

ASR – UK: ELUCIDATING THE HYDROGEOLOGICAL ISSUES ASSOCIATED WITH AQUIFER STORAGE AND RECOVERY IN THE UK

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Author of Report	Gale, I N. Williams, A T. Gaus, I. Jones, H K
Principal Researcher	Gale, I N
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ASR - UK: ELUCIDATING THE HYDROGEOLOGICAL ISSUES ASSOCIATED WITH AQUIFER STORAGE AND RECOVERY IN THE UK

FOREWORD

The term Aquifer Storage Recovery (ASR) was originally coined by David Pyne of CH2M-Hill in the 1980's to describe the technique of creating additional storage at existing water treatment works by injecting surplus potable, treated water into aquifers underlying the works. Injection is via dual-purpose, recharge/abstraction boreholes, often by adapting existing abstraction boreholes. The technique caught on in the US because it provided answers to questions, including:

- How do we meet growing demand without large capital investment in new surface reservoirs?
- How can we design schemes with acceptable impacts on the environment?
- How can we be confident of regulatory approval and avoid lengthy public enquiries?

The steadily increasing number of ASR projects across the US since the late 1980s is testament to the cost-effectiveness of the technique and its operational flexibility.

In the UK, the water industry became interested in ASR* as a real option in the latter part of the 1990s. This was partly, but not wholly, as a result of the higher national profile water resource planning took on since the summer of 1995 and the associated pre-AMP3 regulatory guidance on water resource options. Thus, from OFWAT, we had the useful guideline that options should be small-scale and cost-effective and from the Environment Agency the threefold criteria of sustainability, sustainability and sustainability.

Interest in ASR also came from UK hydrogeologists who realised that because the injected potable water displaces the native groundwater to create a "bubble," the technique could work successfully in unused, non-potable aquifers. Successful development of ASR schemes into saline aquifers in the Florida Keys ably demonstrated this principle. Clearly, an innovative way of utilising the storage potential in hitherto untapped UK aquifers had now opened up. In the era of planning for climate change this indeed represents a huge opportunity for creating sustainable storage.

ASR in the UK is currently progressing up the steep part of the learning curve. To date there have been several ASR pilot investigations but, as yet, no schemes have been commissioned. This initial phase of work has met with mixed success and has drawn out several technical issues and uncertainties, most of which have become the focus of this study.

From the water resource planning perspective, ASR has many plus points and is a key option in the Environment Agency's Water Resources Strategy for England and Wales. However, from an asset investment perspective the current technical uncertainties

around ASR mean that other options, more certain of success,** albeit possibly more costly and environmentally less sustainable, may be given higher priority. The outputs from this project will help greatly in providing a better understanding of the technical problems that may be encountered and thereby enable the design of an appropriate feasibility study, and staged development programme, to minimise risk of failure.

The outputs from this project consist of this summary report, a web site, interim technical reports and models on the web site and the attached CD, as well as several scientific publications. Together these provide greater understanding and analytical elucidation of the environmental impacts, yield potential and quality changes associated with ASR in UK aquifers.

Finally, for those water companies who believe that they will not need to develop new water resources over the next 25 years this project will be of academic interest only. However, for the rest of us I would suggest that assimilating the outputs of the project is a compulsory exercise. Whilst water resource strategists and planners need to understand the key messages herein, it will be the front-line practitioners, largely hydrogeologists, who will gain most from the project's technical findings and analytical tools. With AMP4 on the horizon, this jointly-funded project represents not only good value for money but is also a timely and needed addition to our planning armoury.

Brian Connorton

UKWIR technical member of project Steering Committee

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* The term ASR has been adopted by UK hydrogeologists to describe recharging, hitherto unused or under-used aquifers, via dual-purpose, abstraction/recharge boreholes to create an ASR "bubble." In this regard ASR is a sub-set of artificial recharge. The latter is the generic term applied to any technique for transferring water into aquifers (intentionally or as a result of other activities) either indirectly through surface recharge basins or directly by borehole injection. Thus hydrogeologists tend not to refer to the large scheme in North London (NLARS) as an ASR scheme but rather as an artificial recharge scheme. This is because it has numerous injection points into a widely-used potable aquifer and the management principle is not so much to create a recharge bubble under each site, but to manage the aggregated recharge mound over a large extent of the confined Chalk aquifer.

** Success is here regarded as having achieved the designed deployable output at the budgeted cost and with the necessary regulatory approvals.

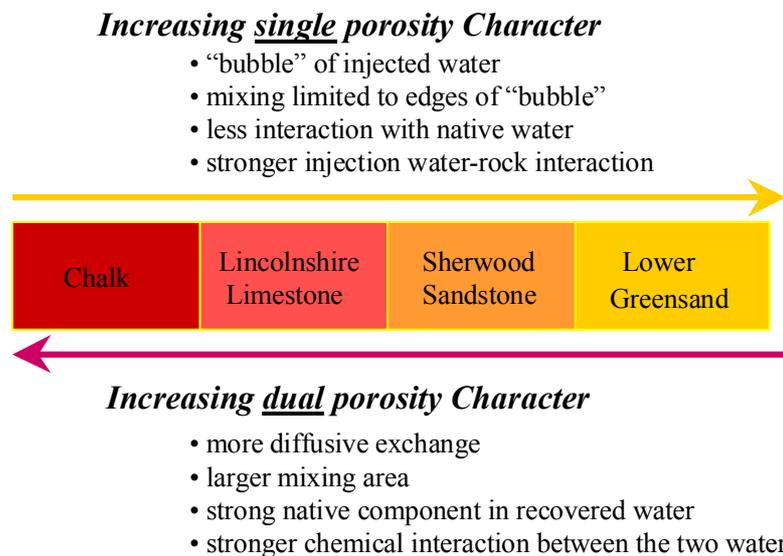
Executive Summary

Objectives

- Promote ASR through research into generic issues related to risks that are currently constraining development of ASR to British aquifers, namely:
 - impacts of the dual-porosity behaviour of the Chalk and other aquifers on recovery efficiency
 - geochemical interactions; water/water and water/rock in different aquifers
 - environmental impacts in relation to operational cycles
- Facilitate the exchange of information through the usual media as well as the Internet. Convene annual national symposia on ASR for the water suppliers, regulators and researchers and support a strong presence at international fora.

Conclusions

- British aquifers can be considered as part of a continuum ranging from single porosity unconsolidated aquifers (e.g. Lower Greensand) to fractured dual-porosity aquifers (Chalk).
- The response of these aquifers to ASR varies in a predictable manner that is controlled by the proportion of fracture and matrix porosity, both from the physical mixing and geochemical interaction perspectives. This aquifer ‘continuum’ and the impact trends are illustrated below.



- The conclusions from the work undertaken in this study give generic indications of how aquifers will respond to ASR. Because of geological variability, it must be stressed that this is only intended to act as a basis for site-specific assessments. Some modelling tools have been developed to assist in these assessments.

Recommendations

- As data become available from trials in the Sherwood Sandstone and Lower Greensand aquifers it should be used to refine the physical and geochemical models, provide relevant feedback and compare results with those from the trials.
- Apply relevant models developed by this project to current and prospective ASR schemes to assist in decision making, particularly in relation to the environmental impact assessment.
- The modelling has identified the parameters that are of most significance in a variety of hydrogeological environments. There are very few measurements of some key parameters, such as aquitard properties, and a database needs to be compiled in order to facilitate future developments.

Benefits

- A clearer understanding has been gained of the response of British aquifers to ASR from both the physical and geochemical perspectives.
- Simple modelling tools have been developed (supplied with this report) to assist in decision-making from the early stages of ASR schemes. These tools are designed to help assess the environmental impacts of the scheme and to identify the parameters that need to be determined in order to proceed.
- Physical (SWIFT) and geochemical (PHREEQC) models have been applied to ASR schemes and an initial attempt to link the two was made. This will form the basis for improved understanding of the processes determining the efficiency of ASR scheme as more are developed in different aquifers.
- Regulatory considerations have been reviewed with respect to ASR in the light of new and forthcoming EU Directives and their implementation.

**For further information please contact UK Water Industry Research Limited,
1 Queen Anne's Gate, London SW1H 9BT quoting the report reference number**

Contents

Page Number

Foreword

Executive Summary

1	Background and aims of the project	1
2	ASR – using aquifers to store water in the UK	2
2.1	Introduction	2
2.2	Application of ASR	3
2.3	Drivers for the use of ASR in the UK	4
2.4	Potential for ASR in the UK	5
3	Environmental benefits and constraints	10
3.1	Introduction and modelled scenarios	10
3.2	Confined ASR aquifer overlain by unconfined aquifer	11
3.3	Effect of an ASR scheme on the outcrop area of the target aquifer.	12
3.4	Conclusions	15
4	Significance of aquifer parameters in major aquifers	16
4.1	Introduction	16
4.2	Impacts of the main British aquifers on mixing during ASR	17
4.3	The dual-porosity issue	17
4.4	Impact of mixing in British aquifers	20
5	Water – groundwater – rock interactions	25
5.1	Introduction	25
5.2	Geochemical assessment for the different aquifers	28
5.3	Case Study: fluoride modelling of the ASR trial in Lytchett Minster	31
6	Regulatory considerations	33
6.1	Regulations affecting Artificial Recharges to groundwater	33
6.2	Guidance on dealing with authorisations for artificial recharge and recovery (ARR) schemes	33
6.3	Phased development/authorisations	35
6.4	Environmental Impact Appraisals (EAPs)	35
7	Summary and conclusions	40
8	References	43
	Appendix 1	45

1 Background and aims of the project

The results of a study, funded by BGS, UKWIR and EA, and undertaken between 1995 and 1998, indicated that there is a large potential for development of Aquifer Storage Recovery (ASR) schemes in the UK at a wide range of scales (Jones et al., 1998). The study stimulated considerable interest and, at that time, three water companies had initiated field trials. Several other schemes were at the desk study stage.

Experience from the United States shows that ASR is frequently a cost-effective solution to a wide variety of water supply, as well as water quality problems. The potential of the technology in Britain needed to be more fully evaluated, in order to optimise the use of our water resources, both fresh and wastewater. However, several uncertainties were highlighted by this study that could constrain the acceptance and utilisation of ASR as a component in strategies for sustainable management of water resources.

The project reported here, ASR-UK, addresses some of the issues identified through generic research and the development of guidelines and models to assist in decision-making throughout the investigative, development and implementation stages of schemes. The issues addressed are:

- Research into the controls on dispersion and diffusion of injected water and their impacts on recovery efficiency. Models were developed to predict the extent and movement of “bubbles” of stored water in aquifers as well as changes in water quality with time. Work initially focused on the Chalk aquifer but other aquifers were also investigated.
- The impacts of geochemical interactions between native and injected water as well as water-rock interactions. The adverse as well as the beneficial effects need to be predicted in order to address them in the most cost-effective manner. The project therefore aimed to assess the significance of these reactions on the permeability of aquifers, the quality of recovered water and the efficiency of schemes.
- During the life of the project, the importance of the environmental impacts of ASR schemes, in relation to the operational cycle used, became increasingly apparent. Quantification of these impacts is required throughout all phases of a study, from the initial assessment of the potential of a site, through the staged testing to implementation and commissioning of a scheme.

The project was designed to address these issues through review of current knowledge and the application and development of models, some of which can be used as tools in current and future investigations. It was initially planned that the development of these models would be closely aligned with on-going investigations and field trials, which would provide data to validate the models as well as providing feedback to the investigation. This proved to be the case with the investigations in the Chalk aquifer but little data were available from other aquifers.

2 ASR – using aquifers to store water in the UK

2.1 Introduction

Artificial Recharge (AR) is a means of supplementing natural infiltration into an aquifer; it involves storing water in an aquifer at times when there is a surplus, potentially for use at times of scarcity. Traditionally, the technique has utilised aquifers containing potable water.

Artificial recharge is defined here to include any technique used to introduce water into an aquifer, via boreholes or basins, for any purpose. It can be used strategically where an aquifer is already over-exploited, such that no further abstraction would be allowed without artificial recharge taking place, or where lack of natural recharge prevents its utilisation. In these cases, the water could be abstracted from the same wells as used for injection, or from additional wells or natural discharges down hydraulic gradient. This technique is employed only in potable aquifers, as the water being abstracted is not necessarily the same water that was injected. The net water balance would however be zero or positive.

Artificial recharge has been carried out in many forms for centuries. A more recent development occurred around 50 years ago, when the first investigations of injecting potable water into saline aquifers were carried out (Cederstrom, 1957). This involved recharge of freshwater into the brackish aquifers of the coastal plains of Virginia in 1946; the injected water forming a lens or bubble of potable water within the saline body of groundwater. This practice has become known as Aquifer Storage Recovery (ASR).

Pyne (1995) defines ASR as:

The storage of water in a suitable aquifer through a well during times when water is available, and recovery of the water from the same well during times when it is needed.

The use of non-potable aquifers greatly increases the potential for using artificial recharge, and in the last couple of decades, the concept has been developed widely in the United States. However ASR is not a proven technology for UK aquifers, which can be quite distinct from aquifers that are used in the USA. The risks have been perceived as being too great to consider development of such schemes in the UK but, in recent years, several trial schemes have been undertaken, and the data and experience gained from these, are providing valuable support for the technique.

Areas where ASR are most likely to be considered in Britain would be in confined parts of aquifers that have not been used for productive water supply because of poor quality. It follows that little hydrogeological information is available, so embarking on a scheme becomes more speculative and the risk factors increase. Use of potable aquifers at outcrop for ASR schemes is less likely to occur as the aquifer may be fully licensed, natural recharge will occur and the groundwater-surface water interaction will be more immediate, and hence have environmental impacts. However, each case needs to be judged on its individual merits as ASR schemes can store valuable quantities of water in relatively small volumes of aquifer, so local hydrogeological environments could prove to be suitable.

2.2 Application of ASR

There are a large number of possible applications for ASR, some of which are listed in Table 2.1. It is envisaged that in the UK, ASR will typically be used to meet short-term peaks in demand by recharging aquifers during periods of low demand and recovering the water during periods of high demand. ASR should always be viewed in the context of alternative methods of resolving water supply or quality issues, as part of a broader strategy. In particular, the environmental benefits should be factored into the decision process.

