Migration of polluted mine water in a public supply aquifer

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Abstract

Following the cessation of dewatering in the southern part of the Durham Coalfield in 1974, groundwater levels rose in both the mine workings and the overlying Magnesian Limestone Aquifer (MLA). A pollution plume has since been migrating through the MLA, with sulphate concentrations greatly exceeding potable water standards. Prediction of future plume development requires specification of whether upflow of mine water to the MLA occurs from a small number of point sources or in a diffuse manner (i.e. distributed over most of the area in which the MLA is underlain by mineworkings). A programme of conceptual and numerical modelling has addressed this issue. Mine plans and geological information were used to characterize the flooded mine system and its interface with the MLA. Piezometric and meteorological data were used to constrain finite difference simulations of groundwater flow in the MLA. Flow velocity vectors derived from these flow simulations allowed modelling of plume migration by solution of the advection–dispersion equation. The results of these simulations illustrate that it is necessary to invoke both point and diffuse upflow to explain the observed patterns of pollutant migration to 2003. Predictive modelling of further plume migration indicates that sulphate concentrations are likely to rise to unacceptable levels in the most proximal public supply wells within the next two decades.

It is now widely appreciated that mine workings can remain sources of water pollution long after their abandonment and flooding, as a result of continued leaching of metals and sulphate into groundwaters (Younger *et al.* 2002). Discharge of polluted waters from abandoned mine workings is an extensively documented source of surface water pollution (e.g. Banks *et al.* 1997; Younger *et al.* 2002). However, there appear to be no papers in the open literature that document the migration of polluted mine waters into public supply aquifers. Although it has long been postulated that such migration is likely to occur (see, for instance, Adams & Younger 2001; Booth 2002), and there is a single recorded case in which an unused sandstone aquifer in southern Poland was heavily polluted after flooding of ironstone mines in subjacent strata (Razowska 2001), it would appear that there is rarely sufficient head in flooded mine workings underlying major aquifers for upflow to actually occur. The forthcoming abandonment of a number of European coalfields looks set to change this pattern. In Spain, France, Germany, the Czech Republic, Poland and the UK, for instance, mined strata underlying public supply aquifers extend up-dip beyond the aquifer outcrops into recharge zones at higher elevations, where it is likely that sufficiently high heads will develop to facilitate upflows into public supply aquifers after abandonment (Dumpleton *et al.* 2001; Wolkersdorfer & Bowell 2004, 2005*a*,*b*). This being the case, there are a significant number of important aquifers in Europe in which future maintenance of 'good status', as demanded by the European Union Water Framework Directive (WFD), is already in jeopardy, even though water quality in the aquifers may be good at present.

In this paper we provide the first documented example of this particular manifestation of mine water pollution. In this case, the mine water pollution has already led to the Environment Agency (England & Wales) classifying the groundwater body in question as being 'at risk' of failing to meet WFD requirements. As such, this case represents a timely precedent for many such similar problems elsewhere in future (Wolkersdorfer & Bowell 2004, 2005*a*,*b*). Using archival water quality data we have been able to track plume migration in the aquifer over a period of more than 20 years. We have then used mathematical models to discriminate between the following competing hypotheses for the mine water upflow mechanism.

(1) Upflow is restricted to 'point sources', such as unlined mine shafts and boreholes, and areas of intense fracturing caused by mining activities (which are known to have produced connections between the mine workings and the MLA during the mining period, most notably at Mainsforth Colliery; Clarke 1962).

(2) Upflow occurs predominantly in a 'diffuse' manner, i.e. by widely distributed upflow across the entire body of unworked Coal Measures strata lying between the extensive (15 km^2) array of old mineworkings and the overlying MLA.

(3) Significant upflow occurs in both 'point' and 'diffuse' manners.

Finally, the model is used to predict likely future development of the plume over coming years, revealing

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Fig. 1. Location of the study area.

the existence of a significant risk to public water supply sources in this area within the next two decades.

Study area

Location and topography

The study area lies in County Durham in the NE of England (Fig. 1). The area of interest extends eastwards from the vicinity of the town of Bishop Auckland to the North Sea coast, some 30 km away. In the north–south direction, the study area is some 17 km wide. The topography of the region is such that the highest elevations (measured in 'metres above Ordnance Datum' (mAOD), OD being the UK national mean sea-level datum) lie in the west, reaching about 200 mAOD in the hills surrounding Bishop Auckland, declining eastwards to a coastal plateau with an average elevation of about 100 mAOD, which eventually meets the sea in a cliffline, and southwards to around 10 mAOD along the River Tees at Stockton-on-Tees (Fig. 1).

