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Hydrogeological impact appraisal for groundwater abstractions

Science Report – SC040020/SR2

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Executive summary

This report provides practical guidance on how to assess the hydrogeological impact of groundwater abstractions, for those who are preparing applications to the Environment Agency for 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 proposed groundwater abstraction. 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 aquifer and the abstraction itself. The steps of the methodology are followed iteratively, within a structure with three tiers, and the procedure continues until the required level of confidence has been achieved. Advice is also given on how to undertake HIA in karstic aquifers and fractured crystalline rocks.

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1 Introduction

1.1 Purpose of this report

This report describes a methodology and suite of tools for assessing the hydrogeological impact of groundwater abstraction. With the coming into force of the Water Act 2003, 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 impact of existing or proposed groundwater abstractions, for those who are preparing technical material to support applications to the Environment Agency for full licences. Similar guidance for those preparing applications for transfer and/or full licences in connection with dewatering operations at quarries, mines and engineering works can be found in a separate report (Boak *et al* 2006).

Both reports build on work carried out under an earlier Environment Agency R&D Project (W6-071) on risk-based decision making for water resources licensing, reported by Faulkner *et al* (2003).

1.2 The regulatory context

The Water Act 2003 has introduced significant changes to the abstraction licensing system in England and Wales. It has changed the licensing system in six key areas (Environment Agency 2003a):

i. All small abstractions, generally under 20 m^3/d, will not need a licence.

ii. 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 used to be exempt now need a licence.

iii. Administration for making applications, transferring and renewing licences will be made simpler, reducing barriers to the trading of water rights.

iv. All abstractors now have a responsibility not to let their abstraction cause damage to others. From 2012, the Environment Agency will be able to amend or revoke permanent abstraction licences without compensation if they are causing serious damage to the environment.

v. There is an increased focus on water conservation. Water companies now have duties to conserve water, and all public bodies need to consider how to conserve water supplied to their premises.

vi. Water companies must develop and publish water resources management and drought plans. The Environment Agency can encourage transfer of water resources between water companies, and recover costs associated with drought orders and permits.

Even with all these changes, several other pieces of legislation and regulatory regimes remain highly relevant to the assessment of the impacts of groundwater abstraction 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 can be found in Appendix 1.

1.3 The abstraction licensing process

The methodology for hydrogeological impact appraisal (HIA) described in this report is designed to fit into the Environment Agency's abstraction licensing process. This process can be summarised in terms of a typical sequence of events (for new applications), as follows:

i. *Initial enquiry to the Environment Agency:* Many enquiries for new abstractions are received by the Environment Agency, and a significant number are dropped at an early stage. Enquirers are usually discouraged from submitting formal applications for licences until the Section 32(3) procedure has been carried out (see below). If there is no chance of an abstraction licence being granted (if the enquiry concerns a groundwater management unit that is already over-abstracted, for example), the enquirer is informed at this point. If necessary, the Environment Agency will give advice on the type of licence that should be applied for (temporary, transfer, or full).

ii. *Application for Section 32(3) consent:* This procedure (under Section 32(3) of the Water Resources Act 1991) is triggered by an application from the applicant. Among other things, the Environment Agency needs to know the proposed maximum daily and annual average rates of abstraction, when within the year the water will be abstracted, whether and where water will be returned to the environment, what the water is to be used for, and the proposed borehole design.

iii. *Water features survey:* The applicant then carries out a water features survey within a radius specified by the Environment Agency, looking for boreholes, wells, ponds, lakes, springs, seepages, wetlands, watercourses, etc. The findings are reported using a standard format. The applicant is usually provided with details of existing licensed abstractions within the search radius by the Environment Agency.

iv. *Section 32(3) consent issued:* Assuming that existing water users and conservation sites are adequately safeguarded, a consent is issued to construct the borehole and carry out a pumping test. The results of the water features survey are used to specify which features should be monitored during the pumping test. Conditions are also laid down on maximum pumping rates, required duration, and arrangements for discharging the water during the test.

v. *Pumping test:* The objectives of the test pumping programme include proving the yield of the borehole, and providing enough information for the effects of the abstraction on the environment and other water users to be determined (see Appendix 2 for an extended discussion of test pumping). The applicant is usually expected to analyse the data from the test, and to submit an interpretative report.

vi. *Licence application and determination:* Assuming there have been no fatal flaws in the process so far, the applicant now submits a formal licence application, to be processed by the Environment Agency. This involves (for proposals over 20 m^3/d anyway) advertising the application, consideration of any objections from the public, and consultation with statutory consultees such as water companies. As mentioned above, the Water Act 2003 has exempted all abstractions below a generic threshold of 20 m^3/d from licensing. However, this threshold may be varied up or down, depending on local conditions, provided it is technically appropriate to do so and certain legal requirements are fulfilled. Finally, the application is either rejected or the licence granted (often with conditions attached).

In practice, the actual process may vary from case to case, depending on the local circumstances. It is important to realise that at no stage until the final decision does the Environment Agency commit itself to granting a licence (or indeed refusing it). However, it is in everybody's interest for the process to be stopped as early as possible if refusal is likely to be the final outcome. In practice, the main hydrogeological work (both by the applicant and by the Environment Agency) is carried out during the Section 32(3) procedures. By the time the formal licence application is made, the process is largely concerned with legal procedures of consultation and receiving objections.

So where does the HIA methodology fit in? The exact approach will of course depend on many factors, such as the water resource availability status of the groundwater management unit, whether the application is for a new abstraction or a variation to an existing one; and whether reliable conceptual and/or numerical models of the aquifer are already available. However, generally speaking, the HIA methodology can be commenced as soon as the basic details of the proposed abstraction are known, and it will be seen that the water features survey and the pumping test play an important part. Links between these and the steps of the HIA methodology will become clear in Section 4.

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 groundwater abstraction; the emphasis is 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, 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 in the model into one or two layers with averaged properties.

Sample: results from test pumping usually only represent a small sample in time and space of the overall behaviour of an aquifer. Depending on the length of the test, it 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 equipment used to monitor the test (for example, the calibration of the flow meter used to measure discharge during the test).

Data: 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 is going to be used in subsequent calculations. 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 a simple 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 upconing of the saline water. 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 summarised 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).

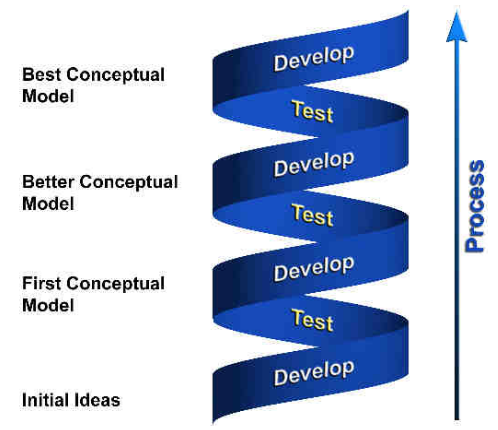

Figure 2.1 The development process for a conceptual model
(adapted from Environment Agency presentations)

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 still too great.