Table 2.1 Applications of ASR (adapted from Pyne 1995)

Category	Application
Water storage	<ul style="list-style-type: none"> - Seasonal storage and recovery - Long term storage - Emergency storage - Diurnal storage
Water quality	<ul style="list-style-type: none"> - Chlorination by-product removal - Nitrate removal - Stabilisation of aggressive water - Control of contaminant plumes
Environmental improvement	<ul style="list-style-type: none"> - Restoration of groundwater levels, 'low flow' rivers and groundwater fed wetlands - Reduction of subsidence - Reduce/prevent saline intrusion - Enhancement of baseflow to streams
Operational advantages	<ul style="list-style-type: none"> - Maintenance of distribution system pressure and flow - Deferment of the expansion of treatment facilities - Deferment of the development of new sources

Ideally, ASR sites should be located in a confined, single porosity aquifer at sufficient distance from the outcrop to have an acceptable impact on flows in streams. The delay of this impact can be engineered to have maximum effect during periods of high stream flow, when it becomes insignificant. Sub-vertical hydraulic boundaries or a moderate value of transmissivity, are needed to constrain the distance at which the effects of injection and recovery are significant. A readily accessible source of water for injection is also needed and access to an existing treatment works could be important if the recovered water is not wholly potable. As with the development of a conventional groundwater supply, the site-specific factors determine its impact on the environment, the resource and other users. These issues also need to be addressed in ASR schemes,

together with those specifically pertinent to ASR, in order to optimise the management and use of water resources.

2.3 Drivers for the use of ASR in the UK

The aquatic environment is regulated by the Environment Agency in England and Wales, the Scottish Environment Protection Agency (SEPA) and the Department of the Environment for N. Ireland. Their agendas are determined by implementation of the regulations established by government, in order to comply with EC Directives, notably the Water Framework Directive (2000) and the Habitats Directive (1997).

Demands for water from a wide variety of users have to be accommodated. These include not only the Water Companies for potable supply and effluent dilution but also navigation, fisheries, irrigation as well as private and industrial abstractions. Environmental demands also have to be met to ensure acceptable river flows and maintenance of wetland habitats.

Demand management techniques such as leakage reduction, metering, industrial reuse, consumer use reduction and other water saving techniques have been successful but become less applicable as the cost/benefit ratio increases. In many areas the available water resources are fully licensed and in some cases the impacts of abstraction will result in the need to reduce or surrender a licence to abstract groundwater.

The options for replacement of these sources are limited, particularly where environmental concerns are valued highly. Construction of surface impoundments tend to meet strong opposition on the grounds of loss of land, habitats and negative environmental impacts and can take decades to come to fruition. Although construction of an additional borehole source can be relatively rapid, this option is often precluded on the grounds that aquifers are fully licensed or that the impacts on surface water and wetlands are unacceptable. Construction of pipelines to improve distribution networks, or importing water from considerable distances need to be assessed on their cost-effectiveness. The number of viable schemes will tend to diminish with time as the most cost-effective schemes are implemented.

Schemes such as desalination and wastewater reuse are high cost options that are usually only viable in small, limited and exceptional circumstances. In comparison, ASR becomes an increasingly attractive option, the main advantages being:

- Minimum environmental impact as the net water abstraction is zero or negative
- Installations can usually be located at or near where storage/demand problems need to be addressed
- Staged development to spread costs and to meet increasing demand
- Costs are usually less than alternative schemes
- *In-situ* improvements in water quality can reduce the need for treatment on abstraction

ASR is therefore a sustainable water resources and quality management technique that can be applied to address a wide range of problems as part of a broader water management strategy.

2.4 Potential for ASR in the UK

2.4.1 Introduction

There is a limited history of artificial recharge schemes in Britain. These include trial schemes on the Triassic sandstones at Clipstone and Edwinstowe (IWES, 1986) and at Stourbridge (Jones, 1983), and on the Lower Greensand at Hardham (O'Shea, 1984). However, the North London Artificial Recharge Scheme (NLARS), which recharges the Chalk and the overlying Tertiary sands, is the only major operational scheme. In more recent years, trial ASR schemes have provided data and experience on British aquifers, most notably the trials in the Chalk in East Anglia and the extensive testing programme in the Chalk at Lytchett Minster, Dorset.

Currently trials are being undertaken on the Sherwood Sandstone aquifer at Loftsome Bridge in Yorkshire and on the Lower Greensand aquifer at Stockbury in Kent. In addition, several agricultural supply schemes have been developed in the Nottinghamshire where groundwater is abstracted from the Permian limestone and stored in the overlying Sherwood Sandstone for later use as irrigation water. Yields from boreholes in the limestone are insufficient to meet the short-term intensive requirements and, as the Sherwood Sandstone aquifer is fully licensed, abstraction can only be permitted subsequent to addition of water; the net abstraction being zero or negative.

A regional assessment of ASR potential of the aquifers in England and Wales was carried out by Jones et al. (1998). Within the scale of the assessment, it was not possible to consider the potential for ASR in each of the minor aquifers, and therefore the study focused on the major aquifers (Table 2.2). This does not mean that all minor aquifers should be dismissed, as they can be important on a local scale. Figure 2.1 shows the locations of the major aquifers in the UK together with the locations of ASR, and other Artificial Recharge investigations.

2.4.2 Chalk and Upper Greensand

The Chalk, in hydraulic connection with the Upper Greensand in some areas, is an important aquifer over much of southern and central England. Although porosity is high (up to 40%), the pore throats are narrow, and most of the usable storage is within the fractures and larger pores. As a result, the Chalk frequently has high transmissivity, but low storage coefficient, resulting in high diffusivity, so the impact of ASR schemes on the aquifer at outcrop needs to be carefully assessed. Where the Chalk is in hydraulic continuity with overlying deposits (e.g. the drift in East Anglia, or the Palaeogene in the London Basin) or underlying formations (e.g. the Upper Greensand), storage is increased, and the combined aquifer may have enhanced potential for ASR.

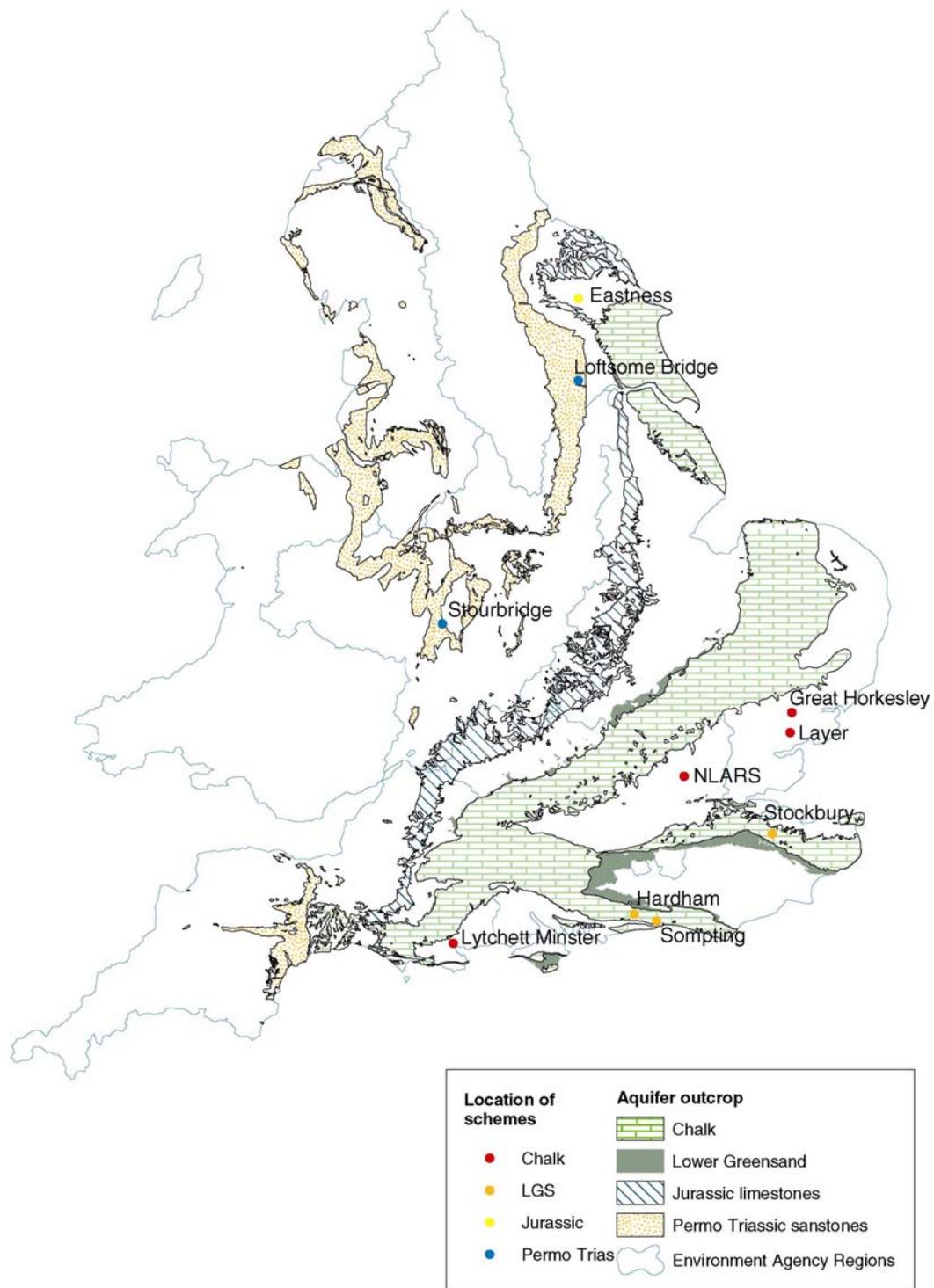


Figure 2.1 The location of ASR and other artificial recharge schemes in relation to the major aquifers in the UK

ASR may be constrained in the Chalk aquifer where the native groundwater is saline or brackish. When fresh water is injected it will tend to flow through the fractures, rather than through the matrix of the rock. Saline native groundwater in the matrix will then diffuse across a large interface and thus reduce the quality of the water subsequently recovered from the fractures. The significance of this process is discussed further in Section 4.3 of this report. In some cases the lack of environmental impacts of ASR may outweigh this water quality issue, which can be dealt with through treatment or dilution on recovery. Where the native groundwater has no serious limitations in terms of water quality, and the hydraulic parameters are satisfactory, the Chalk aquifer will have good potential, as illustrated by the success of the North London Artificial Recharge Scheme (NLARS) (O’Shea et al., 1995).

The Upper Greensand may also have potential for ASR due to its high storage capacity. Impermeable horizons within the formation can act as confining layers, thus creating a multi-layered aquifer and offering the opportunity for stacking. The degree of hydraulic continuity between the Chalk and Upper Greensand varies but where there is continuity, the combined aquifer may have greater potential. High iron in solution in some areas could pose a problem with plugging of boreholes due to the precipitation of iron oxides when injecting oxygenated water into a reducing environment. However, if this process is understood, it can be controlled and managed.

Table 2.2 Potential for ASR in England and Wales

Region (Env. Agency)	Chalk	Lower Greensand	Jurassic Limestones	Permo-Triassic Sandstone
Anglian	Moderate	Good (south of Region)	Lincs Lst: Mod. in south	
Midlands				Moderate to good
North East	Moderate to low (south of Region)		Corallian: Low to moderate	Moderate to good (best in south)
North West				Moderate to good
Southern	Moderate UGS: good in southwest	Good (southwest of Region)		
South West	Moderate to good (east of Region)		Moderate to low (east of Region)	Moderate to good (east of Region)
Thames	Good (East London)	Good (west of Region)	Moderate?	
Welsh				Low

n.b. Empty cells indicate absence of, or insignificant aquifer in region

2.4.3 Lower Greensand

The Lower Greensand is an important aquifer in parts of southern England, mainly in Bedfordshire and Cambridgeshire, and to the south of London around the Weald and on the Isle of Wight, as well as other smaller outliers elsewhere. It has a limited area of outcrop, although its high storage and generally good water quality render it important locally. Water quality can be exceptionally good, even where the aquifer is deeply confined, for example at Slough.

The Lower Greensand has good potential for ASR as transmissivity values, even where confined, are generally reasonable, and storage coefficients are high. It is most likely to be of use where it attains maximum thickness, and where intergranular flow is dominant, for example in the west of the Weald.

2.4.4 Jurassic Limestones

The Jurassic sediments outcrop in a broad band from the southwest to the northeast of England. The two main limestones are the Great Oolite Limestone and the Inferior Oolite. The minor limestones are frequently relatively thin, and rarely extend over large areas. Intergranular permeabilities are generally low, and the dominant flow mechanism is through solution-enlarged fractures. The high transmissivity combined with low storage results in the aquifers having a high diffusivity and hence rapid response times to pumping or injection. This may result in impacts in the unconfined part of an aquifer, even where boreholes are several kilometres from outcrop.

Water quality is good close to outcrop, but tends to become more saline down dip, beneath confining strata. Thus, although the limestones may have some potential for ASR, the potential is only likely to occur at a local scale.

2.4.5 Permo-Triassic Sandstones

The Permo-Triassic sandstones crop out across southwest, central, northeast and northwest England, and the Vale of Clwyd. They comprise a thick (up to 1000 m), variable sequence of predominantly sandstone, with inter-layered fine-grained horizons. Although matrix porosity and permeability are fairly high, the aquifer frequently relies on fractures for much of the groundwater flow close to boreholes. Although fractures generally close with increasing depth of burial, they have been reported even at depth, to be sufficiently open to maintain a high transmissivity. For example, during investigations for the Stourbridge scheme, Jones (1983) reported a fracture at 158 m depth in the recharge borehole. Faults have been reported to compartmentalise the aquifer in some areas, for example the West Midlands. Such isolated aquifer blocks could be particularly useful in terms of ASR.

The great thickness of the sandstones combined with their high matrix porosity and the influence of fractures, may result in injected water being “lost” unless very large quantities are injected. To minimise this loss, ASR schemes should be located in sections of aquifer that are confined by marl, or other low permeability strata, with as few fractures as possible. The injected water will then tend to displace inter-granular water around the borehole, thus increasing recovery efficiency.