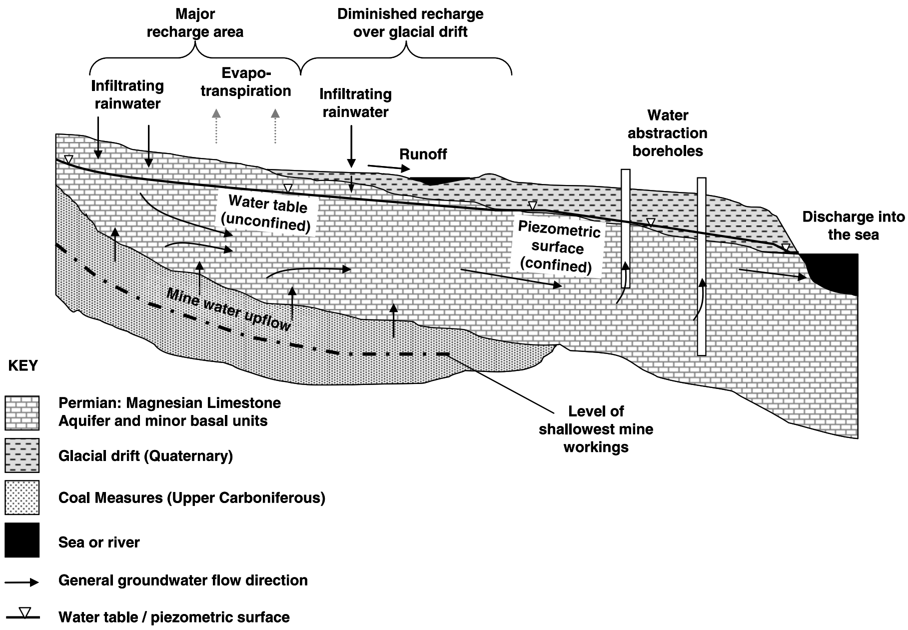

Fig. 2. Diagrammatic cross-section summarizing key features of the conceptual model for flow through the Magnesian Limestone Aquifer (MLA), including mine water upflow from underlying flooded coal workings.

Hydrogeology

The oldest strata exposed in this region are the Upper Carboniferous (Stephanian: Westphalian) Coal Measures, comprising cyclothemic sequences of sandstones, siltstones, mudstones, seat earths and coals (Johnson 1970), which in their natural state are of low permeability (Younger 1995). As many as 13 individual coal seams have been mined in various parts of the study area (Smith & Francis 1967), giving rise to the presence of voids in the strata, which have drastically increased the overall permeability of the sequence. As such, the Coal Measures in this area (Younger 1995), as elsewhere in the coalfields of Europe and North America (e.g. Booth 2002), may now be regarded as being an 'anthropogenic aquifer' (Adams & Younger 2001); that is, although the rock strata are natural, their present aquifer properties are almost wholly a result of human activities. Severe fault-drag associated with the Butterknowle Fault (Fig. 3), which is a major east–west extensional fault, resulted in mining being uneconomic in a zone of 100–200 m either side of the fault trace. As a consequence, the Butterknowle Fault now functions essentially as a low-permeability boundary, hydraulically separating the flooded workings of the South Durham Coalfield (the subject of this study) from the still-pumped workings of the Central Durham Coalfield to the north (Younger 1993), save in a few specific localities where unproductive mine roadways were driven through the fault plane.

The Carboniferous strata are unconformably overlain by Permian strata (Fig. 2). In much of eastern England, the lowermost Permian strata comprise unconsolidated quartz arenites of aeolian origin, generally known as the Basal Permian Yellow Sands (BPYS) (Johnson 1970). Where present, the BPYS tend to form an important aquifer unit (Younger 1995). However, the present study area is somewhat unusual in that the BPYS are only sporadically present, in small pockets of limited hydrogeological significance (Younger 1995). Likewise only sporadic in its occurrence in this area is the Marl Slate, a thin mudstone bed that elsewhere separates the BPYS from the overlying Magnesian Limestone Aquifer (MLA), which is the most important aquifer in the NE of England (Cairney 1972; Younger 1995). For the most part, therefore, the Carboniferous units in this area are directly overlain by the dolomite and dolomitic limestone formations that together comprise the MLA (namely the Raisby, Ford, Concretionary Limestone, Roker Dolomite and Seaham Formations; Smith 1994). The MLA displays widely ranging transmissivities, ranging between 15 and 4600 $m^2\,day^{-1}$ (Cairney 1972), although a more restricted range of 60–800 $m^2\,day^{-1}$ is more commonly encountered in practice (Younger 1995). The unconformable contact between the MLA and the Carboniferous Coal Measures is exposed in the western part of the study area (Fig. 3), where it neatly defines the western boundary of the MLA groundwater flow system (Fig. 2).

Quaternary deposits, primarily of glacial origin, overlie the Permian strata (Fig. 2), and are particularly thick in the south and east of the region. These deposits, which are mostly diamicts of low permeability, significantly reduce aquifer recharge rates, and hydraulically confine groundwater in the MLA in some areas near the River Tees (Fig. 1). Significantly, the glacial deposits are thin or absent over the outcrop of the Carboniferous

