated from make-up water if the system involves recycling water.
- Dust suppression: usually assumed to be 100 per cent consumptive (through evaporation), and estimated from bowser capacity multiplied by number of bowsers required, for example.
- Mineral washing: with actual consumption estimated from make-up water if the system involves recycling water, or from the estimated moisture content of final product leaving the site.
- Product processing: water consumed as an essential part of a concrete batching process, or concrete block manufacture, for example.

- Evaporation or seepage from open or unlined water storage facilities, balancing ponds, settling ponds etc: difficult to quantify, but can be estimated from mini-water balances conducted on the pond itself. If attempting to quantify evaporation, the equilibrium temperature method is recommended for open water evaporation (Environment Agency 2001).

When defining the components of the operational water balance, the sources of the water being used must obviously be taken into account. Many quarries and mines have sophisticated water management systems that deal with water from different sources. For example: water for concrete block manufacture may be obtained from a completely separate borehole, which is already fully licensed; make-up water for a wheel-washing machine may be taken from the mains supply; and water for mineral washing may be supplied by a separate surface water reservoir. Hence the importance of defining the boundaries of the system carefully.

It is usually helpful to prepare schematic diagrams of the way water is used or managed within a quarrying or mining operation. Examples of such diagrams can be found in Figures A9.4 and A9.6 in Appendix 9.

Water balances are best conducted in spreadsheets. For convenience, a simple tool for calculating operational water balances has been included in the spreadsheet of Tier 1 analytical tools referred to in Section 3.4 of the main report.

Allowing for surface run-off and direct rainfall

As just described, the operational water balance ignores one very important question: How much of the abstracted water (the input to the water balance) really comes from groundwater, as opposed to pumping surface run-off and direct rainfall out of the excavation? In terms of the operational water balance itself and assessing the consumptiveness of the dewatering operation, this question does not really matter. Heavy rainfall may result in an increase in dewatering abstraction as the sump pumps deal with run-off and direct rainfall, but this usually reappears as an increased discharge at the other end of the system, and the system balances out over the long term.

However, the question is of great importance when it comes to assessing the impact of the dewatering abstraction on flows and water levels in the surrounding area. If the full quantity of water abstracted for dewatering purposes were to be used to assess impacts (as the parameter Q in the Thiem or Theis equations, for example), then the impacts would be significantly overestimated. This is because no account would be taken of the fact that a proportion of the water is derived from direct precipitation into the excavation, and surface run-off from the catchment of the excavation.

Determining the true impact of a dewatering abstraction would involve a detailed knowledge of the rainfall, run-off, evapotranspiration, recharge, etc before the excavation existed, then comparing that to the situation with an excavated void and the dewatering operational. This is likely to be completely impracticable because of lack of data, and it can be difficult to define exactly what the 'natural' conditions are, especially if the excavation has been there for a very long time. Similarly, while it may be possible with detailed study to determine the proportions of the abstracted water that come from groundwater, surface run-off, direct precipitation, etc, in most cases there will be insufficient data.

A practical approach is therefore required, which makes use of easily available information, and which is likely to be applicable in most cases (although there will always be exceptions that need to be treated differently). The suggested approach is based on the concept of baseflow. The baseflow can loosely be defined as the amount of flow in a river that is contributed by groundwater, as opposed to surface run-off. Baseflow is what enables rivers to keep flowing through extended periods without rainfall, when all run-off has long since ceased. The baseflow index (BFI) is a measure of the baseflow as a proportion of the total flow in a given river. There are various standard techniques for the smoothing and separation of river flow hydrographs to obtain the BFI. Values of BFI are published for every gauging station in England and Wales (CEH 2003). The suggested approach is as follows:

i. Measure or estimate the total quantity of water abstracted for dewatering purposes over a given period such as a year.

ii. Express this total quantity as an average daily abstraction quantity, with units such as m^3/d. This is the basic parameter Q for use in the Thiem and Theis equations, for example, when assessing the impacts of the abstraction.

iii. When estimating the worst-case impacts, use this value of Q without alteration (representing 100 per cent of the actual quantity abstracted).

iv. However, if there is good reason to believe that a significant proportion of the abstraction is actually derived from run-off or direct precipitation, then look up the BFI for the nearest gauging station (or the nearest gauging station with a catchment that has similar characteristics to the catchment in which the dewatering operation is located). If in doubt, or if unable to obtain the BFI, consult the Environment Agency.

v. Multiply the average daily abstraction quantity (Q) by the BFI (which is always a dimensionless factor less than one), to obtain an adjusted value for Q to use when assessing impacts.

This is admittedly a crude adjustment, and professional judgement should still be used, for example, when choosing which BFI to use if there is a choice of nearby gauging stations, or if there are no nearby gauging stations. Also, as mentioned above, there will always be exceptions that need to be treated differently, and again, professional judgement should be used. There may be situations where a sensitive water-dependent feature is close to, and in complete hydraulic continuity with, the dewatering abstraction, and it is necessary to use the unadjusted value of Q, or the spot value of abstraction at any given time.

Practical problems with water balances

It was mentioned above that it is good practice to attempt to quantify all the components of a water balance separately (both inputs and outputs), rather than relying on the 'out-of-balance' to quantify the poorly understood components, but this may not always be possible or practicable. In the list of inputs and outputs when discussing water balances for operational water management above, most of the components can be measured or estimated reasonably well, apart from the last one, namely, evaporation or seepage from open or unlined water storage facilities. Many dewatering operations will have very little idea how much water is 'lost' in this way, and it is tempting just to include this in the 'out-of-balance' component. If a detailed investigation is being undertaken, the usual approach is to do a mini-water balance on the pond itself, but this often turns out to be a smaller version of the same problem. It is relatively easy to quantify the water discharged into and pumped out of the storage pond, and to allow for the change in storage, but other potential components include open-water evaporation, seepage (both outputs), direct rainfall, surface run-off and even seepage again (all inputs), all of which are very difficult to quantify.

Common sense must dictate how far to take this kind of investigation; as always with HIA, it is a question of risk and uncertainty, with the level of effort being matched to the implications of the situation. Ask yourself the questions: How important is each component to the overall water balance (taking into account relative magnitudes and accuracy of measurement/estimation)? Do any quantities arrived at by 'out-of-balance' still seem physically reasonable? Does it really matter if a certain component cannot be quantified (bearing in mind the regional water resources situation, and observed or predicted local impacts)?

Some dewatering operations are part of a very complex water management system spread over a wide area, with many abstraction points, water being constantly pumped from one storage facility to another, several separate sources of water (groundwater, surface water, mains water, etc), and lots of potential for recirculation between storage and abstraction. Just such a case is the china clay industry in Cornwall, which is discussed in Case Study 4 in Appendix 9.

Notes on conceptual modelling of dewatering operations

The main types of information likely to be needed for developing a conceptual model of dewatering operations were described under Step 2 of the HIA methodology (Section 4.2 of the main report). There now follow some more notes on conceptual modelling. As described in Section 3.1 of the main report, conceptual models can be developed at different scales, to different levels of detail. Although the emphasis of the HIA is usually on impacts at a local scale, the regional picture is still required, to provide the context for the local conceptual model. If a completed CAMS document is available for the CAMS area in which the dewatering operation is located, then that is likely to provide sufficient information at a regional scale. If no such document is available, then information needs to be collected for an outline regional conceptual model, covering at least the following components:

- A definition, based on the regional geology and hydrogeology, of the extent of the study area (groundwater management unit) and its subdivision into appropriate zones (vertically and horizontally).
- A description of the hydrogeological conditions and flows at the boundaries of the unit (including vertical boundaries, where the adjoining strata should be identified as aquitards, aquicludes, leaky aquifers, etc).
- An estimate of the plausible range of aquifer parameters in the unit, and a description of the likely groundwater flow paths or flow patterns.
- Identification of the important water-dependent features of the area, such as rivers, ponds, wetlands, springs, seepages, estuaries, etc.
- Identification of the major water resources and water quality pressures on the unit (such as other abstractions, and point sources of pollution).
- A description of the likely mechanisms and locations of interaction between groundwater and surface water features.
- Interpretation of available hydrochemical data.
- A description of the limitations of the current conceptual understanding, and the major sources of uncertainty.

The outline regional conceptual model should then be refined by adding more detail about the local area around the dewatering operation, to form a local conceptual model. The information in the local conceptual model should of course be consistent with that in the regional conceptual model. Information should be collected on, and an understanding gained of, local factors including:

- *Geology:* use borehole lithological logs and large-scale geological maps (1:10,000 for example), to build up a three-dimensional picture of the local geology. Useful information can be obtained from site investigation, geotechnical, mineral exploration, and abstraction boreholes. At the local level, it is important to include drift and other superficial deposits, as they may have considerable significance. Try to construct several cross-sectional diagrams of the local geology, to refine your understanding of the structure of the aquifer.

- *Hydrogeology:* refine your understanding of the location and nature of hydrogeological boundaries (vertically and horizontally), local groundwater flow directions, the hydraulic properties of the aquifer and surrounding formations, and interaction with surface water features. Look for reports of test pumping already carried out in the area, and examine hydrographs from observation boreholes to gain information on local trends in groundwater level. Comparison of the hydrographs with annual recharge estimates or abstraction records from nearby boreholes may help identify whether the trends are natural or artificial.

- *Hydrology:* refine your understanding of the local surface water system, including catchment boundaries, losing and gaining stretches of streams and rivers, seasonal flow variations, behaviour of springs, and relationships to wetlands, lakes, meres, etc.

- *Groundwater quality:* collect information on local groundwater quality (including trends over time), and historical, existing and potential sources of groundwater pollution.

Further guidance on sources of information and data, and useful methods of processing, interpreting and displaying information during the development of a conceptual model can be found in Environment Agency (2002).

For ordinary groundwater abstraction from boreholes, relatively little information is required about the borehole itself, and the conceptual model concentrates on the issues just described. However, for dewatering, it may be necessary to go into a lot more detail on the excavation and the dewatering system. The types of information that would be useful to the conceptual model are as follows:

- Physical configuration of the quarry or mine, including overall footprint, and topography of the various pits, levels, benches, working faces, ramps, etc, relating all these features to the local three-dimensional geology.

- Other relevant physical features that may have been installed or constructed, such as any low-permeability barriers, cut-off trenches, recharge trenches, drainage galleries, soakaways, etc, assessing what effect each of these features is likely to be having on the local hydrogeology.

- Areas or locations in the excavation where there is ingress or seepage of water (that can reveal clues about the local groundwater levels and flow directions).

- Bodies of standing water in and around the excavation, including settling ponds, water storage reservoirs and naturally-flooded areas, assessing the losses from open-water evaporation, and the nature and degree of connection between the water body and the local groundwater levels.

Accurate elevations of the water levels in these various bodies of water are critical to understanding their relationship to each other and to the local groundwater.

- Dewatering arrangements themselves, including sumps, French drains, wellpoints, boreholes, adits, etc, estimating the abstraction quantities and how the quantities vary with time. Is the dewatering intermittent or continuous, constant or variable quantity, seasonal or all-year-round?
- Run-off within the excavation from rainfall, including how run-off is managed, and whether any alterations have been made to the natural catchment boundaries (including diversions of watercourses). Note that run-off is sometimes increased by the floors of the excavation becoming partly sealed due to the packing of fractures, fissures and joints with dust and rock chippings compacted by vehicle movements (Hobbs and Gunn 1998).
- Other sources of water within the overall water management system, such as separately-licensed boreholes or surface water sources, or even mains water supplies, and how these relate to the water derived from the dewatering system itself.
- Discharge arrangements into surface watercourses, including an assessment of the influence of the discharge quantities and timing on the natural flow regimes. Also relate the discharge quantities back to the abstraction quantities.
- Details of other uses of water within the quarry or mine, such as dust-suppression, gravel-washing, wheel-washing, etc, focussing on the amount of water actually consumed, and therefore helping to account for differences between abstraction and discharge.
- Extension plans for opening up new areas or deeper levels for mineral extraction, including any geological or hydrogeological information that can be gleaned from exploration boreholes, feasibility studies, and planning permissions.

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Appendix 7: Tier 2 tools

Introduction

This appendix, first referred to in Section 3.4 of the main report text, provides more information on tools and techniques that are likely to be used at the level of Tier 2 (apart from detailed water balances, which have already been discussed in Appendix 6).

Analytical models

The use of a Tier 2 analytical model allows greater flexibility in the assessment of the impact of dewatering on selected features while not requiring the same time and effort as a full numerical groundwater model. In particular, analytical models allow the user to examine a wide range of dewatering and mitigation scenarios, in two dimensions, and to simulate impacts in time-variant mode. Examples of analytical models include:

- WinFlow: an interactive analytical model developed by ESI Ltd, that simulates 2-dimensional (that is, in a horizontal plane) steady-state and transient groundwater flow (for both confined and unconfined aquifers). The principle of superposition is used to evaluate the effects of multiple analytical functions (such as wells) in a uniform regional flow field.
- TWODAN: an analytical model developed by Fitts Geosolutions with very similar theoretical basis and capabilities to WinFlow.

A range of analytical elements can generally be simulated in two-dimensional flow, including: wells, uniform recharge, circular recharge/discharge areas and line sources or sinks. Any number of these elements may be added to the model, including a uniform regional hydraulic gradient. The models generally depict the flow field using streamlines, particle traces, and contours of hydraulic head. The streamlines are computed semi-analytically to illustrate groundwater flow directions. Particle-tracking techniques can be implemented numerically to compute travel times and flow directions.

Steady-state groundwater flow is usually computed using analytical functions developed by Strack, while in transient mode, the hydraulic heads are usually computed using the Theis equation for confined aquifers and the Hantush and Jacob equations for leaky aquifers. Most models can import a Drawing Interchange Format (DXF) file (from AutoCAD for example) to use as a digitised base map. The digitised map gives the modeller a frame of reference for designing the analytical model.

Analytical models do of course have limitations. Recharge features cannot usually be made head-dependent, so it is difficult to cope with river-aquifer interaction. Also, the aquifer is usually assumed to be of infinite extent, so it is difficult to simulate no-flow boundaries. In these cases, analytical models may be inappropriate. Their real strengths are the investigation of abstraction from multiple points, how flow directions and times change, and the effect of proposed mitigation measures. Typical applications of analytical models are as follows:

- *Investigate the extent of dewatering impacts:* a) Establish the conceptual model of the site, using realistic hydraulic parameters. b) Run the model in steady-state mode to develop a flow situation that approximates the conceptual model. A fully-calibrated model will not be realised by using an analytical model, but a close approximation of flow directions and levels can usually be achieved. c) Introduce the dewatering; abstraction can generally be simulated by placing wells at the edge or the middle of the excavation. The initial dewatering rates may be those calculated as part of initial studies or a range can be applied to assess the rate required to dewater the base of the excavation. d) The model can then be run in transient mode (using the life of the quarry or mine as one of the time steps). The resulting drawdown at any point in the model can then be established by reference to the baseline conditions. The software can prepare a contour plot that will show the areal extent of the dewatering influence and the potential receptors at risk. e) The model can be re-run with a range of typical parameters and times to assess potential impacts under a range of feasible scenarios.

- *Assess mitigation measures:* Analytical models typically include elements that can be used to simulate mitigation measures. In WinFlow, these include flux line sinks, head line sinks, and ponds. Once an impact has been identified in the dewatering model, any of these elements can be introduced into the model to simulate mitigation. For example, a head line or flux line element can be used between a dewatering operation and a receptor to model the impact of a recharge trench.
- *Assess closure impacts:* Once a dewatering model has been run, the abstraction elements can be turned off and the model run for further time periods to calculate how long it will take water levels in the aquifer to recover.
- *Particle tracking:* Particles can be introduced into models like WinFlow to look at the pattern, direction and times for groundwater flow.

IGARF

IGARF is a spreadsheet-based tool developed by ESI Ltd under contract to the Environment Agency (Environment Agency 1999 and 2004). IGARF allows the user to:

- consider the impact of a groundwater abstraction on a single river;
- consider the impact of a no-flow boundary on a single river system;
- compare the impact of a groundwater abstraction on each river in a two-river system;
- specify the relative positions of the river(s), boundary and well;
- consider continuous and periodic pumping regimes;
- design a pumping test;
- obtain drawdown predictions;
- obtain river flow depletion predictions in time and space;
- provide an audit trail for their model.

IGARF is most likely to be used to examine specific suspected impacts. In effect, IGARF is similar to the analytical solutions presented for Tier 1, but has vastly increased functionality and can include a range of greater analytical elements, such as no-flow boundaries and rivers. When IGARF is applied to dewatering situations, the dewatering abstraction is represented by a well. In IGARF, the diameter and position of this well can be specified. The abstraction impact can therefore be assessed either as a single well in the centre of the excavation with a standard well diameter (say 0.1 m), or as a single well with the diameter of the proposed excavation. A judgement needs to be made on a case-by-case basis as to which method produces results that are best applied to the real-life situation. Generally, a single well in the centre of the excavation is used for most analytical dewatering calculations, but this can underestimate the size of the eventual cone of depression if most actual dewatering is achieved at the side slopes.