2.4.6 Magnesian Limestone

The Magnesian Limestone occurs in a narrow north-south strip from Sunderland to Nottingham. It comprises two or three limestone horizons, separated by marls and siltstones. In the north, the aquifer is divided into three, with the Middle Magnesian Limestone having the greatest porosity. The transmissivity of the Magnesian Limestone is controlled by fracturing, hence aquifer properties are very variable and unpredictable. The reliance on fracture flow may limit its potential for ASR.

2.4.7 Carboniferous Limestone

The Carboniferous Limestone occurs in Derbyshire, the Mendips, north and south Wales, and in northwest Yorkshire. The matrix of the Carboniferous Limestone has a very low porosity and permeability, and the formation functions only as an aquifer by virtue of the secondary network of solution-enlarged fractures. It is characterised by very rapid groundwater flow, the direction of which may be difficult to predict. Boreholes yields are unpredictable and may have very high yields if they intercept fractures, and be non-productive if they fail to do so. Much of the yield from the aquifer is from spring discharge so, in general, the aquifer is unsuitable for ASR.

2.4.8 Minor aquifers

Other **minor aquifers** should be considered on an individual basis. ASR schemes require areas of aquifers in the order of tens of hectares so suitable, local geological settings can be adequate. Aquifers identified as having some potential include the **Fell Sandstone, Millstone Grit, Kellaways Rock** and some **Upper Lias Sands**. When considering these aquifers their potential can be reviewed in respect of the “aquifer continuum”, ranging from unconsolidated arenaceous deposits with high matrix porosity (Lower Greensand), through consolidated sandstone aquifer with high matrix porosity and some fracture flow (Sherwood Sandstone), fracture-flow dominated, high diffusivity aquifers (Jurassic Limestone) to fracture flow dominated aquifers with dual porosity mixing being the dominant process (Chalk).

3 Environmental benefits and constraints

3.1 Introduction and modelled scenarios

ASR schemes can utilise aquifers, at relatively small cost, that would otherwise be considered as marginal or useless. ASR can also have benefits when used in conjunction with potable aquifers in situations where the resource within the aquifer is limited. Examples of these situations are when an aquifer has limited recharge, or where an aquifer is already being exploited fully and where environmental impacts may restrict the amount of groundwater that can be abstracted during certain seasons. In this chapter, the factors that might affect the operation of an ASR scheme are examined and methods for their quantification presented.

Of particular concern is the impact that an ASR scheme may have on water levels in adjacent aquifers or in the unconfined zone of the target aquifer. As ASR schemes are designed to not result in a net abstraction from the aquifer, the normal water resources considerations about environmental impacts are not relevant and different factors need considering. In the long-term, there may be some net addition of water to the aquifer, but this should be small in relation to the overall resource. However, it does not necessarily follow that environmental impacts (e.g. water levels at outcrop) are negligible during the injection or abstraction phases.

A scheme that injects and then abstracts the same volume of water will, as a long-term steady state average, have no impact on local or regional groundwater levels. However, the seasonality of the scheme means that the local water levels will first be increased and then reduced. The timing and absolute value of these changes are of importance in assessing the eventual impact of a scheme. The work described here investigates the short-term effect on water levels and flows caused by the addition of water to the aquifer, and its subsequent removal.

Several situations have been considered and numerical and analytical models have been developed to examine these situations. The first is the case when the target aquifer is confined but is overlain (above the confining layer) by another aquifer (in this case, an unconfined aquifer). The impact that injecting water into the underlying aquifer has on the upper aquifer will be considered in terms of a change in flow between the two aquifers. Another case, which is investigated, is that of a target (confined) aquifer which outcrops at some distance from the proposed ASR scheme. The effect on water levels at the point where the aquifer becomes confined is estimated using a numerical model, for a range of transmissivity values and aquifer dimensions. Consideration has also been given to the time at which the maximum effect occurs. Descriptions of the models that have been used to investigate these situations are included in this report. It is anticipated that they will be used at an initial stage of ASR investigations, using parameters appropriate to the particular hydrogeological setting. The software is appended on disk and the models are discussed in greater detail in Williams et al. (2001).

It is envisaged that these ‘first-pass’ hydrogeological models will give one of three indications for a particular site:

- The impact on the adjacent water table is negligible
- The impact is definitely significant and the site is not suitable

- Further investigation is required; perhaps including more detailed modelling.

3.2 Confined ASR aquifer overlain by unconfined aquifer

Before work on assessing ‘environmental impact’ can be started it is important to decide what exactly is meant by the term and how it is best measured. On initial inspection it would appear that the best way to measure the ‘impact’ on an overlying aquifer (as depicted in Figure 3.1) would be in terms of a change in head caused by the operation of the ASR scheme. However, further consideration suggests that this may not be the case. The first problem that is encountered is the question of where this change in head should be calculated. It is obvious that the maximum change will be directly above the abstraction point i.e. at the ASR well. However it is equally obvious that the change at this point is not of any great interest as not only will it be in an area controlled by the operators of the ASR scheme but it will also be very much the worst case.

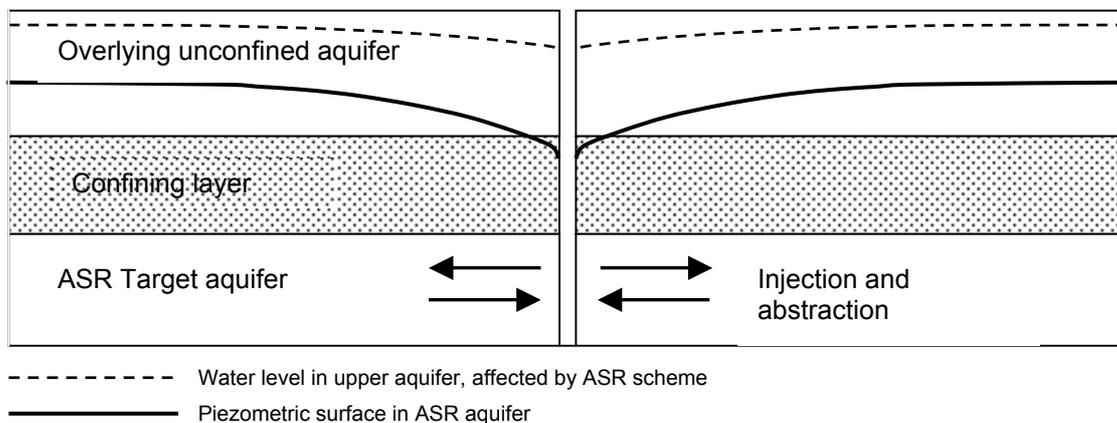


Figure 3.1 Schematic of modelled situation

The change in head at a specified distance (say 100 m), or distances, from the ASR well might be more useful and appropriate. The choice of distance should probably be influenced by the hydraulic properties of the overlying aquifer, as a cone of depression will have a different shape depending on the permeability (a lower permeability aquifer will have a deeper, less extensive cone than a higher permeability aquifer). The estimated effect, in metres, at the chosen point could then be used to decide whether or not the ASR scheme would have a significant impact on the overlying aquifer and the surface environment. The significance of the impact would probably be compared to the natural groundwater level fluctuations expected on an annual basis, or some similar criterion. Thus the decisions, which have to be taken are: a) where to estimate the change in head and b) with what to compare it?

Both of these decisions depend in part on the properties of the overlying aquifer. It would be more convenient to have a measure of the environmental impact of an ASR scheme that is independent of these properties. An ASR scheme will ‘impact’ on the overlying aquifer if it alters the water resources available in that aquifer or alters surface water features such as streams and wetlands. Thus a useable measure of impact would be the flux created into or out of the upper aquifer as a consequence of operating the ASR scheme. This value could sensibly be compared with the water resource available

in the upper aquifer before starting the ASR scheme, as measured by the annual recharge rate, for instance. Alternatively, the leakage out of the upper aquifer could be considered as a new ‘abstraction’ within the aquifer, where the magnitude of the leakage is compared to an abstraction rate.

The equations, which describe the flow of water within the aquifer system, have been solved analytically and the solution (effectively an extension of the Theis solution) has been incorporated into a spreadsheet. The spreadsheet allows the impact of an ASR scheme operating under variable cycling regimes to be estimated. The relevant parameters, which are required, are the storage coefficient of the aquifer containing the ASR well, and the vertical permeability, thickness and specific storage of the intervening aquitard.

Interestingly, the transmissivity of the target (ASR) aquifer is not a required parameter. This is because the focus is on *total* leakage through the aquitard, and not on how it is distributed areally. Another point of interest is that it is the properties of the aquitard that dominate. This is important, as these parameters are often not well known and this suggests that some effort should be focussed towards measuring these parameters, as there is no database of aquitard parameters for the UK.

The spreadsheet model has been used to show how the leakage from the upper aquifer varies during an ASR scheme run for five annual cycles (Williams et al. 2001). The effect of different aquitard hydraulic parameters and physical parameters can also be shown. The key findings are:

- Aquitards with low vertical hydraulic conductivity (k_v of less than 10^{-3} m/d) result in an impact in the upper aquifer which is of the same magnitude as that in the target aquifer with effectively no time delay, i.e. the system behaves as one aquifer.
- The higher the storage coefficient of the ASR aquifer, the less the impact on the upper aquifer.
- If the aquitard is compressible (i.e. high aquitard specific storage) the impact on the upper aquifer becomes negligible.

3.3 Effect of an ASR scheme on the outcrop area of the target aquifer.

In order to give an indication of the effect that an ASR scheme might have in the outcrop area of a target aquifer, a numerical model has been developed. It is possible that an ASR scheme may have an adverse environmental effect on streams or wetlands in the area where the target aquifer becomes unconfined.

Streams and wetlands are influenced by the water level in the aquifer that feeds them. Therefore, in this case, the head change within the aquifer near where it becomes unconfined *is* a useful measure of environmental impact of an ASR scheme. This can be related easily to the heads required to maintain stream flow or wetland function. The numerical model has been used to show the drawdown (i.e. the lowering of the water level) which will occur at the outcrop at the closest point to the ASR well. This will give a worst-case as the drawdown will be smaller further from the ASR well. Figure

3.2 shows the layout simulated in the numerical model. This is described in more detail in Williams et al. (2001).

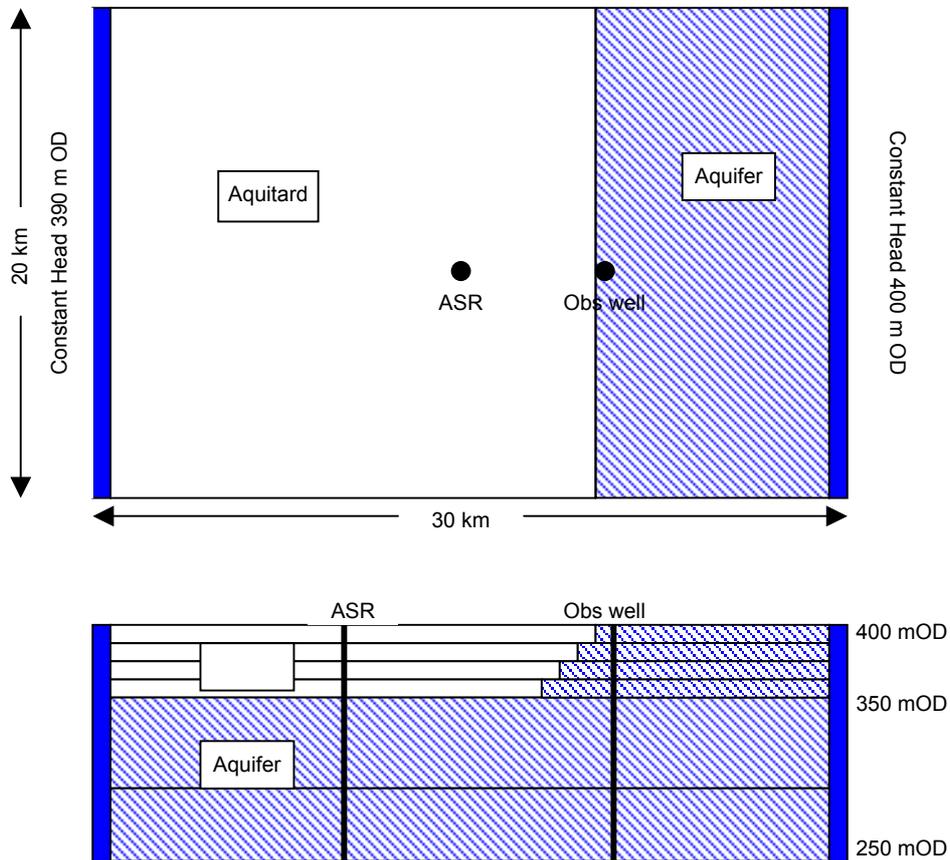


Figure 3.2 Schematic plan and cross-section of the modelled scenario.

The numerical model makes it possible to estimate the drawdown at the edge of the outcrop for different values of the aquifer and aquitard properties. It is also possible to change the thickness of the layers within the model. This makes it a very useful tool for initial investigations of the potential environmental impacts. Examples of the influence of the various aquifer parameters are given in Williams et al. (2001) and the model itself is appended to this report on disk.

For example, the modelling shows that the *distance* from the ASR well to the outcrop makes a significant difference to the potential impact of the ASR scheme. The estimated drawdown at the edge of the outcrop area, after five annual ASR cycles, changes from 0.25 m to 0.04 m as the ASR well is moved from 2 km to 15 km from the outcrop (Figure 3.3).