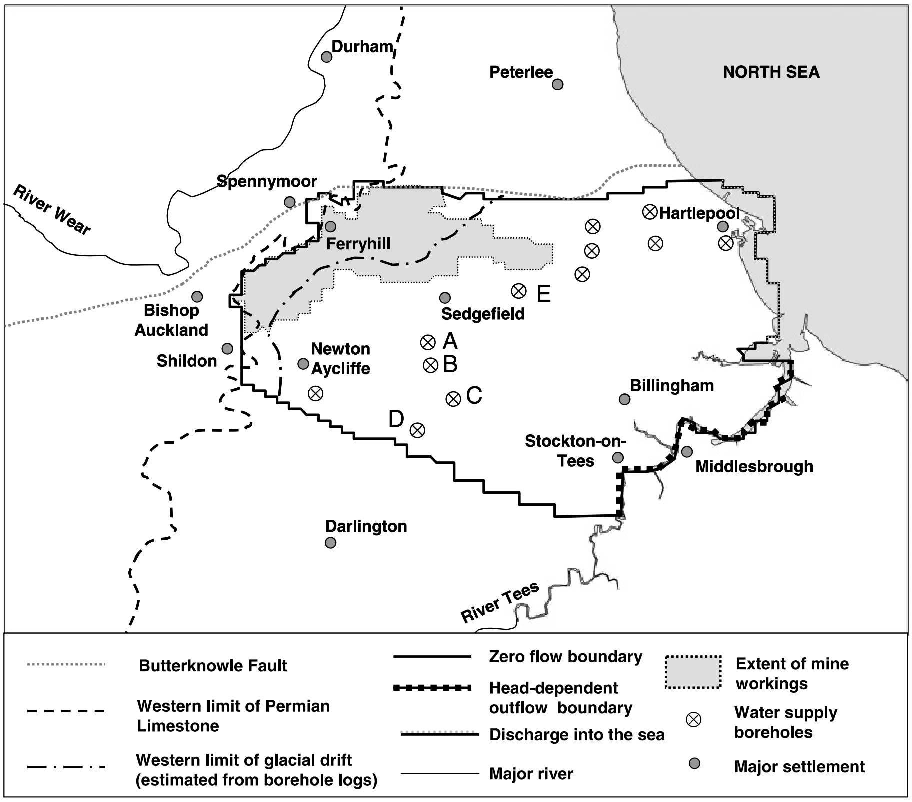

Fig. 3. Map showing groundwater system layout and boundary conditions assumed in the conceptual model. A selection of the public water supply boreholes (all of which are operated by the Hartlepools Water Company) are specifically identified as follows: A, Hopper House; B, Hope House; C, Stillington; D, Great Stainton; E, Waterloo Plantation. (The same letters are used in Table 2.)

strata, so that recharge to the Coal Measures is naturally more vigorous than to the MLA.

Analytical methods

To analyse flow and solute transport in this hydrogeological system, we developed a conceptual model (*sensu* Rushton 2003), which comprises a number of justifiable simplifying assumptions concerning the key hydrogeological features of the area. The first step in building the conceptual model is to collect and critically review relevant data describing the layout and physical properties, such as:

(1) geological data delineating the extent of the aquifer (many of which have been published by Smith & Francis 1967);

(2) transmissivity data for the aquifer (Cairney 1972; Younger 1995);

(3) boundary conditions of the groundwater flow system (derived in accordance with the methods described by Anderson & Woessner (1992) and Rushton (2003));

(4) groundwater head data (available from various observation wells maintained by the Environment Agency);

(5) abstraction rates (reported by the local water companies and other abstractors);

(6) recharge rates (calculated from the results of a public-domain water budget model known as ‘MORECS’ (Hough & Jones 1997), which takes into account rainfall, soil moisture status and actual evapotranspiration); reduced recharge rates were ascribed to southeastern areas with thick glacial diamict cover.

Once the conceptual model had reached a relatively mature stage of development, it was converted into its mathematical equivalents using the well-known computer codes MODFLOW (McDonald & Harbaugh 1988), for flow calculations, and MT3D (Zheng & Wang 1999), for solute transport simulations. With regard to the latter, only the transport of sulphate was simulated, as this is the only parameter that severely limits the potability of groundwater in the MLA in this case. Once both models had been adjusted to achieve satisfactory matches with observed piezometry and solute distributions, within the constraints of recharge estimates and feasible transmissivity values, predictive applications of

MT3D were undertaken to predict the future spreading of sulphates within the aquifer.

Rebound: groundwater level rises in the mine workings and the MLA

The history of mining and associated pumping in the study area is complex (e.g. Armstrong *et al.* 1959; Clarke 1962). In summary, the situation was that large-scale mining began in the 18th century in the west, where the Carboniferous Coal Measures are exposed at surface (Figs. 2 and 3), and proceeded gradually eastwards beneath Permian cover (Fig. 2), with nearly all 20th century mining taking place below the MLA. Occasional problems of excessive water ingress from the MLA were recorded during the 20th century (e.g. Clarke 1962; Saul 1970), most notably to Mainsforth Colliery (near Ferryhill; Fig. 3) and to Fishburn Colliery (north of Sedgefield; Fig. 3). The December 1968 inflow to the Harvey Seam workings of Fishburn Colliery was so substantial that it stopped production from that district of the mine for 3 weeks, an incident so notorious that it became immortalized in a popular humorous dialect song (Draycott 1978). Notwithstanding that incident, by the early 1970s Fishburn Colliery was the only mine still in production in that part of the Durham Coalfield south of the Butterknowle Fault. To prevent down-dip migration of groundwater to the working areas, pumping was maintained from a number of disused mine shafts in the western part of the study area, particularly between Ferryhill and the western fringes of Bishop Auckland (see Parkin & Adams 1998). When coal production finally ceased in 1974, pumping was discontinued, and a period of water-level recovery ('rebound') commenced. The recovery of water levels in the worked Coal Measures was sporadically interrupted on a number of occasions by renewed pumping from higher levels (Cairney & Frost 1975), but after 1976 rebound was allowed to occur unhindered. The recovery of water levels in the coal mine workings was recorded at several locations, as discussed by Younger & Adams (1999). After this process was well advanced, it was noticed that water levels had also begun rising in the MLA near Ferryhill (Brassington 1990), a process that eventually resulted in more than 10 m of water-level rise throughout most of the present study area (Younger 1995). Hydrochemical sampling of observation boreholes in this district revealed that much of the water making up the increased volume in storage in the MLA was heavily contaminated with sulphate, with concentrations ranging up to 800 mg l^{-1} in the vicinity of Ferryhill (Fig. 1; Younger & Adams 1999). Given that background concentrations of sulphate in the MLA are generally low (<45 mg l^{-1}), whereas the coal mine waters are highly enriched in sulphate (>1000 mg l^{-1}; as a result of dissolution of the residues from the oxidative weathering of pyrite (FeS_2) during the period in which the mine voids were force-ventilated; Younger & Adams 1999), it was reasonable to infer that much of the groundwater that had been added to storage in raising the head in the MLA near Ferryhill by more than 10 m was upwardly migrating mine water. That the hydraulic head in the mine workings had risen to locally exceed that in the MLA is readily explicable by the high elevation of the Coal Measures recharge area ($\leqslant$200 mAOD) in the area to the west of Bishop Auckland (see Figs. 2 and 3), where a number of surface outflows of mine water attest to heads in the Coal Measures close to ground level in the valley axes (Parkin & Adams 1998).

