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Assessing and Improving Sustainability of Urban Water Resources and Systems
AISUWRS Work-package 4: Field investigations final report

Groundwater Systems and Water Quality Programme
Commissioned Report CR/05/028N

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BRITISH GEOLOGICAL SURVEY

GROUNDWATER SYSTEMS AND WATER QUALITY PROGRAMME

COMMISSIONED REPORT CR/05/028N

Assessing and Improving Sustainability of Urban Water Resources and Systems

AISUWRS Work-package 4

Field investigations final report

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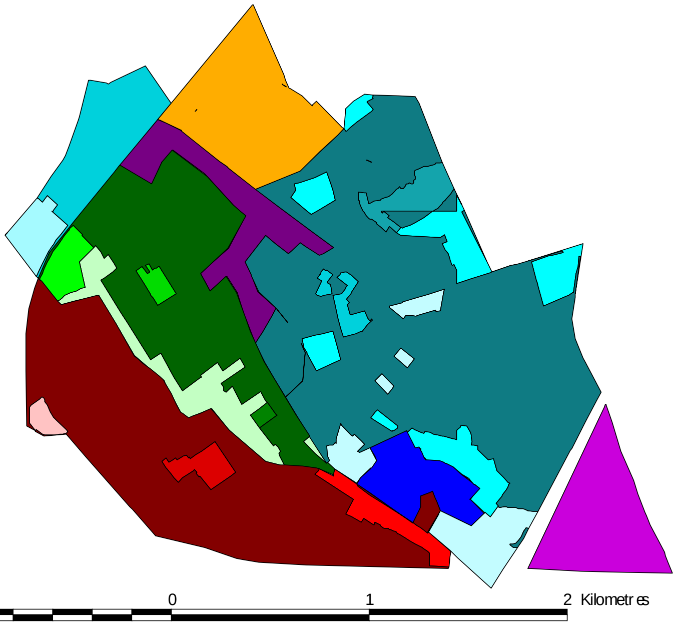

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20 proposed UVQ model neighbourhoods in Bessacarr-Cantley study area

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Foreword

This final field investigations report comprises the second part of Deliverable D10 of the project "Assessing and Improving the Sustainability of Urban Water Resources and Systems" (AISUWRS). It is jointly produced by the UK partners the British Geological Survey and the Robens Centre for Public and Environmental Health of the University of Surrey. The AISUWRS project is a 3-year urban water research programme partly funded by the European Community 5th Framework Programme-Shared Cost Research Technological Development and Demonstration. It aims to develop an innovative modelling system of the urban water infrastructure that can inform decision support systems for cities that depend on underlying or nearby aquifers for their water supply. Doncaster is one of the four case study cities being examined in Work Package 4 of this project; the others being Rastatt (Germany), Ljubljana (Slovenia) and Mt. Gambier (Australia).

Since the publication of the interim report (CR/04/022N), the UK project team at the Robens Centre and the BGS have completed the field investigations phase and used the results to write a number of technical papers for publication in peer-reviewed journals or conference proceedings. This report brings together the drafts of these papers as they provide most of the key results of the field investigations in a concise form. The key findings of the field-based investigations described in these papers is brought together at the end as an Outcomes and Conclusions section, while the new data (analytical results) collected from Work Package 4's field monitoring and surveillance activities in the Doncaster area are listed in Appendices 1 and 2.

Acknowledgements

The authors would like to thank the project's stakeholders for their practical support in this on-going project, especially the following:

David Craven, Doncaster Metropolitan Borough Council (DMBC) Environmental Health Dept and Bernard Russell, DMBC Neighbourhood Services; Gerd Cachandt, Brian Humphries, Ian Mitchell, Meryl Knapp, Andrew Bardsley, Matilda Beatty and various colleagues at Yorkshire Water (YW); Gordon Smith, Andrew Pearson, and Martin Shepley, Environment Agency (EA), Midlands Region.

Contents

1 Introduction

1.1 BACKGROUND

This project report is produced jointly by the UK partners of the AISUWRS consortium: the British Geological Survey and the Robens Centre for Public and Environmental Health at the University of Surrey. The 3-year AISUWRS urban water research project is partly funded by the European Community 5th Framework Programme for Shared Cost Research, Technological Development and Demonstration. The 5th Framework Programme was conceived to help solve problems and respond to major socio-economic challenges the European Union is facing. It focuses on a number of objectives and areas combining technological, industrial, economic, social and cultural aspects.

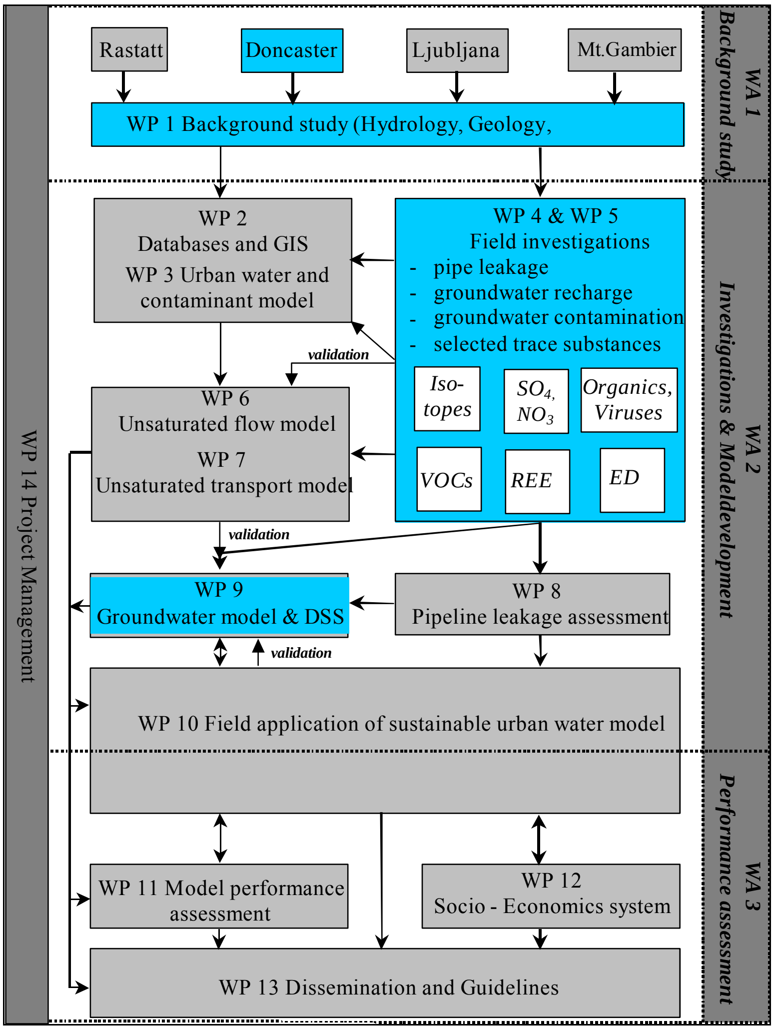
The project is one of a number of European research projects on integrated urban water management that are clustered as the CityNet group. The AISUWRS project aims to develop innovative new modelling techniques and a pilot decision support system (DSS) for cities that depend on underlying or nearby aquifers for their water supply. The objective is to assess and improve the sustainability of urban water resources and systems with the help of computer tools. The AISUWRS project ("Assessing and Improving the Sustainability of Urban Water Resources and Systems") is using case studies of the cities of Doncaster England, Rastatt Germany, Ljubljana Slovenia and Mount Gambier Australia to test and develop an integrated suite of models for urban water management purposes. The European case study cities represent examples of common urban, hydrogeological and water infrastructure settings, so successful application of the models to these situations will be a test of the system's robustness for wider use in the many other cities in Europe and elsewhere that depend on local groundwater for public and private water supply. The roles of the different partners in the project are described in detail in the project's Description of Work (Eiswirth, 2002) and summarised in Table 1.

Table 1 Roles of partners in AISUWRS project

Country	Case study city	Partner	Role
Germany	Rastatt	University of Karlsruhe	Rastatt case study, unsaturated zone flow model, groundwater model, development of DSS and application to Rastatt, dissemination, project management
		GKW Consult	Model performance assessment, socio-economics
Slovenia	Ljubljana	Institute for Mining, Geotechnology and Environment	Ljubljana case study, database development, groundwater model and DSS application to Ljubljana
UK	Doncaster	Robens Centre for Public & Environmental Health (Univ. of Surrey) British Geological Survey	(jointly) Doncaster case study, groundwater model and DSS application to Doncaster, dissemination
Australia	Mt Gambier, inputs to the 3 European cities	Commonwealth Scientific and Industrial Research Organisation	Urban water & contaminant model, comparison investigations in Mt Gambier, unsaturated transport and pipeline leakage models and DSS application to 4 case study cities

1.2 OBJECTIVES OF THIS REPORT

This final report follows the interim report CR/04/022N issued in May 2004 and provides the results of the field investigations in Doncaster, part of Work Package 4, as shown schematically in Figure 1.



WA = working areas, WP = workpackages

Figure 1 Graphical representation of AISUWRS's components (interconnection diagram).

The objectives of the field investigations reported in this document were as follows:

1. Assess relative magnitude of sources of urban groundwater recharge and their effects on the quality and availability of water for public and private supply.
2. Detail the distribution and persistence of standard sewage indicators and sewage-derived viruses and their seasonal fluctuations.
3. Describe vertical variations in lithology, structure and vertical hydraulic gradients in the aquifer.
4. Provide key information to design and calibrate models to assist the quantification of recharge sources.

To meet these ambitious goals the following action plan was performed:

1. *Identify a suitable study area in Doncaster for detailed monitoring and modelling.* The initial phase, when available data were collected, covered a quadrangle of almost 400 km^2. This large area of coverage was necessary to establish what already-existing information might be available, to populate the groundwater model, to assess where project field efforts should be concentrated and to identify an urban area that was both logistically and technically feasible to separate out as a detailed study area. This phase successfully identified the district of Bessacarr-Cantley as suitable because it is well defined in terms of landuse and located directly down-gradient of the old city centre of Doncaster. Subsequent monitoring activities gradually concentrated on sampling points in and around this 6.3 km^2 area of detailed study.

2. *Establish the groundwater setting of Bessacarr-Cantley and characterise the urban water infrastructure.* The project sited and constructed five multilevel research boreholes at four locations and instituted a comprehensive programme of piezometric, microbiological and hydrochemical sampling and analysis. A parallel regional programme of sampling of selected private wells in the general vicinity was established, together with sampling from the district's sewer and piped pluvial drainage network. Over 280 individual sample sets, each consisting of 6 field parameters, 33 hydrochemical parameters and 6 microbial parameters were acquired. A number of selected samples were additionally analysed for isotopes, residence time indicators and enteric viruses.
3. *Collect, collate and refine groundwater level information around the study area in order to inform the urban water budget models.* The project collected a comprehensive and wide-ranging array of datasets from stakeholders and data licensers in order to populate the model array. An assessment of each model's requirements was conducted in consultation with the project partner responsible for developing the model code in question. This enabled >300 separate data input fields to be prioritised and categorised in terms of data quality and availability
4. *Characterise urban recharge qualitatively, then compare with the water quality in the pipe infrastructure in order to inform the contaminant transport models.* The field teams in each case-study city selected a range of possible indicators. There are no universally applicable urban recharge indicators, and the species selected for appraisal varied between different cities according to the availability of background data, sampling and analytical feasibility, and the hydrogeological setting. For the Doncaster study, the UK team tentatively chose Cl, B, and faecal coliforms from a literature search as indicators for eventual contaminant transport modelling, but ensured that a much wider range of determinands was analysed in recognition of the uncertainty whether they would be the most suitable. This proved to be a wise precaution, as K, Na and HCO_3 appear to be more reliable major ion indicators and faecal streptococci and sulphite-reducing clostridia better microbial indicators in the Doncaster setting.
5. *Interpret all available data and provide the necessary basis for the subsequent modelling task.* The large number of new results from the field campaigns, together with historic datasets, water supply, landuse and other data collected under task 3 above, were analysed to obtain a better understanding of groundwater flow and its interaction with the urban water supply network.

This report now contains a detailed discussion of the results, provides the necessary information for the modelling task and reports the major conclusions of the research team.

1.3 REPORT LAYOUT

The field data collected during the field investigation phase is tabulated in Appendix 1 (microbiological) and 2 (hydrochemical). Its interpretation has been explored in a series of six papers, the titles and abstracts of which are collated into the first part of Section 2 in order to provide an overview of the topics addressed. The second and third parts of Section 2 contain the draft papers themselves, organised to cover groundwater and sewage issues in general, then tracers to detect and assess leakage, then the new conceptual picture of the aquifer. A key findings box is included at the start of each paper to provide a concise description of outcomes. The Outcomes and Conclusions section summarises and draws together the conclusions from these papers and provides notes to inform the modelling activities being undertaken during the final year of the project.

2 Interpretation of results of field investigations

2.1 TITLE AND ABSTRACT OF PAPERS INTERPRETING VARIOUS ASPECTS OF CASE STUDY FIELD DATA

#	Title and abstract	Authorship	Destination
1	*Monitoring groundwater quality: multilevel against fully penetrating boreholes to assess groundwater quality and resources* Common strategies for groundwater monitoring aim to resolve both the temporal evolution and the spatial distribution of groundwater composition by using existing fully-screened groundwater wells. This approach fulfils the requirements of most national legislations. The proposed new EU Water Framework Directive introduces the concept of trend observation and reversal as a criterion to promote good quality groundwater resources for future generations. However, a better understanding of the vertical distribution and movement of contaminants can be obtained by depth-specific sampling, and to this end, many multilevel wells have been designed and constructed over the past few years. Direct comparisons between the two types of monitoring wells are scarce, and this paper presents results from a study in the vicinity of the UK city of Doncaster, where a regional groundwater monitoring network of existing fully-penetrating boreholes was set up to resolve the spatial distribution of groundwater quality. As an extension and improvement to the network, 5 multilevel boreholes were installed to sample groundwater at up to 7 different depth-specific intervals between 10 and 60 mbgl. The sampling included field physico-chemical measurements, with samples for major and minor hydrochemistry and bacterial and viral indicators of faecal contamination. Results of the first four sampling campaigns of the regional monitoring network revealed a complicated pattern of groundwater quality. When historical data are added from the Environment Agency's monitoring wells the picture becomes even more complicated. However, temporal trends in groundwater quality observed in some areas can be ascribed to human impact on the urban water resource originating either from pumping-induced leakage or urban contamination. The multilevel boreholes, however, revealed clear signatures of urban contamination. Boron, potassium and microbial faecal indicators were found to be favourable qualitative tracers of groundwater contamination by wastewater due to the high differences between source and natural background concentrations. Urban contamination was mainly detected in the top 20-30 metres. The nearby regional wells, however, which mostly cover the deeper layers (30 to 70 mbgl) showed infrequent and/or low levels of urban contamination. The multilevel monitoring wells demonstrate that depth-specific groundwater sampling can improve the understanding of groundwater flow and, therefore, help predict potential risks to the groundwater resources. Additionally, the understanding of vertical groundwater flow can support future abstraction well design (including screen interval). Furthermore, comparison of both sampling well designs shows that multilevels offer a significant advantage if low concentration and high-risk contaminants need to be detected at an early stage because open boreholes would reveal diluted measurements that may be below the detection limit.	J. Rueedi, A.A. Cronin, R. Taylor	IWRA XIIth World Water Congress, 22-25th November 2005, New Delhi, India

2	*Daily patterns of micro-organisms in the foul sewer system of Doncaster, United Kingdom* Much is known about short-term changes in sewage volumes because they are often measured in pipeline design studies. Usually, sewage volumes peak in the morning and again in the evening, and leakage rates are expected to change correspondingly. To assess the contaminant load leaking into the subsurface, however, it is important to know the daily evolution of contaminant concentrations. These data are scarce, particularly for pathogenic micro-organisms. This study describes results of raw sewage sampling at three pumping stations for total coliforms, faecal coliforms, faecal streptococci, sulphite-reducing clostridia, coliphage and enteric viruses. The results show that all analysed micro-organisms follow a similar daily pattern, namely high concentrations during the morning, dropping around noon, rising again during the afternoon and then falling again in the evening and night time. Concurrent measurement of sewage volumes enabled assessment of total contaminant loads leaving the urban area, estimation of likely daily patterns of toilet/grey water contribution to raw sewage and calculation of daily average loads for urban mass balance modelling. The paper shows how important it is to measure not only daily patterns of sewage volumes but also sewage concentrations because both vary significantly during the day due to different household usage patterns.	J. Rueedi, A.A. Cronin, B.L. Morris	10th International Conference on Urban Drainage, Copenhagen Denmark, 21-26 August 2005
3	*The effectiveness of selected microbial and chemical indicators to detect sewer leakage impacts on urban groundwater quality* Sewer and stormwater pipe leakage can lead to the degradation of urban groundwater quality. This groundwater may be subsequently used for public water supply and so the resulting water treatment and public health consequences can be serious. To understand the impact of sewer exfiltration on groundwater quality, suitable indicators need to be sampled and analysed. This study examined potential sewer-derived inorganic and microbial parameters in the UK city of Doncaster. Sulphite reducing clostridia, faecal streptococci and boron were all detected in groundwater with reductions compared with sewer values ranging from 1 to 6 orders of magnitude for the former two to 1 to 2 orders of magnitude decrease for boron. The correlation between these two different indicator types suggests that groundwater quality is being adversely affected by sewer leakage in the study area. The employment of several and varied indicators can better demonstrate the effect than use of single parameters.	A.A. Cronin, J. Rueedi, B.L. Morris	10th International Conference on Urban Drainage, Copenhagen Denmark, 21-26 August 2005
4	*The use of $\delta^{13}C_{TDIC}$ as a tracer of groundwater evolution influenced by both natural and anthropogenic processes in four UK Permo-Triassic aquifers* This paper examines the use of stable isotope ratios of dissolved inorganic carbon ($\delta^{13}C_{TDIC}$) as a tracer of both vertical and horizontal groundwater movement in four UK Permo-Triassic sandstone aquifers. Seven multilevel piezometers were constructed in the English Midlands (Nottingham, Birmingham and Doncaster) to evaluate the depth to which urban contaminants have penetrated into the underlying urban aquifers. In addition, 11 open well and 3 sewer sampling points were looked at in Doncaster for additional comparative purposes. Furthermore, forty open abstraction boreholes were sampled in the Sherwood Sandstone of the Lagan Valley,	A.A. Cronin, J. Rueedi, R.G. Taylor, B.L. Morris, T. Elliot, R.M. Kalin	Geochimica et Cosmochimica Acta