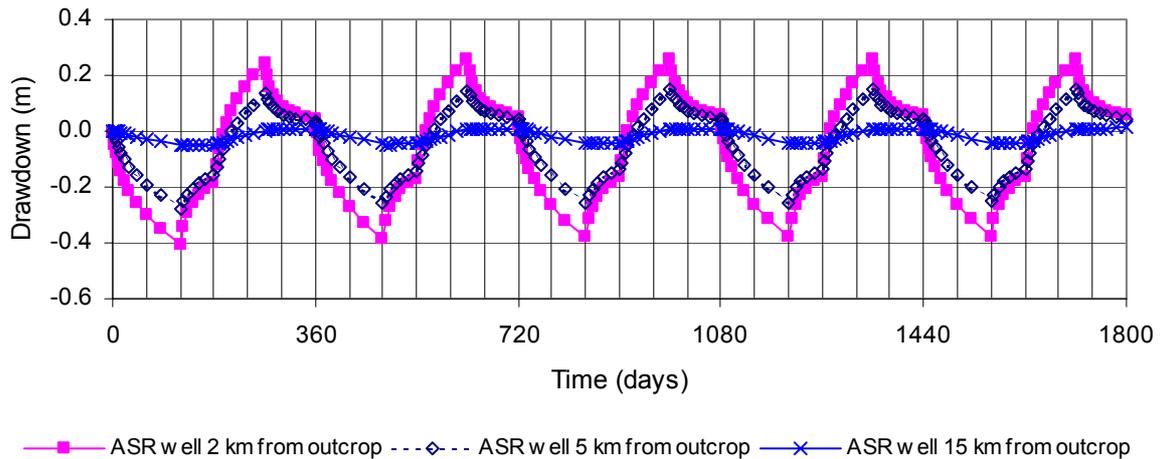


Figure 3.3 The impact of changing the position of the ASR well relative to the outcrop of the aquifer over five annual ASR cycles

It is also interesting to note that the maximum drawdown at the outcrop area occurs sometime after the maximum effect at the ASR well, i.e. the maximum impact of the ASR scheme is delayed beyond the end of pumping. The magnitude of this delay is greater when the aquifer transmissivity is smaller and when the storage coefficient is larger. The delay may be more than several months for an ASR well 10 km from the outcrop area in an aquifer with a transmissivity of $100 \text{ m}^2/\text{d}$ and a storage coefficient of 10^{-3} (Figure 3.4). This factor may be very important when considering whether a change in head in the aquifer is significant in relation to annual variations. A change in water level that is significant in the summer months, when stream flows are low, may be insignificant in the winter.

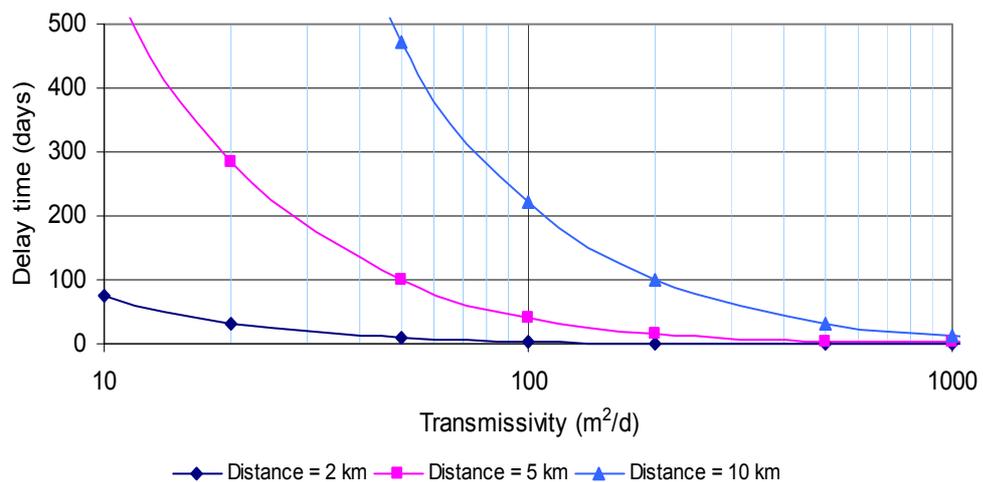


Figure 3.4 Time after the end of a 60-day pumping period when the maximum effect occurs ($S=10^{-3}$).

The model has also been used to demonstrate the difference between an ASR scheme and a conventional abstraction-only scheme (Figure 3.5). The model was set-up with some effective recharge across the outcrop area, and so may not be representative of

regions where ASR might be used (where it is assumed that all the available water resources are already ‘allocated’ to abstractions or environmental requirements). However, even in this ‘best-case’ (for the abstraction-only scheme), it is clear that abstraction-only results in much greater drawdown at the outcrop compared to the ASR scheme. The maximum drawdown simulated is 0.3 m at the end of the fifth year of abstraction and this drawdown is slowly increasing year-on-year. However, as the net abstraction with the ASR scheme is zero, the maximum drawdown does not increase year-on-year and is only half that of the abstraction-only scheme.

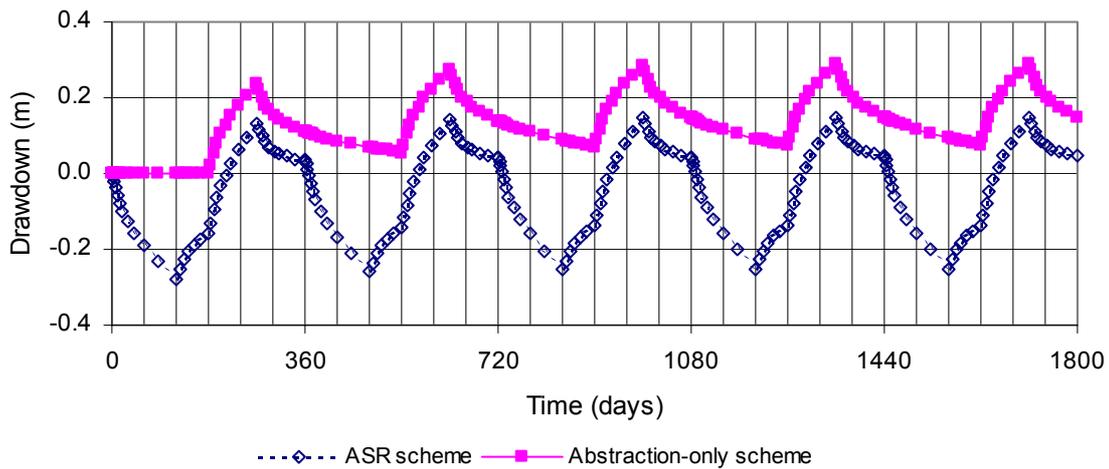


Figure 3.5 Comparison of ASR scheme with an abstraction only scheme.

3.4 Conclusions

Modelling tools have been developed which can give a useful insight into the possible effect of an ASR scheme in a variety of different hydrogeological settings. These first-pass tools can be used to indicate which aquifer parameters have the greatest effect on the impact and thus which parameters it is important to measure. As the tools are easy to use they can be applied to a wide variety of possible ASR sites very simply and thus reduce the cost of initial investigations.

Using the models has shown that under some circumstances there is a significant time delay between the maximum drawdown at the abstraction well and the maximum effect at outcrop. This means that in some cases ASR schemes could well have an advantageous environmental impact. This would occur when the drawdown at outcrop was delayed by several months so that maximum drawdown occurred at high river flow, possibly inducing recharge, and maximum water level rise occurs during low flow periods, contributing to river flow.

4 Significance of aquifer parameters in major aquifers

4.1 Introduction

The main mechanisms for the ‘loss’ of injected fresh water are dispersion (hydrodynamic mixing and molecular diffusion), density stratification (gravitational segregation) and lateral movement down hydraulic gradient. Mixing occurs at the interface of injected and native waters, and a dispersion or buffer zone develops. The degree of mixing is partly dependent on the type of flow in the aquifer. Harpaz (1971) showed that sandstones are likely to have higher recovery efficiencies than limestones due to the relatively slow groundwater movement in sandstones. The recharging water follows a radial expansion pattern and tends to retain the shape of a coherent water body, except in highly stratified formations.

The amount of mixing between native and injected water in an ASR-scheme, and the quality changes that occur as a consequence of this mixing, will largely depend on the physical properties of the aquifer in which the ASR-scheme is carried out. Building up a bubble of injected water in the aquifer in which little or no mixing with the native water occurs is one of the key aims of a successful ASR-scheme. If extensive mixing occurs, the recovered water will be influenced by the native water quality, which can be undesirable.

In order to assess the suitability of the main aquifers in Britain for ASR in terms of mixing, a sensitivity analysis has been carried out. The aquifers considered are the Chalk, Lincolnshire Limestone, Lower Greensand and Sherwood Sandstone, these being the aquifers most likely to be used for ASR (Figure. 2.1).

For each aquifer, a range in values of the major parameters (e.g. porosity, permeability, thickness) was defined based on the known properties of the aquifer. The impact of this variation on the amount of mixing between native and injected water was then assessed, using a 3D-dual porosity model, SWIFT (see Box). The impacts of operational and test cycles on the recovered water quality were also modelled. No chemical reactions between the injected water and the native groundwater and the aquifer material were included. The changes therefore solely reflect mixing processes and are what would be expected of a non-reactive solute such as chloride. The results (represented by the concentrations measured in the well) show the possible ranges in recovered water qualities and the differences between the four main aquifer types.

The SWIFT Model

SWIFT is a fully transient three-dimensional model, which simulates the flow and transport of fluids (brines), heat (energy) and radionuclides in fractured geologic media. It is particularly appropriate for modelling ASR in fractured media because of its ability to model dual porosity aquifers. Such aquifers are widespread in the UK (Chalk and Lincolnshire Limestone) and where present are likely to be important in controlling the recovered water quality in an ASR-scheme. When a dual porosity medium is modelled in SWIFT, the modelled system is described in two parts: the fractures and the porous blocks. Both parts have their own properties and must be fully defined.

4.2 Impacts of the main British aquifers on mixing during ASR

The Chalk and the Lincolnshire Limestone are both best described as dual porosity aquifers. The main difference between them is the matrix porosity (2 to 40 times greater in the Chalk). In dual porosity aquifers, the bulk of the flow takes place in the fractures while storage is provided by the matrix. The important parameters in this type of aquifer, as far as solute transport is concerned, are the fracture and matrix porosity, the spacing between fractures and the diffusion coefficient. This is a measure of the rate at which solutes can penetrate the matrix blocks as a result of concentration gradients. Of these parameters, only matrix porosity is readily measured, although estimates of fracture porosity and fracture spacing can be made from pumping tests and from geophysical logging. The diffusion coefficient of some solutes into clean chalk samples have been measured and so estimates (order of magnitude) for this parameter are available. However the relevance of laboratory measurements on clean uniform surfaces to processes occurring at natural fracture surfaces is still poorly understood.

In contrast to the Chalk and the Lincolnshire Limestone, the Lower Greensand behaves as a single porosity medium. This means that the same void spaces (as reflected by the porosity) control both the permeability and the storage. In this case, the parameters that are of most relevance to transport of solutes, are the porosity and dispersivity. Dispersivity is a measure of how much a plume of solute will spread as it moves and is controlled by the heterogeneity of the aquifer. Dispersivity can be measured in the laboratory under controlled conditions but the values obtained are often found to be much smaller than those estimated from field data. This is because of the large-scale heterogeneities present in the aquifer but not represented in the small blocks of aquifer material that can be tested in the laboratory. The dispersivity is therefore usually derived during model calibration with field data.

The behaviour of the Sherwood Sandstone lies somewhere between the dual-porosity and single-porosity ideals and is probably best described as a dual-permeability medium. This is because the matrix is permeable enough to transmit water at the low rates which are relevant in regional flow systems, but probably not in the near-well situation where fracture flow is likely to dominate. The SWIFT model cannot simulate this type of aquifer, and in the work reported here, the Sherwood Sandstone has been modelled as a dual-porosity medium.

4.3 The dual-porosity issue

4.3.1 Conceptual model

In some cases, the effects of molecular diffusion may be highly significant and this is considered to be the dominant mixing process in the Chalk aquifer in the UK (Jones et al., 1998). The Chalk is unusual in that it has a high porosity (up to 40%) and may contain saline or brackish water in the pore spaces. Due to the small pore throat diameters, groundwater flow is dominantly through fractures (1-2% porosity) and not through the pores. The ratio of matrix water to fracture water is high and so diffusion of solutes from the matrix water can therefore have a strong influence on the chemistry of the mobile fracture water.

If a Chalk aquifer contains poor quality water then injection of potable water will mix with and displace water from the fractures. However, diffusive mixing will be the

dominant process between the fracture and the matrix waters. The rate at which this occurs depends on the surface area of contact between the two waters, their relative volumes and the concentrations and ionic species under consideration. The scale and rate of this process is therefore a key to the success of ASR schemes in aquifers that contain non-potable native groundwater.

It is therefore worthwhile examining the processes that occur during an injection-recovery cycle in some detail. This has been modelled by considering a cylinder of fractured aquifer centered around a well. The numerical grid used for these calculations extended to a radius of 2000 m about the well and has a variable mesh spacing beginning at 0.25 m close to the borehole and increasing logarithmically with distance away from the well. Initially, the aquifer was assumed to be full of water, both in the fractures and the matrix, with a nominal solute concentration ('salinity') of 1.0 representing native groundwater. The injected water was assumed to have a concentration of 0.0. Therefore the relative concentration in the recovered water lies between 0 and 1 and is a direct measure of the fraction of the native water found in the recovered water – this is one measure of the efficiency (or inefficiency) of an ASR scheme. The response of the model to a typical annual cycle is shown in Figure 4.1. The relative concentration plotted is that found at the well.

During the injection phase (Figure 4.1A), the concentration at the well is 0.0, that is, the injected water concentration. If the well is left standing after injection stops (Figure 4.1B), the concentration in the well gradually increases due to diffusive exchange between the injection water in the fractures and the native groundwater in the matrix. When the recovery (pumping) phase starts (Figure 4.1C), the concentration in the well increases further as higher salinity water from greater distances in the aquifer is drawn into the well along fractures. This is the key phase of the cycle as predictions can be compared directly with measured concentrations in the recovered water. When the recovery stops, the now relatively fresh water in the matrix close to the well causes a dilution of the fracture water by diffusive mixing. As soon as injection starts again, the salinity in the well drops to 0.0, i.e. back to the concentration in the injected water. If the volume of injected water is greater than that taken out during recovery, such a cycle builds up a 'bubble' or buffer zone of injected water in the aquifer, albeit with considerable mixing with native groundwater at the interface.

It is the moderately slow rate of exchange between matrix and fracture water that is the problem with dual porosity aquifers such as the Chalk. If this exchange were either very fast or very slow then the ideal for ASR – a distinct bubble of injected water close to the well – could be established. When significant exchange takes place within a few months or years, the timescale of a typical ASR scheme, this ideal is prevented.