Summary of conceptual model

Extent of domain and boundary conditions

Groundwater heads measured in observation boreholes in the MLA indicate a general groundwater flow direction from west to east (Younger 1995). The resultant groundwater flow field is summarized in Figures 2 and 3, which show the principal geometrical features and assumed boundary conditions encompassed within the conceptual model. Bottom and top elevations of the MLA were defined using data from the drillers' logs of numerous boreholes in the area (Smith & Francis 1967). The extent of the worked Coal Measures were taken from statutory mine plans held by the Coal Authority (the UK government body responsible for regulating the coal industry and managing liabilities associated with abandoned coal mines). Mapped workings underlie about 15 km^2 of the MLA.

To the west of the MLA outcrop, hydraulic head in the worked Coal Measures is effectively maintained at a relatively constant elevation by the decanting of 'excess recharge' (i.e. recharge over and above that which makes its way into the MLA across the unconformity) through several prolific surface mine water outflows to the River Wear to the west of Bishop Auckland. Prior modelling of the Coal Measures in this area revealed that the high permeability of flowpath connections to these surface outflows results in an extremely subdued response to annual recharge events (Parkin & Adams 1998). It was therefore concluded that, for purposes of modelling plume migration in the MLA, it could be safely assumed that flow in the coal workings could be accounted for adequately by representing them with a head-dependent flow boundary condition applied across the Carboniferous–Permian unconformity. There was no need on this occasion to increase model complexity and run times by explicitly simulating heads and flows in the network of flooded mine voids.

The lateral boundaries of the model domain comprise (Fig. 3):

(1) a fault to the north (the Butterknowle Fault), which is known to be associated with a discontinuity in water levels in the MLA of several metres (assumed to be a zero-flow boundary);

(2) the westward limit of the MLA outcrop (also assumed to be a zero-flow boundary);

(3) the River Tees in the SE (assumed to be a head-dependent outflow boundary);

(4) a groundwater flow divide in the SW (zero flow boundary);

(5) the coast line between the city of Hartlepool and the River Tees (specified head boundary).

Sources and rates of recharge

Mine water migrating into the MLA from the Coal Measures is joined by natural recharge. In areas where glacial deposits are thin or absent, so that the MLA is present immediately below the soil surface, recharge rates were estimated from MORECS data (Hough & Jones 1997) to average around 0.77 mm day^{-1}. In areas with thick (>1 m) mantles of glacial deposits, recharge rates were assumed to be considerably less. Application to the local setting of reasoning suggested by Brassington (1998) resulted in reductions in estimated infiltration rates to an average of only 0.22 mm day^{-1} in these areas.

Aquifer properties

Initial property values for transmissivity, storativity and effective porosity were collated from test-pumping analysis results reported by Cairney (1972), Younger (1995) and Allen *et al.* (1997). These sources yielded a range of transmissivity values for this area in the range 25–450 m^2 day^{-1}. Taking into account typical saturated thicknesses, this in turn implies hydraulic conductivity values ranging from about 0.5 to 6 m day^{-1} (see Cairney 1972). The effective porosity was assumed to range between 0.03 and 0.07, judging from analogies to specific yield values obtained from pumping tests in unconfined portions of the aquifer. In the absence of any evidence upon which any alternative formulation might be based, all aquifer properties were assumed to be isotropic. Reasonable heterogeneities were introduced during the process of model refinement, where these were found to be needed to explain spatial variations in groundwater head gradients.

Mathematical and numerical models

MODFLOW simulations

The 3D movement of groundwater through porous material may be described by the following partial differential equation:

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_{zz}\frac{\partial h}{\partial z}\right)-W=S_s\frac{\partial h}{\partial t} \quad (1)$$

where K_{xx}, K_{yy} and K_{zz} are values of hydraulic conductivity along the x, y and z coordinate axes, which are assumed to be parallel to the major axes of hydraulic conductivity; h is the potentiometric head; W is a volumetric flux per unit volume and represents sources and/or sinks of water; S_s is the specific storage of the porous material, and t is time. Together with specifications of flow and/or head conditions at the boundary of the aquifer system, and recharge rates and aquifer property values in accordance with the conceptual model, the solution of equation (1) constitutes the complete mathematical representation of the groundwater flow system.