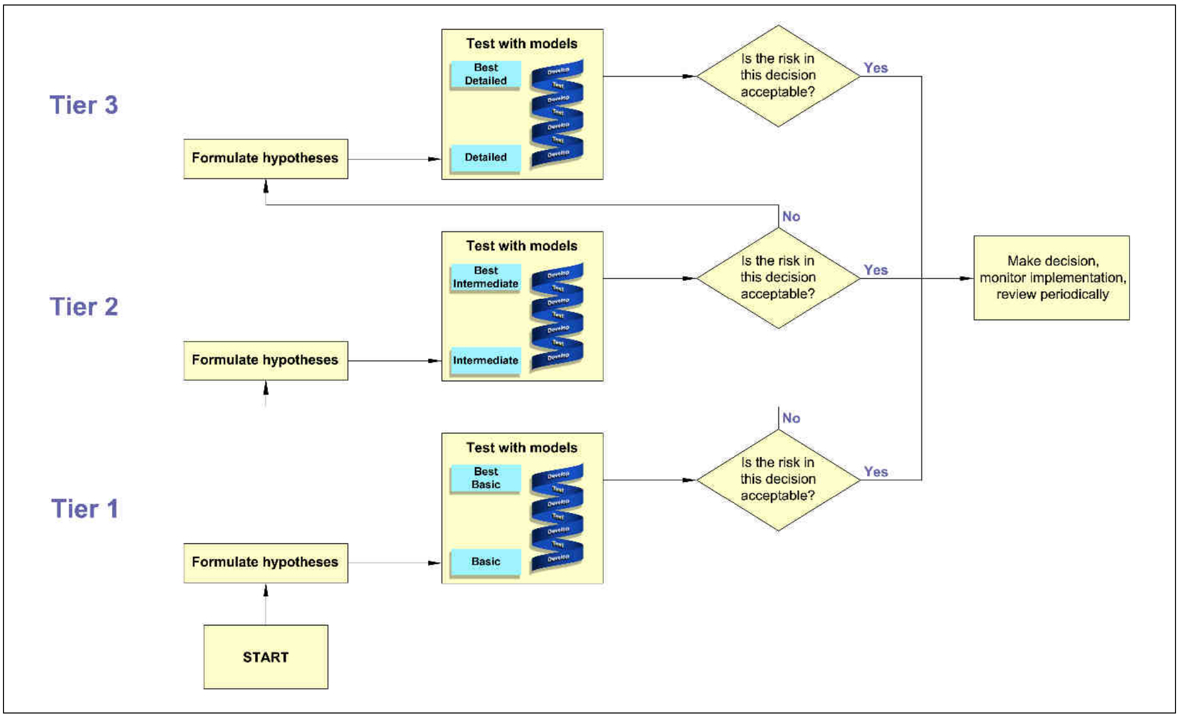

Figure 2.2 Tiered approach to conceptual modelling
(adapted from Environment Agency presentations)

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. However, 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).

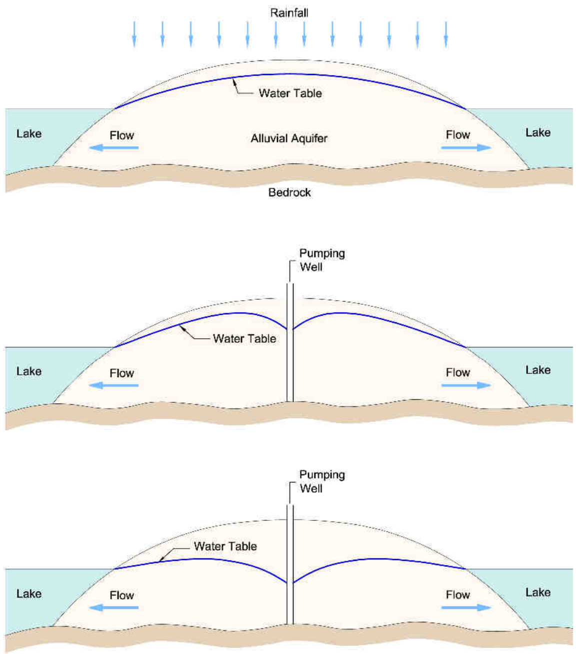

Figure 2.3 Impact of groundwater abstraction on aquifer discharges
(after Bredehoeft *et al* 1982; note that the rainfall continues in the middle and lower diagrams)

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.

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[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.

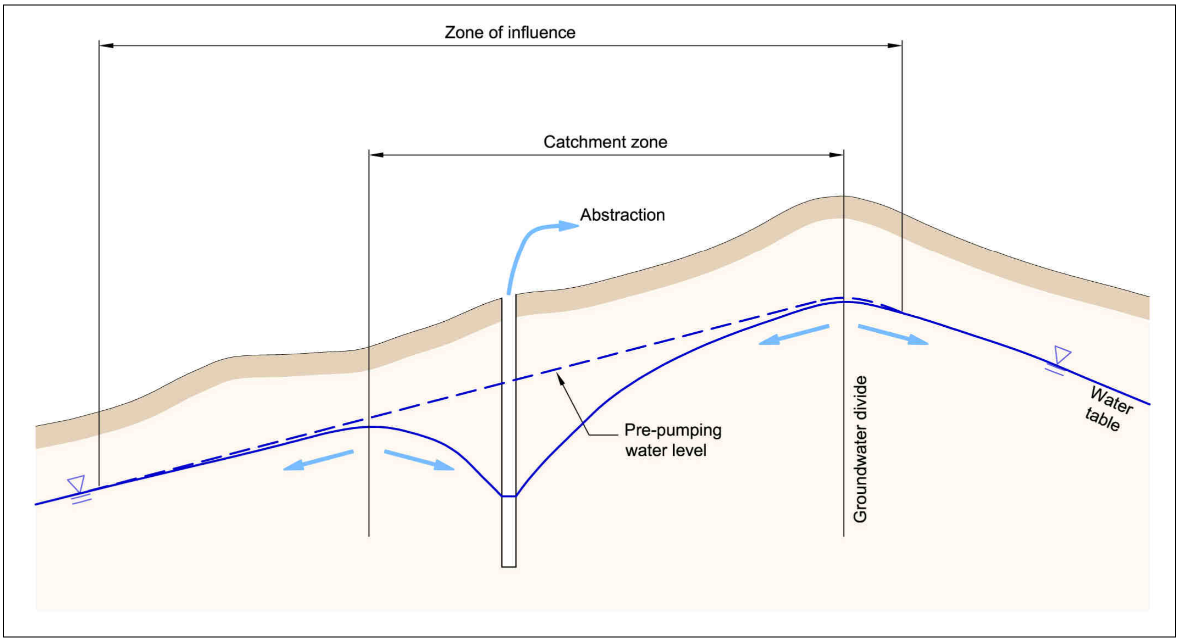


Figure 2.4 Catchment zones and zones of influence
(after Environment Agency 1996)

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 only stop spreading 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.

3 Background to the methodology

3.1 Development criteria

In developing the HIA methodology for groundwater 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.
- result in an appropriate level of on-going monitoring, targeted at the issues of real concern.
- if relevant, take into account the mitigation of impacts by the return of water to the groundwater or surface water system.
- 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 will occur close to the abstraction point, 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 very 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. It 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 public water supply abstraction in a major aquifer, close to some Ramsar sites, pumping large quantities of water, compared to a small abstraction for domestic supply, in an unproductive aquifer, with no sensitive conservation sites in the area.
- 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 abstractor's own historical monitoring data. The conceptual model is typically tested using simple analytical equations, to arrive at a 'best basic' conceptual model. A Tier-1 assessment is likely to be required in virtually all cases.

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. 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).

3.3 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.