If boundary conditions are important (rivers or no-flow boundaries, for example), then IGARF may be more appropriate than the analytical models described earlier. However, IGARF cannot be used to look at mitigation, cannot simulate unconfined conditions (so does not perform well when saturated depth is of a similar magnitude to drawdown), and cannot look at flow directions and times. Version 4 of IGARF has time-series of drawdown at any point as an output, as well as a graphical representation of the cone of depression. Therefore, the drawdown at any point of interest can be calculated and a general feeling for the lateral extent of dewatering can be gained.

Radial flow models

Radial flow models generally solve the horizontal radial time-varying flow equation to a well in a confined, semi-confined or unconfined aquifer. In a dewatering situation, the excavation would be represented by a large diameter well. Two very similar radial flow programs have been produced, both called RADFLOW:

- By Rathod and Rushton from Birmingham University in the early 1980s. This program, written in BASIC, is partially documented in Rushton and Redshaw (1979) and was sold by the International Groundwater Modelling Centre. It is a one-dimensional radial flow model, although it was also expanded to a two-dimensional r-z model.

- By Johnson and Cosgrove of University of Idaho in 2001. This package includes an MS Excel spreadsheet as a user interface and is a two-dimensional, r-z model. It can be down-loaded free from http://www.if.uidaho.edu.

Either RADFLOW program calculates drawdowns resulting from a pumping well in the centre of a circular homogeneous, isotropic aquifer subject to uniform areal recharge. The radial dimension is discretised using a logarithmic function starting at the outer edge of the well and increasing towards the outer boundary. A grid is automatically generated based on the radius of the well and the distance to the outer boundary. The model includes a well of finite radius with allowance for the free water initially contained within the well, various conditions on the outer boundary, a change between the confined and unconfined states, allowance for leaky confined aquifer condition, and variations in saturated depth of an unconfined aquifer. The outer boundary may be: 1) impermeable (no flow boundary), that is, assuming that all water pumped by the well comes from storage or areal recharge; or 2) at a constant value for head (recharge boundary), that is, assuming zero drawdown.

The advantage of radial flow models is that they are usually very accurate in the representation of water levels near the abstraction point, so they could be useful if derogation is an important issue, especially in layered aquifers. Radial flow models are not typically used within either the mining or quarrying industry. The RADFLOW packages take a little more time to build, run and interpret than more user-friendly packages, some of which have increased functionality.

Recharge assessment – spreadsheet method

An estimate of recharge is required for most Tier 2 tools, although a detailed site-specific assessment is probably not warranted. The Environment Agency has developed a spreadsheet-based "WFD recharge calculator", which incorporates a water budgeting approach, coupled with a soil moisture balance (Environment Agency 2006). It entails a tiered level of data requirements, depending on the level of data available and the stress on the groundwater body. The key variables used in the model include:

- Precipitation – daily to annual figures can be used. The software includes a contouring algorithm to average precipitation between gauges. MORECS (or MOSES) and Low Flows 2000 data can also be used, as can long-term averages.
- Urban recharge – input as a crop factor or estimated from population density.
- Snow melt – determined from available precipitation information. Typically a proportion of precipitation is 'held' until the thaw.
- Bypass flow – site-specific and included as a percentage of effective rainfall.
- Surface water leakage – included as a steady-state hydraulic calculation.
- Surface run-off – the HOST (Hydrology of Soil Types) methodology is adopted and is therefore dependent on the availability of HOST maps.
- Actual evapotranspiration – calculated using the FAO crop parameter method and using available data.
- Interflow – percentage factors are assigned depending on the drift geology.

The data are input to a number of worksheets that can be integrated with a GIS database. The results are output numerically or graphically. The recharge calculator is very useful for: a) estimating open-water evaporation in comparison to grass evapotranspiration, for looking at post-closure impacts; b) estimating open-water evaporation losses from balancing tanks or settling ponds, etc; and c) estimating the recharge in the catchment of a mine or quarry, and therefore developing and refining the regional conceptual model, including the influence of features such as urban areas and canals. See also Environment Agency (2001) for more detail on open-water evaporation.

Other useful techniques

Many other investigative techniques may be useful when undertaking HIA at Tier 2 or indeed at any other level. If considered appropriate, and used carefully, these techniques can provide additional information that may help with the development of conceptual models, and the prediction of impacts. Such techniques include the following:

- *Tracer tests:* which involve adding a suitable tracer (such as a fluorescent dye) to groundwater, with the aim of establishing a flow connection between the release point and a sampling point. While the detection of the tracer at the sampling point proves a connection, it is important to realise that failing to detect the tracer at the sampling point does not prove that there is no connection. A review of the theory and practice of groundwater tracing can be found in Ward *et al* (1998).

- *Test pumping:* pumping water from a borehole under controlled conditions, with collection and analysis of appropriate monitoring data from the pumped borehole, and ideally from observation boreholes as well, is the method most commonly used by hydrogeologists to determine aquifer properties. Most standard hydrogeological textbooks contain discussions of how to conduct and analyse pumping tests. See also BS6316:1992.

- *Geophysics:* which involves the measurement of physical properties of soils and rocks (and the groundwater they contain), such as electrical resistivity, the response to gamma or neutron radiation, conductivity, temperature, seismic response, etc. Geophysical surveys are carried out either over the surface or down wells and boreholes. Again, most hydrogeological textbooks contain an introduction to the subject, and see Guérin (2005).

- *Geochemistry:* the study of the chemistry of groundwater in relation to the chemistry of the surrounding soils and rocks can reveal a great deal of useful information, such as the origin and mode of groundwater recharge, and flow paths within an aquifer. A good summary of this subject can be found in Glynn and Plummer (2005).

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Appendix 8: Tier 3 tools

Introduction

This appendix, first referred to in Section 3.4 of the main report text, discusses the main tools that can be used at Tier 3, which are numerical groundwater models, and explains how specific aspects of dewatering can be simulated in the models.

Groundwater models are typically applied at two levels in relation to open-pit mining and quarrying operations: a) the sub-regional impact assessment scale that may include the design of dewatering systems; and b) the local scale considering dewatering or depressurisation issues with regard to geotechnical aspects such as slope stability. Modelling at the sub-regional scale for environmental impact assessment is considered here.

The numerical simulation of mining or quarrying operations for environmental impact assessment is not radically different to typical model applications for water resource or contaminated land requirements. A conceptual understanding of the hydrological and hydrogeological system is required that is translated into a numerical representation of aquifer(s) and associated units. Appropriate boundary conditions are applied to represent mechanisms such as recharge, regional throughflow and abstraction. Models representing mining or quarrying activities do have differences in that:

- Mining and quarrying operations frequently remove parts of the aquifer under simulation, or radically change the hydraulic properties of parts of the model domain. Overall boundary conditions and/or boundary conditions within the mine or quarry may need to change with time.
- Upon closure, sub-water table mines or quarries are frequently left to flood, creating permanent changes to the hydrological and hydrogeological regime.

This appendix considers the specific issues associated with modelling mining and quarrying operations with numerical models, the representation of voids, dewatering systems and simulation of post-closure conditions. This is followed by comments related to specific modelling codes that may be applied by mine or quarry operators and their consultants, or the Environment Agency, to assess the impact of dewatering. As mentioned in Section 3.4, the main modelling codes likely to be used for simulating the impacts of dewatering are:

- *MODFLOW:* a freely available code developed by the United States Geological Survey, which has become the industry standard. Many pre- and post-processors and other useful software modules have been developed for MODFLOW.
- *ZOOMQ3D:* a relatively new code being developed jointly by the University of Birmingham, the Environment Agency, and the British Geological Survey.
- *MIKE-SHE:* a package of models and graphical user interface developed and marketed by DHI Water & Environment.
- *FEFLOW:* a finite-element model (the other three are finite-difference models) developed by WASY GmBH.

Representation of mining and quarrying operations

There are usually a number of elements required for the impact assessment of a mine or quarry using a numerical groundwater model. During representation of the operational phase it may be necessary to represent the forming void and its boundary conditions. Where dewatering schemes are used, these need to be represented in a realistic manner. During the closure phase, voids may be left to flood, changing the hydrological regime, or be back-filled with overburden, waste rock or other inert material.

Representation of the mine/quarry void

A mine/quarry void below the water table represents a removal of aquifer material and is accompanied by dewatering of the aquifer. The void also alters the recharge to the aquifer by removal of the soil and unsaturated zones.

Whether a direct representation of the void is necessary within a groundwater model depends upon the operation of the mine or quarry. Whilst workings are maintained in a dry state, particularly if this is achieved by a dewatering scheme located outside the mine or quarry, then the actual void plays no direct role in groundwater flow. In this situation, if the dewatering scheme is adequately simulated and heads are drawn below the void floor, it may not be necessary to include boundary conditions to represent the void or to modify aquifer parameters.

Where mine or quarry operations are not kept completely dry, with water entering the void either via vertical flow to the base of the void or by seepage on the walls of the void, the representation by the model of groundwater discharges is required. There are several techniques that may be employed with differing degrees of complexity:

- Discharge to the floor of the excavation that is then pumped away can be represented with a head-dependent outflow mechanism. An example is the Drains mechanism in MODFLOW. The elevation of the drain is set as the elevation of the floor with a relatively high conductance term permitting fluxes out of the model without a significant build-up in groundwater head.

- Discharge to benches and slopes can also be represented using individual drain-type boundary conditions, or by using other mechanisms designed for areal use such as evapotranspiration. In either of these cases the elevation of the discharge is set. If the groundwater level rises above the set elevation then a flow out of the model occurs. This is an approximation to a seepage face condition. At the scale of sub-regional impact assessment the approximation is more than sufficiently accurate.

- Conventional abstraction wells can also be used, although typically the pumping rates from wells have to be set *a priori*.

- FEFLOW permits the setting up of composite boundary conditions that can change from fixed outflow, for example at the capacity of a sump pump, to head-dependent, depending on the elevation of the water table.

- More complex and composite boundary conditions may be applied, for example in MODFLOW the stream routing package can be used to represent the head-dependent removal of variable quantities of water from the base of a mine or quarry, and to route these fluxes elsewhere into a recharge trench.

Under unconfined conditions the reduction of groundwater heads to the floor elevation or below leads to decreased aquifer transmissivity. Groundwater models should automatically represent this change in transmissivity provided the aquifer is defined as unconfined.

Some groundwater models, for example FEFLOW, have a mechanism for a deformable upper surface that follows the phreatic surface. In this case, as groundwater levels are reduced to void floor level or below, the upper surface of the model declines to below the void. In this case the void itself is then outside the model domain. The main advantages of the moving mesh are that dry cells are avoided when groundwater levels rise, and adequate representation of vertical gradients is maintained, since the number of layers in the simulation is not reduced.

Stress relief at the base of workings caused by the removal of overlying material can cause heaving of the floor and opening of fractures. Evidence for this may include base-of-pit wells with greater than expected capacity or higher than average permeabilities. In this case, it may be appropriate for the model to represent a change in hydraulic parameters for the floor of the void. These effects are more usually found in hard-rock deep open excavations.

Mining and quarrying change recharge to aquifers. Where areal recharge is applied to a model it may be necessary to alter the recharge values within the mine or quarry area. Where this area is only a small percentage of the overall model domain, ignoring changes to areal recharge may be permissible.

Time-variant simulations of mine or quarry operation that include periods of significant groundwater recovery, such as very high recharge or cessations in dewatering, may experience problems such as dry cells and re-wetting of model layers frequently encountered when representing post-closure scenarios.

The vertical discretisation of a model can influence how a mine or quarry is represented. With additional layers within a model, the representation of vertical flows is improved. This can also be of benefit for representation of the mine or quarry using boundary conditions, for example placing drains in individual layers that represent benches. With additional layers it is possible to increase the accuracy of post-closure simulations that consider the flooding of the void or re-saturation of backfill. Vertical flows into the base of an excavation need to be represented by models to adequately represent dewatering, therefore the model domain should continue to a significant depth below the base of the excavation.

Representation of dewatering schemes

When excavations extend below the phreatic surface, a number of techniques are used to dewater including:

- Wells equipped with surface suction pumps, jet pumps or conventional submersible pumping for deeper well operation.
- In-pit water collection in sumps with associated pumps.
- Horizontal or inclined drains drilled from the face of the excavation, either gravity-draining or airlifted.
- Wellpoint systems connected to a common suction riser, frequently operating on multiple levels or benches.
- Pumped interceptor drains and ditches.

The adequate representation of dewatering schemes in numerical models requires a number of issues to be addressed:

i. Scale issues related to model cell size compared to the size of actual abstraction wells. This can be addressed by using variable mesh spacing or refined elements. ZOOMQ3D and FEFLOW can refine meshes and elements to a fine resolution without extending the refinement to the edges of the model domain. It is possible to apply well corrections in finite-difference models to estimate the drawdown within a pumping well from the head in a square model cell. Scale issues are often of more concern when predicting dewatering system performance than for environmental impact assessment. With refined model meshes dry cell problems may be more of an issue. In a coarse grid the averaging of heads and elevations may stop a cell from drying, whilst in a refined-mesh representation of the same area some cells would have greater drawdown and would become dry.

ii. The flow rates from dewatering systems are head-dependent whilst the conventional representation of wells in many models, such as MODFLOW, requires abstraction rates to be fixed *a priori.*

iii. Dewatering systems are frequently operated with float switches that start and stop pumps to maintain heads at a desired level. Variable-head boundary conditions are more appropriate to represent dewatering systems with these characteristics. Variable-head boundary conditions, such as the Drain mechanism in MODFLOW, should be applied with caution since the flux out of the system is related linearly to the head difference between the model cell and the drain elevation. With high groundwater heads this can lead to very large fluxes, in excess of the pumping capacity of the dewatering system.

iv. Wellpoint systems usually have multiple small-diameter wellpoints connected to common risers and are pumped by suction. To achieve drawdowns in excess of suction lift, they are installed on benches of decreasing elevation. Due to their high density, wellpoints are usually best represent using lines of head-dependent boundary conditions. When this technique is applied it is important to verify that the outflows from the head-dependent boundary conditions are similar to that actually abstracted by the wellpoints.

v. Surface interceptor drains or ditches that are pumped can also be represented by head-dependent boundary conditions. In some circumstances a mechanism that permits both inflow from and outflow to the ditch may be appropriate, such as the general head boundary condition in MODFLOW (GHB).

vi. Discharge of dewatering flows may require representation if the water is injected back into the aquifer or is discharged to surface water or recharge trenches. In the case of time-variant simulations of head-dependent dewatering systems, the use of mechanisms that can route and recharge the variable fluxes is extremely useful. An example of this is the streams package in MODFLOW.

When representing dewatering systems using head-dependent boundary conditions, fluxes should be checked to ensure that dewatering flows are realistic.

Representation of closure

The simulation of closure usually means the cessation of dewatering pumping and the rebound of groundwater levels. There are a number of common issues and problems encountered when simulating closure, including issues relating to the re-wetting of cells and changes in the hydraulic properties of parts of the model domain representing the mine or quarry void.

When considering impact post-closure, the presence of the void, whether back-filled or not, has to be taken into account. Groundwater rebound will usually flood the void once dewatering ceases, or if back-filled, will saturate the backfill materials. This requires changes to the boundary conditions employed and/or to the model's aquifer properties. Most model codes do not allow the properties of the aquifer to change during a simulation. The usual way around this is to stop the time-variant simulation at the end of the dewatering phase, to change model parameters and boundary conditions to include the flooding void or re-saturating backfill, and to re-start the simulation using the groundwater heads from the end of the dewatering phase.

If the void is left to flood during closure, then the presence of the lake can be represented, to a degree, within the groundwater model by changing the aquifer properties and boundary conditions for those model cells or elements that represent the void. These changes may include:

- Significantly increasing the hydraulic conductivity of the lake area. Values of 10^3 to 10^6 m/d permit the rapid flow of water without requiring any significant hydraulic gradient. It may be necessary to tighten solver parameters to ensure accurate mass balances when using very high hydraulic conductivity values.
- Setting unconfined storage to 1 to represent standing water.
- If aquifer properties are changed to represent the mine or quarry then it may also be necessary to alter the vertical and horizontal discretisation such that the void can be represented with the required resolution.
- Changes to the recharge applied to the model to take account of precipitation directly into the lake and evaporation from surface water.
- Where it is not necessary for the model to determine the level a lake may reach, due for example to direct connection to a large surface water feature such as a river, it may be appropriate to represent the lake by a variable-head boundary condition such as the MODFLOW GHB or rivers mechanism.
- Application of a dedicated mechanism for representing lakes such as the lake package for MODFLOW (Council 1998).