In accordance with the conceptual model, three hydrogeological layers were set up in the computer model: the glacial diamict deposits, the MLA, and the Coal Measures. The presence of glacial deposits overlying the MLA in large parts of the study area was taken into account by defining the MLA as an unconfined aquifer wherever the groundwater heads within it did not equal or exceed the elevation of the base of the glacial deposits. As mentioned above, recharge rates were also reduced in areas with substantial thicknesses of glacial deposits.

The input of the elevation of the base of the MLA was facilitated by a contour map with a 10 m interval published by Smith & Francis (1967). Elevations of the top of the MLA were defined either by the land surface in outcrop areas, or by the elevation of the base of the glacial deposits from borehole logs in areas where these cover the MLA. The basal layer representing the worked Coal Measures was assigned a uniform thickness (a convenient but harmless fiction in this case). This layer was also assigned a constant head value of 110 mAOD (corresponding to the head value maintained at the major surface outflow at St. Helen Auckland Engine Shaft (Parkin & Adams 1998), to the west of Bishop Auckland; Fig. 3). The leakance rate across the Carboniferous–Permian unconformity was defined as the ratio of vertical hydraulic conductivity to aquitard thickness, which in this case was set equal to the thickness of unworked Coal Measures above the workings, together with the Marl Slate (where present between the mine workings and the MLA). No field measurements of vertical hydraulic conductivity are known for this area; a low value (10^{-3} m day^{-1}) was assigned originally, which was gradually increased during model refinement to reproduce observed head conditions.

The model domain was divided into 50 × 50 grid cells, making a total of 2500 cells covering a total model area of about 350 km^2. Initial heads in the MLA were defined by average values for groundwater head measured in the period January–March 1999, which followed a well-characterized period of winter recharge. Water abstraction boreholes were represented as single-node constant flux boundaries internal to the model domain.

Refinement of the flow model was initiated assuming the Coal Measures to be hydraulically isolated from the MLA. No simulations using this approach were able to explain observed heads in those parts of the MLA underlain by mined Coal Measures. A leakage connection between the workings and the MLA was therefore introduced, with gradual increases in vertical hydraulic conductivity being tested until a reasonable match was obtained between observed and modelled heads (i.e. no differences in excess of 0.5 m).

MT3D simulations

MT3D (Modular 3-Dimensional Transport model) performs contaminant transport simulations by solving the following partial differential equation (Zheng & Wang 1999), which describes the fate and transport of species k in 3D, transient groundwater flow systems:

$$\frac{\partial(\theta C^k)}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial(C^k)}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(\theta v_i C^k) + q_s C_s^{k + \Sigma R_n} \quad (2)$$

where θ is the porosity of the subsurface medium (dimensionless); C^k is the dissolved concentration of species k; t is time; $X_{i,j}$ is the distance along the respective Cartesian coordinate axis; D_{ij} is the hydrodynamic dispersion coefficient tensor; v_i is the seepage or linear pore water velocity, related to the specific discharge or Darcy flux through the relationship $v_i = q_i/\theta$; q_s is the volumetric flow rate per unit volume of aquifer representing fluid sources (positive) and sinks (negative); C_s^k is the concentration of the source or sink flux for species k; ΣR_n is the chemical reaction term.

Solution of equation (2) simulates the transport and fate of contaminants subject to advection, dispersion, sinks, sources and chemical reactions. The advection term of the transport equation ($-\delta(\theta v_i C)/\delta x_i$) describes the transport of contaminants at the same velocity as the groundwater. The advection term dominates over other terms for many field-scale contaminant transport systems.

Dispersion is represented by the term $\delta/\delta x_i(\theta D_{ij}(\delta C^k/\delta x_j))$. This conceptually combines the effects of molecular diffusion and mechanical mixing (i.e. the spreading of contaminants over a greater region than would be predicted solely from the average groundwater velocity vectors; Zheng & Wang 1999).

Solute mass entering the model domain through sources or leaving the model domain through sinks is represented by the term $q_s C_s$, and this was used to represent the influx of sulphate from the mined Coal Measures, with q_s being calculated from the leakance flux obtained in the MODFLOW simulations, with the sulphate concentration in the Coal Measures being set equal to 800 mg l^{-1} (in accordance with values measured consistently over many years at a major overflowing shaft to the west of Bishop Auckland).

Chemical reactions are represented by the term ΣR_n. In this case, transport of sulphate was assumed to be conservative. This is justified by the fact that sulphate concentrations (maximum 800 mg l^{-1}) are too low for precipitation of gypsum or epsomite to occur, and there is insufficient labile organic carbon present in the aquifer to support bacterial sulphate reduction (see Younger *et al.* 2002). This being the case, the chemical reaction term could be simply disregarded.

Sulphate concentration data were available for 14 monitoring boreholes in the MLA for the period 1975–2000, which spans the period in which sulphate concentrations had risen from background values (of around 30 mg l^{-1}) to peak concentrations (since maintained). Using a 'split sample' approach, in refining the transport model simulations were compared with the concentration records for eight of the boreholes, and records for six (typically those with only more recent monitoring) were held back for checking the ability of the model to reproduce concentrations at points not used during refinement. To achieve a suitable match between observed and modelled borehole sulphate concentrations at the cluster of eight sites, leakance (and vertical hydraulic conductivity, upon which it depends) had to be further revised. For vertical hydraulic conductivity, three different property zones were defined, with values ranging from 1 to 25 m day^{-1}. The leakance factor was found to be around 3×10^{-6} day^{-1} for most of the area where the aquifer is underlain by worked Coal Measures. In one area of about 600 m^2 in the west, between the towns of Bishop Auckland, Shildon and Newton Aycliffe, a significantly higher leakance rate proved necessary (1×10^{-4} day^{-1}); this corresponds to an area in which it is known that mine workings were accidentally connected directly into the base of the MLA (Clarke 1962). At the other extreme, a zone of zero leakance had to be assigned to another area underlain by mine workings (covering about 1.5 km^2) to the north of Sedgefield; this corresponds to an area in which the thickness of unworked Coal Measures between the MLA and the mine voids is particularly high. Three different zones of effective porosity, ranging between 0.03 and 0.07, were defined. Longitudinal, transverse and vertical dispersivity could not be determined, as a result of the 'noise' associated with substantial diffuse upflows into the aquifer masking in-aquifer mixing effects. The sulphate distribution shown in Figure 4 shows the fruits of this 'history matching' phase of transport model refinement.