3.3.1 Tier 1 tools

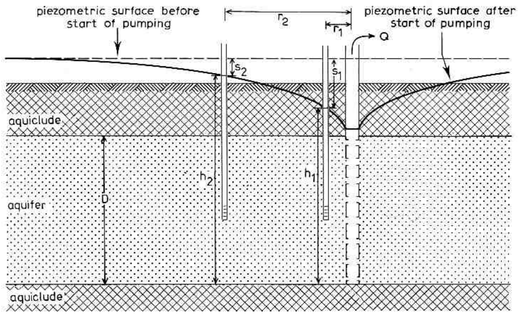
The main tools likely to be used at the level of Tier 1 are simple analytical equations and the analysis of test pumping results. Two good examples of useful analytical equations are the Thiem equation and Thiem-Dupuit equation for steady-state flow in confined and unconfined aquifers respectively (Kruseman and de Ridder 1990). The equations and parameters are shown in Box 3.1.

Such equations must always be used with care, bearing in mind all the assumptions on which the equations are based. As part of this project, over 20 analytical equations have been assembled from various sources (textbooks and other publications), and put into an MS Excel spreadsheet for convenience, for use when assessing the impacts of groundwater abstractions. Many

Box 3.1: Thiem and Thiem-Dupuit equations

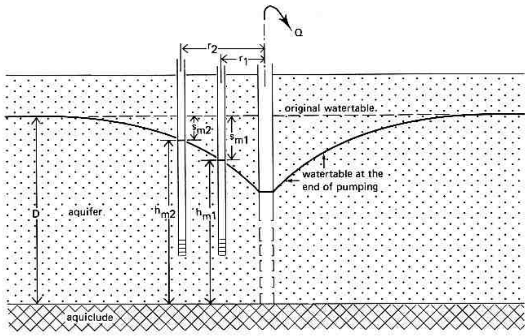
Thiem equation (steady-state confined flow)

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



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.

These equations are given here in their most general form, but they can be used in other ways. For example, if only one piezometer is available (at distance r_2 in the diagrams above), 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. 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.

of the equations are only relevant to groundwater abstraction for dewatering purposes, but some are relevant to all groundwater abstraction, including:

- Thiem and Dupuit-Thiem equations, for steady-state flow to a well in confined and unconfined aquifers (Box 3.1).
- Theis equation, for time-variant flow to a well in a confined aquifer (used in Box 2.1).
- Cooper-Jacob approximations to the Theis equation, for time-variant flow in confined and unconfined aquifers.
- De Glee equation, for steady-state flow to a well in a leaky aquifer.

This spreadsheet will be made available in due course by the Environment Agency.

Various software packages are available for the analysis of test pumping results. The standard package used by the Environment Agency is Aquifer^{win32}. This allows test pumping data to be analysed by a wide variety of methods, and includes details of the assumptions inherent in each method. Virtual pumping tests can also be run, to help plan the details of the actual field test. Assumed values for transmissivity and storage coefficient are used so that the locations of observation points and the time to observable impact (and therefore length of test) can be planned. See Appendix 2 for further discussion of test pumping.

3.3.2 Tier 2 tools

In Tier 2, the conceptual model is tested in more detail, typically using time-variant data instead of steady-state. The focus may be on investigating particular issues of concern, such as river-aquifer interaction. Tools and techniques likely to be used at the level of Tier 2 include the following:

- *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 (see below for more details) 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.

IGARF (Impact of Groundwater Abstractions on River Flows) is a spreadsheet-based tool for assessing the impacts of new groundwater abstractions on river flows (Environment Agency 1999 and 2004). IGARF (version 3) uses the analytical solutions of Theis, Hantush or Stang, whichever is applicable to the aquifer in question, and the user must be aware of the assumptions inherent in each of the analytical solutions. A typical procedure for using IGARF would be as follows:

i. Establish a preliminary conceptual model of the river-aquifer system, and choose whichever analytical solution available in IGARF is closest to the conceptual model.

ii. Using IGARF, make initial predictions of the likely impacts of the abstraction, carrying out sensitivity analysis on the input parameters to give a range of likely impacts.

iii. Design a field pumping test based on the predictions, using IGARF to indicate whether these changes are large enough to be observed, and if so, over what time-period monitoring should be maintained, and how far upstream and downstream the river flow should be monitored.

iv. Conduct the pumping test, analyse and interpret the results to evaluate impacts on the river and to calculate aquifer physical properties.

v. Review the conceptual model, the selection of analytical solution and the range of parameter values used in IGARF.

vi. Make predictions of the long-term impacts, using IGARF as a guide, and taking into account all uncertainties.

3.3.3 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 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 and Environment, Denmark.
- *FEFLOW:* a finite-element model (the other three are finite-difference models) developed by WASY GmBH, Germany.

The Environment Agency's current preferred numerical groundwater modelling software is MODFLOW, with Groundwater Vistas as the user interface.

3.3.4 Other useful techniques

Many other investigative techniques may be useful when undertaking HIA at any 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 Appendix 2).
- *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 also 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).

3.4 Dealing with karst and fractured rock

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 3.1 shows the main areas in England and Wales where there is potential for the development of dissolution features. See the GeoSure service provided by the British Geological Survey for details of obtaining this information at larger scales.

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.

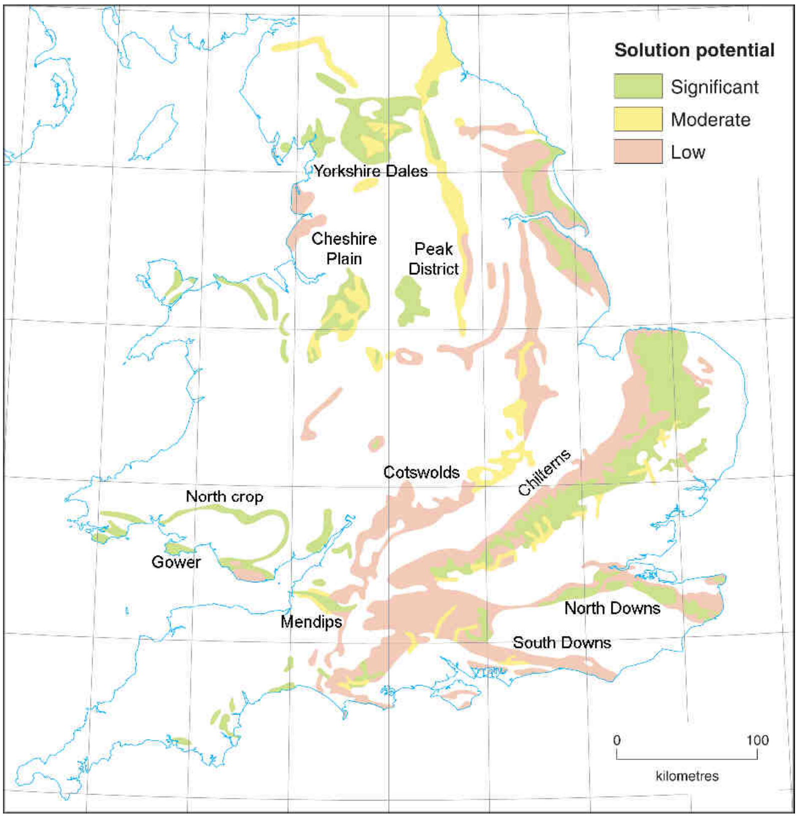


Figure 3.1 Main locations of karst-prone rocks in England and Wales
(adapted from GeoSure information, British Geological Survey)

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.