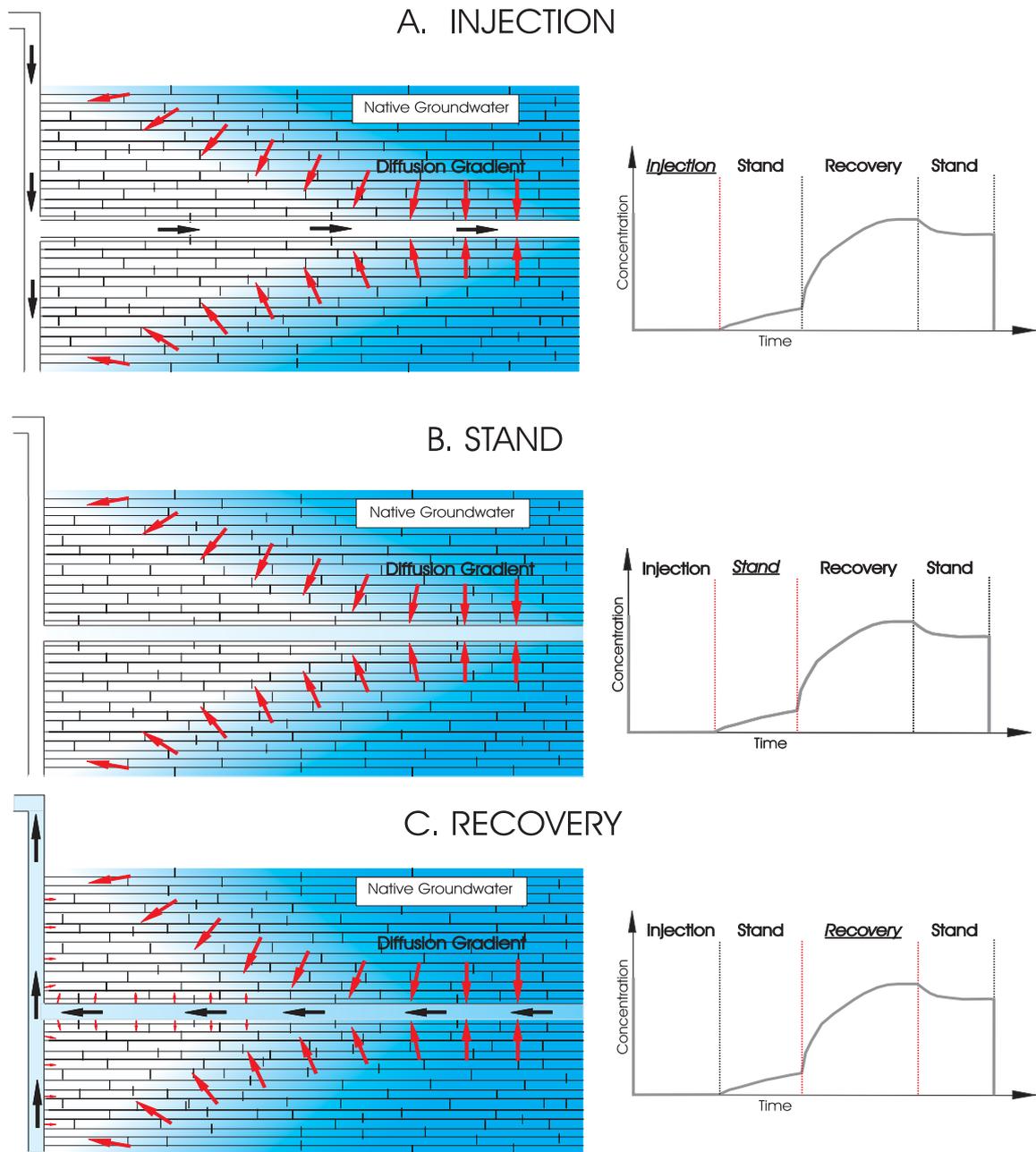


Figure 4.1 Modelled response of a dual-porosity aquifer to a typical injection-stand-recovery cycle. The relative concentration plotted is a measure of the fraction of the native water in the recovered water

4.3.2 Calibration of a dual porosity model against field data collected from a trial in the Chalk aquifer

The modelled results were calibrated against data collected during an extensive cycle-testing programme undertaken by Wessex Water Services at their site in the confined Chalk aquifer at Lytchett Minster in Dorset. Water from a supply main was injected at a constant rate of 2 to 3.2 Ml/d in cycles ranging in duration from a few days initially and increasing to over four months in Cycle 9. Apart from the first cycle, less water was

recovered than was injected in order to build up a buffer zone. The water injected over nine cycles totaled about 980 MI of which 360 MI was recovered. Using a fracture porosity of 0.01, the maximum radius of the residual ‘bubble’ in the fractures is estimated to be about 700 m.

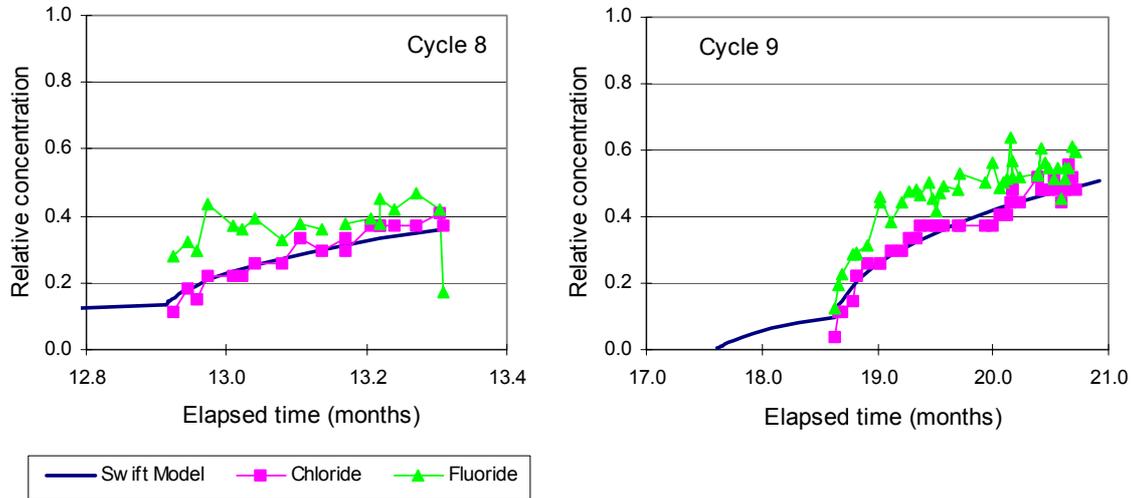


Figure 4.2. Results of dual porosity modelling from Cycles 8 and 9 of the testing programme in the Chalk aquifer at Lytchett Minster. Data for the observed variation in chloride and fluoride concentrations are also shown

The model was matched to the non-reactive solute, chloride. The match between the modelled and observed concentrations is good for Cycle 8 and 9 (Figure 4.2). A good match was also found for the preceding seven cycles. Also shown in Figure 4.2 are the relative concentrations for what proved to be a reactive solute, fluoride. The measured relative concentration of fluoride is greater than the modelled concentration indicating that some fluoride must be dissolving from the aquifer. The impacts of geochemical reactions on the recovered water quality are discussed further in Chapter 5.

4.4 Impact of mixing in British aquifers

In order to assess the impact of ASR-schemes on the recovered water quality in British aquifers, a sensitivity analysis was carried out using the SWIFT-model. The impact of operational and test cycles on the recovered water quality were modelled assuming a plausible range of aquifer parameters (Table 4.1). No chemical reactions between the injected water and the native groundwater and the aquifer material were included in the model. The changes therefore solely reflect mixing processes, both advective and diffusive.

For the sensitivity analysis, two ASR regimes were tested. One regime represents a typical series of test cycles (short cycles), while the second represents a typical series of operational cycles, i.e. long cycles. This was done to determine whether the timescale was important or not.

Table 4.1 Estimated values or ranges of values for some physical properties of the main aquifers in the UK

Aquifer	Permeability (m/d)		Porosity (%)		Storativity (m^{-1})	Thickness (m)
	Matrix	Fracture	Matrix	Fracture		
Lincolnshire Limestone (Jurassic)	10^{-3}	10 to 100	1-10	0.5 to 5	10^{-6}	10-40
Chalk (Cretaceous)	10^{-3}	1 to 100	20-40	0.5 to 2	10^{-6}	50-200
Sherwood Sandstone (Triassic)	1 to 50	10 to 500	10-25	1	10^{-5}	50-300
Lower Greensand (Cretaceous)	1 to 100		10-25		10^{-5}	10-40

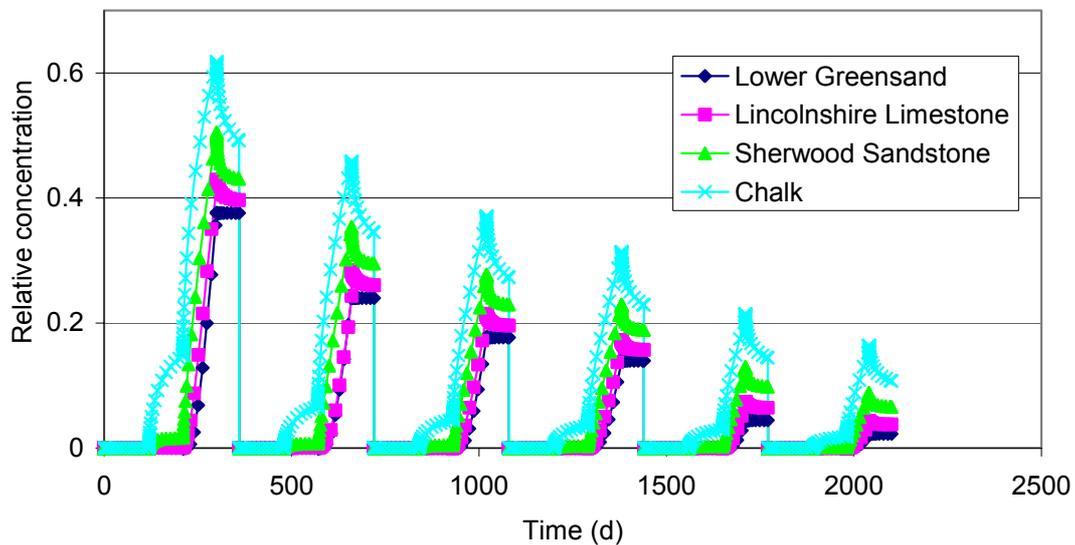


Figure 4.3 Predicted responses of the four major aquifers in the UK to ASR over six annual cycles. Each trace represents a series of injection-stand-recovery cycles of the type illustrated in Figure 4.1

For each aquifer a base case was modelled. The selected parameter values were chosen to be in the middle of the estimated parameter range (if a range was indicated). The base case was calculated for each aquifer over six annual cycles (Figure 4.3). The peak concentration declines with successive cycles because of the gradual increase in the size of the freshwater bubble close to the well. The relative concentration in a given cycle increases in the sequence: Lower Greensand < Lincolnshire Limestone < Sherwood Sandstone < Chalk, reflecting the increasing influence of fracture flow in drawing in native groundwater and reducing the compactness of the freshwater bubble around the well.

As an example of the sensitivity of the results to variations in aquifer properties, a detailed analysis of the Chalk aquifer was undertaken (Figure 4.4). These analyses were

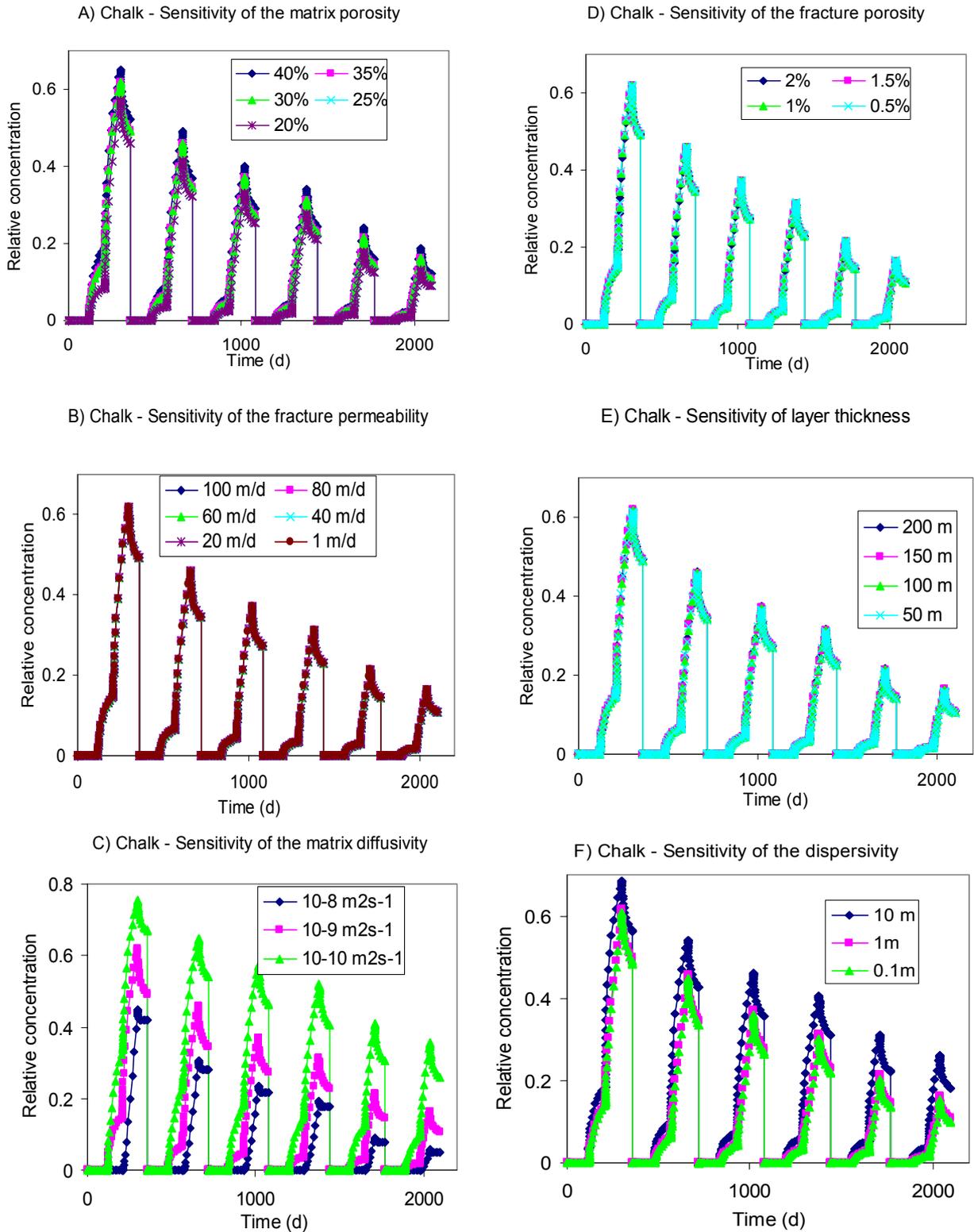


Figure 4.4 Sensitivity analysis of ASR-cycle testing in the Chalk to various aquifer properties: matrix porosity, fracture permeability, fracture porosity and layer thickness, matrix diffusivity and dispersivity

also carried out for the other aquifers except for the influence of matrix diffusivity and dispersivity, which were only evaluated in the case of the Chalk aquifer. These two parameters were assumed to be constant for the Sherwood Sandstone, the Lincolnshire

Limestone and the Lower Greensand because their dependency on the other parameters was unknown. It is, in general, difficult to measure the matrix diffusivity and dispersivity of aquifers.