Having made these refinements to the simulation, their verisimilitude was assessed by comparing modelled

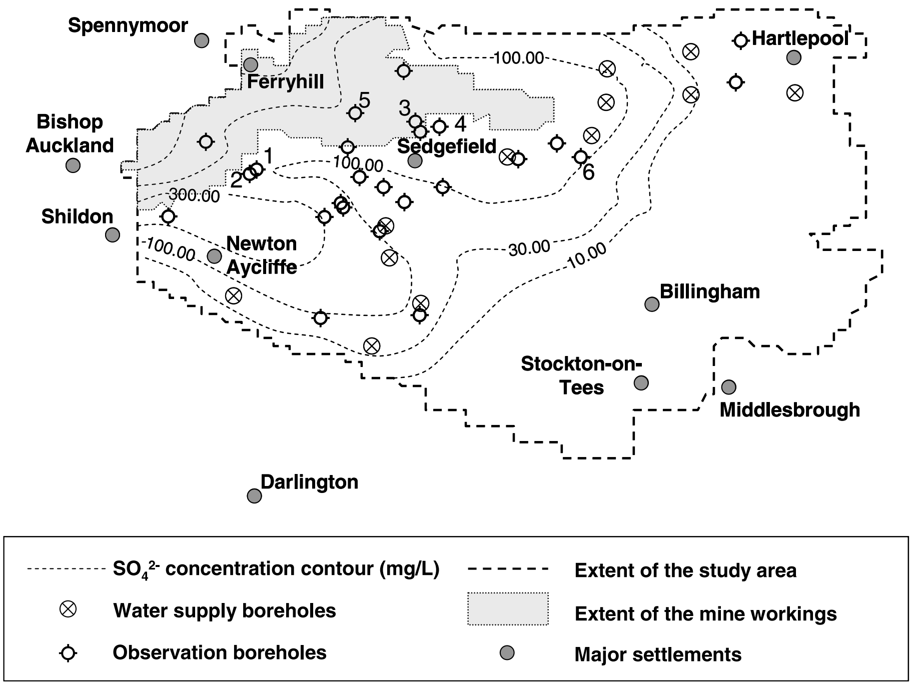

Fig. 4. Extent of sulphate plume in the Magnesian Limestone Aquifer in 2003, as defined by borehole monitoring data reproduced by the model (see Table 1). The observation boreholes specifically identified by numbers on this map are as follows: 1, Rushyford NE; 2, Rushyford A; 3, Lizards Farm; 4, Home Farm; 5, Bishop Middleham; 6, Low Swainston.

sulphate concentrations with values from the remaining six boreholes that had not been used during the refinement adjustments. An encouraging agreement was obtained (Table 1), especially bearing in mind the complexity of the real-world system.

It is evident that the spreading of the plume to 2003 had already begun to affect sulphate concentrations in a number of industrial abstraction boreholes in the west of the study area; fortunately, none of these industrial abstractions uses the water for purposes sensitive to sulphate concentration. However, the limit for sulphate in drinking water is $250\,mg\,l^{-1}$, and the local water supply company (the Hartlepools Water Company) operates a number of water supply wells to the east of the current plume perimeter. Predictive model simulations were carried out to simulate the further spreading of sulphate in the MLA towards these public supply wells. Predicted migration of the plume to 2010 is shown in Figure 5. It is clear that the leading edge of the plume is predicted to begin reaching the westernmost public supply wells around that time, albeit at concentrations that will still be within the drinking water limit ($250\,mg\,l^{-1}$). Table 2 summarizes predicted years in which the $250\,mg\,l^{-1}$ threshold will be exceeded in selected public supply abstraction boreholes, in the absence of any preventive measures being implemented in the mean time. Discussions are now under way between the Coal Authority, the Environment Agency and the Hartlepools Water Company to devise a management strategy to avoid long-term removal of these wells from the local resources base.

Table 1. *Comparison of observed (2001) and modelled sulphate concentrations for selected observation boreholes not used in the model refinement stage*

	Borehole	Sulphate concentrations ($mg\,l^{-1}$)	
		Observed	Modelled
1	Rushyford NE	80	104
2	Rushyford A	134	120
3	Lizards Farm	125	154
4	Home Farm	185	160
5	Bishop Middleham	105	117
6	Low Swainston	143	107

The numbers assigned to the boreholes are those used to denote their locations in Figure 4.