	Belfast in order to assess regional variations in $\delta^{13}C_{TDIC}$. Matrix $\delta^{13}C$ samples have also been analysed each study aquifer and these new data are also presented. $\delta^{13}C_{TDIC}$ signatures proved useful in resolving the principal influences on the geochemical evolution issues of urbanised Triassic sandstone groundwaters. Evolutions were found to be affected by both natural (carbonate and gypsum dissolution) and anthropogenic sources (mainly sewer-derived recharge). Combined application of groundwater pH, TDIC and $\delta^{13}C_{TDIC}$ distinguished between different evolution pathways and helps constrain potential ambiguities arising from the observation of TDIC and pH alone. The findings show that, under natural conditions, the carbonate system evolves in a similar way in each aquifer. An open system evolution during recharge and early groundwater flow largely saturates the groundwater with carbonate. Matrix calcite is not always available in sufficient amounts so that some samples remain undersaturated. Depending upon the nature of the rock matrix passed through during groundwater flow, dissolution of gypsum or dolomite leads to the precipitation of calcite. If sewer exfiltration is added to the recharge the subsequent mixture shows a readily distinguishable signature with lower pH and higher TDIC values but constant $\delta^{13}C_{TDIC}$. However, $\delta^{13}C_{TDIC}$ cannot be used in isolation to determine the extent of geochemical evolution but it can advance the understanding of natural and anthropogenic influences on urban groundwater quality.		
5	*Estimating sewer leakage using hydrochemistry sampling of multilevel piezometers* The need to maintain or regularly replace ageing pipe systems to prevent sewage outflow to the subsurface environment makes leaking sewer systems a financial and environmental burden. In particular, the subsequent contamination of soil and groundwater with sewage-derived pathogens (e.g. viruses), heavy metals (e.g. zinc) or trace substances (e.g. endocrine disruptors) has come under increasing scrutiny. The EU, recognising this problem, has funded research to assist development of management strategies to improve the structural integrity of urban sewer systems. Test methods and leakage simulation models are increasingly used to analyse sewage systems and optimise maintenance efforts. However, calibration of these models is usually done by using available pipe asset information and extrapolating detailed knowledge of single leaks to the entire sewerage system. This paper presents an approach to estimating sewer leakage rates independently by using groundwater quality measurements and demonstrates an example taken from different depths of multilevel monitoring wells specially installed below Bessacarr, a suburb of the UK city of Doncaster. The results show total leakage rates from the sewage system are typically in the range of 20-45mm/y, corresponding to a total leakage of 7-15% of the annual sewage throughput volume. This is more than that shown in previous studies. However, this is partly a result of the large separate pipeline system. Results will enable a better calibration of the decision support system being developed as part of the AISUWRS project. These results underpin the importance of groundwater quality monitoring in general and using depth-specific sampling, in particular, to independently quantify leakage rates from sewage systems.	J. Rueedi, A.A. Cronin, B.L.Morris	Water Resources Research

6	*Assessing the impact of modern recharge on a sandstone aquifer beneath a suburb of Doncaster, UK* A major water quality issue in urban areas underlain by a productive aquifer is the impact of modern recharge. Using a variety of sample sources including multi-level boreholes, this study has found detectable CFCs and SF$_6$ throughout the upper 50 m of the saturated aquifer beneath a suburb of Doncaster, indicating that modern (<50 year old) recharge has penetrated to at least this depth. Additional support for this deep penetration is provided by the detection of sulphite-reducing clostridia and faecal streptococci. Despite the upper aquifer being a poorly cemented sandstone, the residence time indicators suggest that modern recharge has travelled via fracture systems rather than by simple piston flow. However, the overall impact of 80 years of steady urbanisation on water quality in the aquifer beneath this suburb has in general been limited. This is attributed to a combination of factors including previous land use, dilution by direct recharge of rainfall through green-space areas including gardens, and locally high storage in the friable upper aquifer.	B.L. Morris, W.G. Darling, A.A. Cronin, J. Rueedi, E.J. Whitehead, D.C. Gooddy	Hydrogeology Journal

2.2 TITLE AND ABSTRACT OF TECHNIQUE PAPERS USING CASE STUDY FIELD DATA

#	Title and abstract	Authorship	Destination
7	*Geographic Information System Analysis in pipe infrastructure modelling: making the most of available data* GIS analysis of data in the AISUWRS research project is helping develop an array of linked water models that aim to facilitate sustainable management of urban water in groundwater-dependent cities. This international project involves system development in four towns, Doncaster being the UK example. The GIS has proven indispensable to analyse available data in order to populate models and also to help model linkage by permitting spatial outputs to cascade into model inputs downstream. This paper demonstrates how already available data, produced for quite different purposes, is transformed by interpretation and manipulation into inputs suitable for the urban water flow and transport models. Examples (*pipe infrastructure analysis, production of a sewer gain/loss map*, and *land-use analysis)* demonstrate that even for demanding multiple-parameter models like those employed in the AISUWRS project, innovative use of available data can provide many of the-site-specific values required to calibrate models for an urban area.	J. Cunningham, B.L. Morris, J. Rueedi	2nd CIWEM National Conference, Wakefield UK, 5-8 September 2004
8	*Groundwater surcharging of sewers: example from Doncaster, England of a technique for identifying its extent.* A technique is described for the rapid assessment of which parts of an existing sewer or pluvial drain network may be below the watertable and, therefore, in areas of potential gain from groundwater. The technique, which arose from data conditioning for the pipeline leakage part of an urban water balance model array, can be used together with a groundwater flow model to undertake scenario modelling. The extent of effects on the sewer network from different groundwater level rebound scenarios can then be rapidly predicted. Such changes in water level can arise for example from decline in nearby urban/periurban groundwater abstraction or as a consequence of increased urban recharge. An example is given for a 6.3 km^2 suburb of Doncaster, England with a 128 km sewer and pluvial drain network.	B.L. Morris, I. Neumann, J.E. Cunningham, R.L. Hargreaves ,J. Rueedi, A.A. Cronin	10th International Conference on Urban Drainage, Copenhagen Denmark, 21-26 August 2005

2.3 HYDROCHEMISTRY

This section includes 6 papers relating to both hydrochemical and microbiological aspects of the water quality of the Bessacarr-Cantley study area and other sampling points in the vicinity. Each paper is prefaced by a key findings box.

2.3.1 Paper 1: Monitoring groundwater quality: multilevel against fully penetrating boreholes to assess groundwater quality and resources (Rueedi *et al.*)

Key findings

- Comparison shows importance and usefulness of employing multilevel boreholes to assess groundwater quality, particularly if trace substances of public health significance are involved (e.g. micro-organisms, heavy metals, boron, etc.).
- The good agreement between multilevel piezometers and nearby regional wells that are screened over a longer depth interval in terms of major hydrochemical parameters shows that the two monitoring methods produce compatible results but that the multilevel wells provide greater detail.
- Multilevel wells give significantly help the understanding of the three-dimensional movement of water and contaminants as they enable the vertical resolution of groundwater quality (or groundwater contamination). Depth-related quality information makes it possible to predict water quality deterioration trends, choose options for mitigation (e.g. re-siting of abstraction wells or deeper screen intervals) and assess effectiveness of protection strategies (landuse change, control of pollutant flux by changes in agricultural practice, etc.).

Introduction

In most countries, groundwater protection criteria utilise threshold values, i.e. groundwater quality fails to meet the required standard if certain values are above maximum allowable concentrations. The proposed new EU Framework Directive introduces the concept of trend observation and reversal as a criterion to promote good quality groundwater resources for future generations [1]. The proposed monitoring strategy accounts for trends in key water quality indicators but neglects the importance of the contaminant sources and pathways that invariably lead to a dynamic horizontal and vertical distribution of contaminants and, hence, spatial as well as temporal variation in groundwater quality parameters.

The horizontal distributions can be approximated using common fully-penetrating monitoring wells. Fully penetrating wells are either open wells (without screen section) or wells screened over the full saturated depth-range of the borehole. Private and public groundwater abstraction wells are often screened over a large depth interval in order to provide the maximum possible water abstraction. Bundled piezometers are wells with multiple, hydraulically separated and short screen sections. Fully penetrating wells a) may lack comparability between sites because they cover different depth intervals; b) are unable to provide information on vertical contaminant movement; and c) may not detect high-risk pollutants with low toxic doses (e.g. viruses, organics, hormones) in a timely fashion because an exceedance in the shallow zone is being diluted with unaffected water from the rest of the open hole.

One way to overcome this problem is to sample the well with a double-packer assembly [2] or a dual pumping assembly [3]. Double-packer systems can only be applied in open boreholes (without screen) and dual pumping assemblies require a large number of samples to properly resolve the vertical profile. Hence, in order to better constrain variation in water quality with depth, several recent studies have employed bundled multilevel piezometers to assess and monitor both groundwater quality and resources in all three dimensions [4].

This paper analyses and discusses the major differences between fully penetrating wells and multilevel wells by using a recently acquired data set comprising samples from both types of wells located in the vicinity of the UK city of Doncaster.

Methods

RESEARCH SITE

Table 1. Construction details of the wells selected for the regional sampling network.

ID	Name	Drilled depth	Screen interval
Wells within 3 km radius of Multilevels			
RC	Race course	41.1	unknown
Pg	Pegler Ltd.	30.5	unknown
WTL	Warning Tongue Lane	63.4	18.3 to 63.4
GWG	Gatewoood Grange	76.2	unknown
CT	Cantley Water Tower	65.5	27.2 to 65.5
Wells outside 3 km radius of Multilevels			
MQ	Misson Quarry	76.2	24.4 to 76.2
SCF	Sandall Common Farm	63.4	17.4 to 63.4
CTF	Crow Tree Farm	33.5	17.7 to 33.5
BTN	Beech Tree Nursery	30.5	17.7 to 30.5
LF	Lings Farm	14.7	10.9 to 14.7
EF	Elmstone	50	19.5 to 50

The location of Doncaster is shown in Figure 1 together with the locations of the production wells of the local public water supply, the eleven regional sampling wells and the five multilevel monitoring wells. The multilevel wells are located in or near the suburb of Bessacarr-Cantley, which was selected to be the focus study area for the EU 5th FP project AISUWRS (Assessing and Improving Urban Water Resources and Systems).

The production wells are drilled to depths between 140 to 170 meters below ground level (mbgl) and screened to the bottom from various depths below about 27 mbgl. The wells for the regional monitoring network were selected to represent the distribution of shallow groundwater in the vicinity of the urban study area. Construction details are listed in Table 1. The wells are grouped into those that are within 3 km of Bessacarr-Cantley and those that are more distant.

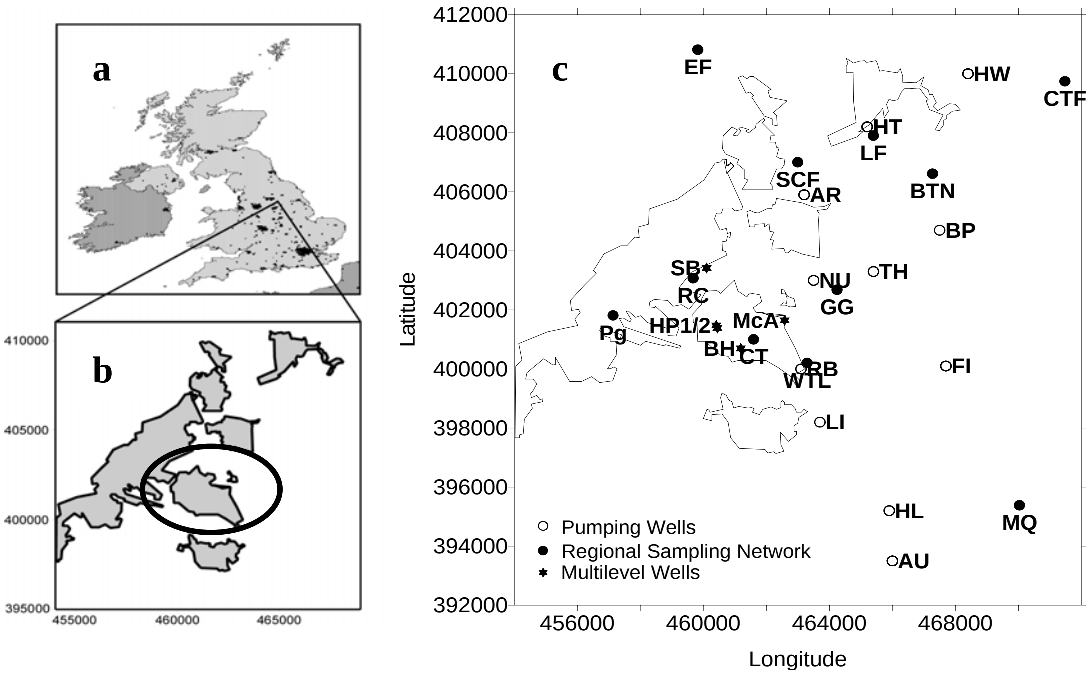


Figure 1. (a) Location of the city of Doncaster within the UK. (b) detailed map showing the urban area of Doncaster in relation to OS grid references, with the study area Bessacarr-Cantley indicated with a black ellipsis. (c) map showing the locations of the production wells of the water supply well field (open circles), the regional sampling network (full circles) and the multilevel wells (stars). Again the co-ordinate system is the UK Ordnance Survey system.

The 5 multilevel piezometers were all drilled using air-flush technique and installed in September 2003 at sites chosen to detect contamination originating from the older parts of Doncaster centre and the suburb of Bessacarr-Cantley, which was constructed mainly post-1945. Planned geophysical logging had to be cancelled because the open holes were found to be unstable due to the poorly cemented nature of much of the uppermost 30m or so. In fact, the first hole drilled at Sandall Beat collapsed at a depth of about 16 mbgl and the multilevel piezometer had to be installed by telescoping through temporary casing. Between drilling and installation, short pumping tests (3-10 hours) were performed and recovery data analysed to obtain a better idea of the average hydraulic conductivity (Table 2).

Table 2. Technical details of multilevel wells.

Name	Drilled depth [mbgl]	Head level [maOD]	Easting	Northing	Hydraulic Conductivity [m/day]	No. of ports
Sandall Beat	36	8.491	460080	403457	-	5
Bolton Hill	51	15.153	461230	400704	8.2	7
Haslam Park 1	60	11.092	460455	401392	1.0-1.7	7
Haslam Park 2	60	10.75	460400	401465	1.5-2.7	7
McAuley School	61	9.621	462597	401786	3.5	7

Installation was conducted immediately after the pumping test (Figure 2a). The larger size pipes are made of PVC and the smaller ones are of HDPE, tied to the centre pipe. Different sizes were used for the top and bottom-most level to enable online monitoring of water levels, temperature and electrical conductivity. The end of each pipe was sealed with a cap and a 30cm screen was constructed 20cm above the bottom of the pipe (Figure 2b).

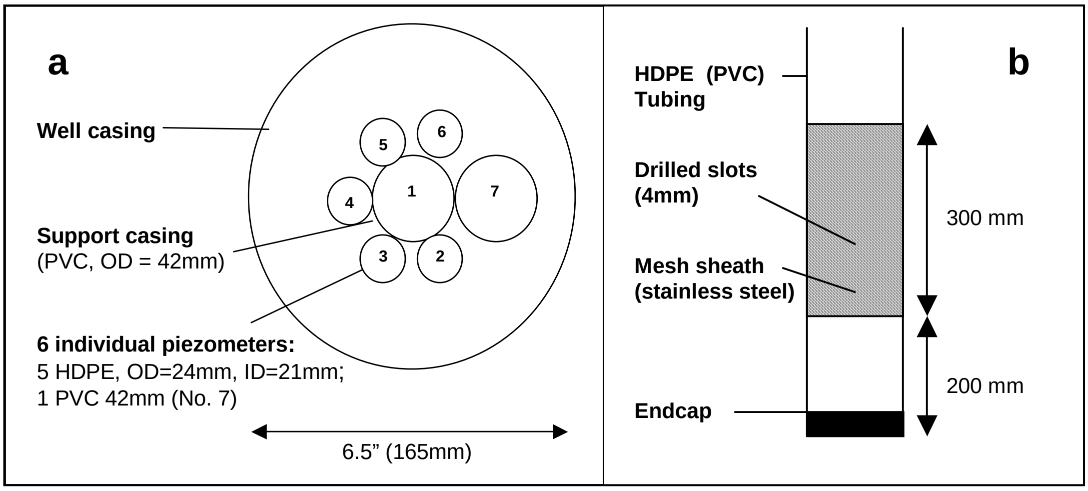


Figure 2. (a) Arrangement of plastic pipes inside the open borehole. b) details of screen section showing lowest part of a HDPE (or PVC) pipe with intake section comprising holes drilled into the pipe and covered with a stainless steal mesh to prevent sand from entering the pipe during sampling.

The holes were refilled with coarse sand (average diameter of 1mm) around the screen and the different levels were hydraulically separated with bentonite clay plugs of 1-3 m. thickness. Before the sampling was commenced the wells were developed to remove potential contamination from the wells by introducing a 50mg/L sodium hypochlorite solution. After leaving it for a several minutes 3-5 purge volumes were pumped out until the electrical conductivity stabilized.

HYDROGEOLOGY

The major bedrock geological formations in the study area of Doncaster are the Mercia Mudstone and the Sherwood Sandstone aquifer (both of Triassic age) and underlying Permian strata of marls and limestones. The outcrops of the formations are shown in Figure 3.

The Sherwood Sandstone Group (formally the Bunter Sandstone) comprises a thick sequence of red, brown and more rarely greenish-grey sandstones of fine to medium grain size with thin layers or lenses of red mudstone and siltstone. Quaternary superficial deposits ranging from glacial sand-and-gravel to peat and lacustrine silty clays overlie the sandstones in many places and these can exert a major control on recharge processes, flow patterns and solute/contaminant transport [5]. The stratigraphy of the bedrock sequence dips consistently to the east at between 1-3°. This uniform dip is the result of tectonic activity in the late Jurassic period. After the tectonic phases, subaerial and submarine erosion exposed the lower stratigraphies resulting in a north to south outcrop orientation [6,7]. Large-scale and extensive faulting of the geological units, including the Sherwood Sandstone sequence, is illustrated in Figure 3 (top map).

Groundwater flow occurs from west to east following the gradients induced by the water supply wells located in an arc to the east of the urban area (Figure 1). Bulk hydraulic conductivities are between 2 and 11 m/day, typically ranging between 3 and 5 m/day [7]. Those for deeper wells tend to be slightly smaller than those for shallower wells. Using an average horizontal hydraulic gradient of about 0.0015-0.005 typical horizontal flow velocities are c. 2-30 m/year. Vertical hydraulic gradients are observed to be 0-0.0015 remaining relatively constant throughout the year.

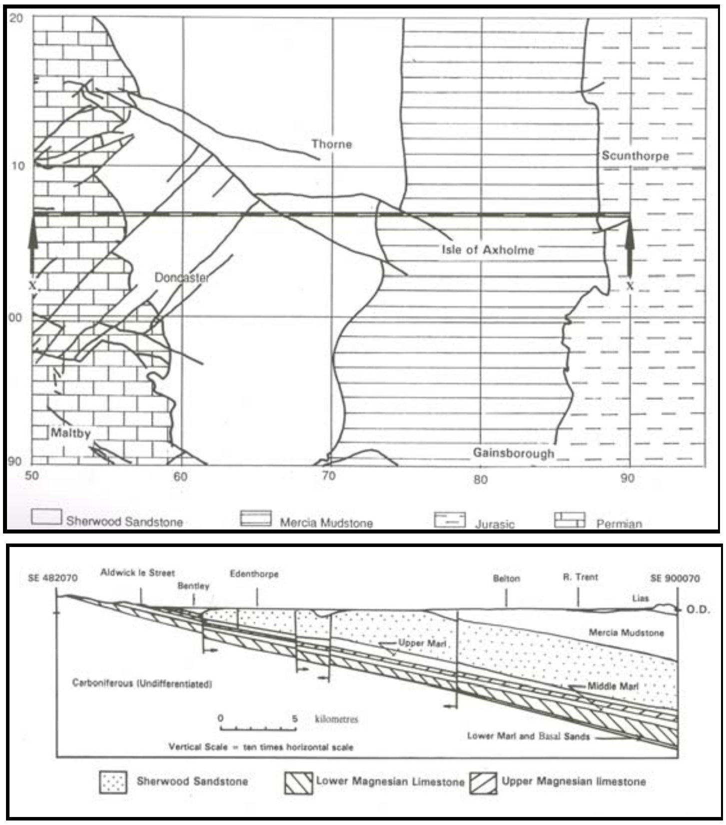


Figure 3. The top map shows the major geology of the area around Doncaster. The lower figure shows a cross section as indicated in the upper map (a vertical exaggeration is applied to show the layers more clearly) from [7].