In many cases, the process will be able to be streamlined. For example, it is recognised that the impacts of some groundwater abstractions are mitigated, by a proportion of the abstracted water being discharged back into the environment, for example. The HIA methodology assesses the impacts as if there were no mitigation, then adds back in the beneficial effects of mitigation. This is done because the locations and timing of the abstraction impacts may be different from the beneficial effects of the mitigation, and the mitigation measures may need to be optimised. Obviously, if all the abstracted water is consumed, and there are no mitigation measures, then Steps 5, 10 and 13 (see Box 4.1) can be omitted.

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.

Box 4.1: The 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.

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 abstraction is located. As described in Appendix 1, the whole of England and Wales has been divided up into CAMS areas, and each 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.

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

iii. *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.

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

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 your abstraction is in, and information on which CAMS areas already 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 groundwater body in which your abstraction is located. 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. 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

The key to HIA is a good conceptual model. As has already been emphasised several times, the process of developing a conceptual model is a continuous and cyclical one, with the process continuing until the level of confidence in the model is sufficiently high to enable a decision to be made. As described in Section 3.1, 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. As mentioned under Step 1, if a completed CAMS document is available for the CAMS area in which the proposed abstraction 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.

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. 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 (2002a).

The outline regional conceptual model should now be refined by adding more detail about the local area around the proposed abstraction, 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.

At this point, readers may be asking the question, for what area should the local 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. If the abstraction is from a karstic aquifer or fractured crystalline rock, the shape and size of the zone of influence is likely to be highly uncertain, and will need careful consideration (see Appendices 3 and 4 for further information).

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 get 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 is less important than the hydraulic resistance of the deposits between the water feature and the aquifer (see Box 4.2).

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.

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 that shown in Figure 4.1.

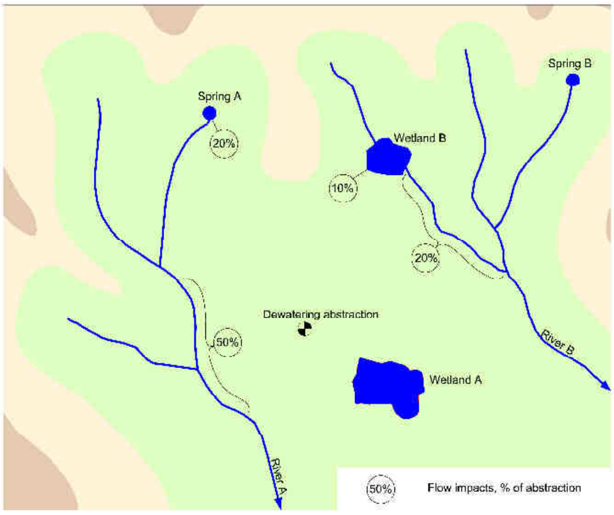


Figure 4.1 Sketch map of water features

Part of the Section 32(3) process described earlier is a water features survey, the results of which are very useful for this step. The default search areas specified by the Environment Agency for water features surveys are shown in Table 4.2:

Table 4.2 Default search areas for water features surveys

Abstraction rate (m^3/d)	Radius of survey area
Up to 20	100 m
2–100	250 m
100–500	500 m
500–1,000	1 km
1,000–3,000	1.5 km
3,000–5,000	2 km
Over 5,000	2 to 4 km

In practice, the search radius is increased where: sensitive abstractions or environmental features are located just beyond the specified radius; the aquifer is confined; or where there is a high degree of uncertainty about the aquifer characteristics. In addition, the search radius, or the shape of the search area, should be varied depending on the understanding of the aquifer in the conceptual model. Note that the cyclical approach is valuable here, as information from Steps 7 and 8 below may contribute to the decision on the search radius. If necessary, a 'dry run' should be conducted through all the steps of the HIA, before returning to Step 3 and reaching a conclusion on the search area for water features.

4.2.4 Step 4: Apportion the flow impacts

Now apportion the flow impacts to those water features likely to be deprived of water by the abstraction, either as portions of the total abstraction quantity, or as percentages (making sure that the percentages add up to 100 per cent). In this example (shown in Figure 4.1), 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, and 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.

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 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. It may be that not all the water abstracted is consumed, but that a proportion is discharged back into the environment (see Box 4.3 for a discussion of 'consumptiveness'). Or it may be that a discharge is required as part of the abstraction licence conditions, as compensation flow in a surface watercourse, for example.

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. However, consumptiveness was rarely studied in any detail, and the percentages were effectively rules of thumb. A recent Environment Agency consultation document has proposed the following loss factors (equivalent to consumptiveness): 100 per cent for spray and trickle irrigation; 60 per cent for public/private water supply, certain industrial uses, boiler feed, bottling, etc; 3 per cent for vegetable washing and non-evaporative cooling; and 0.3 per cent for fish farms (Environment Agency 2005a). Establishing the consumptiveness of the abstraction is important for assessing the net impacts of the abstraction, and may also influence whether or not a licence can be granted (particularly in groundwater management units where water resources are under stress). Consumptiveness is best estimated using a water balance approach, focussing on accounting for all the abstracted water. Another Environment Agency consultation document (Environment Agency 2005b) strongly encourages the use of water audits (which include establishing consumptiveness) to improve the efficient 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 from a borehole at some distance from the river, 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. Further information on net gain and the augmentation of river flows from groundwater can be found in Rippon *et al* (2003).

Assessing the mitigating effects of discharges is not necessarily straightforward, for the following reasons:

- 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 abstraction may be affecting a certain stream, but the water is being discharged into a different stream. The same can apply to other methods of mitigation such as 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 location).

- 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 smoothing 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. The suggested procedure under Step 6 is as follows:

i. If 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.

ii. 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).

iii. If neither a CAMS status nor WFD status is available, and the abstraction is in unproductive strata, then as mentioned earlier (Table 4.1), the focus of the HIA is 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 hydro-ecologist 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 hydro-ecological 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 Sites of Special Scientific Interest (SSSIs) or Natura 2000 sites, there are specific legal obligations associated with assessments for these sites (see Appendix 1 for further details). If you think you may be impacting such a site, contact your local Environment Agency 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 (usually analytical or empirical equations) are in use for estimating R_o, but the HIA methodology takes a different approach, which is to base the definition of R_o on a good conceptual understanding of the 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 abstraction. Draw a circle, centred on the abstraction, passing through this feature (Spring A), as shown by the solid red line in Figure 4.2.

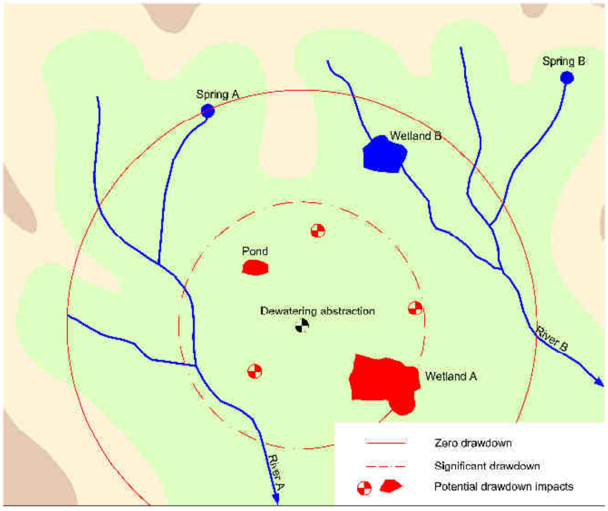


Figure 4.2 Sketch map showing potential drawdown impacts

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. 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 abstraction.
- The presence of impermeable layers between aquifers, such that water features that are clearly hydraulically isolated from the abstraction can be ruled out.
- The presence of a major 'recharge boundary', for example a river in complete hydraulic connection with the 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 below.