Discussion and conclusions

This paper has examined the first documented case of pollution of a public supply aquifer by mine water migrating into an overlying public supply aquifer. Similar scenarios can be expected in future, both in the same aquifer along strike in the English Midlands, and into other public supply aquifers in other European countries. Although the carbonate aquifer effectively prevents acidification of the groundwater, sulphate concentrations are not significantly attenuated by geochemical reactions and reach $800\,mg\,l^{-1}$ in places close to zones of upflow from the Coal Measures.

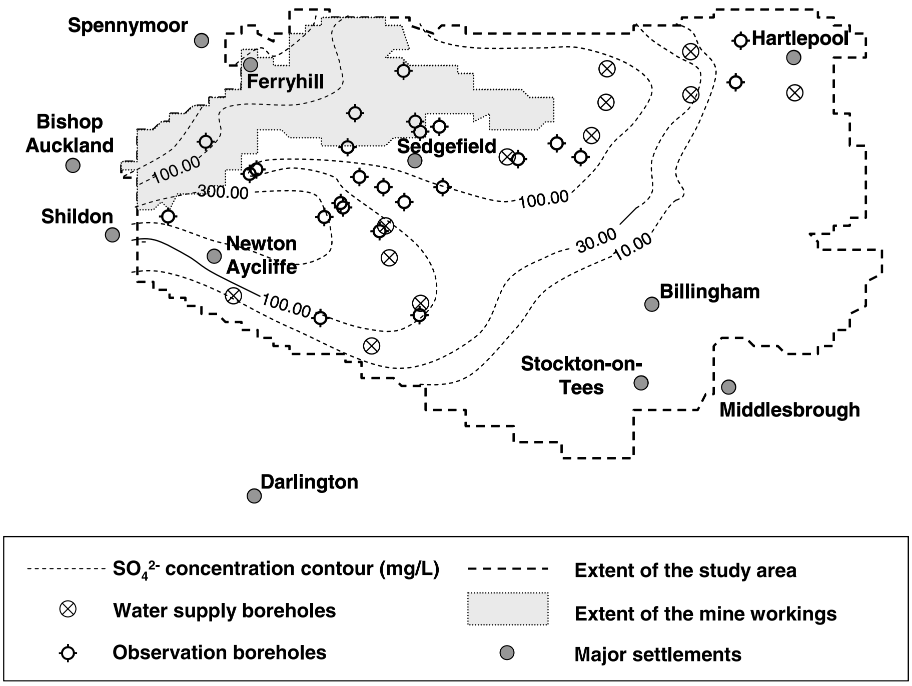

Fig. 5. Modelled expansion of the sulphate plume in the Magnesian Limestone Aquifer to 2010.

The MODFLOW simulations presented here have their limitations, in that only steady-state flow patterns were modelled. One consequence of this is that, during the early stages of upflow, the transmissivity in the MLA will locally have been lower than at present, as a result of the lesser saturated thickness prior to the completion of the rebound process identified by Brassington (1990). This also means that the groundwater head in the MLA (which is the main driver for lateral pollutant migration) was smaller during the onset of minewater migration, so that migration probably occurred rather more slowly in reality than our simulation suggested. A full simulation of the transient dynamics of the rebound processes and the early stages of pollutant migration in the MLA may eventually prove possible, although it was beyond the scope of the resources supporting the work described here.

Table 2. *Predictions of the years in which sulphate concentrations will first exceed 250 mg l^{-1} in selected public supply abstraction boreholes*

	Borehole	Year in which SO_4 is predicted to exceed 250 mg l^{-1} for the first time
A	Hopper House	2024
B	Hope House	2019
C	Stillington	2029
D	Great Stainton	2038
E	Waterloo Plantation	2057

The letters assigned to the boreholes are those used to denote their locations in Figure 3.

The MT3D model simulations were also based on the simplifying assumption that the upflow of mine water into the MLA occurs as a constant loading, representing a steady input of around 6.6 Ml day^{-1} of mine water with a uniform sulphate concentration of 800 mg l^{-1}. Oversimplified as this assumption seems the relative ease with which good correspondence was achieved between observed and modelled concentration distributions (e.g. Table 1) does suggest that inputs are rather uniform over time. This is consistent with the maintenance of a relatively constant groundwater head in the adjacent exposed Coal Measures as a result of overflow from a number of shafts and adits (Parkin & Adams 1998).

A key finding of this study is that upflow of mine water to the MLA occurs (albeit at variable rates) through almost all of the area in which the Permian strata are underlain by mine workings with hydraulic head in excess of that in the MLA. This strongly suggests that future mine water management strategies in similar settings should avoid a general raising of head in the Coal Measures above that in overlying aquifers, rather than focusing only on specific potential upflow 'hot spots' (see Dumpleton *et al.* 2001).

Simulations have shown that sulphate concentrations within the aquifer have significantly increased in areas overlying the contaminant source (i.e. coal workings; Fig. 4). Model results from a simulation carried out for further development of the sulphate plume (Fig. 5; Table 2) indicate that further down-gradient expansion of the plume is inevitable, eventually rendering some current public supply wells unusable (at least without blending with other waters), unless some preventative intervention is made.

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References

Adams, R. & Younger, P.L. 2001. A strategy for modeling ground water rebound in abandoned deep mine systems. *Ground Water,* **39**, 249–261.

Allen, D.J., Brewerton, L.J., Coleby, L.M., *et al.* 1997. *The physical properties of major aquifers in England and Wales*. British Geological Survey Technical Report **WD/97/34**. Environment Agency R&D Publication **8**.