EXISTING HISTORICAL DATA

Historical data were provided from the Environment Agency of England and Wales (EA). Between 1974 and 1993 groundwater was routinely monitored at several wells, all located in the rural area down-gradient of Doncaster. Some of these boreholes are used as drinking water supply wells and not only have a wide total depth range but also highly variable screen intervals, with uppermost screens starting from as little as 11 mbgl to >60 mbgl. Therefore, potential contamination occurring at shallow depths may be diluted and, therefore, would not be detected by surveillance at these wells. The resulting spatial distribution is complicated casting doubts about how representative these monitoring wells are of the aquifer at large.

However, the wells located in the south of the project area (Figure 1) showed consistent temporal trends in major and minor hydrochemistry. These trends are likely to be anthropogenically influenced but it cannot be resolved whether the trends are a consequence of urban contamination or merely a consequence of changed flow regime due to the constant long-term pumping from lower levels of the aquifer to provide the city’s drinking water.

SAMPLING CAMPAIGNS

The regional sampling campaigns were carried out in July and November 2003, February 2004, May 2004 and September 2004. Multilevels were sampled at the same dates apart from July 2003. To ensure a dataset that covers an entire year an additional sampling campaign was undertaken in November 2004 where selected wells were sampled only for field parameters (ph, Eh, temperature, etc.) and micro-organisms.

Electrical conductivity, temperature, pH, redox potential and alkalinity were measured onsite. Major and minor ion chemical parameters were analysed by the British Geological Survey (BGS). Thermotolerant coliforms (TTC), faecal streptococci (FS) and sulphite reducing clostridia (SRC) were isolated from 100 mL sample volumes using membrane filtration. The results from all analyses were recorded as colony forming units (cfu) per 100 mL. Coliphage samples were preserved with 1ml of chloroform to prevent phage re-growth. Enumeration of coliphage was determined by assay of 1 mL of sample using a double agar layer technique.

DATA INTERPOLATION AND CORRELATION

In order to compare regional wells, with their varied screen depths, and multilevel wells the observations at the multilevel wells located in the depth-range of the regional wells were averaged. This procedure assumes that all levels contribute equally to the average concentration observed in the fully-screened regional well. If the screen of a regional well reaches shallower or deeper levels than the depth-range of the corresponding multilevel well the shallowest or, respectively, the deepest levels were used to extrapolate concentrations and cover the unknown depth intervals. The averaging procedure does not account for biases introduced through velocity profiles because the depth profiles of hydraulic conductivities are unknown. In fact, in a fully penetrating borehole, the observed concentrations are dominated by layers with higher hydraulic conductivities, whereas poorly conductive layers contribute much less water to the sampling interval [8].

The agreement between major hydrochemical parameters of the regional samples and the corresponding averages of the multilevels was assessed using a χ^2–test (chi). The hypothesis is that the averaged multilevel concentration is equal to the observation at the respective regional well. The degree of freedom of the χ^2 test was determined by applying a correlation analysis of all involved parameters. I was found that sodium and chloride as well as calcium, magnesium, bicarbonate and sulphate correlate significantly leading to four independent parameters. A χ^2 of less than 7.82 fulfils the hypothesis stating that the deviations are small enough to be by chance only.

Results

As the number of samples taken and analysed during the project are rather large the samples are not listed in this paper. However, the data can be found in Morris et al 2005.

MAJOR CHEMISTRY

The major chemistry parameters were found to show extensive depth-stratification (Figure 4). This stratification is due to both natural processes (e.g. calcite dissolution) [9] and anthropogenic influences [10] (Figure 4). Groundwater contamination by major ions (e.g. nitrate or potassium) is of a dispersed nature via leaking sewers, road salting, fertilizer application etc. [11]. Wastewater contamination indicators of sewer leakage were consistently found to depths of 20 to 30 mbgl with the sewer influence generally decreasing with depth. Surprisingly, the urban contamination signal was elevated again at depths of about 60 mbgl in some of the wells, suggesting more rapid flow paths via fracture systems [12]. Figure 4 shows vertical profiles from all five multilevel wells compared to the average concentrations observed in the regional wells. Averages measured at the nearby regional wells compare reasonably well with the concentrations observed at the corresponding depths of the multilevel wells. The vertical decrease in alkalinity with depth can also be observed in the regional wells, where wells are covering deeper levels tend to show lower concentration.

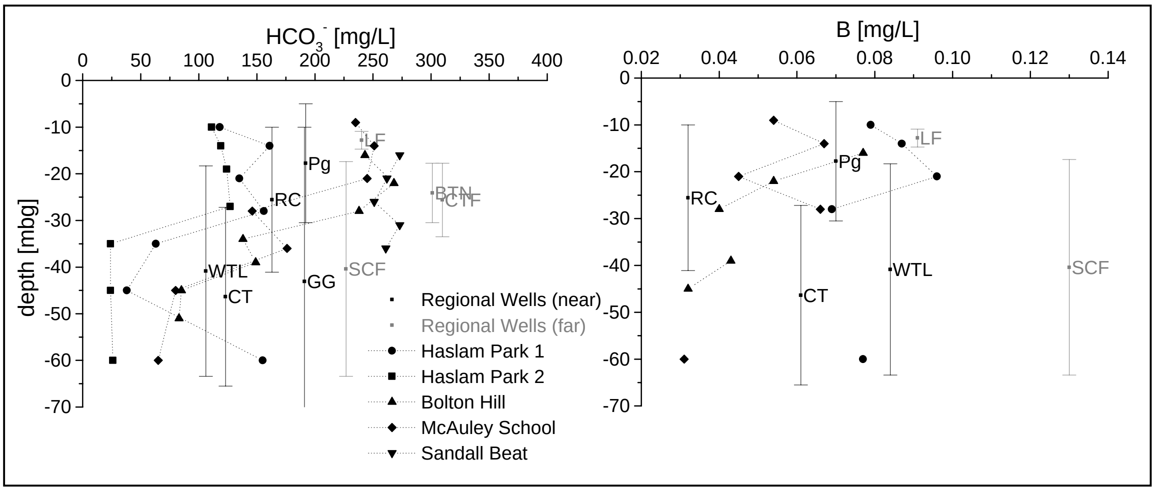


Figure 4. Vertical profiles of total dissolved nitrogen (TDN-N) and alkalinity of multilevel wells compared to nearby (in black) and more distant (in grey) regional wells.

To compare the calculated and the measured concentrations of each regional well, a χ^2-test, was applied and includes all major hydrochemistry parameters. Results are listed in Table 3. The table shows that most nearby wells agree with the hypothesis ($\chi^2<3.84$), i.e. the estimated values using the multilevel results agree reasonably with the observed concentrations at the regional well. As expected, the χ^2-values increase with increasing distance between the wells. Four of the eleven samples strongly disagree with the hypothesis, namely RC, BTN, CTF and EF. RC was found to be very clean low mineralised water, most probably infiltrating through the grassland of the racecourse area. BTN, CTF and EF all originate from the areas with reducing redox conditions and are highly mineralised groundwaters. In fact, the low correlation is a clear indication that the wells further away from the multilevels undergo a different hydrochemical evolution (e.g. water-rock interaction, different susceptibility due to Quaternary deposits, etc.).

Table 3. Results of χ^2-test between regional wells and averaged multilevel wells for all observed wells together with the distances between the regional wells and the particular multilevel.

	HP 1		**HP 2**		**BH**		**McA**		**SB**	
	χ^2	**km**	χ^2	**km**	χ^2	**km**	χ^2	**km**	χ^2	**km**
Pg	2.2	3.34	3.0	3.28	2.7	4.24	2.7	5.46	3.3	3.37
RC	81.3	1.85	94.5	1.76	65.9	2.83	79.4	3.19	107.3	0.56
GWG	4.3	4.01	4.8	4.04	5.5	3.61	3.6	1.88	2.6	4.24
CT	2.5	1.31	3.9	1.38	2.0	0.56	2.3	1.19	3.7	2.94
WTL	2.4	3.08	3.5	3.16	1.7	2.13	2.6	1.73	3.7	4.58
SCF	5.5	6.11	6.0	6.07	4.6	6.49	5.1	5.18	4.7	4.55
BTN	13.7	8.59	14.3	8.59	12.0	8.45	12.2	6.72	15.7	7.86
CTF	13.7	13.83	14.9	13.83	8.6	13.66	11.4	11.92	14.6	13.02
LF	4.0	8.17	4.3	8.15	2.7	8.32	5.1	6.73	4.4	6.93
MQ	3.9	11.31	4.9	11.40	2.7	10.29	3.8	9.82	3.0	12.82
EF	17.1	9.44	15.4	9.36	11.7	10.20	16.3	9.44	13.3	7.36

The results show, except for the race course well, good agreement between the major hydrochemistry of both multilevel and nearby fully screened wells. Discrepancies between them are mainly random, due to land use differences and small-scale heterogeneities in geology and recharge behaviour, possibly compounded by regional trends. As contamination of these parameters is common on quite large spatial scales, vertical trends can be observed with both types of wells even though multilevel wells are certainly more sensitive to both temporal and depth changes. If contamination with major hydrochemical parameters takes place over a small vertical extent it may well miss the dedicated screen sections of multilevel intervals but it would always be detected by a fully screened well though significantly diluted. This implies that, apart from resolving vertical concentration profiles, the major ion hydrochemistry can be sampled representatively using both well types. Open wells are most suitable if drinking water quality compliance is the focus of the groundwater monitoring while multilevel piezometers are more suited to differentiating natural and anthropogenically induced processes and changes in hydrochemistry with depth

MINOR CHEMISTRY

The minor hydrochemical parameters originate, as the major ions, from both natural and human origin. However, their measured concentrations are at times very close to or below the detection limit. This is particularly true for anthropogenic contaminants such as boron or heavy metals, two examples plotted in Figure 6 to show the importance of the detection limit. Good agreement exists between Bolton Hill and Cantley Tower wells (Figure 6/a) with all parameters measured above the detection limit. The second example, (Figure 6/b) compares the McAuley multilevel with the GG well. It was found that the observed Mn and Cr concentrations are far higher in the regional well. This could be because of a localised contamination source that cannot be seen at the nearby McA multilevel. Furthermore, there were three parameters (B, P, Al) found to be below the detection limit at the GG well. However, these three parameters were consistently detected in some of the levels of the McAuley multilevel.

The two examples demonstrate that fully screened monitoring wells detect minor chemical compounds even if a small vertical extent of the aquifer may be contributing this parameter to the overall well as long as the combined final water has a high enough concentration to be detected. Otherwise, the dilution of contaminated inflow levels, potentially detected with multilevel wells, with water from clean levels may lead to mixtures with concentrations below the analytical detection limit even though the contaminant(s) arrived at the well. Potentially, contaminants originating from point sources (e.g. landfills) moving in vertically well-defined plumes could arrive at a fully-screened monitoring well long before the mixed water signal would rise above detection limits because the signal would be diluted by the large proportion of clean water above and below the plume. A multilevel interval, on the other hand, provides less dilution on each level and therefore would detect contamination quicker – provided the contamination plume does not pass between two separate sampling intervals.

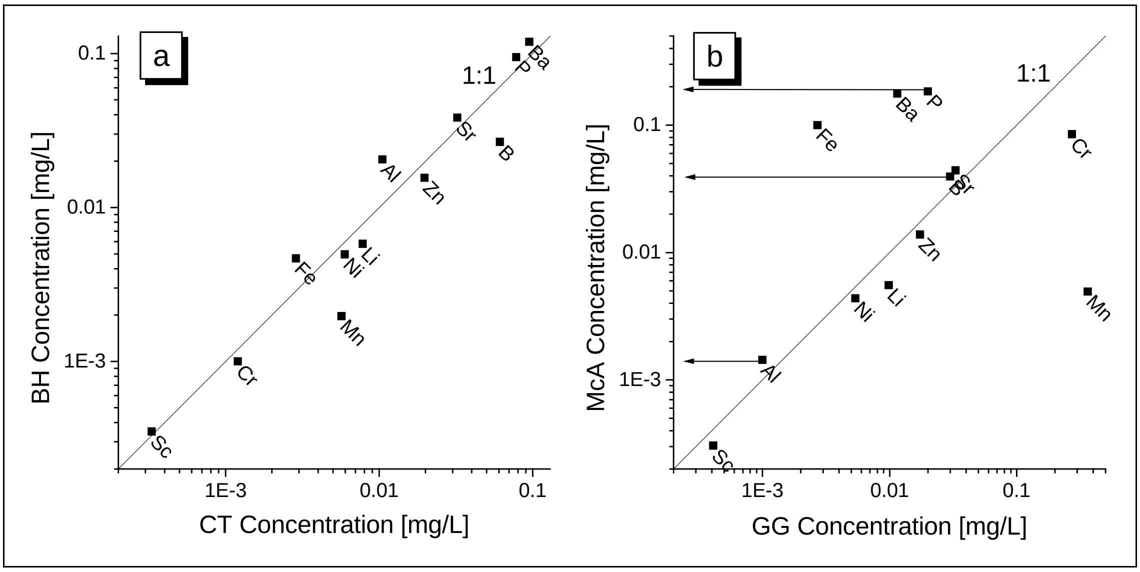


Figure 5. Comparison of minor chemistry composition. a) between CT regional well and Bolton Hill multilevel and b) between GG regional well and McAuley School multilevel.

MICROBIAL CONTAMINATION

Culturing of microbial parameters (here faecal indicators) implies a minimum detection limit of one complete viable and culturable organism. The detection limit can only be reduced with increasing the sample volume or with repeating measurements, both effectively increasing the probability of detection by filtering more water. This is a very important difference compared with the minor hydrochemistry parameters. If a hydrochemistry sample found to be below the detection limit is re-measured the result will, most likely, still be below the detection limit, no matter how often the same volume of sample is measured.

As the faecal indicator organisms measured during this project most likely originate from leaking sewers located underneath the urban areas and due to the fact that most micro-organisms are subject to attenuation processes such as inactivation and adsorption to soils then lower levels of microbial contamination are expected in the regional wells, located outside the urban areas, compared to the multilevel wells, located inside the urban areas. Typical levels of faecal indicator contamination found in this project were between zero and five colony-forming units /100ml and the overall percentages of positive detects in the multilevels are, as expected, slightly higher than the ones in the regional wells. However, as the detected numbers are rather close to the detection limit of one colony per plate comparisons between different wells and well types at specific sampling dates are poor. Increasing the sample volumes would most likely improve this picture.

Overall, both well types should provide similarly representative information. However, taking standard size samples, the multilevel wells are much more sensitive as samples are less diluted. This could be overcome by increasing the sampling volumes of the fully screened wells by a factor of ten to a hundred.

Conclusions

This paper compared conventional fully screened wells with bundled multilevel piezometers to monitor urban groundwater quality. Differences between the two types of monitoring wells were discussed for three types of contaminants, major hydrochemical parameters, minor hydrochemistry or trace substances and microbial parameters. Summarising, there were a number of advantages and disadvantages found for both well types:

1. The major advantage of fully screened wells is certainly the fact that they are usually drilled to install a pump and supply water and monitoring is only a secondary usage. Multilevel piezometers, on the other hand, are installed specifically for monitoring reasons.
2. There are a number of depth-specific samples coming from a multilevel well but only one sample from a fully screened well. However, in a long-term monitoring program this extra sampling effort could be reduced by selecting the most sensitive level(s) based on results of surveillance.
3. Multilevel wells really help to understand the three-dimensional movement of water and contaminants as they enable the vertical resolution of groundwater quality (or groundwater contamination). Depth-related quality information makes it possible to predict water quality deterioration trends, choose options for mitigation (e.g. re-siting of abstraction wells or deeper screen intervals) and assess effectiveness of protection strategies (landuse change, control of pollutant flux by changes in agricultural practice, etc.).
4. In most cases, the agreement between the two types of wells was good, particularly for natural substances and dispersed pollutants. However, multilevel wells have proved to be much more sensitive to changes in water quality because they do not dilute a contaminant plume. The problem becomes particularly apparent for trace substances being introduced (e.g. from landfills). In this case, even though the fully-screened well is affected by the contamination, it would not be detected until the diluted average level in the well rises above the detection limit. A suitably designed and constructed multilevel should detect contamination more readily as the dilution is much smaller. This is particularly important for trace substances posing a high risk to human health even in very small concentrations.
5. If micro-organisms (indicators or pathogens) are found to be close to the detection limit of one count per plate considerable discrepancies can occur between multilevels and fully screened wells, particularly if temporal sampling is not considered. However, this implies that the conventional detection limit is too high or, respectively, the sample volume is too small. Decreasing the detection limit by filtering more water used for samples from fully-screened wells therefore directly increases their lower sensitivity due to dilution.

In summary, for low risk contamination fully-screened wells (if the screen intakes are reasonably positioned towards the potential contamination source) are more economical than multilevel piezometers. Multilevel monitoring wells, on the other hand, increase the understanding of the groundwater system as they can better resolve vertical groundwater quality profiles. If trace substances are released to the groundwater that pose a high risk to human health multilevel wells are certainly the better choice because dilution of a sample with clean water from different levels is small, making them much more sensitive than fully-screened wells. Subsequently, they can detect potential contamination far quicker and therefore enable a quicker response in finding solutions to the problem.

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2.3.2 Paper 2: Daily patterns of micro-organisms in the foul sewer system of Doncaster, United Kingdom (Rueedi *et al.*)

Key findings

- Sewage measurements provided useful information about volumes and loads of potential contaminants, particularly for sewage derived micro-organisms.
- Their daily and/or seasonal variations were found to vary over several orders of magnitude. This has to be considered when using them to quantitatively assess the influence of sewage on microbial groundwater quality.
- Microbial indicators demonstrate a strong daily pattern dominated by toilet use. Therefore, they are useful to assess the influence of toilet use as well as other water use types (greywater: bathroom, laundry, kitchen) on the average pathogen load released from a household. This information will be needed to assess hygiene issues when considering alternative water use strategies such as grey water recycling.

Introduction

The continued expansion of urban areas and increasing cost of replacement of old sewage networks have intensified the discussion about how alternative urban water management strategies can foster sustainable development (Clarke *et al.*, 1997; Grottker and Otterpohl, 1996; Otterpohl *et al.*, 1997) and their impact on contamination of soil and water through leaking sewer systems (Eiswirth and Hoetzl, 1994; Lerner *et al.*, 1994). Assessing the contamination potential of sewer leaks and deciding on the optimal future water management strategy require more scrutiny of contaminant sources and pathways through the urban system (Almeida *et al.*, 1999). The overall scope of the EU 5th FP project AISUWRS (Assessing and Improving the Sustainability of Urban Water Resources and Systems) is to assess the urban water resources, systems and flows within four study cities in Europe (Rastatt, Germany; Ljubljana, Slovenia; and Doncaster, UK) and Australia (Mt. Gambier) (see www.urbanwater.de). This project is developing and applying a series of computer models to analyse the urban water systems and their effect on underlying groundwater.