Varying the parameter values over the plausible ranges expected for the dual-porosity Chalk aquifer, the Sherwood Sandstone and the Lincolnshire Limestone aquifers showed that the effect on the relative concentration of a non-reactive solute was small. Because of these low sensitivities, the conclusions drawn from the short-term test cycles can reasonably be extrapolated to longer-term operational cycles. Of the parameters examined, the matrix porosity had the greatest influence on the modelled relative concentrations. Variations in the other parameters (fracture permeability, fracture porosity and thickness of the aquifer) were negligible and variations in the matrix diffusivity and aquifer dispersivity were small.

However, the sensitivity to the parameters that are difficult to measure (matrix diffusivity, dispersivity) might be greater than shown in Figure 4.4. For example, it cannot necessarily be concluded from these calculations that these parameters never influence the modelled concentrations since they also depend on the values of other parameters such as fracture permeability, rock porosity and matrix porosity. A matrix diffusivity of 10^{-9} m²/s and a fracture dispersivity of 10 m were found to be appropriate for all of the aquifers modelled.

The lack of sensitivity of the mixing to the aquifer thickness and permeability is not unexpected since the same volume of water had been injected in each of the model runs. This, of course, leads to very different imposed heads on the aquifer, some of which would be unlikely in reality. However the volume of the aquifer (and thus the volume of the native groundwater), which comes into contact with the injected water is the same in all cases in which only the permeability or the thickness were changed. For dual-porosity aquifers, the matrix permeability has little effect on predicted concentrations because it has been assumed that there is no flow in the matrix except perpendicular to the fractures. This is a reasonable assumption for the Chalk and the Lincolnshire Limestone, but for the Sherwood Sandstone this assumption might not hold and therefore the results for the Sherwood Sandstone aquifer should be viewed with caution.

Based on the parameter estimates given in Table 4.1 and the calculations described above, the Chalk seems to be the least suitable aquifer for ASR in terms of recovered water quality. This is due to its high matrix porosity in comparison to the other aquifers, leading to a high degree of mixing between injected and native water. Due to the low matrix porosity of the Lincolnshire Limestone, matrix effects are less important. Because of the combination of good matrix porosity and fracture permeability in the Sherwood Sandstone (a dual-permeability aquifer), the response largely depends on the presence or absence of fractures in the target aquifer. If fractures are present, the response tends towards that of a Chalk aquifer, whereas if fractures are absent, then the response tends towards that of the Lower Greensand aquifer.

The absence of dual-porosity effects in the Lower Greensand aquifer means that the quality of the native groundwater has a minimal influence on the quality of the recovered water as it is simply displaced by the injected water with little mixing.

If the quality of the native groundwater in an aquifer is already good, then the extent of mixing is not important and other quality issues and the efficiency of the scheme in terms of recovery of injected water will determine the likely overall benefits.

The main conclusion of this modelling is that where an ASR scheme is to be used in an aquifer with poor quality native groundwater then, all other things being equal, dual-porosity aquifers will need more conditioning (injection of non-recoverable water to build up a buffer zone of mixed waters) than single porosity aquifers. However the suitability of any particular aquifer depends on its physical (dispersive and diffusive parameters) and chemical characteristics (native groundwater and injected water quality) Local variations can be great and will need to be determined on a site-by-site basis from field trials.

5 Water – groundwater – rock interactions

5.1 Introduction

The geochemical processes involved in an ASR scheme involve a range of chemical reactions between the injection waters, the native groundwater and the aquifer (Figure 5.1). Therefore the chemical changes that occur are determined by three end-members:

- chemical composition of the injected water;
- chemical composition of the native groundwater;
- chemical (mineralogical) characteristics of the aquifer.

There are large potential variations in the chemical compositions of both the injection and the native waters, as well as the rock matrix. The source of the injected water can include drinking water, untreated or treated surface water, untreated or treated groundwater or reclaimed water. Native groundwater quality can vary from fresh through brackish water to saline water. The major element chemistry is often dominated by reactions involving carbonate or silicate mineral phases. The redox status may be either oxidising or reducing. In addition, large regional differences exist in native groundwater quality related to land use, residence time and differences in aquifer mineralogy.

The simplest estimate of recovered water quality is that it will lie somewhere between that of the injection water and that of the native groundwater. If there are no chemical reactions, then the recovered water quality will merely reflect the mixing ratio. This would be true of an ion such as chloride and is what has been modelled in Chapter 4. Most solutes do react to some extent with the aquifer – the solid/solution ratio is extremely high in aquifers so that even a small amount of dissolution, for example, can have a major effect on groundwater quality. In most cases, native groundwater will have equilibrated with the host aquifer and so this should give an indication of the presence of relatively soluble minerals such as halite (NaCl), gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$), calcite (CaCO_3), dolomite ($\text{CaMg}(\text{CO}_3)_2$) and fluorite (CaF_2). Geochemical modelling can help to establish which of these minerals is present in the aquifer.

Occasionally, the injection of water may cause ‘new’ reactions to take place, which could lead to the quality of the recovered water quality being outside the range of quality spanned by the injection and native water. For example, if oxidising water is put into a reducing aquifer containing pyrite (FeS_2) then this could lead to pyrite oxidation with the release of sulphate, and probably more significantly, dissolved iron. Other minor elements could also be released. The extent of this oxidation is controlled by both the amount of dissolved oxygen and nitrate in the injection water. Injecting reducing water into an oxidising aquifer could also lead to the release of iron and manganese by dissolving iron and manganese oxides. There can also be some surprising effects when mixing two waters – for example, the mixing of two calcite-saturated waters can lead to a calcite-unsaturated water and thereby lead to further calcite dissolution from the

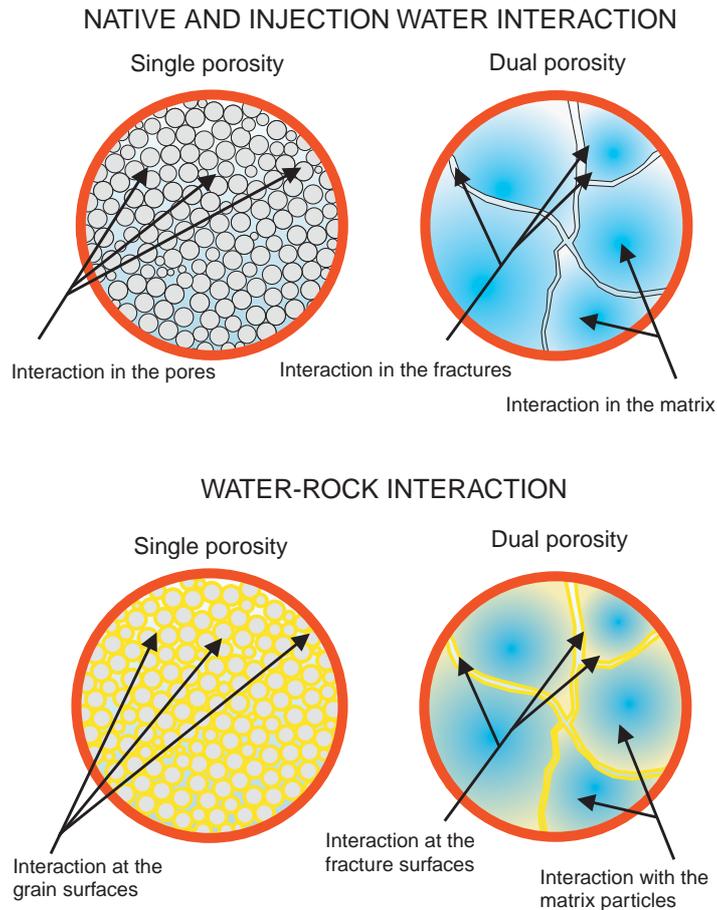


Figure 5.1 Situations in which chemical interactions between groundwater and aquifer can take place during ASR

Aquifer material. Similar effects can occur with other minerals. The danger with these additional reactions is that they may put the recovered waters outside the acceptable range for drinking water or may lead to the development of precipitates that may clog the well. Geochemical modelling can provide estimates of these effects but the rate of most of these dissolution/precipitation reactions in British aquifers is poorly understood and so these estimates are likely to be only approximate.

The PHREEQC Model (originally developed by the USGS)

PHREEQC is based on an ion-association aqueous model and has capabilities for (1) speciation and saturation-index calculations; (2) reaction-path and 1D-transport calculations involving reversible reactions, which include aqueous, mineral, gas, solid-solution, surface-complexation, and ion-exchange equilibria, and irreversible reactions, which include specified mole transfers of reactants, kinetically controlled reactions, mixing of solutions and temperature changes and (3) inverse modelling, which finds sets of mineral and gas mole transfers that account for differences in composition between given waters, i.e. how the water quality may have evolved from one place to another (Parkhurst and Appelo, 1999).

Modelling of the likely major impacts of ASR on the recovered water quality in the four main aquifers in Britain was carried out using various plausible combinations of

injection water quality, aquifer type and native water quality. The aquifers investigated were the Chalk, the Lower Greensand, the Sherwood Sandstone and the Jurassic Limestone. The PHREEQC geochemical speciation and transport model (see Box) was used for these calculations. The objective of this modelling was to provide guidance on possible water quality issues that should be considered when investigating the development of new ASR-schemes. More detailed modelling based on site-specific data is necessary to assess any probable water quality problems in specific situations since conditions are likely to vary greatly from site to site.

Table 5.1 The range of injection and native water combinations tested during the geochemical modelling for various British aquifers

Chalk	Lincolnshire Limestone
<ul style="list-style-type: none"> • Injection of treated Upper Chalk water into the confined Upper Chalk at Lytchett Minster • Injection of treated Upper Chalk water into the confined Upper Chalk at Holton Heath • Injection of treated Blashford Lakes surface water into the confined Upper Chalk at Lytchett Minster • Injection of treated Blashford Lakes surface water into the confined Upper Chalk at Holton Heath 	<ul style="list-style-type: none"> • Injection of River Trent water into the Lincolnshire Limestone at Spalding Bulb Company • Injection of River Trent water into the Lincolnshire Limestone at Lenton PS
Sherwood Sandstone	Lower Greensand
<ul style="list-style-type: none"> • Injection of River Derwent water into the Sherwood Sandstone aquifer at Budby. • Injection of River Derwent water into the Sherwood Sandstone aquifer at Gainsborough B.P. • Injection of Magnesian Limestone water into the Sherwood Sandstone aquifer at Gainsborough B.P. • Injection of Magnesian Limestone water into the Sherwood Sandstone aquifer at Budby. 	<ul style="list-style-type: none"> • Injection of treated Upper Chalk water into the Lower Greensand at Stockbury • Injection of River Medway water into the Lower Greensand at Stockbury

The main geochemical processes relevant to ASR include mixing, adsorption-desorption, ion-exchange, oxidation-reduction and dissolution-precipitation reactions

(Jones et al., 1999). In addition, microbially-mediated reactions (not modelled here) may occur, particularly related to the degradation of organic chemicals and the reduction of nitrate. An overview of the different types of injection and native waters used for the exploratory modelling is given in Table 5.1. The end members were selected to be representative of scenarios that have been, or are likely to be utilised. Details of the water chemistries used are given in Gaus (2001).

Major chemical changes to the quality of the injected water during recovery are to be expected when one or more of the following conditions are met:

- there is a large difference in chemical quality between the injected and the native water; this can cause large differences in pH or redox status.
- the native water or the sediment do not possess a sufficient pH buffering capacity (e.g. in the case of acidic waters where no calcite or dolomite is present for dissolution)
- there is a large difference in elemental concentrations between the injection and the native water (e.g. fluoride) and significant mixing occurs (e.g. in dual porosity aquifers)
- a change in chemical condition of the water having contact with the sediment is able to trigger major (e.g. dissolution of gypsum) or minor (e.g. dissolution of heavy metals) reactions.

5.2 Geochemical assessment for the different aquifers

Depending on the flow regime in the aquifer, different types of chemical interactions are likely to dominate and determine the quality of the recovered water.

In cases where the aquifer has a dual porosity character (as is the case for the Chalk and the Lincolnshire Limestone), the following factors might influence the quality of the recovered water:

- during ASR, the native water in the fractures will be replaced quickly with injected water. However, diffusional interchange between the matrix and fractures is relatively slow with the matrix retaining the native water signature for a long time. This diffusional mixing might lead to additional geochemical reactions.
- the water quality in the matrix pores may differ significantly from that of the water in the fractures. Such differences may have significant effects.

When the aquifer is a single porosity aquifer (e.g. the Lower Greensand) in which intergranular flow predominates, then it is more likely that a ‘bubble’ of injected water will form around the ASR-injection well. This bubble, of almost 100 % injected water, will be surrounded by a mixing zone of native and injected water. The creation of such a bubble (reservoir) of injected water in the aquifer has two major consequences:

- simple mixing between the injected water and the native water will be restricted to the mixing zone, as will the impact of geochemical reactions between the waters, leaving the injected water in the bubble relatively unaffected.
- largely unmixed injected water in the bubble will be able to trigger more intense geochemical reactions with the sediment when there is a large difference in quality between injected and native waters.