When simulating the post-closure recovery of water levels in an underground or surface void in low-permeability fractured rock environments, the MIFIM model of Banks (2001) might prove useful. This model simulates the filling of a void space (whose area can change with increasing water level) by both head-dependent inflows from a limited number of individual fractures or tunnels, *and* by head-independent recharge from the surface.

If a mine or quarry is back-filled the model needs to take account of properties of backfilled material. Property values for backfill are usually unknown. Detailed sensitivity analysis is therefore appropriate to determine how dependent predictions are on the properties estimated. During closure it may also be required to determine the impact of the removal of discharges on surface water bodies that have previously received dewatering flows.

Representation of low-permeability features

Grout curtains or cut-off walls are sometimes employed to reduce groundwater flows into dewatering operations. These can be included in groundwater models as low-permeability features, either simply as thin zones of lower hydraulic conductivity or by the use of specific mechanisms. In MODFLOW the wall package is designed to enable the representation of thin low-permeability structures within an aquifer. These are applied between model cells and act to reduce the conductance term between two cells.

FEFLOW and ZOOMQ3D can be made to change the scale of their meshes in order to represent cut-off walls in detail.

Model-specific comments

Of the models used to predict impact from dewatering, MODFLOW is the most commonly applied, and this is reflected by the comments in this section.

MODFLOW

MODFLOW, a freely available code developed by the United States Geological Survey (McDonald and Harbaugh 1988), is the most widely applied groundwater model. This is partly due to the variety of boundary conditions, or packages in MODFLOW terminology that are available. The potentially useful packages for representing mines and quarries include:

- Wells for standard pumping wells where abstraction rates are known.
- Drains for permitting head-dependent discharge from dewatering schemes.
- Rivers and general head boundaries for representing interception trenches.
- Rivers, general head boundaries or injection wells to represent recharge trenches and ditches.
- The creative use of the streams package to represent head-dependent abstractions from dewatering systems and to route the flows to recharge the aquifer elsewhere. This application permits the model to determine the dewatering rates and to directly recharge the same flows.
- The lake package to represent the flooding of a mine or quarry on closure. This package can be used in conjunction with other MODFLOW packages such as the streams and recharge packages.
- The creative use of composite or multiple boundary conditions within the same cell. By combining more than one boundary condition it is possible to include more realistic representations of dewatering systems, for example, by combining two drain cells, it is possible to represent a head-dependent abstraction with a constraint on the maximum flow.
- The multiple node well package (Halford and Hanson 2002) permits drawdown-limited pumping constraints to be applied. This is a more

flexible method than the conventional wells package and can be configured to more realistically represent dewatering schemes. The drawdown restraint can also prevent cells with pumping wells from going dry.

The re-saturation of model layers that go dry during simulation of the dewatering phase is a common problem for models developed with MODFLOW. Ways around the problem include:

- Using a single or thick model layer to represent the mine or quarry area such that water levels are not drawn below the base of the layer, although this may compromise accuracy of the model.
- The use of the MODFLOW (BCF2) re-wetting option. Although this frequently introduces numerical instability in some cases, it can be made to work effectively.
- Using MODFLOW-SURFACT, which is a proprietary version of MODFLOW produced by Waterloo Hydrogeologic Inc. It converts from saturated flow to an unsaturated flow equation as the groundwater elevation drops below the base of the model cell. The key point of MODFLOW-SURFACT is that the cell does not go dry, is never excluded from the numerical calculations, and as heads rise again the model switches the cell back to saturated flow.
- Application of changes to the MODFLOW code for single layer models that prevent cells from drying by maintaining a minimum depth of saturation.

ZOOMQ3D

ZOOMQ3D is a finite-difference-based flow model developed by the University of Birmingham, the British Geological Survey and the Environment Agency. The principal differences between it and other finite-difference models are the coding using the object-oriented method, and the ability to radically alter the resolution of the finite-difference grid without extending the refined zones out to the edges of the model domain. With relation to assessing the impact of quarry dewatering there are two aspects of ZOOMQ3D that are of interest:

- The ability of ZOOMQ3D to represent large changes in model scale to enable the representation of small-scale features such as individual components of a dewatering scheme.
- The object-oriented design should make the alteration of the model code to include new boundary conditions more straightforward than for traditional linear coding methods.

ZOOMQ3D has a number of boundary conditions that may be applicable to modelling mines and quarries including abstraction wells, rivers, head-dependent leakage and springs. ZOOMQ3D includes a re-wetting routine similar to that of the BCF2 package used with many MODFLOW simulations. Although the developers make no reference to instability issues with the re-wetting routine, given its similarity to MODFLOW, this could experience the same issues. ZOOMQ3D is a relatively new model and the number of people with practical experience is still limited.

MIKE-SHE

MIKE-SHE is a group of integrated models and a graphical user interface that can simulate all land phases of the hydrologic cycle. Included within MIKE-SHE is a bespoke quasi-3D groundwater flow model (that is similar to MODFLOW). The MIKE-SHE model is proprietary software developed and distributed by DHI Water and Environment, Denmark.

Unlike MODFLOW, data are linked to the model rather than being imported. The user interface is based on geo-objects, for a natural object-oriented model design (including pinching geologic layers and lenses). Mesh refinement carries the same problems as MODFLOW. All types of commonly used boundary conditions for groundwater modelling are available with the groundwater component of MIKE-SHE.

MIKE-SHE allows for full integration of both surface water and groundwater, although both components can also be run separately. If the two components are run concurrently, it enables feedback from the groundwater model to the recharge mechanism, making the models ideal for modeling riparian zones and wetlands. However, at present, the preferred Environment Agency FAO recharge code cannot be used with MIKE-SHE. MIKE-SHE is also capable of unsaturated zone simulations and representation of seepage faces.

The authors of MIKE-SHE acknowledge that using a physically-based distributed parameter code like MIKE-SHE can be very data intensive and computationally complex (compared to a code like MODFLOW). The modular nature of the MIKE group of models allows a gradual build up in complexity; for example, the groundwater system can be greatly simplified, or even eliminated from a surface flow simulation, allowing the surface flow details to be evaluated without the additional complexity of the groundwater system. Once a stable configuration for the surface flow is established, the groundwater system can be incorporated.

MIKE-SHE has been used in projects involving dewatering, for example, the development of five small construction dewatering models for the Citytunnel rail line in Malmo, Sweden. The models had to represent groundwater flow around open holes and tunnels. Sheet piling, grout curtains and bleeder wells were represented in the models to predict groundwater inflow under different scenarios, although it is not known how this was done (contact DHI Water & Environment for further details). Drawdown around the construction sites was closely monitored and wells were used to re-inject the abstracted water into the aquifer and prevent the influence of the dewatering from spreading. MIKE-SHE has also been used for the simulation of water levels in a flooded quarry during a pumping test for water resources evaluation in Italy.

FEFLOW

FEFLOW is a finite-element model developed by WASY GmBH of Berlin that includes a highly-developed graphical user interface. The finite-element mesh used by FEFLOW can easily accommodate large changes in scale within the model domain. FEFLOW does not suffer from the dry-cell problem; it offers two methods of representing falling groundwater conditions:

- Simulating groundwater flow using an equation for unsaturated or variably-saturated media based on a moveable model mesh.
- Using a “phreatic slice” configuration, where the model mesh is fixed and scaling conductivity according to the saturated thickness in an element approximates unsaturated flow.

The moveable model mesh option is the most likely to be applied to dewatering problems since this permits a shortcut in defining boundary conditions on the uppermost model slice that then moves with the water table. Boundary conditions available within FEFLOW are similar to the basic functions of MODFLOW such as wells, drains and general head boundaries. More complex boundary conditions can be derived but would require integration with a surface water model such as MIKE-11 or bespoke programming.

Evaluation criteria for bespoke dewatering models

It may be that a bespoke modelling code has been used instead of the off-the-shelf packages just described. The evaluation of such a bespoke dewatering model should include:

- Review of benchmarking / testing of the model against analytical solutions and other thoroughly-tested numerical modelling codes to ensure that the model is capable and accurate.
- Consideration of the boundary conditions applied to represent dewatering systems and whether these are suitable and adequate for the required tasks.
- The zone budgeting function that should be capable of performing local and global flow balances during the simulation of dewatering and recovery. These should be checked for consistency and for overall model mass balance.

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Appendix 9: Case studies

As mentioned in Section 1.5 of the report, the HIA methodology was tested on six case studies. The case studies were selected to illustrate the application of the methodology in different hydrogeological settings. One of the case studies is a proposed quarry on a greenfield site; all the others are existing operations. The six case studies are as follows:

i. Slate quarry at Ffestiniog, Gwynedd, operated by Alfred McAlpine Slate.

ii. Opencast coal mine at Margam, near Bridgend, operated by Celtic Energy Ltd.

iii. Karstic limestone quarry at Dove Holes, near Buxton, Derbyshire, operated by CEMEX UK Ltd.

iv. China clay pits near St Austell, Cornwall, operated by Imerys Minerals Ltd.

v. Proposed sand and gravel quarry on the floodplain of the River Kennet near Newbury, Berkshire, which is being investigated by Tarmac Ltd.

vi. Construction dewatering project near Haverigg, Cumbria, undertaken by Project Dewatering Ltd.

Water Management Consultants Ltd is very grateful to the various operators for permission to publish these case studies. Individual acknowledgements are given near the beginning of each case study.

When reading these case studies, the following points should be borne in mind:

- Within the scope and budget of this Science project, it was only possible to apply the HIA methodology at Tier 1 level (first iteration), and each case study finishes with a comment on what would happen next.
- The case studies have had to be kept fairly general to avoid revealing commercially sensitive information.
- In reality, investigations at some of the case study sites have progressed well beyond the end of Tier 1.

In general, the case studies illustrate the following points about the application of the HIA methodology in practice:

- In most situations, Tier 1 is not onerous, and can be undertaken on the strength of existing knowledge, reports and data.
- The majority of the effort goes into the development of the conceptual model.
- The answers under many of the steps of the methodology can be short and to the point, and may even just consist of "Not applicable".
- Tier 1 serves to highlight the gaps in information, and the issues on which further work would need to be done at Tier 2.

Further points are brought out in the individual case studies.

CASE STUDY 1: Ffestiniog Quarry, Gwynedd, North Wales

Introduction

This case study illustrates the application of the HIA methodology to a quarry in fractured crystalline rock (slate in this case), as described in Appendix 4. Aspects of the case study also apply to the discussion on the influence of underground workings in Appendix 5. The Ffestiniog Slate Quarry, incorporating the Gloddfa Ganol and Oakeley workings, is located around 1.5 km north-west of Blaenau Ffestiniog in North Wales (Figure A9.1). It lies at an elevation of around 300 maOD on the steep slope of the western side of the upper reaches of the Afon Barlwyd valley.

Acknowledgement: Water Management Consultants Ltd is very grateful to Alfred McAlpine Slate Ltd for permission to publish this case study, and thanks are due to the staff at Ffestiniog Quarry, especially Mark W Jones (SHEQ Manager) and Dafydd Williams (Quarry Manager) for contributing their knowledge and time.

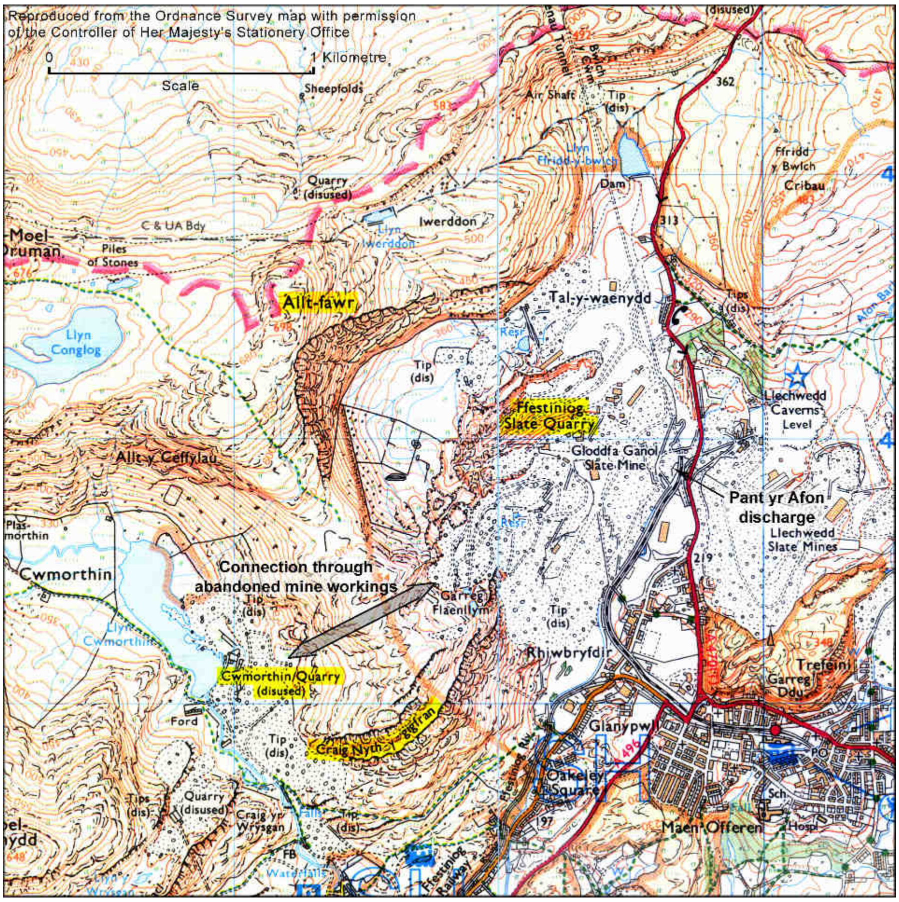

Figure A9.1 Location map for Ffestiniog Slate Quarry

Step 1: Regional water resource status

The Ffestiniog Quarry lies within the Llyn and Eryri CAMS area (No.103). At the time of writing, the CAMS was still in preparation, but the Environment Agency staff responsible for its production were able to predict that the Afon Goedol (into which the Afon Barlwyd flows) catchment would be classified as ‘Over-licensed’. Under the initial Water Framework Directive characterisation, the Afon Goedol was classified as ‘At

Risk'. It therefore falls to this HIA to demonstrate whether or not the dewatering abstraction at Ffestiniog Slate Quarry is part of the regional water resources problem.

Step 2: Conceptual model

The local geology is Ordovician Caradoc Series siltstones and mudstones, hosting igneous intrusions and associated metamorphics, such as slate. The dominant rocks, siltstones and mudstones, have low primary porosity, which contributes little to permeability or storage. Groundwater flow and storage are predominantly within joints and fractures, the occurrence of which varies within formations. It is assumed that before quarrying commenced, groundwater and surface water catchments were more or less coincident in the area.

Extensive historical workings at the site now form a network of adits (and drifts and shafts) within the rocks beneath the higher ground between the Craig Nith-y-gigfran and Allt Fawr peaks which rise to almost 700 maOD to the west and south-west of the quarry. Indeed, the underground workings connect with those from the abandoned Cwmorthin Quarry which is located in the Cwmorthin valley to the south-west (Figure A9.1). These adits act as high-permeability conduits for groundwater flow, and their presence increases the bulk hydraulic conductivity of the rocks beneath and adjacent to the quarry by many orders of magnitude.

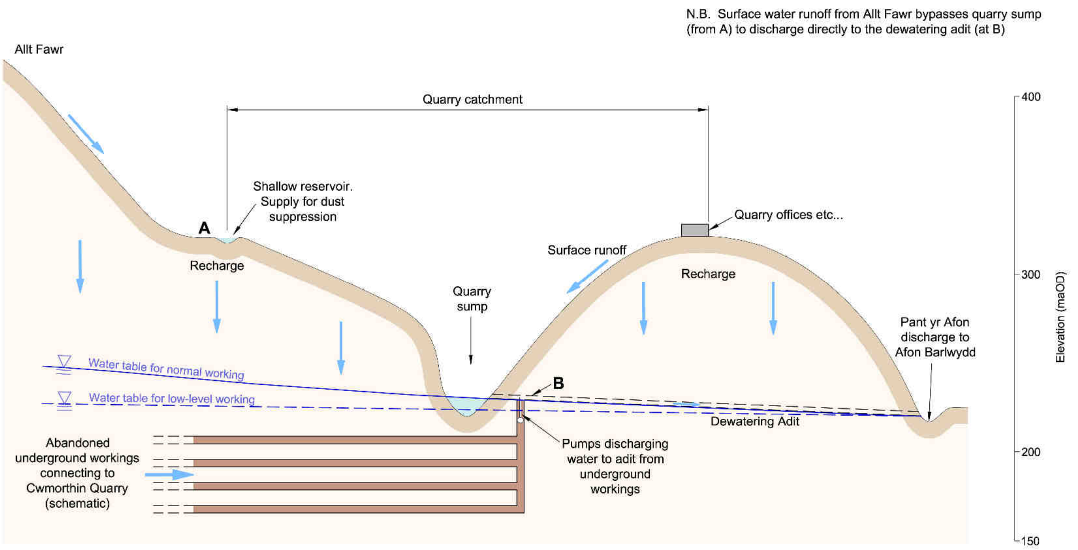

Figure A9.2 Schematic cross-section

The arrangements for dewatering the quarry are shown on the cross-section in Figure A9.2. Water is removed from the quarry sump (Photo A9.1) by gravity flow along a gently-sloping adit which discharges at Pant yr Afon (Photo A9.2, and location shown on Figure A9.1), before flowing immediately into the Afon Barlwyd.