Faecal contamination can reach groundwater through many routes. Point sources, such as septic systems or leaking sewer networks are of primary concern because they release concentrated pathogen loads into the subsurface. Contamination of groundwater with faecal pathogens is known from many studies but only few are reported (summarized in Macler and Merkle, 2000). Pathogenic viruses were thought more likely to be found in karstic areas but recent research has shown that pathogenic viruses and indicator microorganisms can be found in consolidated sediments too (Abbaszadegan *et al.*, 1998; Powell *et al.*, 2003). These cases show the potential of micro-organisms to contaminate groundwater and the still limited understanding of transport pathways and processes. Most importantly, reliable estimations of pathogen levels, variabilities and diurnal patterns in sewage systems or septic tanks are poorly known even though these are the input values and therefore the starting point of all attempts to assess transport and fate of pathogens in the subsurface (Ginn *et al.*, 2002).

Sewage volumes and contaminant fluxes from households are usually highly variable between different countries and even between households (Almeida *et al.*, 1999). For human pathogens, the variability was found to be particularly large, varying over orders of magnitude (Eriksson *et al.*, 2002). Seasonality of faecal contaminants in sewage systems is poorly known, but a seasonal variation was found for gastroenteritis outbreaks caused by norovirus (Mounts *et al.*, 2000). This pattern should be directly represented in sewage. Sewage flow and chemical quality varies diurnally, following the different distribution of water usage among different sources (toilet, bathroom, kitchen, utility room, etc.) (Almeida *et al.*, 1999) but little is known about the daily pattern of faecal indicator organisms or pathogens. Strong daily fluctuations would at least partly explain the high variability of observations taken as spot samples on different days and at different times of the day. This paper presents new results from the urban water-monitoring program in the city of Doncaster, UK.

Methods

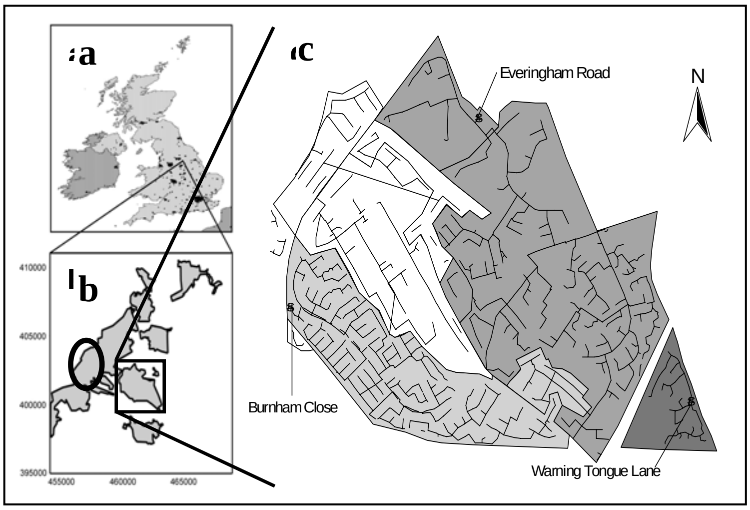

Figure 1. (a) Location of Doncaster. (b) Urban area of Doncaster with the old town centre indicated by the black circle. (c) Detailed map of focus study area of Bessacarr-Cantley. Black lines indicate the foul sewer system and black dots indicate sewer sampling points.

The study city of Doncaster is located in the heart of the England (a and b). Due to the large size of the town a single suburb, Bessacarr-Cantley, was selected as the focus study area.

Three sewer pumping stations/inspection chambers (c) were spot-sampled quarter-annually to observe potential seasonal trends. An intensive one-day monitoring program was run to reveal the daily patterns of sewage concentrations and volumes at two of the sites. All three sites are sewage outfalls from areas with a fully separated sewer and stormwater system.

Table 1 contains some domestic characteristics for the catchments of the three monitoring points. Land use maps and population statistics were obtained from Doncaster Metropolitan Borough Council. Water supply data and sewage network information were obtained from Yorkshire Water. The raw data had to be adapted and split among the different catchments. Average occupancy was calculated to be 2.52 persons per household leading to an average per person water usage of about 180 l/person/day (including water mains leakage).

Table 1. Sampling point catchment key statistics

Catchment	**Area [ha]**	**Households [-]**	**Imported Water [m^3/year]**	**Water usage [l/person/day]**
Everingham Road (EVR)	296.5	4383	718,283	178.2
Burnham Close	132.2	2176	363,540	181.6
Warning Tongue Lane (WTL)	28.6	442	76,019	187.0

During the one-day survey, conductivity, temperature and pH were sampled on-site. Thermotolerant coliforms (TTC), faecal streptococci (FS) and sulphite reducing clostridia (SRC) were isolated from 100 mL sample volumes using membrane filtration. The results from all analyses were recorded as colony forming units (cfu) per 100 mL. Coliphage samples were preserved with 1ml of chloroform to prevent phage re-growth. Enumeration of coliphage was determined by assay of 1 mL of sample using a double agar layer technique (Adams, 1959). 1 litre samples were taken for enteroviruses, where each sample was analysed by PCR and plaque assay. More details about sample analysis can be found in Cronin *et al.*, (2005). An automated data logger was installed in the Warning Tongue Lane pumping station chamber to continuously record water levels, temperature and electrical conductivity. Water level records were used to estimate sewage flows using the geometric dimensions of the holding chamber and assuming that the installed pump can only pump with one dedicated pumping rate (only switching on and off as required).

SOURCES OF SEWAGE AND PATHOGEN LOADS

Household water usage can be split into different sources with different contaminant loads. However, the Urban Volume and Quality (UVQ) model applied in the AISUWRS project to assess urban water and contaminant fluxes splits household water supply among four usage types namely toilet, bathroom, kitchen and laundry (Mitchell *et al.*, 2003). Table 2 lists ranges of household water usage as documented in the literature. A discussion on the uncertainties in these values can be found in Friedler and Butler (1996). The table shows that the grey water volume (bath, kitchen and laundry) accounts for approximately 60-75% of the total water usage. US EPA (1980) seems to overestimate toilet volumes. This could be due to increased awareness and reduced flush-devices that have subsequently led to large improvements of toilet water saving over the past 20 years.

Table 2. Household water use [% of total volume] as documented in literature.

Reference	Toilet	Bath	Kitchen	Laundry
US EPA, 1980	40.7	23.1	10.6	25.2
Almeida et al., 1999	30.8	40	13	16.2
ABS, 2004	26.8	35.7	23.2	14.3
Heaney et al., 2002	30.2	20.5	19.9	25.9

The daily pattern of water use is central to this study as the sampling campaign measured sewage over one typical weekday. Not only the total amount of each separate usage type has to be considered but also its occurrence and magnitude during the day. Unfortunately, there is little known about average laundry or dishwashing times. However, (Friedler *et al.*, 1996) studied the daily pattern of toilet usage in the UK on weekdays and weekends, respectively. As expected, the fluxes are highest between 6 and 9 am during weekdays and between 8 and 11 am on weekends. As micro-organism loads are expected to be mainly linked with toilet flushing, this information will be used as best approximation of total toilet flow.

Table 3 contains typical ranges of the bacteria and viruses in raw sewage. Enterovirus numbers in raw sewage are much lower than bacterial counts or coliphage and they were not detected for all sampling campaigns, While this suggests that virus concentrations vary largely depending on the health status of the population, it should also be remembered that lower viral recoveries are achieved in comparison with bacteria, due to more complicated sampling and culturing processes.

Table 3. Typical numbers of measured bacteria and viruses found in raw sewage.

Micro-organism	No. per 100ml	Reference
Bacteria		
Total Coliforms	$2.1 \cdot 10^4 - 5.2 \cdot 10^8$	Al-Omari and Fayyad, 2003; Kouraa *et al.*, 2002; Rovirosa *et al.*, 2004
Faecal Coliforms	$5 \cdot 10^6 - 1.6 \cdot 10^8$	Tartera *et al.*, 1989; Yates *et al.*, 1987
Faecal Streptococci	$1.1 \cdot 10^6 - 6.4 \cdot 10^7$	Tartera *et al.*, 1989
SRC	$6 \cdot 10^4$ - 10^8	Long and Ashbolt, 1994; Mocé-Llivina *et al.*, 2003; Tartera *et al.*, 1989
Viruses		
Coliphage	10^4-10^8	Mocé-Llivina et al., 2003; Tartera et al., 1989
Enteroviruses	0-$2.4 \cdot 10^3$	Tartera et al., 1989
Polioviruses	$1.8 \cdot 10^3$ -$5 \cdot 10^6$	Yates et al., 1987
Rotaviruses	$4 \cdot 10^3 - 8.5 \cdot 10^5$	Yates et al., 1987

There are a number of publications available and summarized in (Eriksson *et al.*, 2002) where micro-organisms were measured on different grey water sources. The ranges found for micro-organisms are rather large but this is only because some of these references seem to show extremes.

Results

DAILY EVOLUTION OF SEWAGE CONCENTRATIONS

In the following section, measurements are interpreted assuming different scenarios of assumed source-volumes and concentrations and their specific occurrence during the day. Source volumes are plotted in Figure 2F and source loadings are listed in Table 4. The choice of the scenarios is based on the level of agreement between modelled and observed concentrations. Note that the models are fitted manually to meet the observations. This is because many records display large variations complicating an automated fitting procedure.

Two major assumptions were made to constrain the rather complex problem. Firstly, relative contributions from the toilets are assumed to be as documented by Friedler (1996) who measured toilet flush volumes on weekdays and weekends. Secondly the proportion of black and grey water is assumed to be the average of the stated values in Table 2, which is 32% and 68% respectively.

Scenario 1: The first Scenario assumes that sewage is comprised solely of a combination of black and grey water runoff with both having certain average concentrations using concentrations of micro-organisms as listed in Table 4. Figures 2A-F show a poor agreement between modelled and observed concentrations, especially during the morning period. The modelled peak is quite broad, lasting from 6 to 9 am approximately but the observed peaks, indicated with black lines are consistently quite narrow and distinct. As the faecal indicator micro-organisms are linked with faecal matter this disagreement must be linked with a large influx of faeces at that time. This suggests that there is a favoured time for people to defecate between 9 and 11 am – about 1-2 hour after having had breakfast.

Scenario 2: The second scenario, therefore, separates toilet runoff into a favoured faecal usage and an average daily use with mostly urine input rather than faeces (black area in Figure 2F). As we are looking at samples that average the sewage from 1000 and more persons, toilet usage will always represent an average behaviour. All observed indicator micro-organisms originate mostly from faeces rather than from urine and therefore the ratio of the two sources should remain the same for all 5 micro-organisms. The best fitting ratio between average daily toilet use and faeces-dominated use was found to be in the region of 1:10.

Figure 2 shows the results of applying the source concentrations as listed in Table 4 where this scenario is depicted with blue lines. It can be seen that the agreement during the morning hours is much better but the fluctuations observed in the afternoon and evening cannot be easily reproduced. Most indicators show a common pattern of a decrease just after lunchtime with a considerable increase between 3 and 6 pm followed by another drop between 7 and 10 pm. The fact that both decreases are right after common meal times indicates that the source concentrations of kitchen runoff (mainly dishwashing) are considerably lower than all the other sources. Even though some references from direct source measurements seem to disagree with this argument, it is expected that other sources of faecal contamination, besides the toilet, are mainly laundry and bath and shower usage.

Scenario 3: The third scenario thus assumes two major and one minor times where dishwashing is dominant (grey areas in Figure 2F). The resulting evolution of microbe concentrations as shown in Figure 2A to E agree quite well with the observations even though some extremes cannot be resolved yet. Possibly these disagreements can be resolved using measurements of chemical sewage composition.

Principally, the two catchments seem to react very similarly even though the sizes are quite different. Most likely, the larger size of the EVR catchment explains the considerably smaller variance observed for these measurements, i.e. there is a damping effect. Furthermore, WTL is an entirely residential area whereas EVR has schools and a health care centre in its catchment. The hospital possibly explains the higher SRC and coliphage peaks observed in this area.

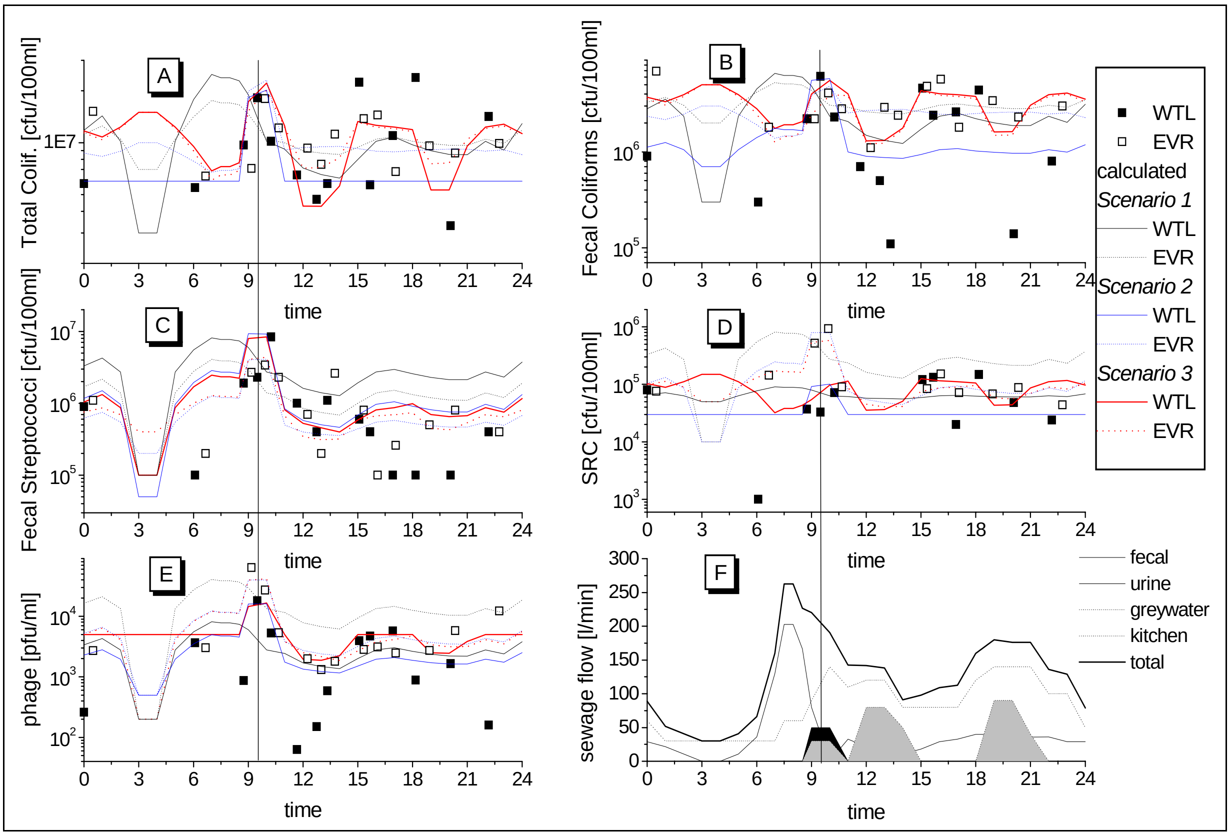

Figure 2 A-E: Results of Scenario 1-3 for both sampling points. Full and open squares indicate measurements from Warning Tongue Lane (WTL) and Everingham Road (EVR). Solid and dotted lines are modelled concentrations for WTL and EVR, using the source concentrations as listed in Table 4. The black line indicates the observed peak of micro-organisms. F: sewage flows assumed to calculate concentrations. Thin black lines indicate black water, the dotted line indicates grey water and the thick black line is the total flow. The black area shows faeces-dominated toilet use and the grey areas show kitchen water use.

Table 4. Best estimations of micro-organism concentration for each scenario and each distinguished water source. The two numbers in each cell refer to sampling point WTL and EVR respectively.

<table>
<tr><th></th><th colspan="2">Black Water</th><th colspan="3">Grey Water</th></tr>
<tr><th></th><th>Feces</th><th>Mixed</th><th>Kitchen</th><th>Laundry</th><th>Bath</th></tr>
<tr><td>Total Coliforms</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Scenario 1</td><td colspan="2">$3\cdot10^{7}$, $2\cdot10^{7}$</td><td colspan="3">$3\cdot10^{6}$, $7\cdot10^{6}$</td></tr>
<tr><td>Scenario 2</td><td>$6\cdot10^{7}$, $6\cdot10^{7}$</td><td>$6\cdot10^{6}$, $6\cdot10^{6}$</td><td colspan="3">$6\cdot10^{6}$, $1\cdot10^{7}$</td></tr>
<tr><td>Scenario 3</td><td>$3\cdot10^{7}$, $3\cdot10^{7}$</td><td>$3\cdot10^{6}$, $3\cdot10^{6}$</td><td>$5\cdot10^{6}$, $8\cdot10^{6}$</td><td colspan="2">$1.5\cdot10^{7}$, $1.5\cdot10^{7}$</td></tr>
<tr><td>Faecal Coliforms</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Scenario 1</td><td colspan="2">$8\cdot10^{6}$, $6\cdot10^{6}$</td><td colspan="3">$3\cdot10^{5}$, $2\cdot10^{6}$</td></tr>
<tr><td>Scenario 2</td><td>$2\cdot10^{7}$, $2\cdot10^{7}$</td><td>$2\cdot10^{6}$, $2\cdot10^{6}$</td><td colspan="3">$7\cdot10^{5}$, $2\cdot10^{6}$</td></tr>
<tr><td>Scenario 3</td><td>$1\cdot10^{7}$, $4\cdot10^{6}$</td><td>$1\cdot10^{6}$, $4\cdot10^{5}$</td><td>$1\cdot10^{5}$, $1\cdot10^{6}$</td><td colspan="2">$5\cdot10^{6}$, $4\cdot10^{6}$</td></tr>
<tr><td>Faecal Streptococci</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Scenario 1</td><td colspan="2">$1\cdot10^{7}$, $5\cdot10^{6}$</td><td colspan="3">$1\cdot10^{5}$, $1\cdot10^{5}$</td></tr>
<tr><td>Scenario 2</td><td>$3.5\cdot10^{7}$, $1.5\cdot10^{7}$</td><td>$3.5\cdot10^{6}$, $1.5\cdot10^{6}$</td><td colspan="3">$5\cdot10^{4}$, $2\cdot10^{5}$</td></tr>
<tr><td>Scenario 3</td><td>$3\cdot10^{7}$, $1\cdot10^{7}$</td><td>$3\cdot10^{6}$, $1\cdot10^{6}$</td><td>$1.5\cdot10^{4}$, $1.5\cdot10^{4}$</td><td colspan="2">$1\cdot10^{5}$, $5\cdot10^{5}$</td></tr>
<tr><td>Sulphite Reducing Clostridia</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Scenario 1</td><td colspan="2">1E5, $1\cdot10^{6}$</td><td colspan="3">$55\cdot10^{4}$, $15\cdot10^{4}$</td></tr>
<tr><td>Scenario 2</td><td>$3\cdot10^{5}$, $3\cdot10^{6}$</td><td>$3\cdot10^{4}$, $3\cdot10^{5}$</td><td colspan="3">$35\cdot10^{4}$, $15\cdot10^{4}$</td></tr>
<tr><td>Scenario 3</td><td>$5.5\cdot10^{4}$, $2\cdot10^{6}$</td><td>$5.5\cdot10^{3}$, $2\cdot10^{5}$</td><td>1000, 1000</td><td colspan="2">$1.5\cdot10^{5}$, $5.5\cdot10^{4}$</td></tr>
<tr><td>Coliphage</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Scenario 1</td><td colspan="2">$1\cdot10^{4}$, $5\cdot10^{4}$</td><td colspan="3">200, 200</td></tr>
<tr><td>Scenario 2</td><td>$6.5\cdot10^{4}$, $1.5\cdot10^{5}$</td><td>$6.5\cdot10^{3}$, $1.5\cdot10^{4}$</td><td colspan="3">500, 500</td></tr>
<tr><td>Scenario 3</td><td>$5.5\cdot10^{4}$, $1.5\cdot10^{5}$</td><td>$5.5\cdot10^{3}$, $1.5\cdot10^{4}$</td><td>0, 0</td><td colspan="2">4000, 100</td></tr>
</table>