Anderson, M.P. & Woessner, M.W. 1992. *Applied Groundwater Modelling—Simulation of Flow and Advective Transport.* Academic Press, San Diego, CA.

Armstrong, G., Kidd, R.R. & Buchan, S. 1959. Dewatering scheme in the south Durham Coalfield. *Transactions of the Institution of Mining Engineers,* **119**, 141–152.

Banks, D., Younger, P.L., Arnesen, R.-T., Iversen, E.R. & Banks, S.D. 1997. Mine-water chemistry: the good, the bad and the ugly. *Environmental Geology,* **32**, 157–174.

Booth, C.J. 2002. The effects of longwall coal mining on overlying aquifers. *In:* Younger, P.L. & Robins, N.S. (eds)(eds) *Mine Water Hydrogeology and Geochemistry.* Geological Society, London, Special Publications, **198**, 17–45.

Brassington, F.C. 1990. Rising groundwater levels in the United Kingdom. *Proceedings of the Institution of Civil Engineers (Part 1—Design and Construction),* **88**, 1037–1057.

Brassington, F.C. 1998. *Field Hydrogeology* 2nd. Wiley, Chichester.

Cairney, T. 1972. Hydrological investigation of the Magnesian Limestone of south east Durham, England. *Journal of Hydrology,* **16**, 323–340.

Cairney, T. & Frost, R.C. 1975. Case study of mine water-quality deterioration, Mainsforth Colliery, County Durham. *Journal of Hydrology,* **25**, 275–293.

Clarke, A.M. 1962. Some structural, hydrological and safety aspects of recent developments in south-east Durham. *Mining Engineer,* **122**, 209–231.

Draycott, B. 1978. The Watter's Brokken In. *In:* Lloyd, A.L. (ed.) *Come all ye Bold Miners: Ballads and Songs of the Coalfields* 2nd. Lawrence & Wishart, London, 196–197.

Dumpleton, S., Robins, N.S., Walker, J.A. & Merrin, P.D. 2001. Mine water rebound in South Nottinghamshire: risk evaluation using 3-D visualization and predictive modelling. *Quarterly Journal of Engineering Geology and Hydrogeology,* **34**, 307–319.

Hough, M.N. & Jones, R.J.A. 1997. *The United Kingdom Meteorological Office rainfall and evaporation calculation system: MORECS version 2.0—an overview.* Meteorological Office, Bracknell.

Johnson, G.A.L. 1970. Geology of Durham County. *Transactions of the Natural History Society of Northumberland, Durham and Newcastle upon Tyne,* **41**, 1–152.

McDonald, M.C. & Harbaugh, A.W. 1988. A modular three-dimensional finite-difference ground-water flow model. *US Geological Survey, Techniques of Water-Resources Investigations,* **6**, A1.

Parkin, G. & Adams, R. 1998. Using catchment models for groundwater problems: evaluating the impact of mine dewatering and groundwater abstraction. *In:* Wheater, H. & Kirby, C. (eds) *Hydrology in a Changing Environment. (Proceedings of the International Symposium organized by the British Hydrological Society, 6&10 July 1998, Exeter, UK),* **II**. Wiley, Chichester, 269–279.

Razowska, L. 2001. Changes of groundwater chemistry caused by the flooding of iron ore mines (Czestochowa Region, southern Poland). *Journal of Hydrology,* **244**, 17–32.

Rushton, K.R. 2003. *Groundwater Hydrology. Conceptual and Computational Models.* Wiley, Chichester.

Saul, H. 1970. Current mine drainage problems. *Transactions of the Institution of Mining and Metallurgy (Section A),* **79**, A63–A80.

Smith, D.B. 1994. *Geology of the country around Sunderland.* Memoir of the British Geological Survey, Sheet 21 (England and Wales), **21**. HMSO, London.

Smith, D.B. & Francis, E.A. 1967. *Geology of the Country between Durham and West Hartlepool.* HMSO, London.

Wolkersdorfer, C. & Bowell, R.J. (eds) 2004. Contemporary reviews of mine water studies in Europe, Part 1. *Mine Water and the Environment,* **23**, 162–182.

Wolkersdorfer, C. & Bowell, R.J. (eds) 2005*a*. Contemporary reviews of mine water studies in Europe, Part 2. *Mine Water and the Environment,* **24**, 2–37.

Wolkersdorfer, C. & Bowell, R.J. (eds) 2005*b*. Contemporary reviews of mine water studies in Europe, Part 3. *Mine Water and the Environment,* **24**, 58–76.

Younger, P.L. 1993. Possible environmental impact of the closure of two collieries in County Durham. *Journal of the Institution of Water and Environmental Management,* **7**, 521–531.

Younger, P.L., 1995. Hydrogeology. In: Johnson, G.A.L. (ed.) *Robson's Geology of North East England.* (*The Geology of North East England*, 2nd edn). *Transactions of the Natural History Society of Northumbria*, **56**(5), 353-359.

Younger, P.L. & Adams, R. 1999. *Predicting mine water rebound.* Environment Agency R&D Technical Report, **W179**.

Younger, P.L., Banwart, S.A. & Hedin, R.S. 2002. *Mine Water: Hydrology.* Kluwer, Dordrecht.

Zheng, C., and Wang, P. 1999. MT3DMS: A Modular Three-Dimensional Multispecies Transport Model for Simulation of Advection, Dispersion, and Chemical Reactions of Contaminants in Groundwater Systems. Documentation and User's Guide. World Wide Web address: www: http://www.us.net/envisim.

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