Photo A9.1 Quarry sump

When working is taking place above the level at which the adit enters the quarry (230 maOD), water is removed by gravity alone. When working is below 230 maOD

Photo A9.2 Adit discharge point

in the quarry, water is pumped into the adit by submersible pumps installed in a vertical shaft which intersects the adit a short distance from the quarry sump. There is no direct connection (through mine workings) between the quarry sump lake and the underground workings from which water is pumped, and the intervening strata are thought to act as a filter for water which flows downwards from the quarry sump lake.

The connection to the Cwmorthin Quarry to the south-west through underground workings almost certainly has the effect of extending the groundwater catchment of the Ffestiniog Quarry into the Cwmorthin surface water catchment. It would be very difficult to quantify the amount of flow which derives from this extended catchment. Operations staff at the quarry suggested that the 'water make' from the extended underground workings is relatively small and, therefore, for the purposes of this case study the possible effects of the dewatering of Ffestiniog Quarry on the stream in the Cwmorthin catchment have been ignored.

In order to reduce the amount of water which needs to be discharged from the lowest level within the quarry, a system for surface run-off management has been installed. Historically, this took the form of mountainside surface water drains running across the flanks of Allt Fawr (Photo A9.3). The modern expression of this system is a surface water drain running around the side of the quarry (Photo A9.4) which discharges to the dewatering adit downstream of the quarry sump.

Photo A9.3 Old surface water drains

Photo A9.4 Surface water drain

The effect of the mine workings and drainage adit is simply to increase the bulk hydraulic conductivity of the system and to establish a groundwater level (or drainage level) which, under gravity drainage conditions, is lower than natural conditions by (very) approximately 50 m. Since the system is in 'steady-state' in this condition, the total discharge flow profile (surface water and groundwater) will be very similar to what it would be under natural conditions, albeit with slightly more 'flashy' surface run-off because of the lack of soil and vegetation (shallow storage) within the quarry itself.

An exception to the above is when work is undertaken below the 230 maOD adit drainage level within the lowest part of the quarry. Before this work commences, the water level in the quarry sump lake needs to be lowered by around 15 m. This involves also lowering the water levels, that is, removing the stored groundwater, in the extensive underground workings at the site by pumping. Operations staff at the quarry estimate that for every 1 m reduction in groundwater level below 230 maOD, around 16,350 m^3 of water needs to be removed.

The discharge from the quarry along the adit is not gauged, and the form of the channel emerging from the adit makes it relatively difficult to estimate the discharge (Photo A9.5). During the site visit in early August 2005, the discharge was estimated at around 20 l/s (1,750 m^3/d).

Photo A9.5 Discharge channel

Photo A9.6 Water source for dust suppression

Water for dust suppression is taken from a small artificial lake at the north-western margin of the quarry (Photo A9.6) which is fed by surface run-off. Dust suppression is taken to be a wholly consumptive use of water within the quarry. It is estimated, by consideration of bowser volume, number of refills, number of active days, etc, that the average rate of water use for dust suppression is 31 m^3/d.

The quarry holds a separate abstraction licence (surface water) for water used in the production of slate products. This is taken from a small, high elevation lake (Photo A9.7) just to the south of the quarry. The water is used during cutting and washing (Photo A9.8) of slate, and is extensively recycled through a filter bed system. As this is already licensed, it is not considered further in this HIA.

Photo A9.7 Water source for processing

Photo A9.8: Wetting cut slate

Step 3: Water features susceptible to flow impacts

From the conceptual model, it is considered that the only water feature susceptible to flow impacts from the dewatering activities at Ffestiniog Quarry is the Afon Barlwyd (into which the flow from the dewatering adit discharges).

Step 4: Apportion the flow impacts

Since the Afon Barlwyd is the only water feature identified under Step 3, the flow impacts are apportioned 100 per cent to that feature.

Step 5: Mitigation of flow impacts

As already mentioned, the flow from the dewatering adit is discharged into the Afon Barlwyd, the water feature that is being impacted. The only consumptive activity within the quarry (not already licensed) is dust suppression, so in terms of overall volume, the net flow impact will be the quantity of water used for dust suppression. The average rate of water use for dust suppression is 31 m^3/d.

Step 6: Significance of net flow impacts

Within the scope of this case study (equivalent to the first iteration of Tier 1 of the HIA), there are insufficient flow measurements to allow a full quantitative assessment of flow impacts. However, the following conclusions can be reached:

- The estimated net flow impact represents around 1.8 per cent of the estimated dry weather dewatering discharge in early August 2005. It is also noted that the dewatering discharge flows into the main Afon Barlwyd immediately downstream of the adit mouth, where the water used for dust suppression will represent an even smaller percentage of total flow.
- Since the groundwater catchment of the Afon Barlwyd is assumed to be unaltered by the dewatering operations, and unlicensed consumptive water use within the quarry almost certainly represents less than 1 per cent of the dry weather flow of the Afon Barlwyd, the effect of quarry dewatering on the aggregate discharge of the Afon Barlwyd over the long-term can be considered to be negligible.
- The dewatering discharge is increased temporarily during lowering of water levels to allow working in the lowest levels of the quarry, and will decrease during recovery of water levels after this working has finished. The pump capacity at the quarry is around 19,600 m^3/d, which is slightly more than the dry weather dewatering rate of the quarry under normal conditions. The total dewatering discharge during pump operation will consist of the pump discharge (groundwater and a small amount of surface water) and the water bypassing the quarry sump through the surface water management system. It is necessary to run the pumps for around 13 days to lower the water level by the required 15 m. Once the groundwater storage within the abandoned underground workings has been removed, the pumps are activated only to remove groundwater seepage in order to maintain the new level. It has not been possible during the current study to assess the significance of these increases and decreases in flow in the Afon Barlwyd or the Afon Goedol.

From the evidence available, it is concluded that the impacts on average downstream flows of the dewatering system at Ffestiniog Quarry are probably of small significance. However, the impacts of the increases and decreases in the dewatering discharge before and after working at the lowest levels in the quarry would need to be assessed in more detail if this study were progressing to Tier 2.

Step 7: Search area for drawdown impacts

The search area for drawdown impacts is assumed to be the western side of the upper reaches of the Afon Barlwyd valley. In the same way as for the assessment of flow impacts, it is recognised that there could be drawdown impacts within the Cwmorthin surface water catchment, propagating through the disused underground workings between the Ffestiniog and Cwmorthin Quarries. However, the uncertainty attached to the hydrogeological effects of the disused underground workings mean that a realistic estimate of these drawdown impacts is beyond the scope of the current study (that is, a Tier 1 assessment).

Step 8: Water features susceptible to drawdown impacts

Since the quarry is located in an upland area with exaggerated topography and high rainfall, it is almost certain that any need for small water supplies will be met from surface waters, and no evidence of domestic or other small groundwater abstractions has been found.

The main water body close to the quarry is Llyn Ffridd-y-bwlch which is located at the northern edge of the quarry (Figure A9.1). This llyn is retained by an earth and stone dam (Photo A9.9), and it discharges through quarry waste material and into the quarry dewatering adit, bypassing the lowest level of the quarry through the surface water management channels. The dam which retains the llyn was probably constructed at an early stage during quarrying, and it is not known whether a llyn existed in this location before the dam was constructed. It is thought probably to be an artificial feature.

Photo A9.9 Llyn Ffridd-y-bwlch

There are no further features susceptible to drawdown impacts. Indeed, the remainder of the search area for drawdown impacts is taken up either by steep mountain slopes or the quarry itself.

Step 9: Predict maximum drawdown impacts

In this geology, it is not possible to use the Thiem of Theis equations, so the HIA must fall back on the conceptual model. Consideration of the relative water levels of the llyn and the quarry sump and their horizontal separation reveals an extremely steep apparent hydraulic gradient of 0.1 between the two. This suggests either that the rocks separating the two have an extremely low permeability, or that the llyn is perched above the local water table. In either case, this would suggest that drawdown impacts on the llyn are probably minimal. This is confirmed by the fact that the llyn was still relatively full during a dry period during Summer 2005.

Step 10: Mitigation of drawdown impacts

Not applicable.

Step 11: Significance of drawdown impacts

As already mentioned, the drawdown impacts on Llyn Ffridd-y-bwlch are considered to be minimal.

Step 12: Water quality impacts

A high proportion of surface water run-off from the quarry catchment flows through well-established channels which form the surface water management system, and this avoids any impacts on water quality through increases in suspended sediment.

The quarry sump acts as a settling pond for surface water run-off which evades the surface water management arrangements. The quality of the water which is pumped from the underground workings at the quarry, when working is below the 230 maOD adit level, has not been witnessed during this case study. However, it is known that the current dewatering arrangements have been in place for a long period of time, and therefore it is likely that any sediment, etc, has already been removed from the underground workings. The discharge is monitored and recorded on a daily basis, with a maximum allowable turbidity specified in the discharge consent.

Step 13: Redesign mitigation measures

Redesign of the mitigation measures is not possible within the scope of this case study, but further work at this quarry would be likely to focus on the flow variability in the receiving watercourse, and the impact of the dewatering discharges on the flow variability (rather than the overall flow volume). Any redesign of the mitigation measures would therefore concentrate on this aspect, with the aim of controlling the discharge rate and timing.

Step 14: Monitoring and reporting plan

If this HIA were being taken further, the monitoring plan for Ffestiniog Slate Quarry would focus on the following:

- Quantifying more accurately the consumption of water for dust suppression, simply by keeping a count of how many bowsers are used and when.
- Quantifying the variations in the discharge from the dewatering adit, both the gravity flow component and the pumped component.
- Quantifying the flow variability in the receiving watercourse.

After Tier 1

As already mentioned, if this study were to proceed to Tier 2, the work would concentrate on the flow variability in the receiving watercourse, and the impact of the dewatering discharges on the flow variability (rather than the overall flow volume). It would also be worth assessing the hydrogeological effect of the connection, through abandoned workings, to the Cwmorthin quarry and valley.

CASE STUDY 2: Margam Opencast Coal Mine, South Wales

Introduction

This case study illustrates the application of the HIA methodology to an opencast coal mine, taking into account the issues raised in Appendix 5. Margam Opencast Mine is operated by Celtic Energy Ltd, which has planning permission from Neath, Port Talbot and Bridgend County Borough Councils to extract coal by opencast methods. The mine is located north-east of Kenfig Hill and north of Cefn Cribwr, near Bridgend. The mine is operated as an opencast site, and the method of working involves the excavation moving from east to west, with the active face moving steadily westwards (Photo A9.10). An earlier phase of opencasting, to the east of the site, is now backfilled and restored to ground level (Photo A9.11). Current workings are expected to continue for another 18 months, followed by 2.5 years of restoration. An application has been made to advance the workings further westward, which will extend the workings across the current (natural) course of the Afon Kenfig. The extension will have a working life of approximately 7 years.

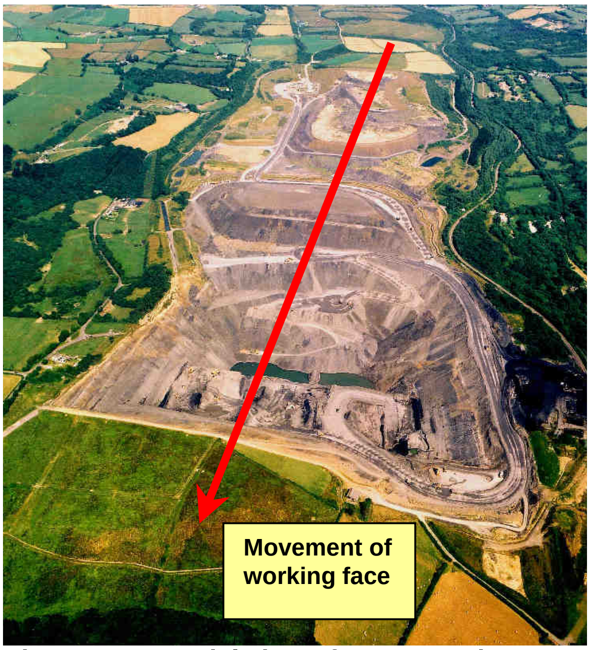

Photo A9.10 Aerial view of Margam mine, looking eastwards

Photo A9.11 Restored former workings

Acknowledgement: Water Management Consultants Ltd is very grateful to Celtic Energy Ltd for permission to publish this case study, and thanks are due to the staff at Margam, especially Adrian Helmore (Mine Manager), Dr Michael Gandy (Planning Manager) and former Principal Geologist Brian Thompson, for contributing their knowledge and time, and the aerial photographs.

Step 1: Regional water resource status

Margam mine lies within the Neath, Afan and Ogmore CAMS area (No.96). The Environment Agency published a consultation document for the CAMS area in March 2005, and a copy of this was obtained for the case study. Margam mine is located in Water Resource Management Unit 4 (Kenfig). Key points relating to this unit are:

- There are two assessment points on the Afon Kenfig: AP12 is near the mouth of the stream near Kenfig Burrrows/Margam Moors; AP13 is at Pyle, approximately 2 km south-west of the mine.
- The upstream reaches of Afon Kenfig (upstream of AP13) are classed as having a Very High sensitivity to abstraction.
- The resource availability status of the stream upstream of AP13 is ‘No water available’. Between AP13 and AP12 the stream is 'Over-abstracted', which reflects high demands by Corus on the lower reaches of the stream.
- Kenfig Pool SSSI is located within the catchment, on the edge of Kenfig Burrows.

No mention is made in the CAMS document of quarries or mines, as they were not licensable at the time. Discussion with Environment Agency staff indicates that it is assumed that dewatering is non-consumptive, with water from dewatering being returned directly to surface waters.

Step 2: Conceptual model

The natural system

The geology in the vicinity of Margam mine comprises Millstone Grit, Lower and Middle Coal Measures and Upper Coal Measures. The mine is excavated in, and follows the strike of, the Lower and Middle Coal Measures where the strata crop out in the bottom of the valley. Ground elevation around the mine is about 70 maOD. In addition to the coal, these strata comprise interbedded shales, siltstones and sandstones (Photo A9.12). The bulk of the strata comprise low-permeability shales and siltstones. Groundwater occurrence is likely to be mainly within thin sandstones, and groundwater flow is predominantly via faults and fractures. In the vicinity of the mine, groundwater flow is highly modified by the presence of abandoned underground mine workings.

Photo A9.12 Working face

The strata within the mine are underlain by strata of the Millstone Grit which crop out to the south of the site forming Kenfig Hill, rising to elevations of around 125 to 130 maOD. To the north, the Lower Coal Measures are overlain by the Upper Coal Measures, forming hills rising to around 250 to 300 maOD. The mine is bisected by a small stream, Nant Craig-y-Aber, the natural course of which has been diverted between the main overburden tip and the current surcharge tip (Photos A9.13, A9.14). The Afon Kenfig follows a course to the west of the current working area. The course of this stream will be diverted during Phase 9. Key features of the conceptual model are summarised in Figures A9.3 and A9.4.

Inflows to the Coal Measures are likely to include:

- Direct precipitation.

- Surface run-off from the surrounding hills. Little or no direct run-off will reach the mine itself, being intercepted by natural watercourses and artificial drainage ditches.
- Stream leakage.
- Upward flow from the underlying Millstone Grit.
- Potential downward leakage from the overlying Upper Coal Measures.

Photo A9.13 Nant Craig-y-Aber, diverted channel around surcharge tip

Photo A9.14 Nant Craig-y-Aber, to the south of the surcharge tip

Outflows include:

- Evapotranspiration.
- Groundwater flow along the strike to natural discharge area (presumably somewhere in the vicinity of Kenfig Burrows/Margam Moors). Some natural discharge may occur to surface watercourses (for example Coal Brook) to the west of the mine.
- Mine discharges. A substantial discharge (estimated at 117 l/s) sometimes occurs from the now disused Bryndu Engine Shaft, located just west of the existing pit. Current dewatering operations intercept this discharge so that while Margam mine is actively worked there is no discharge from this point. The Bryndu shaft is believed to be the lowest point in the system and natural overflow via this point is expected to resume following cessation of dewatering operations.