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.
- Abstraction licence and protected rights maps held by the Environment Agency. Bear in mind that there may be small abstractions exempt from

licensing (that 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 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 is 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 3.1, 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 abstraction in an unconfined aquifer, the simplified situation is as shown in Figure 4.3.

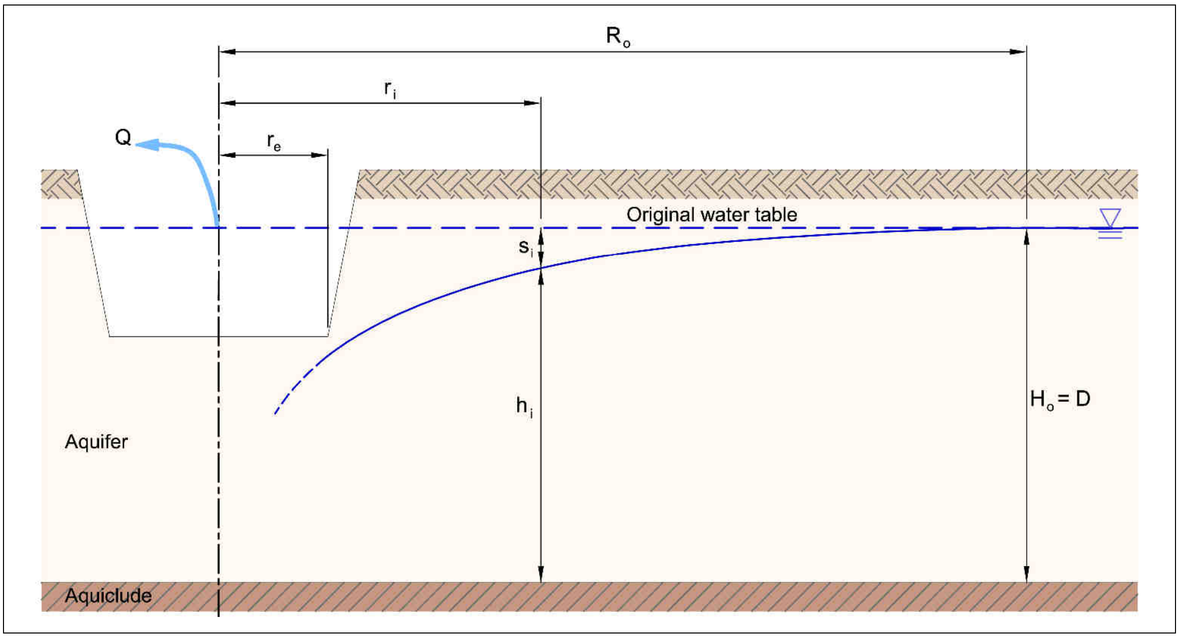


Figure 4.3 Parameters for the Thiem-Dupuit equation

The proposed 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:

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

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 operational life of the abstraction, 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 most hydrogeology textbooks, 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 (see 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 below. Several 'problem' cases will now be explained:

- ***Seasonal abstractions:*** For seasonal or periodic abstractions (only used for irrigation in summer, for example), there are two scenarios that 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.

- ***Intermittent abstractions:*** Many abstractions, particularly public water supply boreholes, pump intermittently, often controlled by water-level switches on service reservoirs. Judgement needs to be used on whether to handle this type of abstraction like a seasonal abstraction (albeit with a very short season), or whether just to use the average pumping rate. This will depend on the relative lengths of the pumping and resting periods.

- ***Groups of closely-spaced boreholes:*** It is common in public water supply for a groundwater source to consist of say four, closely-spaced boreholes, with pumping alternating between two pairs of boreholes. This type of arrangement can usually be treated as a single, large-diameter borehole, using the principles of effective radius explained in publications such as Preene *et al* (2000).

- ***Multiple abstraction points:*** Sometimes, a single licensed abstraction may actually consist of multiple abstraction points, such as an array of boreholes tens or even hundreds of metres apart. Again, judgement needs to be used, as beyond a certain spacing, it becomes unrealistic to treat two or more boreholes as a single borehole with larger effective radius, and they need to be treated as separate boreholes. There are recognised techniques, explained in most hydrogeological textbooks, for calculating the combined effect of several boreholes, using the principle of superposition.

- ***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.

- ***Abstractions from 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, Thiem-Dupuit 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 that have been 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 abstraction 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.

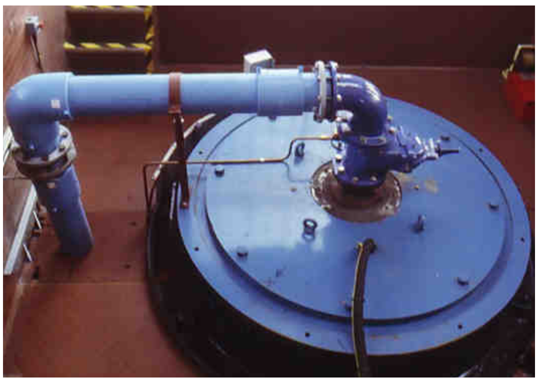
In the same way as for mitigation of flow impacts, the measures for mitigation of drawdown impacts need to be considered carefully, to arrive at a net drawdown impact, while bearing in mind any peculiarities of the local conceptual model. In particular, it is worth remembering that the timing and location of the drawdown impacts from the 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 discussed and agreed in advance with Natural England or the Countryside Council for Wales (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.



Large public water supply borehole

A small domestic supply on the other hand, pumping several hundred litres per day, may only require 1 m or so. The basic details of typical PWLs, and existing pump intake levels 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 by 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 time-limited licence.

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 in 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 in 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).

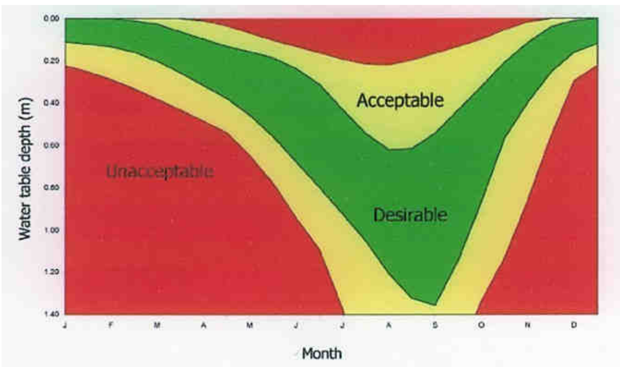


Figure 4.4 Example of water table threshold requirements for particular plant community
(from Brooks *et al* 2004)

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. Note that 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. Guidelines on the management of groundwater to maintain wetland ecological character can be found on the Ramsar website (http://www.ramsar.org).

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. Note that 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 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 a 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.

Potential water quality impacts of groundwater abstraction are largely related to changes in the groundwater flow pattern in the aquifer. In some cases a numerical groundwater model will be available that 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 or industrial sites) 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. Groundwater abstraction can therefore draw pollution into previously unpolluted parts of the aquifer, as reported by Morgan-Jones *et al*

Potential point source of pollution

(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 fresh and saline water, and cause the boundary to move. Note that 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 proposed abstraction are still unacceptable. However, designing or redesigning effective mitigation measures may allow 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 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:

- Planning carefully the point(s) at which water is discharged into surface watercourses, so that the mitigation is targeted at the most sensitive or most impacted reaches. This may involve 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, sometimes 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 abstraction point) to maximise benefit and minimise recycling of water.