Although the behaviour of the Sherwood Sandstone lies somewhere between dual-porosity (the bulk of the flow takes place in the fractures and storage is provided by the matrix) and single-porosity (flow takes place in the matrix), it is probably best described as a dual-permeability medium. The matrix is permeable enough to transmit water at the low rates relevant to regional flow systems, but is probably not permeable enough to affect water flow in some near-well situations where rapid fracture flow is likely to be dominant. Because the extent of mixing will be even greater in the dual porosity case, this is seen as the worst case and was therefore assumed in the modelling.

The main chemical reactions for each aquifer have been classified into issues related to the native water quality, issues related to the injected water quality and issues related to the chemical reactions within the aquifer. Interactions with the sediment are based on the anticipated major components from geochemical descriptions for the different aquifers. The results of the modelling study are discussed in detail, and tabulated in Gaus (2001) and are summarised here in Table 5.2

These conclusions are based on the modelled cases only, and highlight the main chemical reactions likely to occur when implementing an ASR-scheme. Other chemical interactions may determine the quality of the recovered water when other injection waters are used, when the native water has a different quality, or when the geochemistry of the aquifers is different from that assumed here. Also reactions involving some minor elements such as cadmium, nickel and arsenic have not been considered.

In general the changes in the quality of injected water which may occur following recovery will depend strongly on the injected water quality and the native groundwater quality as well as the physical (transport) and chemical properties of the aquifer concerned. Recovered water quality will usually lie somewhere between the two end-members and for reactive solutes, may be closer to the native groundwater quality. Rarely, there may be interactions between injected groundwater and the aquifer that give rise to poorer recovered water quality than found in either of the two end-members. The most likely example of this is where aerated, high-nitrate groundwater is injected into a reducing aquifer containing pyrite (FeS_2). This can lead to the release of significant quantities of iron, sulphate and other minor elements.

Geochemical modelling can be used at different levels when planning and developing an ASR-scheme. During the initial desk study, it can be used to provide an initial assessment of the chemical viability of the scheme. During the trial and implementation stages of an ASR scheme as additional water quality data become available, geochemical modelling can be used to assess the impact of specific geochemical reactions. This is illustrated for the case of fluoride, which was monitored during an ASR trial in the Chalk in southern England (see Section 5.3).

Table 5.2. Possible impacts of ASR on recovered water quality in the UK

Inorganic chemical parameter	Possible groundwater reactions which may change the injected water quality
pH	In aquifers containing free calcium carbonate (all Chalk limestone aquifers and usually the Sherwood Sandstone and Lower Greensand), the pH of the recovered water is likely to be close to that of the native groundwater.
Chloride (Cl)	Expect injected water to mix with native groundwater and the final Cl concentration will reflect the extent of this mixing and the two concentrations, i.e. conservative mixing. This should be relatively rapid in well-mixed aquifers but will be slow (months to years) in dual porosity aquifers such as the Chalk.
Sodium and potassium (Na and K)	As for Cl but some cation exchange will occur on clays, which will tend to buffer concentrations to be closer to those in the aquifer. The effect will be greatest where the clay content is greatest, e.g. Lower Chalk and parts of the Lower Greensand.
Calcium and magnesium (Ca and Mg)	In the presence of free calcium carbonates, these concentrations are likely to be controlled by carbonate equilibria but cation exchange may also be important where the clay content is high. Generally concentrations will tend to approach those of the native groundwater.
Iron, manganese, ammonium and nitrite (Fe, Mn, NH ₄ and NO ₂)	Most injected waters will be aerobic and therefore low in these constituents but may be injected into a reducing aquifer where they are elevated (in the UK, most confined aquifers are reducing). This will increase their concentration in the recovered water but it should not be greater than in the native groundwater and will tend to be less if the injected water contains dissolved oxygen.
Sulphate (SO ₄)	Similar to Fe and Mn but injection of aerated or high nitrate water into a reducing aquifer containing pyrite could lead to release of sulphate and acidity (which could lead to enhanced release of CO ₂ when it reacts with carbonates).
Fluoride (F)	Similar to Cl but there can be an additional slow release from fluoride-containing minerals, especially where Ca concentrations are low. High native concentrations of fluoride are occasionally found in carbonate aquifers in the UK, but not in the sandstone aquifers.
Trace constituents	Most metals (Zn, Cd, Pb, Al etc) which are present as cations or carbonate complexes tend to be present at low concentrations at the near-neutral pH's of most UK groundwaters. Exceptions can arise in slightly acidic (some shallow Sherwood Sandstone and Lower Greensand groundwaters) and strongly reducing waters but such instances are relatively rare. Arsenic can be enhanced near mineralised areas and close to the redox boundary in semi-confined aquifers, both in the Chalk and Sherwood Sandstone. Boron can be high particularly in areas associated with residual salinity.
Nitrate (NO ₃)	Likely to be reduced if high nitrate, aerobic water is injected into a reducing aquifer due to denitrification.

Although beyond the scope of the modelling undertaken, the potential improvements in water quality, resulting from ASR, are considerable. Reduction of nitrate, when injected into a reducing aquifer may be microbially mediated, as may many other reactions involving organic chemicals. Attenuation of these chemicals during artificial recharge, is poorly understood and is the topic of much current research. Examples of attenuation of disinfection-by-products (DBP) (Pyne et al., 1996) and the capacity of aquifers for adsorption and biodegradation of organic carbon (Dillon and Pavelic, 1996) have been documented. However the sustainable treatment processes occurring in aquifers need further investigation, particularly in relation to pathogens, natural and synthetic organic chemicals, including endocrine disruptors.

5.3 Case Study: fluoride modelling of the ASR trial in Lytchett Minster

The SWIFT model, applied to the ASR trial in the Chalk aquifer at Lytchett Minster, Dorset, (see Section 4.3.2) demonstrated that the increase in fluoride concentration in the recovered water could not be attributed solely to dual-porosity mixing. An additional chemical reaction must have taken place to give the observed increase of about 10 % above that predicted for mixing alone. Geochemical modelling was used to help to identify the geochemical processes leading to this additional increase in fluoride in the recovered water. A full description of the fluoride modelling at Lytchett Minster can be found in Gaus, et al. (2001).

An example of the excess fluoride in the recovered water is shown in Figure 5.2 (right) for Cycle 9. Data from other cycles showed the same pattern. Chloride (which shows conservative behaviour) was used to calculate the amount of mixing between injection and native water and is shown on the x-axis. The ratio between mixing based on fluoride to that based on chloride is plotted on the y-axis. If fluoride also behaved conservatively, the fitted line would be horizontal; intersecting the y-axis at 1.0. The fact that this ratio is much greater than 1 at the beginning of each recovery cycle, and reduces during the recovery phase, indicates that processes other than simple mixing control the fluoride concentration. Where the main component of the recovered water is injection water, the fluoride concentration is much greater than would be expected from simple mixing alone. With an increasing fraction of native water in the mixture, excess fluoride due to geochemical reactions will influence the fluoride mixing ratio to a lesser degree because of the high fluoride concentration in the native groundwater.

Three mechanisms that could cause the observed increase in fluoride concentrations were postulated and the results expected from each mechanism was modelled and then compared with the field observations. The three mechanisms are:

1. the fluoride concentration in the matrix water is higher than in the fracture water: the recovered water will reflect the difference between the fracture water and the matrix water in the Chalk even with no geochemical reactions
2. the mineral fluorite (CaF_2) is available for dissolution in the matrix and in the fractures, but only a limited amount is present: the concentration of fluoride in the recovered water will be limited by the amount of fluorite in the sediment that is available for dissolution

- fluorite is available in sufficient quantities for the fluoride concentration to reach saturation but the dissolution kinetics are slow: the rate of fluorite dissolution therefore determines the fluoride concentration in the recovered water (a simple kinetic equation was used to describe the dissolution kinetics).

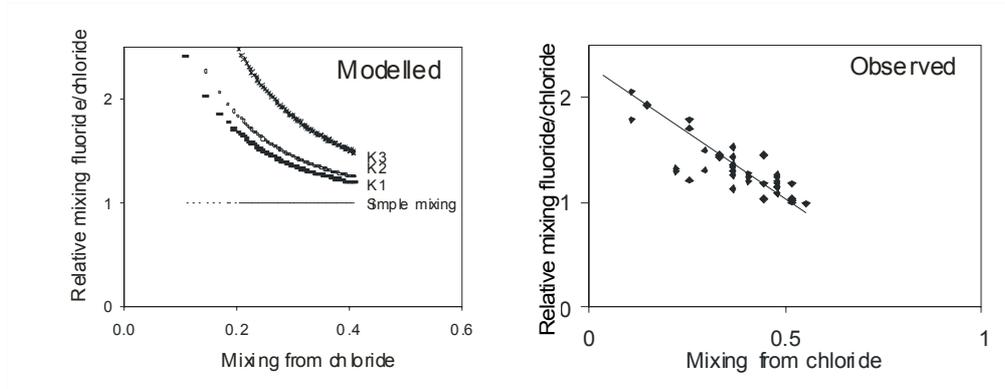


Figure 5.2 Comparison of modelled and observed relative mixing ratios for fluoride and chloride versus the amount of mixing between injection and native groundwater. Left: modelled ratios based on fluorite dissolution kinetics (mechanism 3) for three parameter sets (K1, K2 and K3); Right: optimal fit (line) through observed ratios (crosses) from Cycle 9.

The modelling results according to mechanism 3 are shown in Figure 5.2 (right). Modelling showed that the mechanisms 1 and 2 were not able to fit the observed data. Therefore, it is concluded that the increase in fluoride is probably controlled by the dissolution kinetics of a mineral, probably fluorite, which is present in excess in the Chalk sediments.

If this is the case, a significant decrease in fluoride concentration in subsequent cycles is unlikely to occur until the available fluorite is exhausted and mechanism 2 becomes the dominant process. However, the amount of fluoride dissolved and removed during ASR-cycles is likely to be small compared with that present within the solid phases in the aquifer. No data are available for the amount of fluorite present in the Chalk sediment at the site.

This case study demonstrated the broader view that must be taken when including an ASR scheme as part of a water resources strategy. The cost implications of the additional treatment, or blending, need to be compared to alternative sources of supply, remembering to include the environmental benefits (Eastwood and Stanfield, 2001). The Chalk and Lincolnshire Limestone are known to contain occasional high-fluoride groundwaters (Edmunds et al., 1989). The Sherwood Sandstone aquifer is normally low in fluoride (<0.1 mg/l). Fluorapatite, another F-containing mineral, is known to be present in some Chalk sediments and could provide an additional source of fluoride but it is likely to be released at a much slower rate than from fluorite.

6 Regulatory considerations

6.1 Regulations affecting Artificial Recharges to groundwater

Under the Groundwater Regulations, 1998 (GWR) an authorisation is required for discharges of listed substances to groundwater (the saturated zone). Regulation 6 of the GWR makes provision for the authorisation of artificial recharges, subject to there being no risk of pollution of groundwater. (Note: under the GWR it is not just actual pollution but the risk of such pollution that has to be considered). Direct discharges to groundwater (directly to the water table) may be consented under the Water Resources Act, 1991- these consents also act as authorisations under the GWR.

The GWR, which reflect the EU Groundwater Directive, require that there should be no entry of List I substances into groundwater and no pollution by List II substances. Listed substances (List I and List II) are given in the Schedule attached to the GWR. There is a formal process by which the Environment Agency, as the responsible body in England & Wales, determines whether a substance is listed. The Ecotox Centre should be consulted regarding the status of any particular substance.

Under Regulation 2, the Environment Agency can determine whether a discharge can be excluded from the GWR by virtue of the quantity and concentration of listed substances in the discharge. This is a case-specific assessment, which must be undertaken with no prior investigation of the hydrogeology of the site (following a European Court decision several years ago) it must be obvious from the nature of the source material that it could not cause entry into groundwater of List I substances or pollution by List II substances. The DETR guidance on the GWR makes it clear that although there are no specific concentration limits for pollutants, the drinking water and other similar standards may be used as a benchmark for such assessments.

If a discharge is found to be outside the remit of the Groundwater Regulations by virtue of the *de minimis* provision above, it is unlikely to require any other form of consent under water quality legislation. If the discharge contains only non-listed but nevertheless potentially polluting matter (e.g. bacteriological contamination), it may still possibly require a discharge consent under the Water Resources Act, 1991.

List I substances can be allowed to enter groundwater if the groundwater body can be declared as being "permanently unsuitable for other uses". There is a detailed procedure that describes this within the Agency's Groundwater Regulations Process Manual. This is publicly accessible but is not a published procedure. It can be obtained via Environment Agency offices.

6.2 Guidance on dealing with authorisations for artificial recharge and recovery (ARR) schemes

As part of a project assessing the potential for ASR in England and Wales, the regulatory issues associated with development of schemes were reviewed in the context of the existing guidance. The conclusions of this review are given in Appendix J of the project report (Jones et al. 1998), the main points of which are summarised below.

The purpose of Appendix J was to provide guidance on the licensing and consenting requirements associated with Artificial Recharge and Recovery (ARR) schemes. These can range between:

- Aquifer Storage and Recovery (ASR) schemes as developed in the USA, involving recharge of water into relatively deep, possibly poor quality and little-used aquifers to create a bubble of fresh groundwater for subsequent re-abstraction, and
- Artificial recharge schemes involving injection of water into good quality, generally well-utilised, aquifers to enable better or increased use of water resources (e.g. to store excess winter resources for re-abstraction to meet summer demand, or to support use of an aquifer with limited or heavily-utilised natural resources, etc).

The guidance covers a range of situations encompassed by such schemes, all of which will require authorisations from the Environment Agency.

Nothing in the guidelines modifies the abstraction licensing best practice set out in the Environment Agency's National Abstraction Licensing Manual, to which reference should be made for detailed explanation of Agency requirements, policy and procedures. Unless otherwise stated, references to sections in the legislation relate to the Water Resources Act 1991 ("WRA1991").