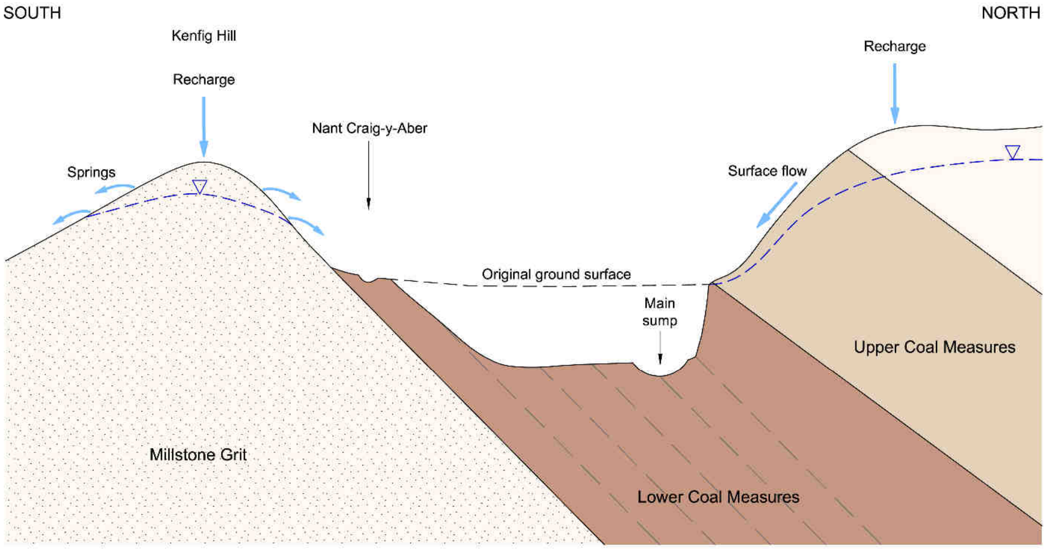

Figure A9.3 Schematic cross-section through Margam mine

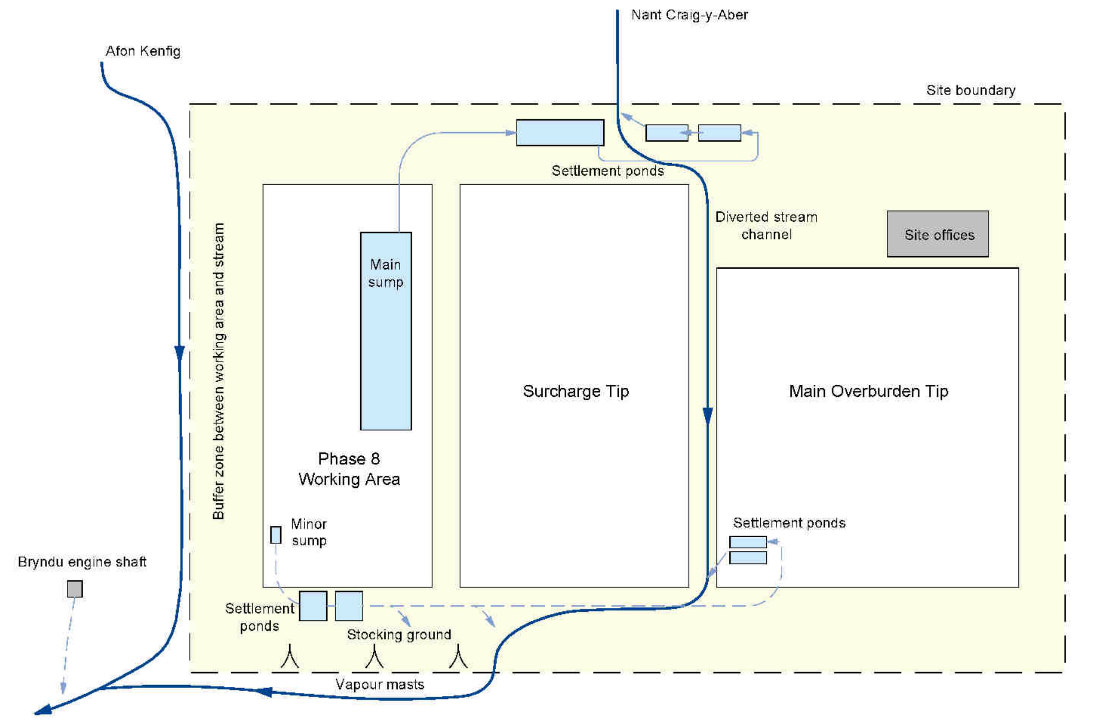

Figure A9.4 Schematic plan showing key elements of the water management system

Dewatering operations

Dewatering operations are simple and well managed. Dewatering of the active pit occurs from two locations:

i. Celtic Energy operates a main sump in the bottom of the active pit, currently in Phase 8 (Photo A9.15). Water is pumped from the sump by a high-head high-capacity pump fed by a submersible pump (Photo A9.16) via treatment/settlement lagoons (Photos A9.17, A9.18) on the north side of the site and then discharged into the Nant Craig-y-Aber (see Figure A9.4 and Photo A9.19). No use is made of this water, and there are no losses from the system between the sump and discharge point.


Photo A9.15 Main sump in 2003


Photo A9.16 Main sump in Phase 8 working area

ii. A second, smaller, sump is located at the base of the west wall in Phase 8 of the workings (Photo A9.20). This sump drains water from old deep mine workings. The pump at this location has a smaller capacity than the main pump and tends to pump to lagoons in the coal stocking area during the night. The pump is switched off during working hours when the sump is used as a source for the site's water bowsers. This smaller sump provides water for dust suppression which is used in two ways: water bowsers to dampen roadways (Photo A9.21); and mist-mast sprays along the perimeter of the site (Photo A9.22) to minimise windblown dust leaving the site. Any water not used for dust suppression is pumped to attenuation lagoons on the southwest corner of the main overburden tip (Photo A9.23), from where it discharges into an unnamed stream which flows into the Nant Craig-y-Aber.

Photo A9.17 Settlement ponds in treatment area on north side of site

Photo A9.18 Settlement pond

Photo A9.19 Ochreous staining in Nant Craig-y-Aber with inflow from treatment area

Photo A9.20 Minor sump with pump; pipeline leading to vapour masts and attenuation ponds

Photo A9.21 Dust suppression on haul roads by bowser

Photo A9.22 Vapour masts at stocking ground/railway loading area

The site and dewatering operations are in general well managed, although little monitoring is undertaken of dewatering volumes. Although the main pump is fitted with a meter, readings are taken only at sporadic intervals. Based on readings taken over a 6-month period (believed to be March – September 2005), Celtic Energy reports that the average daily pumped volume is approximately 9,000 m^3. This water is transferred directly (via treatment lagoons) to Nant Craig-y-Aber and is non–consumptive. The amount of water pumped from the smaller sump is far more difficult to calculate as there is no meter on the sump pump or on the dust suppression plant. Celtic Energy estimates that the average daily volume removed from the sump is 4,500 m^3. The quantities used for dust suppression are estimated to range from nothing on wet days to approximately 360 m^3 on very hot days. Bowsers operate on approximately 30 to 40 per cent of days worked, within which varying quantities of water will be used. There is no estimate of the amount of water used in dust suppression by mist sprays. Mist sprays and bowsers are likely to be regarded as 100 per cent consumptive, although some of the water spread by bowsers will ultimately return to the groundwater system.

Photo A9.23 Attenuation lagoon

From the conceptual model, the main issues relating to Margam mine appear to be: confirmation of the volumes of water used for dust suppression; potential impact on flow in the streams that cross the site (Nant Craig-y-Aber, Afon Kenfig); and potential impact on water quality in streams that flow across or close to the site (Nant Craig-y-Aber, Afon Kenfig).

Step 3: Water features susceptible to flow impacts

From the conceptual model, it is considered that the water features susceptible to flow impacts from the dewatering activities at Margam mine are the streams Nant Craig-y-Aber and Afon Kenfig. There is also a possibility that the marshes in the Margam Moors and Kenfig Burrows areas could also be affected, if their water supply includes natural groundwater discharge from the Lower Coal Measures.

Step 4: Apportion the flow impacts

The abstraction of groundwater via dewatering of the current operational open pit, and overflows through the Bryndu Shaft as a result of historic mining operations must mean that there is a reduction in the natural flow through and discharges from the Lower Coal Measures. This could be manifested as: reduction in flows to streams and watercourses in the vicinity of the mine; or reduction in natural discharge from the groundwater system.

Step 5: Mitigation of flow impacts

The discharge from the main sump is direct to Nant Craig-y-Aber, which then flows into the Afon Kenfig. Some of the discharge from the second, smaller sump is returned to surface watercourses via attenuation lagoons on the main overburden tip. The remainder of this discharge is utilised for dust suppression (bowsers and mist sprays); the relative volumes, and hence the net flow impacts, are not known.

Step 6: Significance of net flow impacts

In terms of flows to streams, it is likely that flows are actually enhanced as a result of dewatering operations. The Afon Kenfig is classed as having 'No water available' in its upper reaches and 'Over- abstracted' in its lower reaches. Hence direct transfer of water out of Margam mine can be viewed as a positive benefit. A potential problem could arise on cessation of dewatering, when enhanced flows in the stream will cease.

However, this may be balanced by resumption of outflows from Bryndu Shaft which also discharges to the Afon Kenfig. The effect of reducing natural discharges in the down-gradient part of the system is difficult to assess. Potential effects could be a reduction in freshwater inputs to the marsh in the Margam Moors area. No assessment has yet been made of this.

Step 7: Search area for drawdown impacts

The search area for drawdown impacts is assumed to extend to the west and east of Margam mine, along the strike of the Lower Coal Measures. Drawdown impacts are unlikely to occur to the north and south of the mine where the land surface rises steeply over the Upper Coal Measures and Millstone Grit respectively. The extent of the search area is poorly defined, but could extend to Tondu / Aberkenfig in the east and towards the M4 motorway in the west.

Step 8: Water features susceptible to drawdown impacts

From the conceptual model, it is considered that there are no water features susceptible to drawdown impacts likely to be affected by dewatering at Margam mine. There are no groundwater abstractions in the vicinity of the mine, and none from the Lower Coal Measures. A number of springs issue from the Millstone Grit on Kenfig Hill, however, these are all well above the level of the pit and will not be affected by dewatering.

Step 9: Predict maximum drawdown impacts

Not applicable.

Step 10: Mitigation of drawdown impacts

Not applicable.

Step 11: Significance of drawdown impacts

Not applicable.

Step 12: Water quality impacts

Water is pumped from the main sump into Nant Craig-y-Aber, after passing through treatment/settlement lagoons. Upstream of the discharge point the stream is adversely affected by historic mine discharge, with ochreous discharges coming from old mine workings in the Upper Coal Measures (Photo 9.19). The current discharge from Margam mine actually improves the water quality, by diluting the poor quality water, resulting in better water quality downstream of the mine.

Step 13: Redesign mitigation measures

The course of the Afon Kenfig will be diverted during Phase 9. Any redesign of mitigation measures should concentrate on ensuring that there is no impact on stream flow or stream water quality. Conservation of the natural ecology of the stream may also need to be taken into account.

Step 14: Monitoring and reporting plan

If this HIA were being taken beyond the limited scope of this case study, the monitoring plan for Margam Mine would need to focus on:

- Quantifying the volume of water used for dust suppression. For water bowsers, this could be achieved by simply keeping a record of how many bowsers are used and when. The volume of water used in mist sprays will be more difficult to quantify, and may require that meters are installed at key points within the water supply system.
- Monitoring of the effect of mining and dewatering on re-routed sections of the Nant Craig-y-Aber and Afon Kenfig. The main question is whether re-routing of the streams results in loss of flow through the stream bed, which could be monitored though a programme of streamflow monitoring at the upstream and downstream ends of the re-routed sections.
- Monitoring the effect of dewatering discharges on water quality in the receiving streams.
- Monitoring the effect, if any, on the ecology of the stream, for example on migration of fish upstream for spawning. A baseline ecological survey may need to be undertaken before the Afon Kenfig is re-routed, with follow-up surveys at suitable times thereafter.

After Tier 1

If this study were taken to Tier 2, the focus would be on collecting and incorporating the data mentioned under Step 14, and on establishing the likely extent, if any, of impacts on wetland areas in the Margam Moors and Kenfig Burrows areas.

CASE STUDY 3: Dove Holes Quarry, near Buxton, Derbyshire

Introduction

This case study illustrates the application of the karst HIA methodology (the revised set of seven steps) to a limestone quarry, as described in Appendix 3. Dove Holes limestone quarry is located around 5 km north-east of Buxton in Derbyshire (Figure A9.5). The site lies on a limestone plateau area incised with steep-sided valleys, some of which are dry. The elevation of the plateau is typically 300 to 400 maOD; the valley floors are typically 100 m lower. The area is known to be karstic, with several recorded stream sinks and shakeholes.

Acknowledgement: Water Management Consultants Ltd is very grateful to CEMEX UK Ltd for permission to publish this case study, and particularly to Chris Pointer (Senior Hydrogeologist) and Chris Firth (Quarry Manager). Much of the information for this case study was taken from various reports prepared by RMC Aggregates (UK) Ltd.

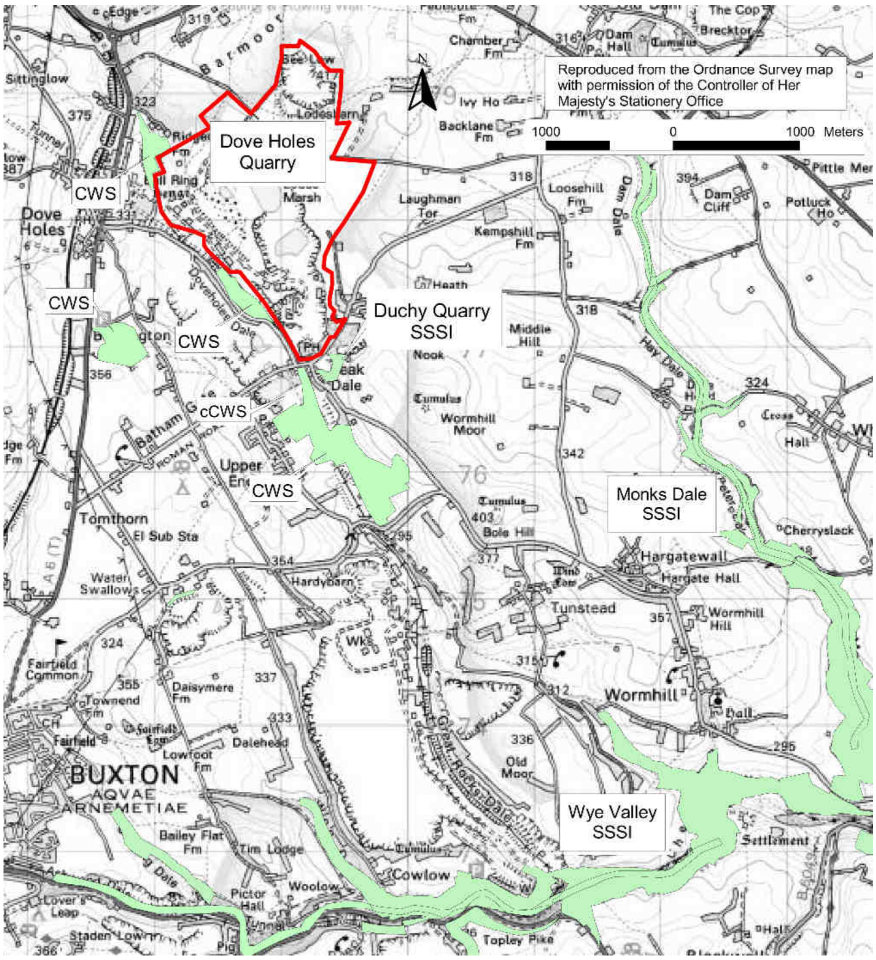

Figure A9.5 Location of Dove Holes quarry and potential sensitive features

The first quarry at Dove Holes opened in 1794, and by the 1840s a number of separate quarries had developed. Lime burning took place at the majority of these. Several have now been restored and are recognised County Wildlife Sites (CWS), many of which are designated for features such as marshland or standing/open water. In the distant past, mineral veins in the area have also been exploited, and there are several infilled shafts in the areas to the north of the main quarry (Photo A9.24). The extent of these workings is not known, but where they are present they may provide preferential pathways for groundwater flow.