These techniques are not new, and are already in widespread use, but the point is that the conceptual model can be used to optimise the mitigation.

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 in particular, 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.
- It will be seen in Section 5 that the monitoring should be focussed on the water features that have been identified during the HIA as being susceptible to flow and drawdown impacts. 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. The Environment Agency maintains a network of such reference boreholes.

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:

i. If any of the water features that have been identified as being impacted (or potentially impacted) is a SSSI, Special Area of Conservation (SAC), Special Protection Area (SPA), or Ramsar site, then the Environment Agency will need to trigger the relevant consultation process with Natural England or CCW.

ii. 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.

iii. 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 you with missing data, or advise you on aspects that need a closer look.

iv. 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.

Note that 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 test pumping. 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, to act 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, because for future reference 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

5.1 Purpose of monitoring

The subject of monitoring has been mentioned briefly several times already in this report, but will now be discussed in much more detail. This chapter is by no means an exhaustive treatment of the subject, but serves to highlight the main issues of relevance to HIA. The focus of this section is on monitoring water levels and flows, but monitoring can also include water quality sampling and ecological surveys. In the context of groundwater abstraction, the main purposes of monitoring are as follows:

- To establish the baseline environmental conditions before the commencement of abstraction.
- To fill gaps in the knowledge of the hydrogeology and hydrology of the area around an abstraction. In other words, to improve the conceptual model by reducing the uncertainty.
- To demonstrate compliance with conditions attached to relevant abstraction licences or discharge consents.
- To trigger mitigation measures or temporary cessation of abstraction, if the water level in a receptor (such as a wetland) falls below an agreed trigger level, 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.

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 groundwater abstraction, 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. 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 abstraction 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 abstraction. 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 very well 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.

Gauging stream flows

- 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 systematically (including the quality control procedures themselves).

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, and discharge quantities (if the abstraction is not fully consumptive). Parameters such as rainfall and evapotranspiration are more likely to be obtained from sources such as the Environment Agency and the Met Office, as opposed to being measured. For more detail on the design and installation of monitoring facilities, the reader is referred to, 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. Note however, that 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, abstraction quantities 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) and Environment Agency (2003b).

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

- Each monitoring point should be clearly labelled with a unique identifier (for example, a number painted on the borehole cover), so that there is no 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, 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 of 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 abstractor. 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).

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

- 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, and working in or adjacent to: deep or flowing water; unprotected large-diameter wells; busy roads; and confined spaces (below-ground chambers or basements are often classified as confined spaces, into which access may be required to dip a borehole). The relevant searches for buried services and pipelines etc must be conducted before drilling boreholes.

Drilling a monitoring borehole

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. It is also a problem in terrain with significant topographic gradients, especially in low-permeability rocks, where 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 some 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 abstraction, will be available, to reveal the natural seasonal behaviour of the groundwater levels. This is especially important when dealing with periods of drought.

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 abstraction is likely to affect, and this should become apparent during conceptual model development.

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 groundwater abstraction, if a certain conduit does not happen to be penetrated by the monitoring borehole, for example (see Appendices 3 and 4 for a discussion of karst and fractured crystalline rocks).

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List of abbreviations

AP	Assessment Point (on rivers as part of the RAM Framework)
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
DETR	Department of the Environment, Transport and the Regions
GIS	Geographical information system
GWMU	Groundwater Management Unit
HIA	Hydrogeological impact appraisal
IGARF	Impacts of Groundwater Abstraction on River Flows (spreadsheet tool)
maOD	Metres above Ordnance Datum
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
SAC	Special Area of Conservation (under the EC Habitats Directive)
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

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

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)

Groundwater body: A distinct volume of groundwater within an aquifer or aquifers (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. Even with the coming into force of the Water Act 2003, several other pieces of legislation and regulatory regimes are still highly relevant to the assessment of the impacts of groundwater abstraction 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 the Directive's 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 abstraction 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:

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 that 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 effect: 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 an appropriate 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 integrity of the European site, 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 integrity of the European site 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).

As for where HIA fits in with the Habitats Regulations, the methodology would contribute to Stages 2 and 3.

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, about 25 sites have been confirmed as affected or potentially affected by abstraction (although not necessarily abstraction from groundwater).

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 a Resource Assessment and Management (RAM) Framework, which 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 can be summarised as consisting 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 hydro-ecological 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 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 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 the 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 now 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 programme of measures to make sure 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 100 per cent consumptive (UKTAG 2003). Where the use turns out to be non-consumptive or less than 100 per cent consumptive, this will be picked up by the further characterisation.

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Appendix 2: Test pumping

Introduction

This appendix, first referred to in Section 1.3 of the main report, discusses in greater detail the subject of test pumping. Conducting pumping tests on boreholes and wells has long been a standard technique for investigating borehole characteristics and aquifer properties, and much has been written on the subject. In particular, the reader is referred to the following publications:

- *British Standard BS6316:1992:* This is the code of practice for test pumping of water wells, and it provides good descriptions of how to plan, carry out, and present the data from pumping tests.
- *Kruseman and de Ridder (1990):* The standard textbook on the analysis and evaluation of pumping test data, covering all conditions: confined, unconfined, leaky, steady-state, unsteady-state, anisotropy, multi-layered systems, partial penetration, etc.
- *Driscoll (1986):* Found on the bookshelves of most hydrogeologists, this book gives practical descriptions of all aspects of designing, drilling, developing, test pumping and equipping boreholes and wells.
- *Price (1996):* Very accessible introductory textbook on groundwater, which includes a section discussing pumping tests against the theoretical background of groundwater hydraulics.
- *Brassington (1998):* Another popular textbook, concentrating on practical methods for hydrogeological fieldwork, including a whole chapter on test pumping.

This appendix does not attempt to compete with these publications in describing the theory and practice of test pumping, but instead focuses on how test pumping fits in with the HIA methodology. It was seen in Section 1.3 of the main report that test pumping is an integral part of the licensing decision-making process, coming after the Section 32(3) consent has been issued and a water features survey conducted, and before a licence application is made. However, the exact point in the HIA methodology at which the test pumping is undertaken can vary from case to case.

Types of pumping test

BS6316:1992 identifies five main types of pumping test, as follows:

i. *Equipment test:* designed to ensure that all the pumping equipment, discharge measuring devices, water level recorders, etc, are working properly and safely. It also enables those pump or valve settings to be determined that will give an appropriate sequence of pumping rates for the step test.

ii. *Step test:* designed to establish the short-term relationship between yield and drawdown for the borehole being tested. It consists of pumping the borehole in a series of steps, each of which is at a different discharge rate, usually with the rate increasing with each step. The final step should approach the estimated maximum yield of the borehole.

iii. *Constant discharge test:* carried out by pumping at a constant rate for a much longer period of time than the step test, and designed to provide information on the hydraulic characteristics of the aquifer. Information on aquifer storage coefficient can only be deduced if data are available from suitable observation boreholes.

iv. *Constant drawdown test:* carried out by pumping at variable rates to maintain a constant drawdown in the pumping borehole. This is far less common than a constant discharge test, and is mainly used for tests with suction pumps, when designing dewatering schemes, or for artesian boreholes.

v. *Recovery test:* carried out by monitoring the recovery of water levels on cessation of pumping at the end of a constant discharge or constant drawdown test (and sometimes after a step test). It provides a useful check on the aquifer characteristics derived from the other tests (but is only valid if a footvalve is fitted to the rising main, otherwise water surges back into the borehole).