SEASONAL EVOLUTION OF SEWAGE CONCENTRATIONS

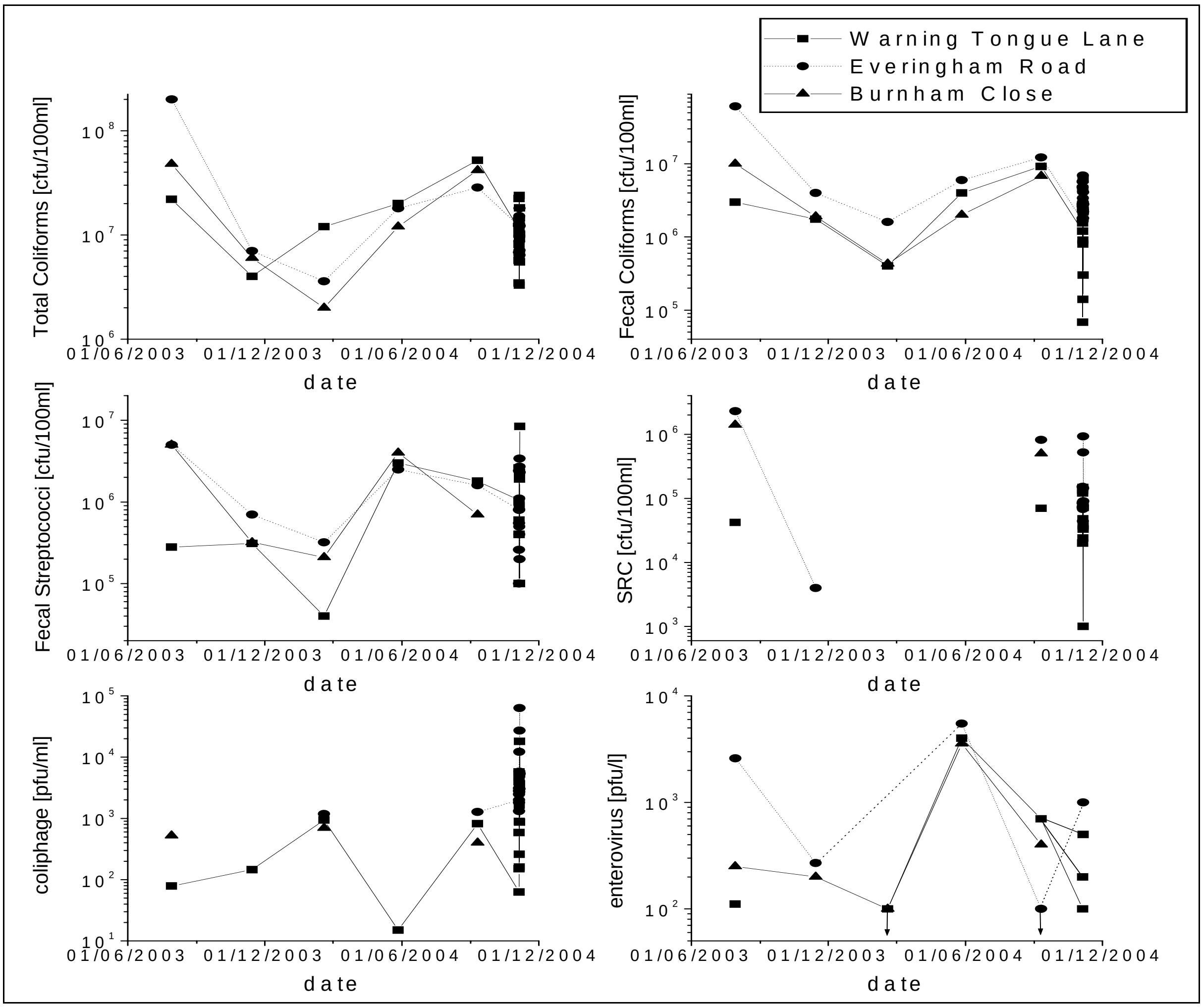

Figure3. Seasonal evolution of pathogens and indicator micro-organisms at the three sampling points.

The concentrations of different pathogens and indicator micro-organisms do not only vary on a daily basis but Figure 3 shows that most of them show significant seasonal variations too. It was mentioned earlier that the samples were not always taken at the same time of the day making the different sampling rounds more difficult to directly compare. Therefore, only approximate estimates of correction factors between average daily concentrations (from observation day) and average annual concentrations can be calculated. Reasonable estimates of annual average concentrations would need higher sampling frequency, both on a seasonal and daily basis. Figure 3 shows that total coliform and faecal coliform counts vary largely during the year, showing a maximum in the summer months and a minimum in winter. A similar pattern can be observed for total enterovirus numbers even though the variations between the three sampling points can be rather large. However, there was no regular behaviour found for the different strains of Coxsackie virus both in time and between sampling points. Again, a higher temporal resolution would probably improve the picture and conclusions could be made on predominance of different strains in sewage. The apparent seasonal pattern for faecal streptococci could well be due to the different sampling times as the daily variations are large. Finally, there is no conclusion possible for either SRC or coliphage because a) the daily variations are larger than the seasonal one and b) some results are reported as minimum counts rather than real numbers.

Conclusions

The paper highlights the importance of assessing the daily (and seasonal) evolution of microbial sewage concentrations because their daily and/or seasonal variations can be over several orders of magnitude. Hence, a reasonable sampling strategy should therefore cover daily, seasonal and spatial variations. Concentration ranges documented in literature are often based on spot samplings, not considering short-term fluctuations. Therefore, numbers are usually linked with high uncertainties, spreading over several orders of magnitude. Applying an adequate sampling strategy gives better confidence levels to actual average concentrations, as required for contaminant mass balance calculations or risk assessments, and ranges can be quoted within much narrower bands. However, a reasonable averaging requires a better understanding of the processes driving the large variations (e.g. water use patterns, loading concentrations, seasonal behaviour). This work shows that using even a simple model to interpret the data can improve the understanding of the impact on loadings and concentrations from different domestic water use patterns and sources (e.g. toilet, kitchen, bathroom/laundry). Faecal indicator micro-organisms are particularly useful because they originate from distinct sources with quite different concentrations and the main source, black water, follows a reasonably well defined pattern. Furthermore, despite their non-conservative behaviour, their commonly huge numbers introduced into the environment make them a very useful tracer to track sewage-derived contamination and assess potential risks to human health.

Acknowledgements

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2.3.3 Paper 3: The effectiveness of selected microbial and chemical indicators to detect sewer leakage impacts on urban groundwater quality (Cronin *et al.*)

Key findings

- Leakage from the sewage system clearly affects microbiological groundwater quality in the Sherwood Sandstone aquifer underlying the study area
- Faecal indicator micro-organisms as well as pathogenic viruses were detected down to depths of 60 meters below ground level (mbgl) in the multilevel wells. This supports findings from previous monitoring of urban groundwater in Nottingham and Birmingham, where urban contamination was also detected to significant depths.
- The high detection rates of sulphite reducing clostridia, faecal streptococci and, to a lesser extent, boron have shown their value as qualitative markers for sewage derived recharge.

Introduction

Sewer and stormwater pipe leakage can have important implications for urban groundwater quality. Serious water quality deterioration can occur with excessive pipe leakage and this has treatment and possibly public health consequences if the groundwater is subsequently used for potable supply (Powell *et al.*, 2003). Leaking sewers have caused public water supply contamination and associated gastric illnesses in Britain and Ireland (Misstear *et al.,* 1996). Water-borne outbreaks of intestinal disease in public supplies are often due to a physical breakdown of the treatment processes used, and in private supplies are typically due to contamination of untreated groundwater supplies with human and/or animal waste (CDSC, 2000). Hence, the quantification of sewer leakage impacts on groundwater is important when analysing a range of urban water supply and disposal scenarios because such scenarios can differ greatly in their handling of contaminants (including pathogens) and in their efficiency to transport them.

The European Union 5FD initiative termed AISUWRS (Assessing and Improving Sustainability of Urban Water Resources and Systems) is addressing this issue of comparing leakage rates across a range of urban water supply and disposal scenarios by producing a Decision Support System underpinned by a linked array of flow and solute transport models (see www.urbanwater.de). In order to evaluate both existing urban water systems and alternative strategies, the sources of contaminants, their flow paths and volumes (*e.g.* recharge from pipe leakage, soakaway disposal) and their sinks need to be identified for different urban settings. Modelling of such volume and contaminant fluxes is being informed by an extensive field sampling campaign of groundwater (depth-specific and open boreholes), sewer flow, and stormwater in the case-study cities of Rastatt (Germany), Ljubliana (Slovenia), Mt. Gambier (Australia) and Doncaster (UK). This paper focuses on results from Doncaster (~300,000 residents) which was chosen as a study city because it depends heavily on groundwater drawn from the regionally important Triassic Sherwood Sandstone aquifer from a wellfield located down the hydraulic gradient from the urban area (Figure 1). Though many potential categories of sewage indicator exist (Table 1), cost and expertise favoured the use of selected inorganic chemical and microbiological markers. The concentrations of these indicators were compared between pipe and groundwater concentrations.

Table 1. Potential indicators of sewer leakage influence on groundwater and their associated constraints (modified from Barrett et al, 1999).

Category	Potential Indicator	Constraints
Inorganic major/minor ions	Metabolites (N, Cl), industrial effluents such as arsenic, silver	*Multiple sources*
Organics, pharmaceutical residues	Chlorinated solvents, THMs, BTEX, endocrine disrupters, clofibric acid	Analytical cost, non-specific to sewers
Detergent components	optical brighteners, boron, phosphate, EDTA, d-limonene, sulphur compounds	Rapidly attenuated in subsurface
Sewage-derived microbes	*Bacteria, viruses, protozoa*	*Time-limited, sampling and analytical cost*
Faecal steroids	*Coprostanol*	*Rapidly attenuated in subsurface*
Isotopes and environmental tracers	Nitrogen, Sulphur, CFCs, SF_6	Analytical cost, interpretation of results
Others	Iodated X-ray contrast media, caffeine,	Analytical cost

The most widely used microbial indicators of faecal contamination are thermotolerant coliforms (TTCs) with the majority of this grouping being *E. coli*. TTCs are often used to document the extent of the microbial degradation of urban water quality. However, several authors have expressed reservations about the effectiveness of TTCs as faecal indicators in the subsurface (e.g. Gleeson & Gray, 1997). Viruses, for example, have much lower inactivation rates than bacterial indicators and pose a much more significant public health threat due to their much smaller size that facilitates transport (Collins et al., in press). However, sampling and analysis of viral parameters is significantly more time-consuming and expensive than for bacterial indicators. Hence, a cheap, easy to analyse faecal indicator with a low inactivation rate is needed in order to reliably assess the impact of sewer exfiltration into aquifer systems. Several microbial indictors were sampled and analysed for the Doncaster study to find a simple, but consistently-detected, faecal indicator in urban groundwater. A tentative comparison is made with boron, which is a constituent of some detergents and so has been used in several studies as a chemical indicator of sewage impact on groundwater (e.g. Metzner, 1999). Other potentially useful inorganic indicators were found to be potassium, sodium and bicarbonate and these are proving valuable parameters to distinguish and quantify different recharge sources. However, their interpretation is complicated by other anthropogenic and natural sources and are discussed elsewhere (Rueedi *et al.*, in prep.). This paper therefore focuses on potential indicators for sewage contamination.

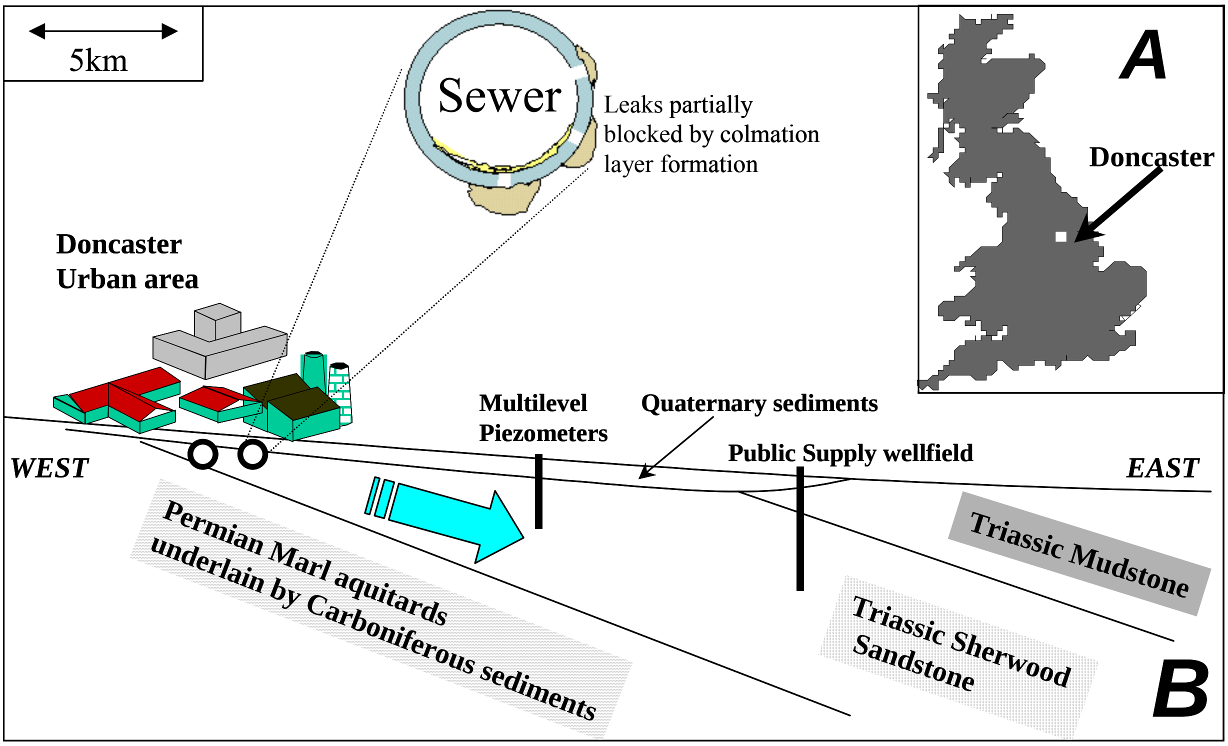

Figure 1. (A) Location of Doncaster in the UK (white dot). (B) Representation of the Doncaster urban area that is underlain by the Triassic Sandstone aquifer. This aquifer contains both the multilevel piezometers and, down-dip, the public supply boreholes. The blue arrow, indicating groundwater flow direction, is towards the wellfield. The expanded view of the sewer pipe shows how exfiltration can occur though the quantity can be reduced by the development of colmation layers. Sewer joints are the other major leakage point.

Methodology

Previous urban groundwater studies in the UK have generally used shallow monitoring piezometers and/or pre-existing boreholes, the depths and construction details of which are often uncertain. Sampling results can, therefore, be masked by mixing of waters from several horizons (Parker *et al.,* 1982). Five bundled multilevel piezometers were installed from depths of 9 m to 60 m below ground level to monitor the depth-specific variation in concentrations of both solute and microbial contaminants in the urban aquifer. The number of discrete sampling intervals was determined by depth and the thickness of clay seal (1.2 to 3m) between intervals. The number of sampling intervals varied between 5 and 7 per borehole. The shallowest and deepest levels contained automated data loggers for monitoring groundwater level, temperature and conductivity and these ports employed 42mm PVC pipes with an internal diameter of 35mm. All other levels used HDPE pipes with an internal diameter of 21 mm (for further details see Rueedi et al., 2004). Groundwater levels varied between ~2.5 and 10 m below ground level. Two shallow hand-augured piezometers were also installed (to 1.30 and 3.15 m depths) to estimate contaminant loading in the unsaturated zone and these are included in the multilevel results below. A selection of privately-owned regional wells was also sampled to improve spatial resolution. Sewers were sampled at three inspection chambers to determine spatial and temporal variations in sewer quality (see Rueedi *et al.*, this volume).

Several indicators of faecal contamination (faecal coliforms, total coliforms, faecal streptococci, sulphite reducing clostridia, coliphage, and enteric virus) were analysed for in Doncaster during sampling campaigns in July and November 2003 and February, May, September and November 2004. Thermotolerant coliforms (TTC), faecal streptococci (FS) and sulphite reducing clostridia (SRC) were isolated from 100 mL sample volumes using membrane filtration and selectively enumerated by culture on membrane lauryl sulphate broth (TTC), Slanetz and Bartley agar (FS) and perfringens agar (SRC) respectively (Anon., 1994). The results from all analyses were recorded as colony forming units (cfu) per 100 mL (membrane filtration). Field blanks and randomly selected duplicates were used as control procedures for all sampling rounds at all sites. All field blanks were found to be free of bacterial analytes. Enumeration of coliphage was determined by assay of 1 mL of sample using a double agar layer technique (Adams, 1959). Two methods were employed for the analysis of enteric viruses (norovirus and enteroviruses) in sample eluates. Buffalo Green Monkey (BGM) kidney cells were used for the quantification of infectious enterovirus by plaque assay, both by the confluent monolayer and suspended cell culture methods (SCA, 1995). Viruses were also analysed using RT-PCR. PCR was used for the analysis of eluates for all other enteric viruses. Replicate 140 µL aliquots of each eluate were assayed with negative and positive controls included in each PCR set. PCR methods. All PCR products (RNA and DNA) were analysed by agarose gel electrophoresis stained with ethidium bromide. Field blanks were used as control procedures for all sampling rounds at all sites. All field blanks were found to be free of viral analytes. Results of coliphage and enteric viruses are given as plaque forming units (pfu) per ml and 10 L respectively while all others are given as colony forming units (cfu)/100 ml. Boron samples were filtered though 0.45 µm and analysed using a Perkin-Elmer Optima 3300DV ICP-OES.