Photo A9.24 Infilled shaft from old mineral working (~2 m diameter)

The consent for Dove Holes quarry was originally granted in 1952. Between 1982 and 1998 the former Peak and Holderness quarries were expanded to form a single operational area. This is being progressed northwards to join up with the Bee Low area. All present workings are above the water table, but the quarry has plans for future phases which will dewater sections of the quarry to 60 m below the current water level. The quarry is rail-linked, and a substantial volume of aggregate leaves the site in this way. Close to the quarry the rail tracks run through a steep cutting and a tunnel.

Step K1: Regional water resource status

Dove Holes quarry lies within the Derbyshire Derwent CAMS area (No.30). At the time of writing the Environment Agency had completed a consultation document for this CAMS area. The River Wye Water Resources Management Unit (containing Dove Holes) has the status of ‘Water available’ overridden to ‘No water available’. The override recognises that although there are no abstraction-related problems within the River Wye WRMU itself, the surplus water is required to meet more critical river flow objectives downstream on the River Derwent. The Tame, Goyt and Etherow CAMS area (No.111), to which some water from the railway tunnel runs, has the status of 'Water available'.

Step K2: Conceptual model

The natural system

The geology of the area is Millstone Grit over Carboniferous Limestone. The Millstone Grit forms a steep unconformable contact with the limestone units 1.5 km to the west of the centre of the quarry. The principal limestone units at Dove Holes quarry are the Miller's Dale Limestone and Chee Tor Rock, which are separated (where it is present) by the Lower Miller’s Dale Lava. The weathered lava is clay rich and has acted as a barrier to the migration of fine material into the underlying Chee Tor Rock, which as a result is much cleaner than the overlying Miller's Dale Limestone. Where the lava is not present it is not possible to distinguish between the limestone formations, and they are collectively known as the Bee Low Limestone.

Dove Holes quarry lies within the western margin of the Derbyshire Dome, and Carboniferous strata typically display gentle folding and low dip angles. The quarry lies within a triangular formation of three folds. The Peak Forest Anticline is located to the east of the quarry, with an axis trending approximately north-south. The Wormhill Moor Syncline is located immediately south-east of the quarry, with an axis trending approximately northwest-southeast. The main trend of bedding at Dove Holes quarry is a gentle south-westerly dip. Superimposed on the main trend is the Bee Low Anticline which trends west-east and passes through Bee Low Quarry.

BGS mapping shows two main sets of faults in the area around Dove Holes. An east-west trending set is associated with vein mineralisation, and a second set is mapped as trending northwest-southeast. In addition to localised faulting within the quarry, drilling has indicated the possible presence of a significant fault running northwest-southeast across the site.

Within the quarry workings there are significant differences in the concentration of fractures in the limestone. Most of these fractures are small and have not opened significantly. The only example of a significant opening in the quarry is seen at the base of a triangular block where a potential cave has opened up due to slippage along a faulted zone (Photo A9.25). The opening could not be approached, so was not inspected closely, but it is above the water table so any potential passageway at this location will probably not contribute to groundwater flow. None of the fractures show signs of enhanced seepage (that is, preferential pathways), but the workings are still mostly within the natural unsaturated zone.

Photo A9.25 Opening in quarry wall

There are 10 known caves and 14 sinks (active or relic) within 5 km of the quarry. In December 2005 a new sinkhole opened near to the entrance to the quarry. The hole is approximately 3 m wide and 1 m deep and has sunk as a competent block containing a small tree (Photo A9.26). It is possible that this subsidence is related to the railway tunnel which passes close by. Within the working quarry no exposures of major cave passages either open or obstructed have been found in recent years. It is not known if any passages were found in the past.

Photo A9.26 Sinkhole that appeared in December 2005

Overburden is thin, on average 1.1 m, and limestone infiltration capacity high; recharge into the limestone is high and surface drainage is generally absent. Where the overburden has been removed it can be seen that there is epikarst developed, the surface of which is uneven and highly fractured (Photo A9.27).

Photo A9.27 Overburden removed, showing surface of epikarst

Although it is difficult to assess groundwater flow direction in karst, the regional groundwater flow is generally to the south-east, but it is thought that flow is affected by the folds surrounding the quarry. Flow seems to be deflected by the anticlines and concentrated along the Wormhill Moor synclinal axis. Baseflow to the River Wye forms the principal discharge mechanism. Groundwater levels across the site, taken in 2001, fall from 320 m (aOD) at the north-west of the site to 270 m at the south-east.

The main sump lies at a level of 292 m. Monitoring of boreholes around the quarry has shown that water levels can vary annually by over 20 m.

The River Wye, 5 km south of Dove Holes quarry, forms the main watercourse of the area (Figure A9.5). This is a gaining river, and thought to be largely sustained by baseflow from the Carboniferous limestone. A minor watercourse rises in Dove Holes village, runs alongside Dale Road, and discharges onto the railway track drainage. Most water then runs southwards towards the River Wye. In the past the watercourse disappeared into shakeholes adjacent to the railway line. It would appear that these are now partially blocked, as flows now continue past this point before disappearing into a shakehole near Tunstead. It is considered that flow in the watercourse is often maintained by surface water run-off from the road and the consented discharges to the watercourse.

In Dove Holes village there are two stream sinks. It is virtually certain that the water originally flowed to the River Wye, however when the Dove Holes railway tunnel was constructed it intersected several fissures that discharged significant flows of water. Some of this water now drains northwards through the tunnel and discharges into the River Goyt via the Black Brook. As a result of the tunnel construction, groundwater levels in the vicinity have been artificially lowered and water has been taken out of the River Wye catchment.

A water features survey from 2003 revealed numerous springs and ponds in the area. Although many of these springs are associated with the River Wye, springs and ponds on the limestone in the vicinity of the quarry are thought to be caused by igneous deposits acting as local aquitards resulting in perched water.

Dewatering operations

The majority of water used within the quarry is taken from the quarry sump, with a small quantity of mains water brought in for potable use. The water collected is a combination of rainfall into the quarry, and groundwater that has moved through the fractures in the limestone. Water is pumped from the deepest quarry sump (Photo A9.28) into a holding tank at the top of the site where it is then piped to the various quarry processes, discharged to the Dale Road watercourse, or returned by gravity to the sump. After being used for the non-consumptive purposes, the water is allowed to settle in a separate lagoon (Photo A9.29) before returning to the main sump.

Photo A9.28 Main quarry sump pump

Photo A9.29 Settling lagoon

Water is used for several purposes within the quarry:

- Washing crushed rock (Photo A9.30) – Most of this water is recycled into a settling lagoon.

- Concrete block, slab, and walling manufacture (wholly consumptive as the water is locked into the finished product).

- Ready-mix concrete manufacture (wholly consumptive as the water is removed in the finished product).

- Dust suppression, on the metalled quarry roads using sprinklers (Photo A9.31), within the quarry using bowsers, and around the railhead (assumed to be 100 per cent consumptive).

- Wheel/vehicle washing (Photo A9.32). Some water will be lost on the vehicles but most is recovered and recycled.


Photo A9.30 Washing crushed rock


Photo A9.31 Dust suppression on roads


Photo A9.32 Wheel washing facility

It is estimated that average total consumption of water is 0.36 Ml/d. In terms of annual quantities consumed by various activities, this breaks down as follows:

Process	*Annual consumption (m^3)*
Washing plant	63,600
Concrete block, slab, and walling plants	9,450
Concrete plant	730
Haul road dust suppression	4,000
Coating plants dust suppression	1,320
Secondary processing and railhead dust suppression	52,000
Total water consumed	**131,100**


Photo A9.33 Dale Road discharge

Water is discharged from the site at two consented discharge points:

- Excess water from the sump is discharged to the Dale Road watercourse (Photo A9.33) via the main holding tank. The volume discharged annually is about 650,000 m^3 (an average of 1.78 Ml/d).

- The run-off generated from the road and car-park near to the site entrance is allowed to settle (Photo A9.34) before being discharged to the Dale Road watercourse upstream of the main discharge point. It is not expected that much of this water will be linked to other processes in the quarry. The volume discharged annually is about 15,300 m^3. (an average of 0.04 Ml/d).

Photo A9.34 Pre-discharge settling tank

In addition to these, an unquantifiable volume of water will be lost due to evaporation from the three large lagoons, and groundwater outflow towards the railway tunnel and River Wye. The way water is managed around the site is illustrated schematically in Figure A9.6.

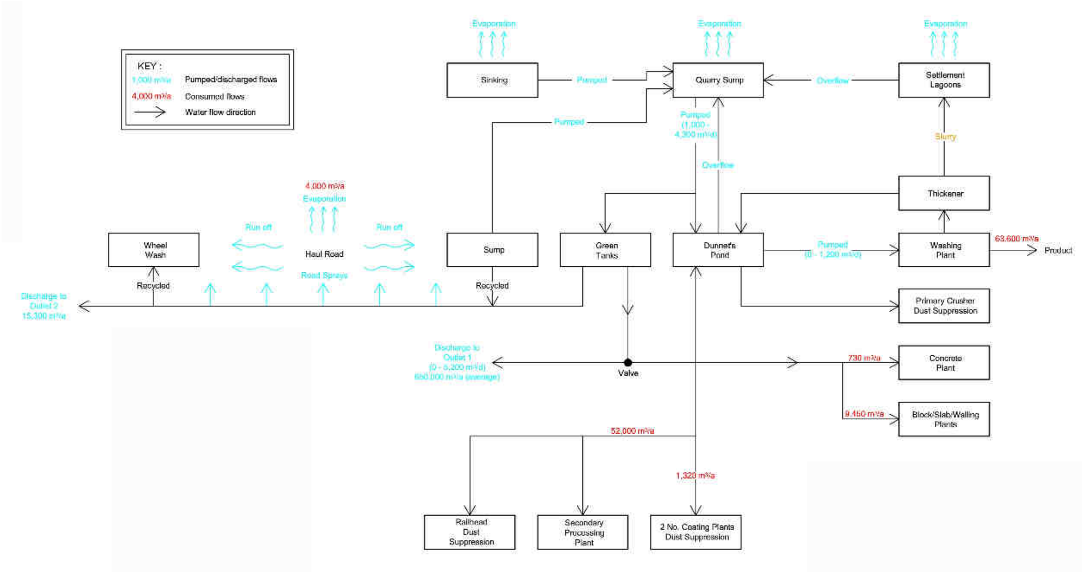

Figure A9.6 Schematic diagram of water management system *(provided by CEMEX)*

Step K3: Identify sensitive sites

Within a karst environment it is very difficult to define an area where impacts could occur. Due to the unpredictable nature of the flow paths it is possible that features near to the site could be unaffected whereas those further away may be impacted. The identification of key sites is likely to involve a more detailed study of each feature. However the following points can be made at this stage:

- The water level within the sump is currently kept at approximately 292 m AOD. Water levels within the quarry have been kept at this level since 1988, so it is unlikely that there have been any major increases in impact since that time.

- At a catchment/CAMS scale the majority of the water from the quarry is not consumed and it will continue to eventually find its way into the River Wye, mainly via the Dale Road watercourse. Therefore the River Wye may be impacted in areas close to the quarry but further downstream the impact of the quarry on flows would be greatly reduced.

- Locally the marshes and ponds making up the County Wildlife Sites close to Dove Holes quarry (Photo A9.35) could be susceptible to drawdown impacts if they are hydraulically connected to the regional water table. However, it is thought that the marshes are rainwater fed and lie on low-permeability material associated with the historical quarrying (and are therefore effectively 'perched'). This will need to be verified before they can be considered free from potential impacts.

Photo A9.35 County Wildlife Site

- There are also a number of licensed abstractions close to the quarry. The two closest licences are within 1 km and both take water from spring sources. The elevations of the springs are above the estimated regional water table, indicating they are likely to be perched. The licensed volumes of the closest sources are less than 10 m^3/d.

- Approximately 3 km to the east of Dove Holes quarry lies Monk's Dale SSSI (Figure A9.5). The site is designated for several features including a number of springs and flushed sites which support tufa-forming bryophyte communities. The stream dries for much of its length during the summer, however lower down the dale the less permeable Miller's Dale Lava is exposed, and in these areas the stream may flow all year.

Step K4: Commence preliminary monitoring at those sites

The quarry is surrounded by monitoring boreholes which already record groundwater levels on a regular basis. If surface water sites are thought to be at risk the monitoring may be expanded to include installation of gauge boards on the ponds, and flow gauging on the streams.

Step K5: Design and demonstrate effective mitigation measures for the sensitive sites

It is not possible within the scope of this case study to design or demonstrate the best option for mitigation, but further work could include

looking into features such as recharge trenches and injection wells to maintain groundwater levels in the area. As the streams and rivers have a large component of baseflow (the River Wye at Ashford has a BFI of 0.76), groundwater recharge may be more effective than stream augmentation.

Step K6: Specify trigger levels for the sensitive sites

This is beyond the scope of this case study.

Step K7: Continue surveillance monitoring at the sensitive sites

This is not applicable at this stage.

Step K8: If necessary, implement mitigation measures when trigger levels have been passed

This is not applicable at this stage.

After Tier 1

If this study were to proceed to Tier 2, the work would concentrate on a more detailed assessment of the potentially sensitive sites mentioned under Step K3 above. Depending on the outcome of this study there may be a need to install additional monitoring equipment, and agree mitigation strategies with the relevant authorities (including demonstrating the effectiveness of the proposed mitigation measures).

CASE STUDY 4: China clay operations near St Austell, Cornwall

Introduction

Rather than running through the steps of the HIA methodology, this case study concentrates on the issues associated with applying water balances to the china clay operations near St Austell, Cornwall, specifically those operated by Imerys Minerals Ltd. Among UK industrial minerals, the extraction and processing of china clay is perhaps unique for the scale and history of the operations within a relatively small geographical area. China clay has been mined in Cornwall for at least 250 years, and there is in fact one pit that has been continuously in operation for about 240 years. The china clay industry plays a vital part in the local and national economy, currently being worth about £250 million in exports. The china clay operations near St Austell dominate the landscape of an area of about 85 km^2.

Acknowledgement: Water Management Consultants Ltd is very grateful to Imerys Minerals Ltd for permission to publish this case study, and thanks are due to the staff in Cornwall, especially Mandy Gore, Roy Taylor and Chris Varcoe, for contributing their knowledge and time.

China clay and water

Water is critical to the mining and processing of china clay. High-pressure water jets are used to extract the clay from the working faces in the pits (Photo A9.36). Water is used to transport the clay from pit to refinery in pipelines in the form of a slurry (Photo A9.37). Water is also used in the processing of the different grades of clay, and is used to flush pipelines. Over the years, a complex network of pipelines, pumps, reservoirs, storage tanks, settlement lagoons, sumps and discharge points has grown up, covering the whole area of operations (Figure A9.7). It is estimated by Imerys that there are about 450 million litres of water in circulation at any one time, being shifted around by many hundreds of pumps, 24 hours a day. The system is also very dynamic, with sump pumps and water jets constantly being moved, with many pipelines able to be pumped in either direction and used to pump clean water, clay slurry, effluent or residue.

Photo A9.36 High-pressure water jet

Photo A9.37 Pumping of slurry

Water is frequently discharged into intermediate storage lagoons or reservoirs, then re-abstracted later for a different purpose (Photo A9.38). Mains water is also used for some purposes, along with separate (and already licensed) surface water sources.

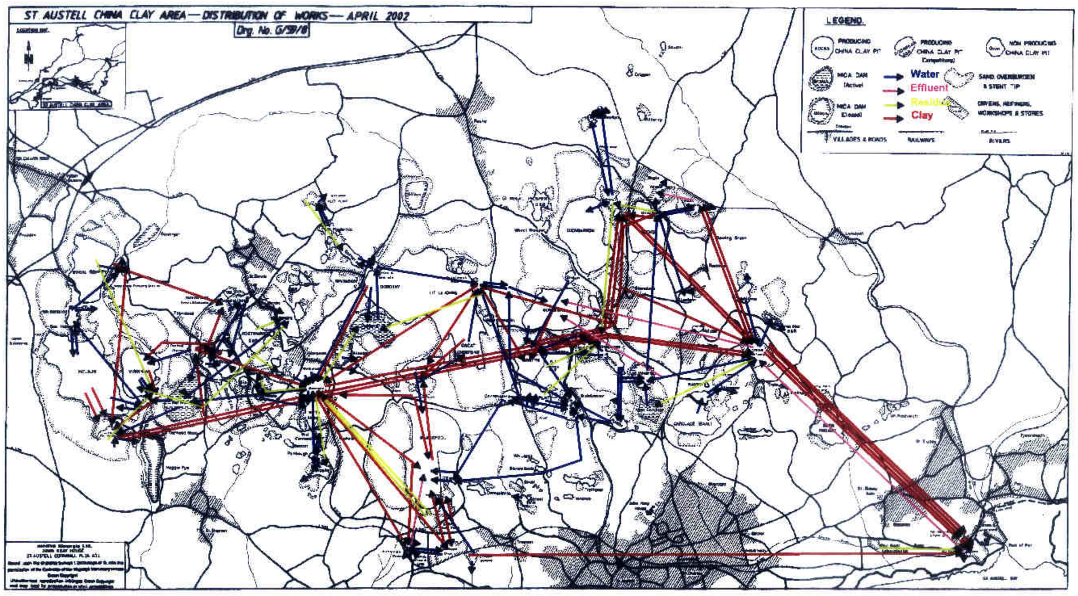

Figure A9.7 Diagram of main water transfers (diagram provided by Imerys Minerals Ltd)

Photo A9.38 Abstraction point in lagoon

Water balance approach

With such a complex and dynamic operation, it is virtually impossible to separate the water already in the system, and being constantly recycled, from 'new' groundwater seeping into the pits and being abstracted by the dewatering pumps. The usual approach to operational water balances described in Appendix 6 (account for the total volume of water pumped from the dewatering sumps or boreholes) is not feasible, for the following reasons:

- The system is so complex and dynamic that it is not practicable to keep track of every single dewatering abstraction and transfer, let alone the hours of operation and quantities pumped by hundreds of pumps.
- Even if all abstractions were monitored, the fact that water is temporarily discharged and then re-abstracted so many times as it moves around the system would lead to a lot of double-counting of 'true' abstraction quantities.