These tests are usually carried out in combination, with a typical test sequence on a new borehole being: equipment test, step test, constant discharge test and recovery test. An operational borehole with known yield-drawdown characteristics might only be subjected to a constant discharge test and recovery test. The value of the recovery test is often underestimated, with not enough effort being put into continuing the monitoring once pumping has ceased. Ideally, the duration of the recovery test should be as least as long as the duration of the pumping phase of the test programme. Recovery tests are valuable for several reasons:

- The water levels in the pumping borehole are easier to measure accurately, in the absence of turbulence caused by the pumping (especially in the early stages of the test, when water levels are changing quickly).
- The start of the test is much 'cleaner'. In practice, the start of a constant discharge test, for example, vary rarely achieves a clean jump from no pumping to the chosen pumping rate. Switching a pump off is usually much easier than starting a pump, and the jump from a constant pumping rate to no pumping can be achieved fairly cleanly. This improves the quality of the water level data significantly in the very early stages of the test.
- They represent a good option for testing operational boreholes that have already been pumping at a constant rate for extended periods. In these cases, the recovery test can be performed first, when the pumps are first switched off, followed by a constant discharge test when the pumps are switched back on again.

Designing a pumping test

In the context of HIA, the test pumping programme should be designed to test and improve the conceptual model, as ultimately it is the conceptual model (and not the pumping test by itself) that will be used to determine whether or not derogation is likely or whether sensitive sites such as wetlands may be adversely impacted. The pumping test by itself cannot normally be used to make the decision, as it is unlikely to have been carried out in dry enough conditions or for long enough to show these impacts adequately. The degree to which the conceptual model can be improved depends on the nature of the pumping test; a major test for a large abstraction with several purpose-drilled observation boreholes and extensive monitoring of springs, wetlands etc, might yield enough information to develop an intermediate model. Bear in mind that the way that the groundwater levels respond to the abstraction provides information on the aquifer boundaries and on the relationship with surface water bodies that may not be apparent from surface water measurements alone. This sort of information is very useful in refining the conceptual model and gaining confidence in its reliability.

It is recommended that the test is designed by first carrying out a virtual pumping test, using the basic conceptual model with initial estimates of aquifer properties, to predict where and when the effects of pumping will be detectable. Standard pumping test software, such as Aquifer^{win32} (the Environment Agency's standard software package for test pumping analysis), can be used for this purpose. A virtual pumping test will enable you to evaluate the pumping period required to affect water levels in key locations. This allows the appropriate length of test to be defined, and avoids the applicant having to measure things that are unlikely to be affected within the test period. You should consider at this point the implications of the initial conditions. For example, if the pumping test is carried out during a period of high groundwater levels, is it possible that the expected impacts will be masked?

Where it is suspected that rivers or other surface water features may be impacted, IGARF (Environment Agency 1999 and 2004) can be used to help design the duration of the test and to identify which watercourses to monitor. For applications affected by the Habitats Directive, the requirements of the appropriate assessment (see Appendix 1) must be considered when planning the pumping test. Note that the pumping test itself must not be allowed to have a detrimental effect on the conservation site.

When designing the test, it is important to ensure that it is possible to collect all the necessary data, including pre-test readings. For example, on a test with several monitoring points, it may not be easy to record water levels at the required frequency during the first few hours of the test if the monitoring is being done manually. However, early data may be important in understanding the aquifer behaviour (and particularly important for observation well data near the abstraction or in low storage systems), so the applicant should ensure that these measurements can be taken satisfactorily. If pressure transducer/data logger systems are used, it is important to specify how the data should be presented to ensure that they are in a suitable format to be used for the interpretation. Monitoring was discussed at greater length in Section 5 of the main report.

Groundwater quality data may also help in the interpretation of the hydrogeology. In some circumstances it may be necessary to take water samples for analysis at a number of stages during the test period, or to record aspects of the quality (such as electrical conductivity) throughout the test. Such data may assist in predicting long-term changes in the groundwater quality, or facilitating a better understanding of the groundwater system in other ways.

The importance of spending time on designing the pumping test is being emphasised here because the test may need repeating if it was wrongly carried out, and this can be very expensive. In addition, it may be necessary to repeat the water features survey over a larger radius if the test reveals a much larger area of influence than was expected.

Interpreting the pumping test

The pumping and recovery test results can be interpreted using techniques such as:

- Manual curve-fitting methods such as Theis (for confined aquifers) and Neuman (for unconfined aquifers).
- Spreadsheet methods for solving the Cooper-Jacob approximation to the Theis equation.
- Software such as Aquifer^{win32}, which has numerous options.
- Comparing the water levels and spring flows monitored during the pumping test.
- In a limited number of cases, using numerical models (radial flow models, for example).

When using analytical equations, it is important for the analysis to be carried out using equations that are appropriate to the aquifer being analysed, and the predicted drawdown from the equation used should reasonably match the observed drawdown. If not, an alternative equation should be tried that assumes a different hydrogeological structure. The assumptions inherent in the equation that gives a good fit to the data must, of course, be geologically plausible. Validation occurs if the geological features can be shown independently to be correct. The reliability of each method depends on how closely the aquifer conditions correspond to the assumptions inherent in that method. For example, the assumptions of the Theis equation are:

- The aquifer is homogeneous, isotropic, of uniform thickness and infinite areal extent.
- The aquifer is confined (although Theis can be a good approximation for unconfined aquifers if the drawdown is small compared with the saturated thickness of the aquifer).
- The piezometric surface is horizontal before pumping.
- Flow in the aquifer is entirely horizontal.
- Storage in the well can be neglected.
- Water removed from storage is discharged instantaneously with decline of head.

In addition, the Cooper-Jacob method assumes that $u < 0.01$ where $u = r^2S/4Tt$ (r is the radius from the pumped well, S is aquifer storativity, T is transmissivity, and t is time). In other words, it is only valid for drawdown observations made close to the well or after a sufficiently long pumping time. If the abstraction is close to a boundary of the aquifer, then the assumption of infinite areal extent is invalid, and the method of image wells should be used. Full information on the analysis and evaluation of pumping test data can be found in Kruseman and de Ridder (1990).

Finally, it is worth repeating that it is vital to ensure that the assumptions inherent in the chosen method of analysis match the geological reality around the borehole in question. It is particularly easy to overlook this step when using software, and the validity of the assumptions should always be checked. Modelled responses fitting observed data is not necessarily validation, as the fit may be due to compensating errors, or there may not be a unique solution.

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Appendix 3: Karst

Introduction

This appendix, first referred to in Section 3.4 of the main report text, discusses the particular characteristics of karst aquifers that need to be taken into account when developing conceptual models and undertaking HIA for groundwater abstractions in such situations.

Acknowledgement on authorship: the content of this appendix has largely been distilled from a longer appendix in Boak et al *(2006), on the hydrogeology of karst, which was contributed by Professor Peter Smart (School of Geographical Sciences, University of Bristol, 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, which 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).