Results and Discussion

Table 2 summarises all results giving the total number of samples collected and analysed for as well as the percentage of positive detects of each parameter in each receptor type. This table shows that sulphite reducing clostridia has the highest number of positive detects for the regional and multilevel groundwater samples; in fact over 40% for both. SRC are anaerobic spore-forming non-motile bacteria exclusively of faecal origin that can survive in water for longer than coliforms or streptococci due to their spore-forming ability (Gleeson and Gray, 1997). Faecal streptococci, common in recreational water monitoring and as a comparison for TTC results, are also detected in 40% of the multilevel analyses. Table 2 indicates high positive detects frequency of faecal indicators in the aquifer. This is surprising from a hydrogeological viewpoint as the Sherwood Sandstone is generally regarded as a high-porosity, slow-moving system in the regional sense. However, positive detects of enteric viruses and faecal indicator bacteria have been previously found in a similar urban setting in the Sherwood Sandstone underlying Nottingham and Birmingham (Powell *et al*., 2003), where they have been explained by a small but rapid flow component transporting sewer-derived leakage to depth (Cronin *et al*., 2003). Approximate die-off rates for the faecal indicators are given in Table 3 along with the maximum and mean of all groundwater (multilevel and shallow piezometer) analyses. The latter show that the groundwater is not grossly contaminated.

Table 2. Faecal indicator sampling results summary. Results are cumulative of all sampling campaigns in Doncaster (July, November 2003; February, May, September, November 2004).

	Depth-specific Multilevel intervals	**Regional wells**	**Sewers**	**~Cost £ Sterling**
N samples	**154**	**45**	**43**	
Field TTC %[1]	18	11	100	<0.3
E. coli %	18	16	100	~10
Total coliforms %	34	24	100	~10
Faecal Strep. %	40	24	100	~15
SRC %	44	47	100	~15
Coliphage %	1	7	100	~25
N samples	**60**	**3**	**17**	
Enteric virus[2] *%*	12	0	100	>100

[1] Analyses of thermotolerant coliforms were undertaken in the field using a portable DelAgua testing kit as well as samples being sent for laboratory filtration and confirmation (shown in the next row named *E. coli*), [2] Combination of results from both methods.

Table 3. Comparison of maximum and mean values of the various marker species outlined in Table 2. All minimum and median values were <1 cfu or pfu/100ml. Results are cumulative of all sampling campaigns in Doncaster (July, November 2003; February, May, September, November 2004). Approx. decay rates are also given.

	Max	**Mean**	***Decay constant (hr^{-1})***	***Half-life (hr)***	***Reference***
E. coli[1]	14	0.4	*0.038* *0.013*	*18.2* *53.3*	*(Yates* et al.*, 1985)* *(Keswick,* et al.*, 1982)*
Faecal coliforms[1]	40	0.9	*0.064*	*10.8*	*(Yates* et al.*, 1985)*
Total coliforms[1]	420	4.9	The huge size of this grouping means huge variability in die-off figures.		
Faecal Strep.[1].	900	17.3	*0.015* *0.0096*	*46.2* *72.2*	*(Yates* et al.*, 1985)* *(Keswick* et al.*, 1982)*
SRC[1]	120	5.6	Very low decay rate; difficult to estimate as spores can remain viable in the subsurface for months to years		
Coliphage[2]	2	0	0.0005		(Collins *et al.* in press)
Enteric virus[3]	2	0	0.01-0.0002 (Coxsackie)	76-3500	(Gordon and Toze, 2003)

[1]units are cfu/100ml; [2]units are pfu/ml, [3]units are pfu/10L

Comparing Tables 2 and 3 shows a correlation between the die-off rates of the micro-organisms and the percentage of positive detects of each micro-organism. Despite low inactivation rates, viruses (coliphage and enteric) have lower positive detection rates due to more complex sampling and culturing procedures than for bacteria. Coliphage and enteric virus transport are also strongly influenced by electrostatic force binding to the aquifer matrix. Table 4 shows the log-reduction in magnitude ranges of indicators at various stages in the urban water cycle and this demonstrates the effectiveness of inactivation, filtration, attachment and dilution mechanisms on sewer exfiltration as it moves into the aquifer. Values in human faeces are also included for comparison purposes (though note the difference in units). Table 4 highlights the order of magnitude differences between the sewer and the groundwater results. The biggest decreases are in organisms with the highest die-off rates, i.e. the coliforms (up to 8 orders of magnitude). When cost, ease of sampling, positive detection rate and order of magnitude difference are taken into account then faecal streptococci and SRC are the two most promising microbial indictors of sewage impact on groundwater quality. The comparison between the various microbial indicators serves to highlight the importance of combining several such indicators in monitoring programmes (e.g. Jagals *et al.,* 1995).

Table 4. Orders of magnitude of indicator bacteria numbers at various stages of urban cycle

	Human faeces[1]	**Sewer sampling (Doncaster) cfu or pfu/100ml**	**Groundwater sampling (Doncaster) cfu or pfu/100ml**	**Orders of magnitude decrease**
Faecal coliforms	10^6 - 10^9	$10^5 – 10^7$	$<10^0$-10^1	4 to 7
Total coliforms	10^7 - 10^9	$10^6 – 10^8$	$<10^0$-10^2	4 to 8
Faecal Strep	10^5 - 10^8	$10^5 – 10^6$	$<10^0$-10^2	3 to 6
SRC	10^3 - 10^{10}	$10^3 – 10^6$	$<10^0$-10^2	1 to 6
Enteric viruses	up to 10^{12}	10^2	$<10^{-2}$	4

[1] Number of faecal indicators commonly found in human faeces expressed as cells per gram of faeces (wet weight) (from Gleeson & Gray, 1997).

It is interesting to compare these two microbial indicators with the boron analysis results. Boron may be present in some detergents in the form of sodium perborate (an oxygen bleach used in the UK for over 50 years) where it can comprise some 5 - 15% of total detergent composition (Barrett et al., 1999). Constraints on the potential use of boron as a marker are the variability of detergent composition (it may not always be present in wastewater) and its use in metal working industries (it may occur as a point-source industrial land contaminant). It is naturally present in low levels in groundwater from clay mineral influences. The median of the regional wells sampled here is 0.05 mg/l (N=30, 8 sites). This is higher than rural Sherwood Sandstone values elsewhere (cited in Edmunds *et al.,* 1989). However, B has proved useful in Doncaster due to the differences between sewer and groundwater results. Typically sewer concentrations were 0.4 mg/L (range 0.1 to 0.6 mg/L for N=29, 3 sites) while the median groundwater concentration for those results above detection limit was 0.06 mg/L (61% of these regional and multilevel samples were above detection limit of 0.02 mg/L). The multilevels boron concentrations suggest enhanced values in the upper 30m of the aquifer. Figure 2 outlines the correlation between both SRC and FS and B. A general increase in both microbial detect values can be seen with increasing B, suggesting a possible link between the microbial and chemical indicators. However the correlation is poor. This could be a result of the varying attenuation processes affecting the two. The B is being diluted, and possibly adsorbed to some extent, in the subsurface accounting for a typical 1 to 2 order of magnitude decrease in concentration from source to detection point. The microbes are particulate and can be physically detained by pore-neck size but also are subject to die-off and this can account for their larger decreases of up to 6 orders of magnitude between sewer and groundwater (Table 4). While the extent of contamination evidenced by both types of marker is not extremely high, the comparison highlights the need for a range of indicator types. This is necessary not only as different indicators have different attenuation and transport processes, as previously noted, but also because the positive microbial detects may be a result of other faecal sources such as animal defecation on park land or road runoff. On balance, B distribution in the urban aquifer supports the microbial results in that it suggests that sewer exfiltration is a likely contributor to recharge.

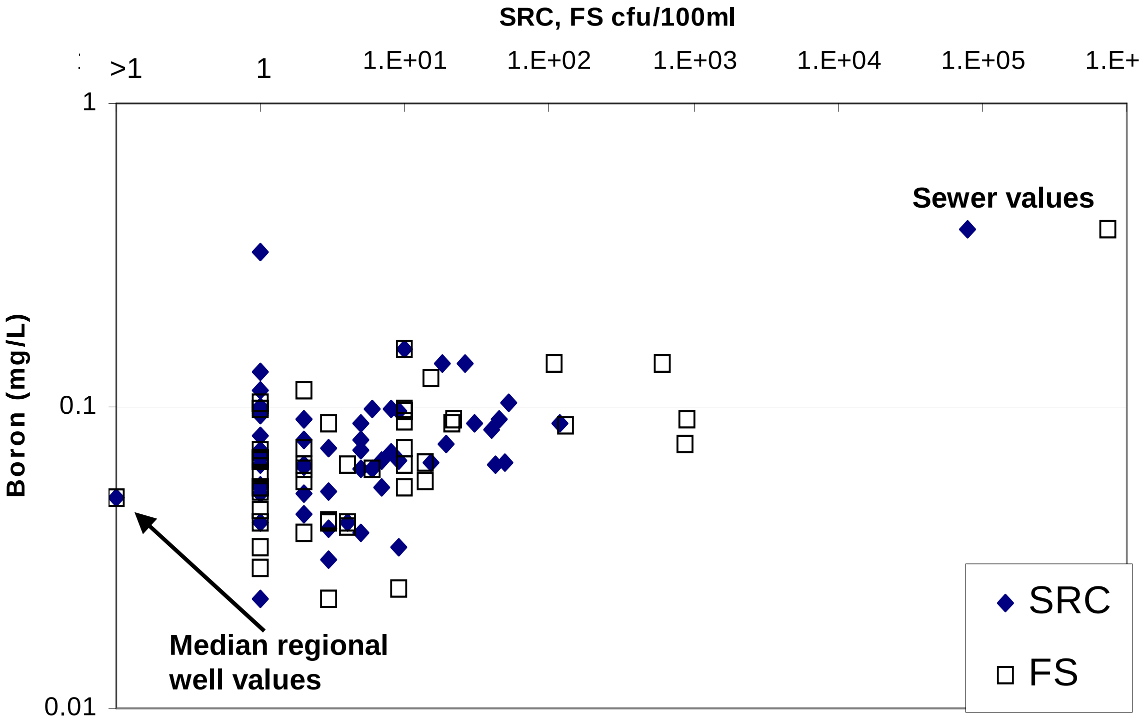

Figure 2. All positive detects of Faecal Streptococci and SRC (given in cfu/100ml) plotted against above limit detects of boron (mg/L).

The qualitative evidence of groundwater degradation by sewer leakage in Doncaster discussed above is currently being transformed to quantitative calculations of leakage rates via mass balance estimates of recharge and the AISUWRS modelling array. This modelling suite consists of an urban mass flux model to split all input urban water components into recharge, runoff or pipe flow, from which a pipeline leakage model estimates exfiltration. Unsaturated and saturated zone models then allow an estimation to be made of the volumes and concentrations of key indicators arriving at the aquifer. Various water management strategies can then be compared via a Decision Support System (e.g. Rueedi and Cronin, 2004).

Conclusions

Sewer pipe leakage is affecting the groundwater quality underneath the city of Doncaster, UK. If improved management structures are to be put in place the full extent of such leakage and the associated impacts must be assessed. To do this a robust set of indicators or marker species is required. Ideal marker species are those which, when present in groundwater, indicate recharge from a specific source. This requires that the species should be uniquely related to one recharge source, and be easily detectable in both the recharge source water and in groundwater. Various common faecal indicators have been assessed here to demonstrate the relative value of each in determining the effects of sewer exfiltration on groundwater quality. The high detection rate of sulphite reducing clostridia, faecal streptococci and, to a lesser extent, boron has shown their value as markers. This has led to the incorporation of these indicators into an urban water modelling suite that aims to assess the flux transfers from the various components of the urban water cycle though the unsaturated zone to the groundwater flow and contaminant transport model.

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2.3.4 Paper 4: The use of $\delta^{13}C_{TDIC}$ as a tracer of groundwater evolution influenced by both natural and anthropogenic processes in four UK Permo-Triassic aquifers (Cronin *et al.*)

Key findings

- Stable isotopic ratios of total dissolved inorganic carbon ($\delta^{13}C_{TDIC}$) in groundwater, pore water and aquifer matrix have proved to be a useful tool when trying to distinguish between natural and anthropogenic processes affecting the carbon system in urban groundwater. Urban recharge from leaking sewage pipelines was clearly able to affect the carbon system of shallow groundwater and was discernible from uncontaminated deep groundwater.
- The carbon signature of sewage, with its lower pH, higher TDIC and similar $\delta^{13}C_{TDIC}$ is clearly distinguishable from that for local uncontaminated groundwater when all three parameters are considered in combination.
- Pore water samples taken during the drilling works have shown that only about 50% of the pore volume is involved in the actual flow process represented by groundwater samples. The balance is non-mobile water mainly acting as a retarding influence. Exchange between the two systems is assumed to be purely diffusive, making it very slow.
- Stable isotopic carbon evolution trends relationships were useful as an indicator of the affects of natural and anthropogenic influences on groundwater geochemical evolution. However the qualitative nature of this parameter means other age dating techniques need to be employed in conjunction with $\delta^{13}C_{TDIC}$ to fully understand recharge processes in urban aquifers.

Introduction

The Sherwood Sandstone and the Chalk are the UK's two most important aquifer types. Fluvially-deposited, red-bed sandstones of Permo-Triassic age form regionally important aquifers throughout western Europe. In the United Kingdom, the tectonic history of this sandstone formation has resulted in a series of separate outcrops and aquifer systems. The formation generally comprises fine to coarse-grained sandstones in which both matrix and fracture flow occurs. Groundwater flowing in these sandstone aquifers has played an important role in the development of several major UK cities such as Liverpool, Birmingham, Nottingham, Doncaster and Coventry, supplying water for industrial and potable use for more than 130 years. In Northern Ireland, the Sherwood Sandstone is the principal aquifer in the Province as it underlies the highly populated and industrialised region around Belfast (Manning *et al.,* 1970; Kalin & Roberts, 1997). Recharge sources and pathways in urban sandstone aquifers remain poorly resolved and are the subject of active research (Morris et al., submitted; Taylor et al., 2003; Cronin et al., 2003; Tellam and Thomas, 2002).

Preliminary interpretation of hydrochemical data involves the description of the concentrations and relative abundances of dissolved major and minor constituents which often can be related to groundwater provenance and flow directions (Domenico & Swartz, 1990). However, groundwater origins and the hydrogeochemical processes affecting their chemical evolution cannot always be deduced from changes in major ion chemistries alone (Elliot *et al*., 2001). This is due to chemical equivalence wherein the same aqueous chemistry can arise through more than one set of processes (Tellam & Lloyd, 1986). To overcome this potential problem, it is often useful to test interpretations further using techniques, such as ion ratios/correlations, characteristic minor/trace determinants, isotopic compositions, downgradient mineralogy and reaction path simulations (Elliot *et al*., 2001). In the case of isotopic signatures, isotopes of a given element differ slightly in mass due to different numbers of neutrons in the nucleus (e.g. ^{13}C). Isotopic ratios have proved extremely useful in resolving multiple sources and sinks for dissolved ions, and identifying the downgradient evolution of groundwaters affected by water-rock interaction (e.g. Elliot *et al*., 1999).

Carbonic acid ($H_2CO_3 \leftrightarrow H_2O + CO_2$) is the most abundant acid in natural water systems and the one most responsible for rock weathering (Langmuir, 1997). The isotopes of carbon are ^{12}C, ^{13}C and ^{14}C and their average terrestrial abundances are 98.89%, 1.11% and 10^{-10}%, respectively (Fritz & Fontes, 1980). Ranges of δ^{13}C in natural materials are very variable (e.g., Langmuir, 1997; Craig, 1953; Burleigh *et al*., 1984) where δ^{13}C is defined in Eqn. 1.

$$\delta^{13}C = \frac{\left({}^{13}C/{}^{12}C\right)_{\text{sample}} - \left({}^{13}C/{}^{12}C\right)_{\text{standard}}}{\left({}^{13}C/{}^{12}C\right)_{\text{standard}}} \qquad [‰\ \text{V-PDB}] \qquad [\text{Eqn. } 1]$$

The δ^{13}C ratio of the total dissolved inorganic carbon content of water ($\delta^{13}C_{TDIC}$) has been used as a tracer in river water systems (e.g., Pawelleck & Veizer, 1994) and lake water systems (e.g., Wachniew & Rozanski, 1997). $\delta^{13}C_{TDIC}$ is an excellent tracer of carbonate evolution in groundwater systems because of the large variations in different carbon reservoirs (Clarke & Fritz, 1997; Evans *et al.* 1997). Iterative modelling of chemical and isotopic changes during carbonate rock dissolution enables theoretical $\delta^{13}C_{TDIC}$ evolution of a groundwater to be mapped as in Figure 1 (Deines *et al*., 1974). This can help to determine whether spring or well waters have evolved under open or closed systems (i.e. with or without a constant supply of soil CO_2). Figure 1 shows how a sample can undergo four different processes during its evolution and how these processes can be distinguished by plotting pH, TDIC and $\delta^{13}C_{TDIC}$.

6. Mineral dissolution increases the $\delta^{13}C$ ratios in the groundwater as mineral C is more enriched in the heavy isotope (^{13}C). Once the groundwater sample is saturated, gypsum dissolution or dedolomitisation can continue the exchange with the rock matrix, increasing the $\delta^{13}C_{TDIC}$. However, provided the sample was previously saturated, these dissolution processes leave both pH and TDIC constant.
7. In an open system, such as the unsaturated zone, dissolved carbon is in equilibrium with the soil CO_2, therefore continuously equilibrating dissolved calcite from the matrix. This leads to a significant increase of TDIC with increasing pH. Due to the much larger reservoir of carbon in the soil, the $\delta^{13}C_{TDIC}$ will be in equilibrium with the soil CO_2. Note that the curves displayed in Figure 1 for open evolution are independent of the initial pH.
8. In a closed system, no CO_2 source is available to permit distinguishable evolution i.e. there are only slight changes of both TDIC and $\delta^{13}C_{TDIC}$ with increasing pH. Note that, in addition to ^{13}C values for the soil and matrix, the curves displayed in Figure 1 require an initial pH, TDIC and $\delta^{13}C_{TDIC}$ of the sample.
9. Admixture of sewage with its lower pH and high alkalinity to a groundwater with a natural background signature leads to an increase in TDIC and a decrease in pH of the mixture. If the recharging water is undersaturated with respect to calcite (Fig. 1) admixture of sewage leads to a hook-shaped curve in the TDIC-pH plot. If the recharge is already saturated the evolution will follow the calcite saturation line.

Despite the fact that groundwater contamination through leaking sewage systems is an emerging issue, little work has been done on the effects of sewage leakage on groundwater ^{13}C. Fernandes et al., (2005) have shown that application of sewage sludge from a wastewater treatment plant in Brazil has increased the C content of the soil and depleted the soil $\delta^{13}C_{CO2}$ values due to a $\delta^{13}C$ value of sewage of –23.7‰. In a temperate climate, such as in the case study settings, where the $\delta^{13}C_{TDIC}$ of recharge is in equilibrium with the soil (open system) the similar value of $\delta^{13}C$ sewage will result in an almost unchanged $\delta^{13}C_{TDIC}$ value for the mixture.