The only realistic approach to an assessment of the hydrogeological impacts of the china clay operations is to concentrate on quantifying the overall consumption of water, and working backwards to the sources of that water. The components of the water balance then reduce down to the following:

Consumptive outputs

- Water leaving the system in the final products (typically about 10 per cent water content).
- Water lost to the atmosphere when drying clay products in kilns.
- Vehicle and wheel washing (the make-up water).
- Dust suppression (if consumptive through evaporation).

Potential inputs (sources of water)

- Mains water.
- Separately licensed surface water sources.
- Groundwater abstracted from separate boreholes.
- Water abstracted from the excavations by sump pumps.

The time period chosen for the water balance should be such that the change in storage in the system can be safely ignored (a 'water year', from October to September, for example), and the inputs should then balance the outputs. For Input 4 (abstraction from the sumps), there is of course the issue of separating out the proportion obtained from groundwater from the proportion originating in surface run-off and direct rainfall into the excavations, which is significant, given the size of the excavations (Photo A9.39).

Photo A9.39 China clay excavation

In fact, water levels can rise so quickly in response to heavy rainfall that pumps have to be mounted on rails so that they can be moved quickly out of harm's way (Photo 9.40). In addition, a significant proportion of this water is recycled. The water balance calculation now proceeds as follows:

Photo A9.40 Sump pump mounted on rails

- Estimate or measure all the components apart from Input 4.
- Quantify that component (abstraction from the sumps) by 'out-of-balance'.
- Estimate the proportion of Input 4 derived from groundwater (as opposed to surface run-off and direct rainfall), using the BFI technique from Appendix 6.
- Add Input 3 to the groundwater-derived part of Input 4 to obtain an estimate of the total consumptive use of groundwater.

Incidentally, the BFIs for the River Fal at Tregony and the River Fowey at Restormel are 0.66 and 0.63 respectively (Hydrometric Register and Statistics 1996-2000, Centre for Ecology and Hydrology).

At the beginning of Appendix 6, relying on 'out-of-balance' to quantify poorly-understood water balance components was discouraged, but this case study shows that it is sometimes the only practicable way to approach the situation. The important thing is to ensure that the logic of the water balance is consistent, with the system well defined, with all the components correctly identified, and with all quantities checked to see if they are physically reasonable.

This water balance would have to be interpreted in the wider context of the HIA methodology, bearing in mind that the surface water catchments in the St Austell area have been anthropogenically altered over a period of 250 years. This is inevitably reflected in the observed flow regimes of the local rivers, which long ago adjusted to the presence of the china clay pits. Mitigation measures in such a situation would probably focus on dealing with specific environmental issues, by careful control of water quality for example, and by optimising the location and timing of discharges.

CASE STUDY 5: Chamberhouse Farm, Newbury, Berkshire

Introduction

Chamberhouse Farm is a greenfield site located approximately 5 km east of Newbury, Berkshire, partly within the floodplain of the River Kennet, and within an area of ecological importance (Figure A9.8). To the west, south and east the site is bounded by the River Kennet, and to the north by the Kennet and Avon Canal. River Terrace and Valley Sands and Gravels form the economic mineral of the Chamberhouse Farm site. The area contains several historic and active sand and gravel pits, and the site has preferred area status in the Minerals Local Plan.

Acknowledgement: Water Management Consultants Ltd is very grateful to Tarmac Ltd for permission to publish this case study, and to Gavin Chaplin of BCL Consultant Hydrogeologists Ltd and Toby Gill of Capita Symonds for provision of various reports which formed the source material for the case study.

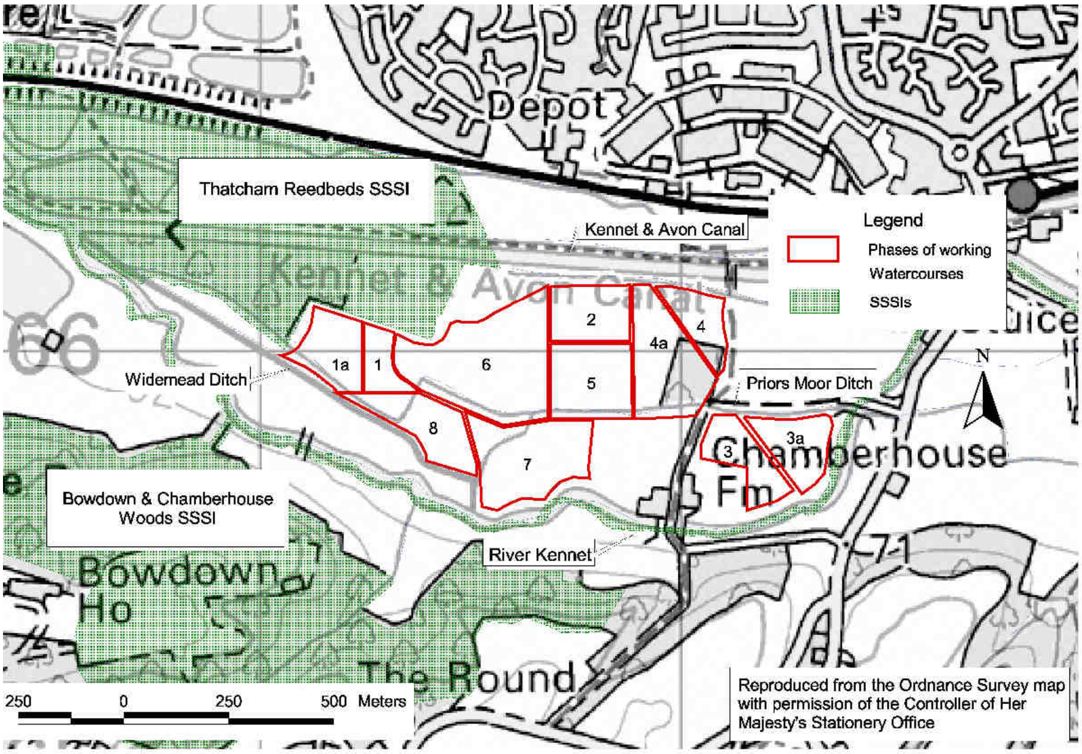

Figure A9.8 Location of Chamberhouse Farm in relation to potentially sensitive features (Phase boundaries are purely for illustrative purposes, without prejudice to any future planning application)

Step 1: Regional water resource status

Chamberhouse Farm lies within the Kennet and Pang CAMS area (No.47). In the CAMS documents of May 2004 the Lower Kennet assessment point is 'Over-licensed' for both surface water and groundwater. In the annual review of June 2005 the resource status has been reclassified as 'No water available' for surface water and 'Over-licensed' for groundwater, following the revocation of a large surface water licence. The water resources management unit contains the River Kennet SSSI, Bowdown and Chamberhouse Woods SSSI, and Thatcham Reedbeds SSSI, part of the Kennet and Lambourn Floodplain cSAC. All contain water-dependent features. It therefore falls to this HIA to demonstrate whether or not the proposed workings at Chamberhouse Farm would add to the regional water resources problem, or have adverse impacts on the conservation sites.

Step 2: Conceptual model

The economically important River Terrace and Valley Sands and Gravels are present within a broad 2-km wide belt with an average thickness of 3.9 m recorded from mineral evaluation boreholes. The sand and gravel deposits are described as poorly-sorted grey flint gravel with 10 - 20 per cent coarse-grained clean sand. They are classified by the Environment Agency as a secondary aquifer with soils of high leaching potential. Alluvium overlying the sand and gravels comprises silty clays with occasional peat, varying in thickness from 0.4 to 1.8 m.

Dipping gently to the east, the underlying solid geology comprises London Clay, underlain by the Reading Beds, which in turn are underlain by the Upper Chalk. BGS mapping of the area indicates that the majority of the site is directly underlain by the Reading Beds, with the Upper Chalk present immediately at subcrop some 1 km west of the site.

The principal surface drainage in the vicinity is provided by the River Kennet, flowing from west to east along the southern boundary of the Chamberhouse Farm site (Photo A9.41). At its closest point the river is some 20 m from the boundary of potential mineral extraction. Data have shown that over the entire reach of the river adjacent to Chamberhouse Farm, groundwater within the sands and gravels is consistently below the stage of the River Kennet, and it is likely that the sand and gravel aquifer receives recharge from the river. The mean flow in the river 4 km upstream of the site at Newbury is 5.02 m^3/s (433.7 Ml/d).

To the north of the site lies the Kennet and Avon Canal (Photo A9.42). Field measurements of water levels have shown that the water levels in, and immediately north of, the canal are approximately 1.8 m above the water level in the superficial deposits. This is a combination of the difference in ground elevation across the canal, estimated as about 1.5 m within the

Photo A9.41 River Kennet

Photo A9.42 Kennet and Avon Canal

Photo A9.43 Reedbeds north of canal

woods, and the thin/absent unsaturated zone among the reedbeds to the north (Photo A9.43). Of course, the differences in elevation between the canal and the surrounding ground change significantly at the locks on the canal, with the heads in the canal varying by over 1 m at each step.

Towards the eastern end of the canal the differences in ground level between the canal and the Chamberhouse Farm site are much less, but the canal stage is always higher. Although there is historical field evidence of leakage along the base of the canal bank, the only leakage seen during a field visit in March 2006 was from a breach in the canal bank which was overflowing and flooding onto the northern section of the Chamberhouse Farm site. The obviously high water level to the north of the canal, and otherwise dry embankment to the south, suggest that the canal is a fairly effective barrier to groundwater flow.

Priors Moor Ditch, which generally flows from west to east through the centre of the site (Photo A9.44), is entirely man-made, being constructed for the purposes of land drainage. Historical monitoring data indicate that locally the ditch acts as a groundwater drain, exerting some control upon piezometric levels within the sands and gravels, with groundwater flowing towards the ditch locally. Flows in the ditch range from 1 l/s in its upper reaches, to 42 l/s at its eastern outflow from the site (Photo A9.45).

Photo A9.44 Priors Moor Ditch

Widemead Ditch runs along the south-western edge of Chamberhouse Farm. The ditch appears to be entirely man-made, with flow derived from an off-take from the River Kennet that is regulated by a sluice gate. Historical monitoring data suggest that Widemead Ditch does not have much influence on local groundwater levels, and flows in the ditch were of the order of 35 l/s. However, when cleared out recently by the Environment Agency, it was found to lose water to the ground, and the sluices have now been closed.

Photo A9.45 Priors Moor Ditch outflow

Within the area of the proposed workings, historical alluvial groundwater levels range from 65.87 maOD to 66.78 maOD. Generally, groundwater flow is from west to east. Pumping tests on the sand and gravel aquifer have produced a range in hydraulic conductivity of 185 to 261 m/d, with an average value of 250 m/d.

The draft proposal is that the quarry be worked in 8 main phases (Figure A9.8), each of approximately one year's duration. The first of these phases is the area of the site closest to the Thatcham Reed Beds SSSI. The draft working plan is to dewater the economic mineral layer to 0.5 m below the saturated mineral surface. In areas where the base of the overburden is below groundwater, this may mean dewatering to 2.2 m below the current water table.

Step 3: Water features susceptible to flow impacts

From the conceptual model, the main water features in the area which are likely to be susceptible to flow impacts are:

- *The Kennet and Avon Canal:* This is lined so flow should not be affected by dewatering at the Chamberhouse Farm site. If there is leakage from the canal to the superficial deposits, it is possible that this will increase. If groundwater heads in the superficial deposits are already below the base of the canal then leakage will not increase.
- *The River Kennet:* If dewatering is to occur then the water table in the gravels will be lowered and the hydraulic gradient between the gravels and the river will increase. The riverbed is likely to be composed of lower permeability deposits than the surrounding aquifer, which will inhibit extra leakage but not necessarily prevent it entirely.
- *Priors Moor Ditch:* The upper section of the ditch will be quarried out completely during draft Phases 4, 5, and 6. It is intended that flows in the lower section of the ditch would be maintained by discharges during the quarry's working life, and by outflow from a series of lakes after restoration. As draft Phase 3 is immediately to the south of the lower section of the ditch, there is potential for derogation of flows when this phase of the quarry is dewatered. There is a licensed abstraction from Priors Moor Ditch within 500 m of the eastern site boundary.
- *Widemead Ditch:* An artificial feature, regulated by sluice gate at its upper end, with two outflows, both to the River Kennet, which will not be disturbed during the mineral workings.

Step 4: Apportion the flow impacts

At this stage of the investigation flow impacts cannot be apportioned with any certainty. If it can be proven that there will be not be increased leakage from the canal then this can be ignored. Locally the flow impacts should probably be split between Priors Moor Ditch and the River Kennet.

The size of the abstraction from Priors Moor Ditch is unknown, but the licence is for spray irrigation from April to October so will be

operating during the driest times of the year. The wider flow impacts can be apportioned 100 per cent to the River Kennet, as Priors Moor Ditch rejoins the main river approximately 2 km from the site boundary.

Step 5: Mitigation of flow impacts

After being abstracted, it is intended that the dewatering water be discharged into recharge trenches to the north of draft Phases 1, 1a, and 6. Any excess water, after provision of augmentation flows to Priors Moor Ditch, is to be transferred to Lower Farm Quarry, where the minerals are to be processed (Photo A9.46), before being passed through a settling lagoon and returned to a surface watercourse, presumably the River Kennet. As the mineral washing is likely to be at least 95-97 per cent non-consumptive, most of the process water will return to the system.


Photo A9.46 Processing of minerals at Lower Farm Quarry

Step 6: Significance of net flow impacts

There is potential for the surface water abstraction from Priors Moor Ditch to be derogated by the dewatering abstraction, but as mentioned above, it is intended that this be mitigated by discharges during the working life of the quarry, and by outflow from a series of lakes upon restoration.

To determine the possible wider flow impacts the daily dewatering volume must be estimated. The hydraulic conductivity of the gravels, estimated from pumping tests, is 250 m/d. From quarry plans the average thickness of the gravels is approximately 3.0 m during draft Phase 1. Given the conceptual model of the gravels, the Thiem-Dupuit equation is perhaps the most appropriate for estimating the dewatering volume. Values used in the equation include a required drawdown of 0.5 m (to 2.5 m aquifer thickness), a radius of 80 m from the centre of the dewatering to the phase boundary, and an unknown radius to the radius of influence.

Using an iterative procedure, it can be shown that there is very little change in the dewatering volume once the radius of influence reaches 1,500 m. This is not an unreasonable distance given the nature of the aquifer (but bear in mind that this is the theoretical, steady-state radius of influence). Using the above parameters gives an estimated dewatering rate of about 740 m^3/d. This assumes that the overlying deposits have low storage and transmissivity. Note that although the analytical equation has been used, the assumptions that the aquifer is of infinite areal extent and is not dewatered have both been broken. This will affect the estimated abstraction rate.

Within the scope of this case study (equivalent to the first iteration of Tier 1 of the HIA), there are insufficient ecological details to allow a full assessment of the significance of the net flow impacts. However, the following points can be raised:

- The worst-case scenario for the River Kennet would be if all the water were to be derived from the river and be 100 per cent consumed. If the dewatering rate is approximately 740 m^3/d, as predicted from the analytical solution, and all water were to come from the river, it would on average be impacted by 0.009 m^3/s.
- As mentioned above, the mean flow in the river at the permanent gauging station at Newbury (approximately 4 km upstream of Chamberhouse Farm) is 5.02 m^3/s, with a Q95 of 1.84 m^3/s. So, in the worst case, dewatering would reduce flows in the river by an average of about 0.2 per cent, and a maximum of about 0.5 per cent during periods of low flow. As it is intended that most of the water be returned to ground using recharge trenches, or discharged back to the river, the actual impacts are likely to be negligible.
- Estimated change in flow as a result of dewatering is therefore small, even during low flows. It is not known how this will affect the river's ecology but it is thought

that the effects would be minimal. A more detailed investigation will be required if the river is found to be ecologically sensitive to such changes in flow.