The phased and individual nature of ARR scheme development (which can extend over several years) makes it difficult to produce guidelines which will cover every eventuality but in general, the developer should ensure that :

- a) Abstraction of water from any aquifer or surface water source is authorised either by an existing licence or by a Section 32 consent, as appropriate, having been discussed with the Agency beforehand.
- b) The recharge of any water to the receiving aquifer is authorised in accordance with legislative and Agency requirements.
- c) The discharge of any water to controlled waters (either surface or groundwater) is authorised in accordance with legislative and Agency requirements.

Discharge consenting requirements for b) and c) above will depend upon whether or not the discharge is classed as trade effluent. Where in doubt, Agency staff should consult their Regional legal departments.

The Agency will seek to control all recharge and subsequent re-abstraction to ensure effective development of water resources whilst protecting the environment and other abstractors. In particular, it will expect schemes to be sustainable in terms of quantities recharged and re-abstracted and may wish to apply water level or water quality conditions to any authorisations issued.

The developer should be aware that, for most schemes, the Agency will issue separate authorisations relating to each main phase of development/implementation, since the monitoring, abstraction and discharge requirements for the initial testing, operational

testing and final scheme implementation phases will differ. It is therefore important that the developer continues to keep in close contact with the Agency as each phase progresses.

Whilst the Agency appreciates that developers will want to know at the outset about the likelihood of obtaining all necessary authorisations for all phases, it is unlikely to be able to give firm indications at that stage. Much will depend upon the findings of investigations as each phase of the scheme progresses. It is possible that later phases will not be able to go ahead if, for example:

- earlier phases show there are insufficient resources to provide the donor water, or
- there are incompatibilities in quality between the donor and receiving waters that cannot be resolved, or
- there are adverse effects on lawful abstractions, or
- the Agency believes that the recharge of water will be environmentally damaging.

6.3 Phased development/authorisations

The phased and individual nature of ARR scheme development (which can extend over several years) makes it difficult to produce guidelines that will cover every eventuality. In general, the developer should obtain the necessary authorisations throughout the six phases of development and ensure that the required Environmental Impact Appraisals (EAPs) are carried out.

Phased development of Artificial Recharge and Recovery Schemes	
Phase 1	preliminary discussions
Phase 2	development/construction of donor source(s)
Phase 3	construction/initial development of recharge borehole(s)
Phase 4	operational testing of recharge borehole(s) and re-abstraction source(s)
Phase 5	applications for authorisations for operational scheme
Phase 6	determination and issue of licences and consent

6.4 Environmental Impact Appraisals (EAPs)

The Agency will require that the effects of the proposed development on the environment are considered. The scope of the impact appraisal will be affected by the nature of the aquifer, the two extremes of which are shown below.

Aquifer Type	Nature of the Aquifer	Type of EAP
Shallow Unconfined (Traditional Artificial Recharge Schemes)	Good quality water present Probability that surface water may be affected Probable impact on existing abstraction rights Limited spread of level changes	Full EAP probably required
Deep Confined (Aquifer Storage and Recovery Schemes)	Poor quality water No direct connection with surface water Existing abstractors likely to be absent Level changes can be widespread	Relatively restricted EAP probably sufficient

In addition to the normal requirements necessary for abstraction licences, environmental impact appraisals for ARR schemes should include consideration of, amongst others, the following effects as appropriate. Some of these issues have been considered further in the ASR-UK project. Discussion of the possible environmental impacts is in Chapter 3 of this report. A series of models are described that can be used to assess and quantify the likely impacts. These models are of varying complexity so they can be applied at appropriate phases of development.

Feature	Possible effects	Possible problems
Existing abstraction boreholes	Increase or decrease in rest water level in the borehole	Existing installation becomes inappropriate
Existing abstraction boreholes where rest water level is close to surface	Increase in level to give artesian conditions	Flooded pump chambers Electrical problems Flooding of nearby farmland
Low lying areas	Increase in groundwater level	Water logged ground Flooding Increased flow of land drains (loss of recharged water)
Springs	Increase in spring flow; Changes in quality of groundwater discharged	Insufficient overflow capacity for new flows Flooding of chamber and pumping equipment Flooding of receiving water course

		Loss of recharge water Change in habitats
Streams	Increase in streamflow/flow in ditches; Changes in quality of groundwater discharged	Flooding Increase in depth of cattle watering areas Loss of recharge water Change in habitats
Water bodies	Change in level	Change in habitats Changes in overflow quantities Loss of recharge water
Change in groundwater quality	Deterioration in quality	Loss of potability or suitability of water for use Corrosion or encrustation on water system - pumps and household systems Quality effects on habitats
Ground instabilities	Wetting of Clays; Lubrication of strata boundaries	Subsidence as clays swell Instability of slopes

ANNEX J3 – The Potential for Aquifer Storage and Recovery in England and Wales. (from Jones et al. 1998)

DECISION MATRIX FOR ABSTRACTION CONSENTING & LICENSING

Details of source		Scheme development stage															
		Initial testing					Operational testing					Operational use					
		Approval needed		Licence variation		S32 consent	Approval needed		New licence		Licence variation	Approval needed	New licence	Licence variation			
Source of supply?	Does abstraction point already exist?	Is it already in use?	Is it already licensed?	Will proposal increase authorised or actual quantities abstracted or alter pattern of abstraction?	S32 consent	New licence	Licence variation	S32 consent	New licence	Licence variation	Approval needed	New licence	Licence variation	Approval needed	New licence	Licence variation	
					✓	✗	✗	See Note 1	✗	See Note 1	✗	See Note 1	✗	See Note 2	✗	See Note 2	✗
Groundwater	Yes	No	Yes	No	✗ (Note 3)	✗	✗	✗ (Note 4)	✗	✗	✗ (Note 4)	✗	✗	✗	✗	✗ (Note 5)	✗ (Note 7)
					✓	✗	✗	See Note 1	✗	See Note 1	✗	See Note 1	✗	See Note 1	✗	See Note 2	✗
Surface Water (Note 10)	No	N/A	N/A	Yes	✓	✗	✗	See Note 1	✗	✗	See Note 1	✗	✗	See Note 1	✗	✗ (Note 7)	✗ (Note 7)
					✓	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 9)	✗	✓ (Note 7)
Yes	Yes	No	Yes	Yes	N/A	✓ (Note 6)	✗	N/A	✗	✓ (Note 6)	✗ (Note 6)	✗	✓ (Note 6)	✗	✓ (Note 6)	✗ (Note 7)	✗ (Note 7)
					✓	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 9)	✗	✓ (Note 7)
No	N/A	N/A	N/A	Yes	N/A	✓ (Note 6)	✗	N/A	✗	✓ (Note 6)	✗ (Note 6)	✗	✓ (Note 6)	✗	✓ (Note 6)	✗ (Note 7)	✗ (Note 7)
					✓	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 8)	N/A	✗	✓ (Note 6)	✗ (Note 9)	✗	✓ (Note 7)

✓ = Normally required

✗ = Normally not required (but subject also to note where indicated)

- Notes:**
1. A Section 32 Consent should be issued at the operational testing stage if initial testing has failed to provide sufficient information to enable a formal licence application to be properly assessed. Otherwise, issue a time-limited licence or variation as appropriate (time limit may need to be shorter than policy would normally indicate).
 2. Either new licence or licence variation, depending on type of approval given for operational testing. If licence was issued at that stage, a variation may be needed.
 3. If characteristics of source or understanding of impact not properly known (e.g. current licence was determined as a Licence of Right or source was never properly tested), or it is necessary to control the testing to be carried out/specify special monitoring, it would be best to persuade potential operator to accept a Section 32 Consent, otherwise the Agency's requirements should be defined in a formal agreement (either under Section 20 (for water companies), or Section 158 of WRA, 1991).
 4. If characteristics of source or understanding of impact still not properly known, you will need to persuade potential operator to accept a further Section 32 Consent, or define the Agency's requirements in another formal agreement (either under Section 20 or Section 158 of WRA, 1991).
 5. The ARR operator will need to apply for a variation if it wishes to take advantage of reduced charges associated with any transfer of water or if earlier testing reveals current licence is inappropriate for the operational scheme.
 6. New or varied licence will usually need to be for a shorter time limit than normal policy dictates, since the conditions included are likely to be test-specific.
 7. Where a licence has been issued/varied for testing purposes, it may need varying for subsequent phases of development/operational use.
 8. If it is necessary to control the testing to be carried out /specify special monitoring in such a way that is not covered by any existing licence, the Agency's requirements should be defined in a formal agreement (either under Section 20 or Section 158 of the WRA, 1991).
 9. If it is necessary to further control testing/specify monitoring, the Agency's requirements will need to be defined in another formal agreement as for Note 8 above.
 10. "Surface water" source of supply refers to those sources for which abstraction must be licensed (i.e. those that are non-exempt). The Water Resources Act 1991 defines those that are exempt (see 3.2.2.1(a) in guidance notes).
 11. ✓= Normally required, ✖=Normally not required (subject also to note where indicated).

7 Summary and conclusions

1. Aquifer Storage Recovery (ASR) is viable water management technique that has the potential to resolve a wide variety of supply issues in an environmentally sustainable manner. The main advantages are:
 - Net abstraction is zero or negative so minimal impact on the surface water environment.
 - Small land area required to store relatively large quantities of water at the point where it is available for injection and needed for supply.
 - Staged development possible to spread costs and at the same time keep up with demand.
 - Improvements in water quality can occur during storage and, once understood, water quality problems can be addressed, on recovery, through treatment or dilution. Cost implications need to be assessed in relation to alternative schemes
2. Several aquifers in the UK are suitable for ASR schemes, their advantages and disadvantages are summarised below. These comments are generic and because the scale of schemes can be of the order of a few hundred metres, local geological conditions may be different. Minor aquifers, or blocks of major aquifers may prove to be ideal for ASR at particular localities even where they are not generally considered to have a high potential.
 - The **Sherwood Sandstone and some Permian sandstone** aquifers appears to have the greatest potential due to their wide geographical distribution, great thickness and high inter-granular porosity. Care is needed to select a section of the aquifer with high porosity and few fractures in order to constrain the bubble of injected water, displace indigenous groundwater and thus obtain a high recovery efficiency
 - **Lower Greensand** aquifer is limited in areal extent and thickness but, in suitable locations, this high matrix porosity aquifer will be a good ASR target. Injected bubbles will displace the native groundwater thus minimising the zone of mixing and hence geochemical problems
 - The **Chalk** aquifer has been demonstrated to be an effective hydraulic target for ASR but if the native groundwater quality is non-potable then diffusion from the matrix will be a long-term problem. This does not preclude the use of these parts of the aquifer where environmental impact considerations may still make the scheme attractive. However, the economic implications of additional treatment or dilution will need to be factored in. Where the Chalk is in hydraulic contact with overlying **Palaeogene** or **Quaternary** arenaceous deposits or the underlying **Upper Greensand**, then the storage in these aquifers can make an ASR scheme potentially more viable.
 - The variability of the **Jurassic limestones** will make their suitability for ASR equally variable. However, the aquifer may be able to provide a

reservoir for ASR where the local hydrogeological conditions are suitable. The high diffusivity of these aquifers means that the environmental impacts need to be evaluated at distances of several kilometres.

- Other **minor aquifers** should be considered on an individual basis. ASR schemes require areas of aquifers of the order of only tens of hectares so suitable, local geological settings can be adequate. Aquifers identified as having some potential include the **Fell Sandstone, Millstone Grit, Kellaways Rock** and some **Upper Lias Sands**. When considering these aquifers their potential can be reviewed in respect of the “aquifer continuum” described in this report. This continuum ranges from unconsolidated arenaceous deposits with high matrix porosity (Lower Greensand), through consolidated sandstone aquifer with high matrix porosity and some fracture flow (Sherwood Sandstone), fracture flow dominated, high diffusivity aquifers (Jurassic Limestone) to fracture flow dominated aquifers with dual porosity mixing being the dominant process (Chalk).
3. The environmental impacts of schemes will be crucial in determining their success or failure and should be assessed at the earliest stages of a scheme. Key issues are the impacts on water levels in overlying aquifers and in the unconfined part of the target aquifer, even at distances of several kilometres. Of particular concern are the impacts on stream-flow and wetlands and these need to be assessed in relation to the time delay of response of the aquifer at outcrop in relation to injection or recovery at the ASR well. Simple models have been developed to address some of these issues and to act as tools to assist decision-making on how to proceed and what additional information is required throughout the review, field trial and implementation phases of a scheme. Optimisation of the operational cycle can be demonstrated to minimise the impacts.
 4. The recovery efficiencies of ASR schemes will be determined by the amount of mixing between the native groundwater and the injected water and this is controlled by the physical properties of the aquifers. In addition to physical mixing, both advective and diffusive, geochemical reactions between different waters and the rock matrix will determine the quality of the recovered water. The physical aspects have been modelled using the SWIFT/486 modelling package to test the sensitivity of the systems to ranges of aquifer properties as well as testing and operational cycles. Within the limitations and assumptions inherent in the model, the main conclusion is that dual-porosity aquifers will need more conditioning (injection of non-recoverable water to build up a buffer zone of mixed waters) than single porosity aquifers before they are suitable for use for ASR. However the suitability of any aquifer depends on the site-specific dispersive and diffusive parameters, which can generally only be obtained from field trials.
 5. Geochemical models have been used to predict likely impacts of injecting a variety of waters into different aquifers containing different groundwaters. The selection of the different components was constrained by the most likely scenarios in the four major aquifers. The PHREEQC model was used and it was

found that, depending on the flow regime in different aquifers (matrix flow or fracture flow dominated), different chemical reactions dominate

- In dual porosity aquifers the injected water in the fissures will continue to react with the matrix water as well as the rock matrix whereas in a single porosity aquifer the pore water is displaced and reactions are restricted to the peripheral mixing zone, the bulk of the injected water being unaffected. However, in the latter case, the reactions, if any, between the injected water and the rock matrix will dominate.
- Geochemical reactions can be expected to be significant if there are:
 - i. Large differences in pH and redox potential between the native and injected water
 - ii. Little buffering capacity in the native water and/or the sediment
 - iii. Large differences in concentrations between native and injected waters, combined with intensive mixing
 - iv. The water mixture triggers dissolution reactions leading to the sediment releasing significant quantities of major or minor elements.

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Project outputs

1. **Web page** www.nwl.ac.uk/gwf/asr/asr_intro.htm

2. **Project reports and models on CD with this report**

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3. **Publications**

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