As a general rule, redox reducing influences give rise to depleted $\delta^{13}C_{TDIC}$ values (*i.e.* more negative $\delta^{13}C_{TDIC}$ values) whereas mineral sources tend to produce $\delta^{13}C_{TDIC}$ signatures that are enriched in the heavy isotope. Anthropogenic influences also affect $\delta^{13}C_{TDIC}$ values, in particular landfill leachate. Many authors have observed significant increases in these values due to the onset of methanogenesis (e.g. Hackley *et al*., 1996). North et al. (1996) measured leachate $\delta^{13}C_{TDIC}$ values in excess of +16‰ adjacent to a New Zealand landfill. Organic contamination can also affect carbon-13 isotopic signatures which have been used to assess the degree of biodegradation; these processes and fraction factors for given compounds are reviewed in Meckenstock et al., (2004). This paper uses major hydrochemical composition and the carbon-13 isotopic signatures of groundwater, pore water and aquifer matrix samples to assess and distinguish natural and anthropogenic processes in four major Permo-Triassic sandstone aquifers. These processes are considered here both in relation to depth and to lateral distribution (of samples)

STUDY AREAS

Belfast: The thickness of the Triassic Sherwood Sandstone aquifer is ~ 300 to 400 m in the Lagan Valley and underlying the Belfast area. The formation thins to the SW and dips at about 14° to the NW beneath the Antrim basalts (Manning *et al* 1970; Fig 2. The aquifer is generally confined in the region under thick Quaternary superficial deposits (GSNI, 1997). The Sherwood Sandstone itself grades up conformably into a series of overlying low-permeability argillaceous sediments, the Mercia Mudstones. To the east of the Lagan Valley, the base of the aquifer is underlain, and its eastern side flanked, by Permian marls which overlie the minor Permian Sandstone aquifer. Rainfall in the area averages about 950mm/yr., with 50% loss to evapo-transpiration (for a more detailed description of the study area consult McNeill *et al.* 2000). Hence, 40 abstraction wells in the aquifer were sampled for both $\delta^{13}C_{DIC}$ and ^{14}C in order to identify possible regional trends in isotopic compositions and hence delineate groundwater flow paths as an aid to aquifer management. All samples represent pumped groundwaters from boreholes screened in the Sherwood Sandstone. Samples, therefore, represent a mixture of flowpaths in the aquifer and different well intake depths. X-Ray diffraction analyses and thin section petrography on sandstone samples near Belfast show that the detrital mineralogy is dominated by quartz and feldspar (McKinley *et al.*, 2001), similar to Sherwood Sandstone samples from the Trent Basin (Edmunds *et al.*, 1982). The sandstones contain a variety of minor cements including quartz, K-feldspar, dolomite, calcite, iron oxide, clay minerals, hydroxyapatite and haematite, (McKinley *et al.*, 2001). Evaporite minerals including gypsum are generally present in the Mercia Mudstones (Manning *et al.*, 1970).

Birmingham: Permo-Triassic sandstones form the principal aquifers also in the Trent River Basin of central England (Fig. 3). The Birmingham Sherwood Sandstone aquifer is overlain by Quaternary glacial deposits of 0 to 40 m thickness in west Birmingham and these can produce locally confined conditions (Ford & Tellam, 1994). The aquifer is confined by the Triassic Mercia Mudstone to the southeast of the Birmingham fault. This fault restricts but does not totally eliminate groundwater flow between confined and unconfined aquifer sections, as evidenced by comparison of piezometric surfaces on either side of the fault (Jackson & Lloyd, 1983). The Birmingham plateau has the Trent and its tributaries to the north and east and the Severn and its tributaries to the west and south. The two sampling sites were chosen to be located in confined (Witton (W)) and unconfined (Bromford (B)) areas.

Nottingham: The city is underlain by a sedimentary sequence (Carboniferous to Triassic) of limestones, marls, sandstone and mudstone that dip generally 1.5 to 4° toward the SE (Charsley *et al.,* 1990). The Sherwood sandstone is underlain by Permian marls and, in the southern and eastern areas, overlain by the Mercia Mudstone, providing similar confining conditions to those southeast of the Birmingham fault. Extensive Quaternary superficial deposits in the Trent valley consists of till, sand and gravel, silt and clay. The thickness of these deposits is generally less than 5m but can be up to 10m locally. Beneath much of the city of Nottingham the aquifer is unconfined and found at shallow depths. Regionally, groundwater in the Sherwood Sandstone flows toward the Trent River system. The two sampling sites in Nottingham, Daybrook (D) and Old Basford (OB) are both located in the unconfined part of the aquifer.

Doncaster: The Sherwood Sandstone has little topographic expression apart from isolated and subdued ridges on its western (basal) margin (Fig. 3). The aquifer increases in thickness from its western edge, reaching about 175 m to the east of Doncaster where the suburbs and nearby former coal mining villages are located. Quaternary superficial deposits ranging from glacial sand-and-gravel to peat and lacustrine silty clays overlie the sandstones in many places and these can exert a major control on recharge processes, flow patterns and solute/contaminant transport (Smedley and Brewerton 1997). Groundwater flow occurs from west to east following the artificial gradients induced by the water supply wells located outside the urban area. Hydraulic conductivities are between 2 and 11 m/day, typically ranging between 3 and 5 m/day. The three multilevel wells Haslam Park (HP), Bolton Hill (BH) and McAuley School (MAS) are all located in Bessacarr-Cantley, a suburb of Doncaster. This area was chosen because it is located downgradient of the historical town centre of Doncaster and because the Quaternary deposits are thin to non-existent across much of the suburb. The sites are part of a set of five, all of which are located where the sandstone is either at outcrop or below thin permeable sands and gravels.

Much research has been carried out in these aquifers in the Trent Basin though these studies usually employed shallow monitoring piezometers and/or pre-existing deep abstraction boreholes, (often of uncertain depth and construction). Large screened intervals and ambient vertical flow in such boreholes masks depth-specific trends through the mixing of waters from several horizons (Parker *et al*., 1982). This deficiency in the ability to distinguish depth specific variations, led to the installation of the three multilevel piezometers in Nottingham, the two in Birmingham (Taylor *et al.,* 2000) and the five in Doncaster (Rueedi et al., 2004). All of the multilevels were installed in open holes 50 to 90m deep, drilled by air-flush and each have 7-10 sampling ports. Bentonite clay seals (1 to 3m thick) separate each sampling interval.

METHODOLOGY

All wells and piezometers were purged for a minimum of three to five well volumes and isotopic samples were only taken after well head parameters (temperature, pH, electrical conductivity, redox conditions) had stabilised. Samples for $\delta^{13}C_{TDIC}$ analysis of the Belfast groundwater samples were taken in air tight 100 ml Wheaton® bottles that were capped with a butyl stopper and subsequently crimped tight. Mercuric chloride ($HgCl_2$) was added in order to stop any biological activity in the sample that could alter the carbon isotope signal under sample storage. The Doncaster samples were collected in 60 ml glass bottles with polycone tops. Samples were then stored at 4°C until analysis, which was within 2 weeks after sampling. The total dissolved inorganic carbon (TDIC) was converted into CO_2 gas by direct acidification of the water sample with 80% H_3PO_4 and the gas was purified under vacuum by a series of cryogenic cold traps. The CO_2 was then isolated and analysed on a dual-inlet isotope ratio mass spectrometer (IRMS, either a Micromass Prism III in Belfast or a VG-Optima at BGS Wallingford) to give its isotopic composition ($^{13}C/^{12}C$).

Birmingham and Nottingham waters were collected in 1.5ml crimp top vials that were analysed by gas chromatography-isotope ratio mass spectrometry (GC-IRMS) on a PDZ Europa 20-20. The reference standard for $\delta^{13}C_{TDIC}$ analyses was NBS-19 (National Bureau of Standards limestone standard) that is traceable to the Vienna Pee Dee Belemnite primary standard (VPDB). Water samples were analysed in triplicate to determine the instrumental precision (1 SD ~ 0.05‰).

Pore waters were extracted by centrifuging at 15000 rpm and then analysed as above by dual-inlet IRMS. $\delta^{13}C_{TDIC}$ analysis was also carried out on the rock matrix from cores taken during borehole drilling in the Sherwood Sandstone in Birmingham, Nottingham and Doncaster. The samples were crushed to a grain size between 250 and 425 µm, then acidified by addition of phosphoric acid. The subsequent gas analyses were carried out by GC-IRMS for the Birmingham and Nottingham sets and by dual-inlet IRMS for the Doncaster samples, as described above.

$\delta^{13}C_{TDIC}$ values in aerobic systems will be strongly influenced by the $\delta^{13}C$ of the soil CO_2 though various constraints did not permit soil measurements to be made as part of this work. Kalin (1999) measured $\delta^{13}C_{CO2}$ as –24.3 ± 1.0 ‰ in Northern Ireland while Smith (1976) quoted an average UK value of -26.1 ± 0.3 ‰. Hence, the range –23.30 to –26.13 ‰ was used in the calculations in this paper.

Results and Discussion

Detailed evaluations of the geochemical trends in each aquifer system are given in Cronin et al., (2000) for Belfast, Taylor *et al.*, (in prep.) for Nottingham and Birmingham and in Morris et al., (submitted) for the Doncaster piezometers. These results are therefore limited to the carbon system, with some additional background material to illustrate various processes occurring in the Triassic Sandstone. New $\delta^{13}C_{matrix}$ values from these areas are also presented here as for future reference e.g. to correct radiocarbon groundwater ages. All results are summarised in Table 1. Such sizeable variation in the matrix values is surprising though not dissimilar to that found in other studies (Table 2). Bath *et al.* (1979) postulate that the wide range in $\delta^{13}C_{matrix}$ values reflects a minor secondary precipitate carbonate being more susceptible to solution than the primary carbonate cement. Other studies showed that the large range is due to the difference in $\delta^{13}C_{matrix}$ for dolomite (~ 0 to -3‰) whereas secondary calcite has more depleted $\delta^{13}C_{matrix}$ values than that of primary calcite, often taken as close to 0‰ (Edmunds *et al.*, 1982; Evans *et al.*, 1983).

SEWAGE

Sewer leakage has been found to influence water quality at the majority of the sites. Regular positive detections of bacterial and viral indicators of faecal origin have been found in all three aquifers (Powell et al., 2003, Cronin et al., 2005) where they have been explained by a small but rapid flow component transporting sewer-derived leakage to depth (Cronin *et al.*, 2003; Taylor *et al.*, 2004; Morris et al., submitted). Rueedi et al., (in prep.) have used mass balance calculations of sewer-derived indicators detected in the Doncaster multilevels to estimate that sewer exfiltration makes up c. 5-20% of groundwater recharge. Yang et al., (1999) estimated sewer leakage rates of 3 to 14% of total annual recharge in Nottingham. Hence sewer leakage is potentially a major influence on carbon evolution trends.

In Doncaster 3 samples of a 30-sample dataset of sewage samples that were analysed for major and minor hydrochemistry were also analysed for $\delta^{13}C$ (Fig. 4). The much higher pH values of these sewerage samples places these results in a field distinct from the multilevel and regional well samples, whose pH and TDIC (derived from the bicarbonate alkalinity measurement) fall within a narrow range. However, both datasets, when compared with the lines of calcite and dolomite saturation, can be seen to be oversaturated with respect to both minerals (Fig. 4). Due to the health and safety issues associated with field measurements of pH and particularly alkalinity of sewage, pH values were measured in only one of the field campaigns though laboratory measurements were undertaken. The range of the laboratory results (N=14) is displayed within the shaded area in Figure 4/A It indicates that the initial composition of sewage is strongly oversaturated with respect to calcite and dolomite, possibly leading to calcite precipitation in the unacidified sample used for alkalinity titration, which is conducted by automated titrimetry. This process is indicated with the lines for closed system evolution. Figure 4/B shows that the $\delta^{13}C$ values spread over a rather large range. The $\delta^{13}C$ ratios in the actual sewer may be marginally lower than the sampled values if calcite precipitation occurred in the sampling bottle leading to enrichment of the $\delta^{13}C_{TDIC}$ values.

DONCASTER

Samples from shallow multilevel intervals in Doncaster were found to be significantly influenced by sewage exfiltration. Rueedi et al. (in prep.) calculated mixing ratios of 5 to 20% sewage in local recharge; Figure 5 (plots A, B and C) graphs pH and TDIC, sulphate concentrations and calcite saturation indices for all three Doncaster multilevel piezometers. The TDIC plot displayed in Figure 5/G demonstrates typical open system evolution (as displayed with dotted lines) to be dominating in the shallow part of the aquifer. There is a general trend of carbonate system evolution with a line of mixing between deep, uncontaminated groundwaters and typical sewage composition as indicated with the shaded circle.. The regional wells, of which the sewage contribution is unknown, seem to represent a mixture between the different intervals of the multilevel. An alternative hypothesis is that the very mixed disposition of the data may be influenced by the degree of cementation present at different depths in the aquifer. The relative positioning of the samples may be associated with the availability of calcite to drive samples towards saturation. However, this is less likely as the presence of cements would strongly affect the pH while only slowly increasing TDIC values (even with open evolution). However the trend line shows large increases in TDIC and little pH change favouring evolution line 4 rather than 2 in Figure 1.

DOC and Sr^{2+} are both related to the carbon evolution and they are both found in sewage in amounts exceeding natural background concentrations. Mean sewage values of DOC and Sr^{2+} are 67 mg/L and 0.11 mg/L respectively; compared with rural well background mean values of 3.66 mg/L and 0.03 mg/L. DOC undergoes adsorption and degradation/oxidation, the latter potentially influencing $\delta^{13}C$ signatures in groundwater. The results displayed in Figure 5/D suggest that the sharp decrease of DOC may be due to adsorption in the top 15 m or so of the aquifer. The natural background of strontium in Triassic sandstone wells may come from incongruent dissolution of impure carbonate cements (Edmunds & Morgan Jones, 1976). Figure 5/E shows a decrease in strontium concentrations with depth that is significantly different from the sewage contribution.

As explained above, $\delta^{13}C$ ratios cannot be used to distinguish between contaminated and uncontaminated groundwater because ratios in groundwater and sewage are very similar. However, the narrow ranges of $\delta^{13}C$ ratios in both groundwater (-15 to -19‰) and aquifer matrix (-3 to -7‰) the pore water samples plotted in Figures 5/F to H do indicate that there is a distinction between mobile (or effective) and immobile porosity. This is because the pore water samples, in contrast to the other groundwater samples, are centrifuged from a core and this is likely to include a proportion of ('pellicular') water held immobile by capillary forces within the matrix. This immobile water is likely to be in equilibrium with the surrounding calcite matrix with a corresponding highly-enriched $\delta^{13}C$ ratio. If the groundwater sample is saturated with respect to calcite and if the pH of both mobile and immobile water is equal (as for steady state conditions) the mixture of mobile and immobile water should not affect the TDIC values but it should linearly increase the $\delta^{13}C$ ratios without affecting the pH (similar to evolution 1 in Fig. 1). This is what can be observed in Figures 5/G and H

In Figure 5/G, the regional wells tend to plot on the top left of the deep multilevel samples indicating some on them as they plot close to the shallow contaminated wells then the deeper multilevel plots. The $\delta^{13}C$ ratios of the regional wells seem to be consistently higher than the multilevels. This increase in $\delta^{13}C$ could originate from permanent but very slow admixture of carbon from the immobile phase by diffusive exchange. However, as discussed above (Evolution trend line 1 in Fig.1), congruent dissolution of gypsum and subsequent precipitation of calcite (dedolomitisation) can lead to the same evolution of the carbonate system. The fact that these integrated samples contain a larger mixture of waters representing different flow paths could add to the discrepancy between samples from regional and multilevel wells.

NOTTINGHAM

TDIC and pH values (Fig 6/E) at Daybrook, Nottingham, indicated that all samples have undergone an open system evolution leading to $\delta^{13}C_{TDIC}$ ratios in the range of -14 to -16‰ (Fig 6/F). A shift to more enriched $\delta^{13}C$ signatures between 30.71 and 34.98 mbgl points to the potential dominance of dolomite dissolution over calcite and/or ongoing calcite precipitation.

At Old Basford, Nottingham, vertical evolution of pH, TDIC and sulphate (Fig. 6/A, B and C) are very similar to the Daybrook site. However, $\delta^{13}C_{TDIC}$ signatures increase slowly (-15.2 to -12.3‰) from 18.0 to 30.3mbgl and then rise significantly to -6.6/-7.5‰ from 35.1 to 39.3mbgl (Fig 6/B). The rock matrix $\delta^{13}C$ signatures show high values in the upper part of the aquifer and these decrease to lower ratios with depth. Congruent dissolution of gypsum cannot be responsible for the enriched $\delta^{13}C_{TDIC}$ ratios because the sulphate concentrations are consistently low in the deep levels. Therefore, $\delta^{13}C$ signatures are considered to reflect enrichment due to dissolution of secondary calcite at shallow depths followed by dedolomitisation in the deepest samples. The absence of matrix $\delta^{13}C$ values at these depths prevents confirmation of this assertion but depth profiles of magnesium and calcium concentrations are consistent with this hypothesis (Taylor et al., in prep.).

BIRMINGHAM

$\delta^{13}C_{TDIC}$ signatures at Witton, Birmingham, fall throughout the depth profile within a narrow range between -13.7 to -14.8‰ that is indicative of calcite or dolomite dissolution (Fig 7/D). Taylor et al. (in prep.) conclude that the middle levels are influenced by sewage recharge (Fig. 7/B). The alkalinity of the groundwater samples is slightly less than the shaded circle indicating sewage samples from Doncaster (Fig. 7/E).

The TDIC-pH plot (Fig. 7/E) indicates that the groundwater at Bromford, Birmingham evolved under open conditions, showing similar values as observed in Doncaster. $\delta^{13}C_{TDIC}$ values decrease slightly with depth from -8.3‰ to -9.9‰ and correlate well with the vertical trend in matrix $\delta^{13}C$ values (Fig 7/D). The higher ratios at this well are a consequence of congruent gypsum dissolution leading to SO_4^{2-} levels in this well of more than 1000 mg/L (Fig. 7/B). Overall, the Birmingham samples seem to show slight, though consistently low pH values (by ~0.1 unit). This is potentially due to a systemic error with the specific pH meter though calibration of the meter was carried out daily.

BELFAST REGIONAL WELLS

$\delta^{13}C_{TDIC}$ values increase along flow paths in the regional aquifer through gradual isotopic enrichment by mineral dissolution. The regional spread of $\delta^{13}C_{TDIC}$ values in Belfast is variable but general enrichment is observed along the groundwater flow lines indicated in Figure 8. . Variability in trends along flow paths is introduced via mixing, variable matrix values, potential contamination sources and overall this points to a complex interaction of several carbon sources and sinks. An inverse geochemical model, such as NETPATH (Plummer *et al*, 1994), can help calculate the relative contributions of these sources and sinks to the final water geochemistry (e.g. Cronin et al., 2001). $\delta^{13}C_{TDIC}$ values were used to help screen a variety of mass-balance hypotheses for the given chemical reactions taking place along a flow path. Differences of ±2‰ were taken as the limit for a model-calculated

$\delta^{13}C_{TDIC}$ match with the field measured value (cf. Kimblin, 1995). $\delta^{13}C_{TDIC}$ results thus serve to constrain better the mass balance as these data quantitatively account for carbon sources and sinks affecting the ^{14}C activity of the waters. In this way, $\delta^{13}C_{TDIC}$ data permit more accurate estimates of groundwater residence times (Cronin *et al.,* 2000).