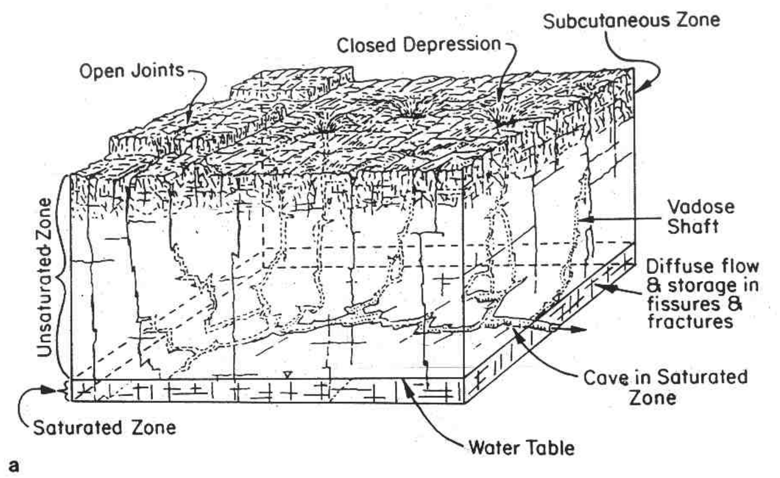

Figure A3.1 Features of a karst groundwater system
(from Bakalowicz *et al* 1995)

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.

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 abstraction point. Smaller impacts occur in areas more distant from conduits where intergranular and small fracture flow dominate – potentially quite close to the abstraction point.

- *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 runoff 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:

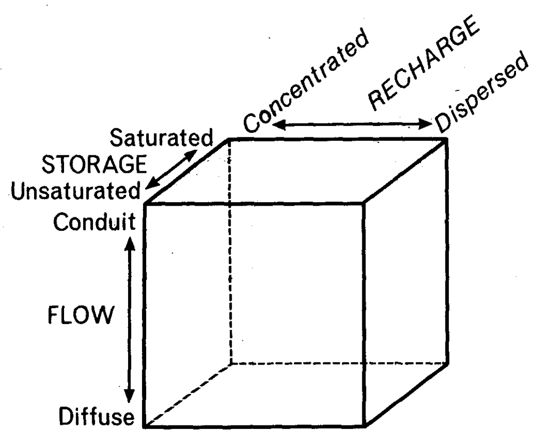

Figure A3.2 Carbonate aquifer model
(from Smart and Hobbs 1986)

- *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. 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 basis for evaluation of the extent of karstic behaviour in a carbonate aquifer. Whilst it is unlikely in practice that the position of an 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 (Figure A3.3) with respect to assessment of potential hydrogeological impacts (of quarry dewatering, in their study):

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.

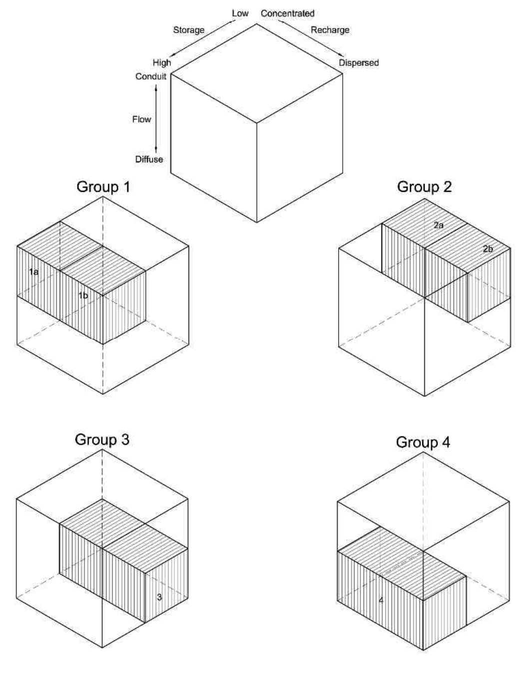

Figure A3.3 Carbonate aquifer classification
(after Hobbs and Gunn 1998)

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. 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 groundwater (including dewatering) may therefore be significant. They are often developed by boreholes rather than by spring abstraction. Examples include Pwlly 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.
- Tracer detection in observation wells is focussed at specific sites, rather than forming a 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.

Great care is needed when using information from boreholes. Water levels 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 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 passages 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).

It is important to recognise that with the possible exception of the results of tracer tests, none of the criteria listed above 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.

Management of potential hydrogeological impacts in karst ('monitor and mitigate')

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.

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 (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.

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.

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 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 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 cases before any authorisation is granted by the Environment Agency).

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.

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. The scale of this problem is demonstrated by the fact that only 40 per cent of monitoring boreholes in the East Mendips exhibit 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 likely to be affected by the proposed abstraction. Given the considerable uncertainties in prediction of the extent of impacts from abstraction 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 the commencement of abstraction. A minimum of three years pre-abstraction 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 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 of Kundzewicz and Robson 2004, and associated papers in this special volume). To date, such techniques have received limited use in monitoring the impacts of abstraction, 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-abstraction monitoring data. Statistical techniques such as multiple regression can be used to relate key properties of the monitoring series (such as annual maximum 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).

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.

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 abstractors 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 still occur, remedial damages may be awarded.

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 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

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.

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.

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Appendix 4: Crystalline rock

Introduction

This appendix, first referred to in Section 3.4 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 for groundwater abstractions in such situations.

Acknowledgement on authorship: the content of this appendix has largely been distilled from a longer appendix in Boak et al *(2006), on the hydrogeology of fractured crystalline rock, which was contributed by David Banks (Holymoor Consultancy, Chesterfield, UK).*

Characteristics of crystalline rock aquifers

Crystalline rocks, such as slates, granites, marbles, basalt and dolerite are not usually regarded as aquifers, but they can represent locally-important sources of groundwater. 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 when making any assessment of the impacts of groundwater abstraction:

- The generally low hydraulic conductivity (K) means that groundwater flow towards boreholes, and therefore borehole yields, 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 may intercept two different fracture systems and thus experience very different yields and groundwater chemistries, and may be in poor hydraulic continuity with each other.
- 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 result 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 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 datasets 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:

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

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:

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

Lithologically-based subsets of large borehole-yield datasets 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 an abstraction in a crystalline rock aquifer.

Specific transmissive features

Of course, a borehole 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. Such features 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 borehole construction, 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. 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 groundwater abstraction.

If it is suspected that the bog is in continuity with the water table, the impact of abstraction 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 A4.1, 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 abstraction or 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.

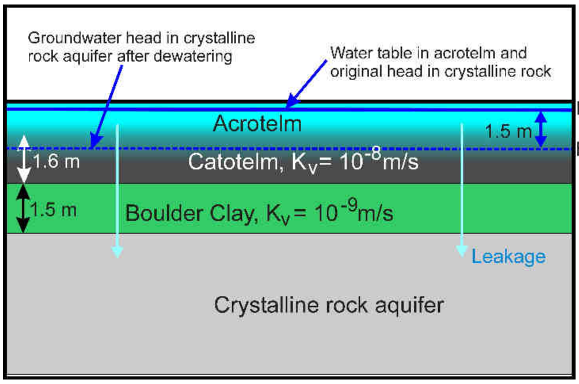

Figure A4.1 Schematic section of a bog developed on a boulder clay till substrate overlying a crystalline rock aquifer
(The bog may be in some degree of hydraulic continuity with the crystalline rock aquifer and a calculation of potential leakage through the catotelm and the till can be performed. It 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 HIA methodology described in this report *can* be used to assess the likely hydrogeological impacts of groundwater abstraction 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 these 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.

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