Step 7: Search area for drawdown impacts

The River Kennet is at the southern limit of the gravel deposits so this will be the southern-most feature that could be impacted, and therefore defines the search radius for potential drawdown impacts. Fieldwork and longer term monitoring (Photo A9.47) has shown that there is a large head gradient across the Kennet and Avon Canal. If this is a barrier to groundwater flow, as suggested by field observations and postulated in site reports, then there will be no impacts north of this feature. For the purposes of this case study it has been assumed that the canal is indeed an impermeable barrier. The search area is therefore limited to the area between the River Kennet to the south, and the Kennet and Avon Canal to the north. The eastern and western boundaries are unknown but impacts would follow the course of the sand and gravel deposits. Using the same equation as in Step 6 give a theoretical steady-state radius of influence of 1,500 m. However, above 1,000 m radius the impacts are less than 0.1 m so there should not be significant derogation.

Photo A9.47 Piezometer

Step 8: Water features susceptible to drawdown impacts

The natural water features that may be susceptible to drawdown impacts are thought to be limited to the southern area of the Thatcham Reedbeds SSSI, which comes within 30 m of draft Phases 1 and 1a. There are not thought to be any groundwater abstractions within the search area.

Step 9: Predict maximum drawdown impacts

The Thiem-Dupuit equation predicts that for a 0.5-m drawdown in the worked area there could be a drawdown of about 0.3 m at the edge of the Thatcham Reedbeds SSSI (before any mitigation measures are put in place).

Step 10: Mitigation of drawdown impacts

It is proposed that there be recharge trenches along the boundary between the quarry and the Thatcham Reedbeds SSSI to mitigate the impacts on the wetland. The height of water in the trenches will be kept up to 0.25 m above the historical undisturbed water table. The effectiveness of the recharge trenches is currently being tested under a separate project.

Step 11: Significance of drawdown impacts

It is expected that once recharge trenches are tested and fully operational there will be very little impact on the SSSI. If a small drawdown in water table were to occur then the trees in this area of the SSSI would be more able to tolerate a short period of water table reduction than if it were to occur in the waterlogged reedbeds to the north of the canal.

Step 12: Water quality impacts

Before being discharged into any surface watercourse the water is to be passed through settling lagoons to remove suspended solids, so water quality impacts are unlikely to be significant. Other than this, it is not believed that the dewatering abstraction associated with the mineral extraction will have any other potential water quality impacts.

Step 13: Redesign mitigation measures

The main mitigation measures at this site are likely to be recharge trenches. Experiments with recharge trenches at this site have shown the infiltration rate to be highly variable over a small area. It may be necessary for the area of the trenches to be increased or additional features such as low-permeability barriers to be installed between the excavation and the SSSI to reduce recycling of water between the trenches and the dewatering abstraction. As there will only be a short time between dewatering commencing and water levels responding it should be possible to adjust the dewatering and mitigation plan quickly to avoid adverse impacts occurring at the SSSI.

Step 14: Monitoring and reporting plan

A schedule of monitoring for groundwater and surface water, along with trigger and action levels for any mitigation, will have to be agreed with the Environment Agency and English Nature. As this is a greenfield site it should be possible to make sure in advance that infrastructure is in place to enable water balances to be calculated. In particular, the following quantities should be measured as accurately as practicable: abstraction from the dewatering sumps; discharges into the recharge trenches; overflows from the recharge trenches if infiltration capacity is exceeded; consumption by the mineral washing plant; off-site discharges; and consumption for dust suppression. If the settling lagoons are excavated into the shallow aquifer and are unlined, they will need to be monitored as if they were large recharge trenches.

After Tier 1

This case study has concentrated on the potential impacts of draft Phase 1 of the proposed workings. In reality, the study would be extended to investigate subsequent phases, and the cumulative effects if more than one phase is being dewatered at any one time. If this study were to proceed to Tier 2, the investigations are likely to focus on the following main aspects:

- The effectiveness of the Kennet and Avon Canal as a barrier to groundwater flow. This initial Tier 1 assessment has assumed that there is no groundwater flow under the canal. If this turns out not to be the case, the search area would have to be widened to include features to the north of the canal.

- The status of the abstraction licence from Priors Moor Ditch. Additional work may be required in order to design the mitigation measures to maintain flows in the lower section of the ditch, so that the abstraction is not derogated.
- The hydroecology of the River Kennet. In particular, the significance of the predicted flow impacts needs to be assessed.
- The effectiveness of the recharge trenches. It is essential that effective mitigation measures be in place before dewatering commences, to protect the SSSIs.

Note that investigations at Chamberhouse Farm have already progressed well beyond the information presented in this case study, and include extensive experiments with recharge trenches, and a groundwater flow model.

CASE STUDY 6: Construction dewatering near Haverigg, Cumbria

Introduction

This case study illustrates the application of the HIA methodology to a construction dewatering project near Haverigg, on the south west coastline of the Lake District, about 1 km from Millom. Temporary groundwater control works were undertaken to enable the construction of a storage tank, 15 m diameter, with a base slab at 10 m below ground level (bgl).

Acknowledgement: Water Management Consultants Ltd is very grateful to Project Dewatering Ltd (PDL) for permission to publish this case study, and particularly to David Wright (Managing Director of PDL) and Professor Paul Younger of the University of Newcastle upon Tyne (who also serves as Director of Design for PDL). Collation of much of the information presented was undertaken by Peter Harker whilst working on an engineering geology MSc project funded by PDL entitled "Technical analysis of full-scale construction dewatering operations" (2004, School of Civil Engineering and Geosciences, University of Newcastle upon Tyne).

Step 1: Regional water resource status

The village of Haverigg lies within the Duddon CAMS area (No.123), which is currently being assessed in combination with the Derwent and West Cumbria areas (Nos.125 and 124 respectively). The consultation document is due for publication in Autumn 2006, so the regional water resource status has not yet been defined. Following the WFD initial characterisation exercise, Haverigg Pool (the small river which runs adjacent to the site, see Figure A9.9 and Photo A9.48) has been designated as 'Probably at risk' (of failing to meet the WFD objectives). The Duddon estuary (about 300 m from the site), Hodbarrow Lake (500 m away), and the groundwater under the site are designated as 'At risk', 'Probably at risk', and 'Probably at risk' respectively. So this HIA needs

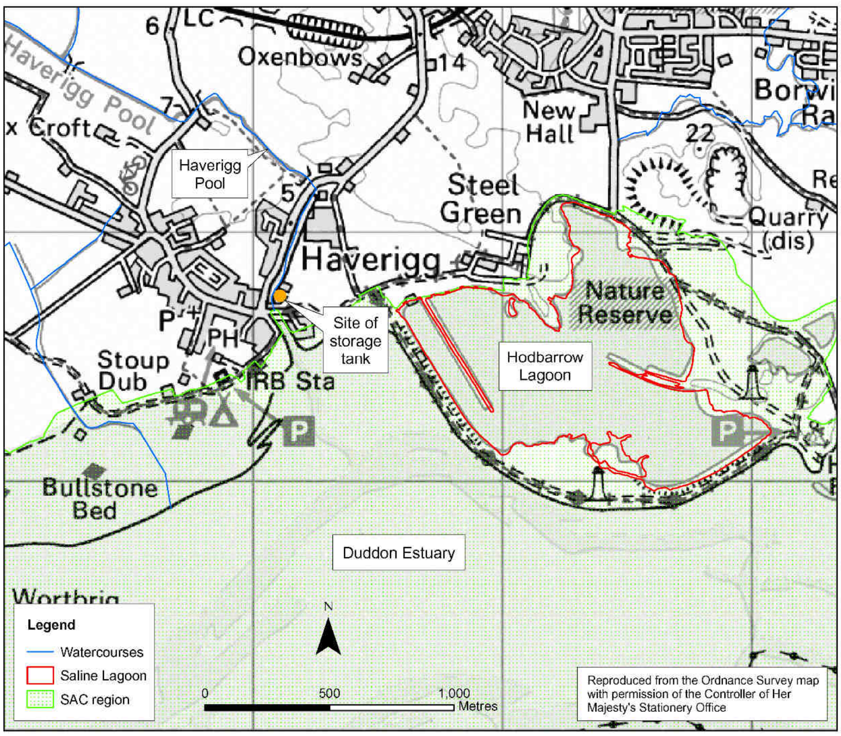

Figure A9.9 Locations of site and potentially sensitive features

Photo A9.48 Completed storage tank adjacent to Haverigg Pool

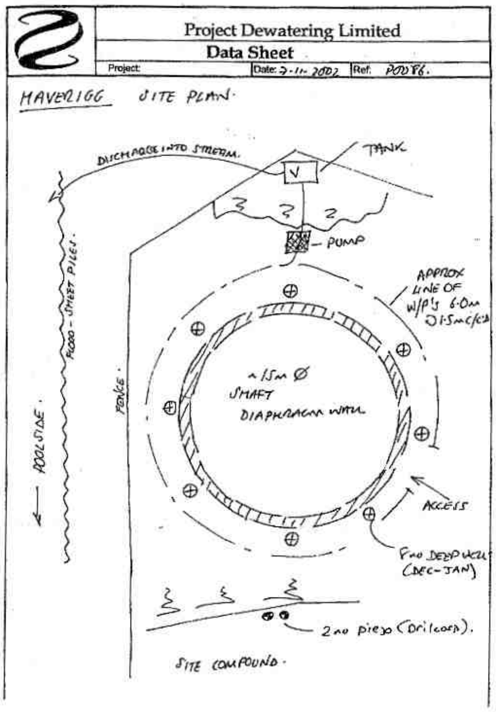

Figure A9.10 Sketch of dewatering system design (Project Dewatering Ltd)

to demonstrate that the dewatering operation is not contributing to the regional water resource problem.

Step 2: Conceptual model

According to the BGS map of the area, the site consists of Quaternary storm beach deposits, with layers of sand and gravel interspersed with layers of clay. Information from six site investigation borehole logs revealed an upper aquifer of gravelly sand and sand, from near the surface to about 10 mbgl, followed by a layer of stiff clay between 2 and 8 m thick. Below the clay is a lower aquifer of gravelly sand and sand, the upper boundary of which varies from 14 to 20 mbgl, but the full thickness of which was not established from the site investigation.

The groundwater level in the upper aquifer was at about 3 mbgl, but there were found to be artesian heads in the lower aquifer of up to 3 m above ground level. Given the dimensions of the excavation, this obviously raised concerns about potential floor heave in the base of the excavation, so the dewatering design focussed on dewatering the upper aquifer and reducing heads within the lower aquifer to below the level of the base slab.

Particle size distribution analyses were available from 23 samples taken from the site investigation boreholes. Hazen's formula applied to these analyses gave average values for the hydraulic conductivity of the aquifer layers of about 5 x 10^{-4} m/s (about 43 m/d). Given the required dewatering depth, the presence of two aquifers, and the permeability of the aquifers, it was decided that the appropriate dewatering system design was a combination of well-points and deep wells (Figure A9.10).

A line of well-points was installed in a circle around the proposed excavation, to a depth of 6 mbgl. These were all connected together via header pipes (Photo A9.49) to a surface suction-pump, which delivered a total of around 190 m^3/d for the duration of the dewatering operation. Calculations on the critical depth for floor heave led to the conclusion that the head in the lower aquifer needed to be reduced to about 5.5 mbgl (that is, a total drawdown within the excavated area on the order of 8.5 mbgl). A pilot well was drilled some 5 m into the lower aquifer and test-pumped, yielding an average transmissivity for the lower aquifer of 1,480 m^2/d. Armed with this information, Theis-based superposition calculations were used to deduce that the necessary drawdown could best be achieved using a system of eight deep wells, disposed in a circle just inside the ring of well-points. These deep wells were cased-off through the upper aquifer and completed to 5 m into the lower aquifer, or a maximum depth of 24 mbgl. Given that drawdowns would not exceed 8 mbgl, it was possible to pump these deep wells also using surface suction pumps (Photo A9.50); an unusual circumstance for deep wells, which often need individual electric submersible pumps.

Accordingly, the deep wells were set up as two groups of four pumps, with each group controlled independently. The total yield from all eight wells was around 480 m^3/d. The entire dewatering system was operated for around 6 months, during which time some 121 Ml of groundwater was abstracted.

Photo A9.49 Wellpoints with suction pump

Photo A9.50 Connecting discharge pipe to deep wells

Step 3: Water features susceptible to flow impacts

The water features susceptible to flow impacts in the area are likely to be the ones mentioned under Step 2 above, namely, the small river Haverigg Pool, the Duddon estuary (Photo A9.51), and Hodbarrow Lake (which is fresh to slightly brackish, although not directly connected to the sea).

The magnitude of the artesian head in the lower aquifer is presumably explicable in terms of lateral connection from the confined portion of the aquifer at this site to an outcrop area at higher elevation inland. This outcrop area may conceivably be that of an underlying bedrock aquifer, which feeds water up towards the Quaternary sequence in this near-coastal discharge zone setting. However, scrutiny of published geological maps and other geological data sources (such as the published Aquifer Properties database for England and Wales) did not shed any light on this. The possibility cannot therefore be ruled out that, were abstraction from the lower aquifer at Haverigg to be sustained for a long period of time, an impact might eventually be exerted on some other aquifer area inland, and that this impact might conceivably deplete natural groundwater discharge to the headwaters of the Haverigg Pool or

Photo A9.51 The Duddon estuary

other coastal streams. However, given the temporary nature of the abstraction, this issue was not pursued further, and the investigation concentrated on local water features.

Step 4: Apportion the flow impacts

Given the relative scale of Haverigg Pool, the Duddon estuary and Hodbarrow Lake, and the temporary nature of the dewatering abstraction, Haverigg Pool is the water feature most likely to be affected, and it is reasonable to apportion the flow impacts 100 per cent to Haverigg Pool.

Step 5: Mitigation of flow impacts

The water abstracted from both the well-point system and the eight deep wells was discharged directly into Haverigg Pool, without any consumptive use. Potential impacts from abstraction (at least from the upper aquifer) should therefore have been fully mitigated. Also, the discharge point was very close to where Haverigg Pool enters the estuary (and it is in fact tidal at this point, Photo A9.52).

Photo A9.52 Haverigg Pool entering the estuary

Step 6: Significance of net flow impacts

The net flow impacts were thought to be insignificant, because of the temporary nature of the abstraction, and the discharges being directly into the affected water feature.

Step 7: Search area for drawdown impacts

The radius of the search area for drawdown impacts is defined by the furthest water feature likely to be deprived of water by the abstraction. From the conceptual model, this is Hodbarrow Lake (Photo A9.53), about 500 m from the site.

Photo A9.53 Hodbarrow Lake

Step 8: Water features susceptible to drawdown impacts

There are no groundwater source protection zones within this search area, so it is assumed that there are no major groundwater abstractions (the nearest one being in Barrow-in-Furness, about 8 km away, on the other side of the Duddon estuary). Local enquiries did not identify any pre-existing groundwater abstractions from the local Quaternary sequence in which the dewatering operations were to be undertaken. It was concluded that there were no water features susceptible to drawdown impacts.

Step 9: Predict maximum drawdown impacts

Not applicable.

Step 10: Mitigation of drawdown impacts

Not applicable.

Step 11: Significance of drawdown impacts

Not applicable.

Step 12: Water quality impacts

Groundwater quality samples collected by the client during site investigations had found no elevated concentrations of any potential contaminants. Therefore the only issue which needed to be addressed was the possibility of silt entrainment in the water pumped to the Haverigg Pool. Careful well design (including ensuring adequate filter packs) and well development (surging and air-lifting before the commencement of regular pumping) ensured that the wells were not pumping silt. The water was all routed via a V-notch weir tank to a pipe line to avoid any mobilisation of surface silt. The V-notch tank also had the potential to serve as a settling tank in the event of any unforeseen silt release from the wells later in the life of the operation.

Step 13: Redesign mitigation measures

Not necessary.

Step 14: Monitoring and reporting plan

Water levels and pumping rates were measured for the duration of the dewatering operations, and daily visual inspections were made in the weir tanks to ensure that no problems were developing with silt mobilisation. No long-term monitoring is necessary, because groundwater levels were allowed to recover on completion of construction.

After Tier 1

No further tiers of investigation necessary, given the temporary nature of the abstraction.

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