SUMMARY - SYNTHESIS OF δ^{13}CTDIC INFERENCES IN TRIASSIC SANDSTONE AQUIFERS

The carbon system of all samples taken from the different Triassic sandstone aquifers starts with dissolution of CO_2 and calcite under open system conditions. Typical partial pressures for the samples in Doncaster, Birmingham and Nottingham were found to be between 0.0032 ($10^{-2.5}$) and 0.00032 ($10^{-3.5}$) bar. The samples from Belfast, however, seem to be somewhat higher but the influence of urban contamination cannot be ruled out entirely for many of these. The range of alkalinity and $\delta^{13}C$ ratios found in shallow groundwaters are also within quite a narrow range (-14 to -19‰) corresponding well with an assumed $\delta^{13}C$ soil gas ratio of -23 to -27‰. Hydrochemical evolution through gypsum dissolution and/or dedolomitisation is subsequently able to influence the carbon system leaving the alkalinity unchanged but resulting in an increase of $\delta^{13}C_{TDIC}$. The range of $\delta^{13}C$ ratios of aquifer matrix are variable but similar for all four sites (Table 1) and correspond well with the variation in ranges documented in literature (Table 2).

Besides the natural processes, urban contamination through leaking sewage systems has been shown to be able to alter the carbon system by adding considerable amounts of bicarbonate leading to a decrease of the pH. This carbon evolution path is not readily distinguishable from natural pathways due to the similarity between soil and sewage $\delta^{13}C$.

Conclusions

Field studies of the Triassic sandstone aquifer (Sherwood Sandstone Group) in four locations in the UK (Belfast, Birmingham, Doncaster, Nottingham) show that the stable isotope of carbon clearly distinguish natural from anthropogenic geochemical processes responsible for dissolved inorganic carbon in groundwater. In Birmingham, Nottingham and Doncaster natural evolution involves an equilibration of recharging water with soil CO_2, having typical partial pressures of 0.0003-0.003 bar and $\delta^{13}C$ ratios between -23 and -27‰. Depending on the aquifer material, local dissolution of gypsum and dolomite can lead to increasing $\delta^{13}C$ ratios. Admixture of sewage, however, is clearly distinguishable because it increases the TDIC and decreases the pH but has a minimal impact on $\delta^{13}C_{TDIC}$. The effect of sewage on the $\delta^{13}C$ ratio of recharging water, however, depends strongly on the $\delta^{13}C$ signature of the soil CO_2 and the $\delta^{13}C$ signature of raw sewage. As the latter varies considerably, further research is required to constrain the isotopic signature of this source of C. For the Belfast study, enrichment of $\delta^{13}C_{TDIC}$ generally proceeds with depth and suggests a down-gradient geochemical evolution (i.e., sample maturity). Such samples from open boreholes have been used in a regional groundwater investigation to constrain geochemical mass-balance models and to account for carbon sources and sinks along delineated flow paths (Cronin et al., 2000, 2001).

Depth profiles from multilevel monitoring wells improve understanding of different geochemical processes affecting groundwater and enabled the depth of contamination via leaking sewer systems to be resolved from natural processes. Fully penetrating wells, in comparison, represent a mixture of waters (integrated over?) from a much larger depth interval, though diluted signals the influence of sewage were still observed. Theoretical evolutions of mixtures of sewer and natural recharge enable observed depth-specific field values to be explained (needs work). The $\delta^{13}C_{TDIC}$ evidence backs up microbiological, mass balance and age indicator work that all points to components of younger recharge with an important sewer-derived component reaching significant depths in the Permo-Triassic sandstone. Hence, the qualitative nature of carbon stable isotopic ratio studies, can, if used in conjunction with quantitative groundwater dating tools such as radiocarbon, tritium/helium, SF_6 and CFCs, lead to a more complete understanding of water quality and age evolution.

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Table 1. Range of $\delta^{13}C$ values (‰ w.r.t. VPDB) of groundwater (GW) and of the Sherwood Sandstone matrix (Ma) for Belfast (B'fast), Nottingham (Nott), Birmingham (B'ham) and Doncaster (Don) aquifers. The total number of sites and the total number of samples analysed are given. Note that the Belfast sample set comprises pumped production wells while the others are averages of discrete multi-level depth intervals isotopic signatures of the Nottingham, Birmingham and Doncaster results. [not sure you need this last sentence as they are all averaged anyway] Reg = regional wells, ML = multilevel, PW = porewater

	B'fast Reg	**B'fast**	**Nott ML**	**Nott**	**B'ham ML**	**B'ham**	**Don ML**	**Don Reg**	**Don ML**	**Don PW**
	GW	**Matrix**	**GW**	**Matrix**	**GW**	**Matrix**	**GW**	**GW**	**Matrix**	**GW**
N sites	40	3	2	2	2	1	3	11	3	1
N sample	40	12	20	13	18	6	22	11	9	5
Min	-19.1	-6.8	-15.3	-10.8	-14.8	-5.9	-20.9	-19.3	-7.2	-13.5
Max	-8.3	-1.2	-6.6	-2.4	-6.4	-0.2	-15.1	-14.4	-3.0	-10.8
Mean	-15.8	-3.1	-11.9	-4.3	-11.3	-2.6	-17.0	-16.4	-5.9	-12.2

Table 2. Comparison of $\delta^{13}C$ matrix ranges (‰ w.r.t. VPDB) from other studies on Permo-Triassic Sandstone aquifers in Great Britain.

Range of $\delta^{13}C_{matrix}$	**Reference**
-0.2 to –10.8	This study
-5.5 to -11.7	Bath et al. (1979)
-7.9 to -10.7	Jackson & Lloyd (1983)
0 to -7	Edmunds, et al. 1982)
-2.2 to -9.9	Evans et al. (1983)
-4.5	Tellam (1994)

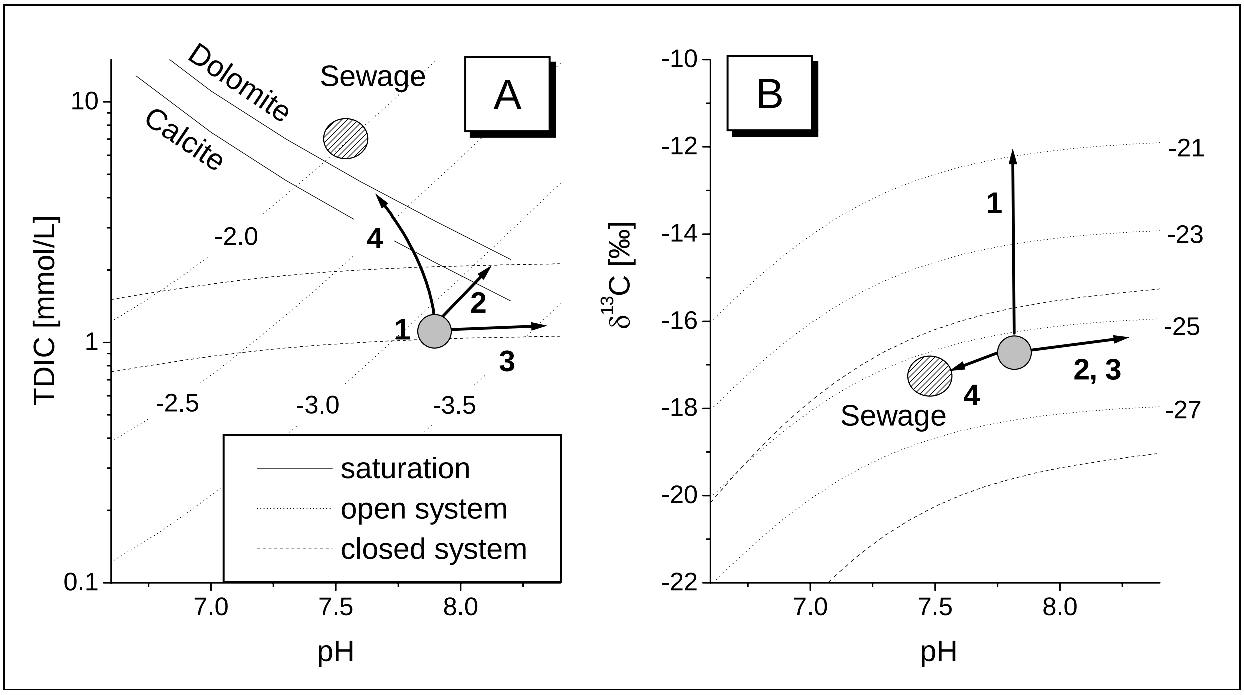


Figure 1. TDIC and $\delta^{13}C_{TDIC}$ values (‰ w.r.t. VPDB) versus pH for four water evolution pathways in the carbonate system. 1) gypsum dissolution or dedolomitisation; 2) open system with indicated log values of CO_2 partial pressure and $\delta^{13}C_{CO2}$; 3) closed system with initial (closing) pH of 7.6 and closing CO_2 partial pressure of $1 \cdot x\ 10^{-3}$ and $2 \cdot x\ 10^{-3}$, $\delta^{13}C$ of soil CO_2 of -16 and -20‰ and a $\delta^{13}C$ of the aquifer matrix of -3‰; and 4) sewage admixture.

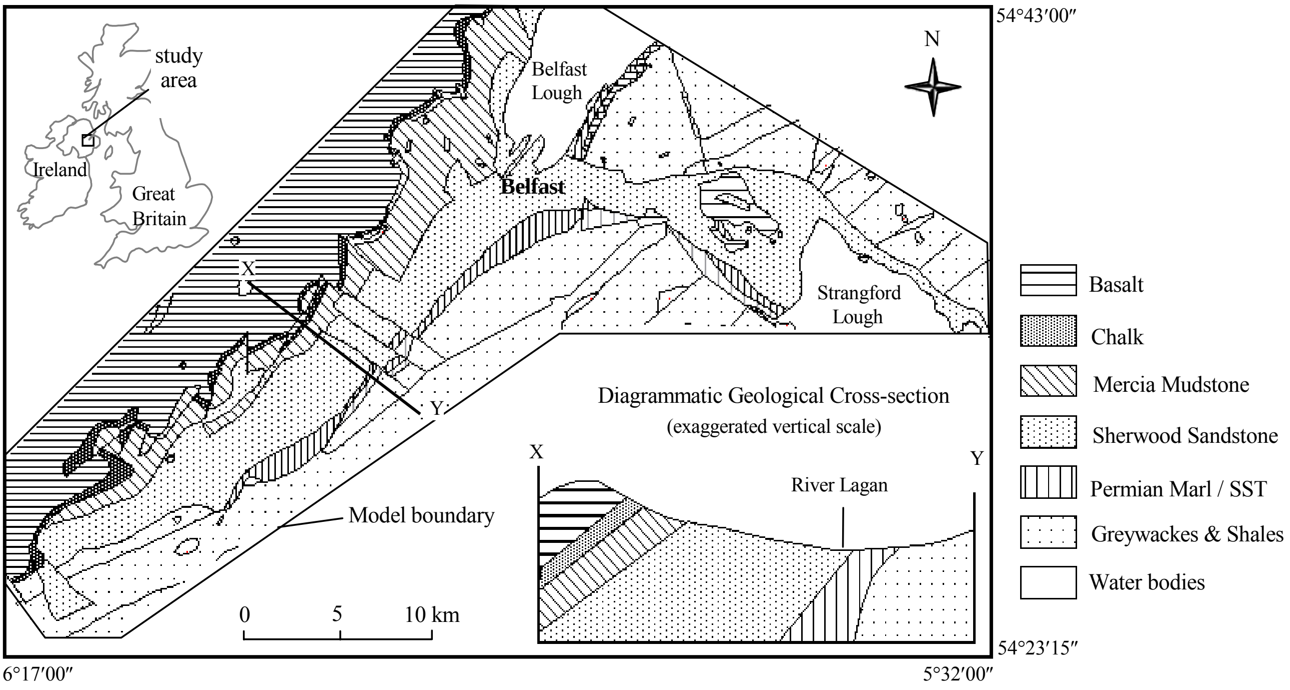


Figure 2. Map showing the Lagan Valley Permo-Triassic aquifer underlying the Belfast region, Northern Ireland, (after GSNI 1997).

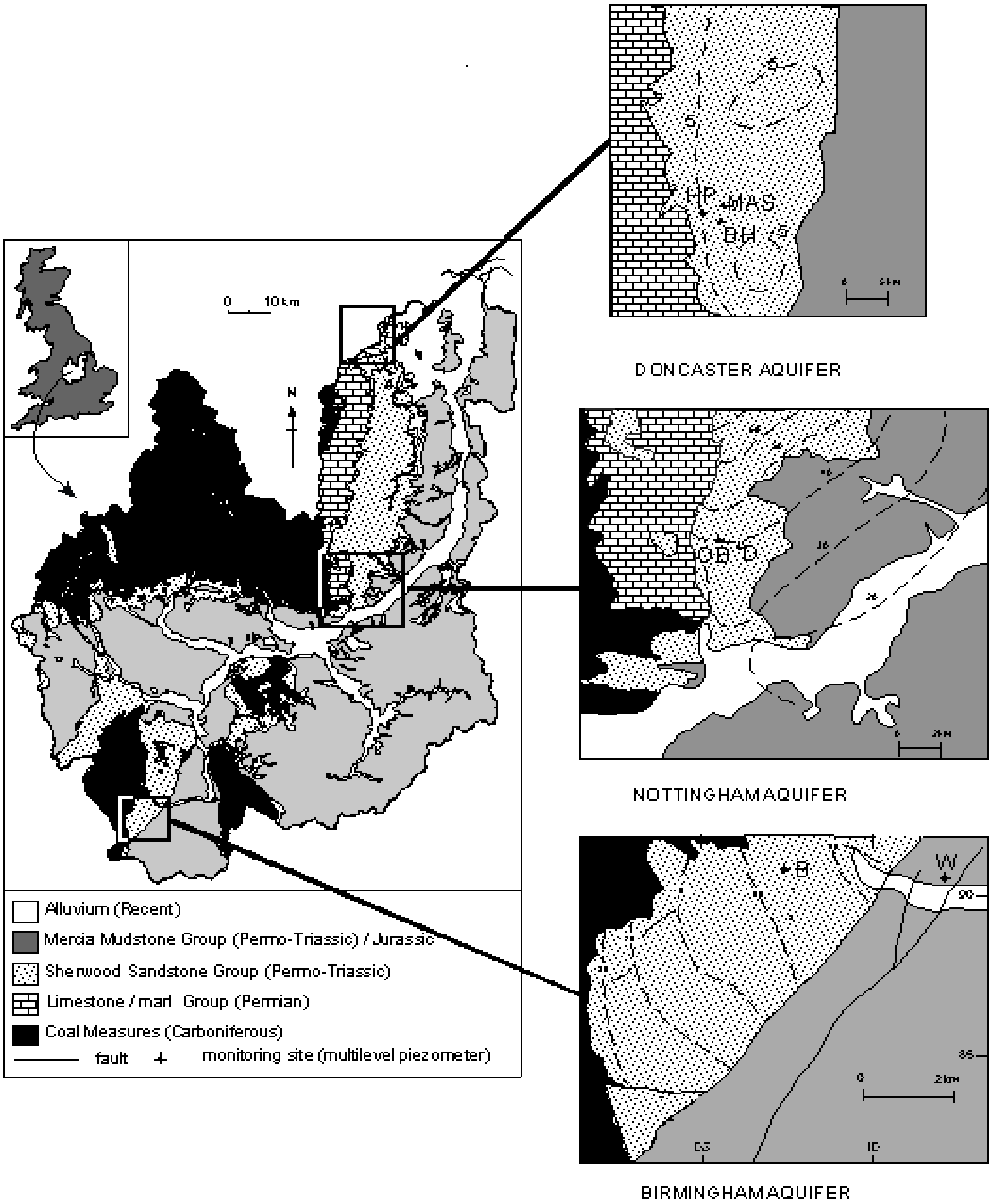


Figure 3 Location and geology of the Trent River Basin, England. The upper right hand box shows the geology of Doncaster at an expanded scale, the middle box denotes Nottingham while the lower right hand box shows the geology of the Birmingham area at the same scale. Piezometric water levels are denoted by the dashed lines with associated labels m relative to Ordnance Datum. These expanded boxes also show multilevel piezometer locations as (+) crosses.

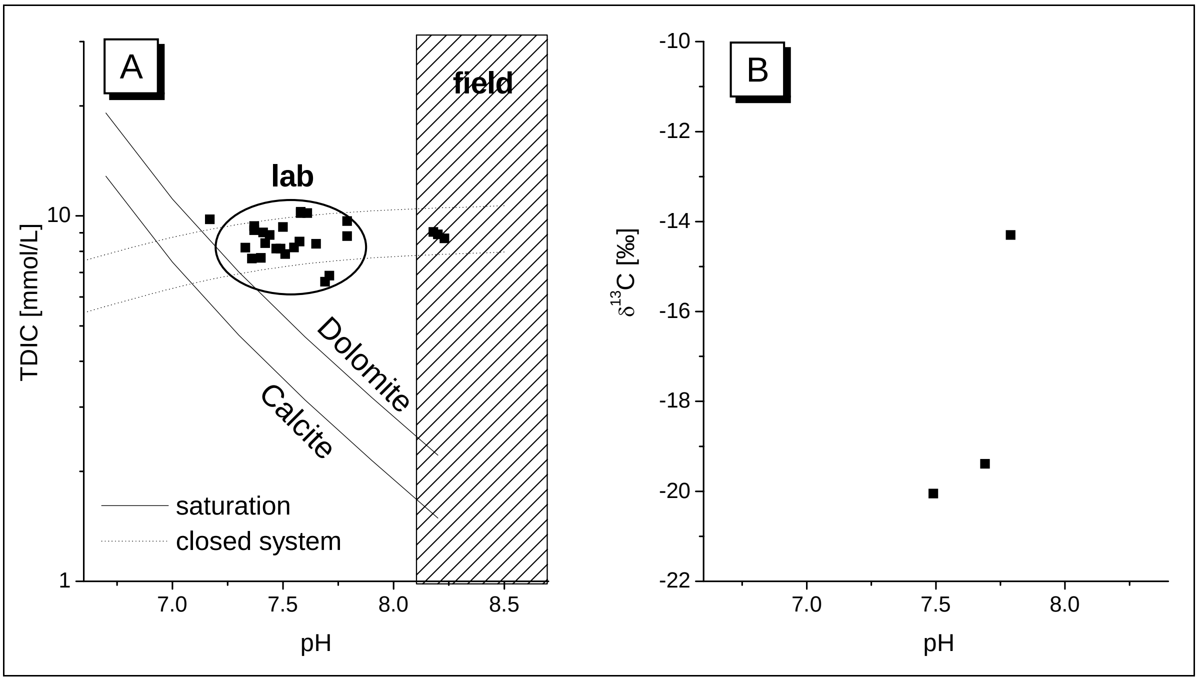


Figure 4. Plot of TDIC and $\delta^{13}C_{TDIC}$ versus pH of sewage samples taken in Doncaster and measured in the lab. The shaded area indicates the range of pH values measured in the field (N=14). Full lines indicate calcite and dolomite saturation and dotted lines indicate a closed system evolution when calcite is precipitated. Initial values of TDIC at pH 8.4 (of field samples) were chosen to be 8 and 10.4 mg/L.

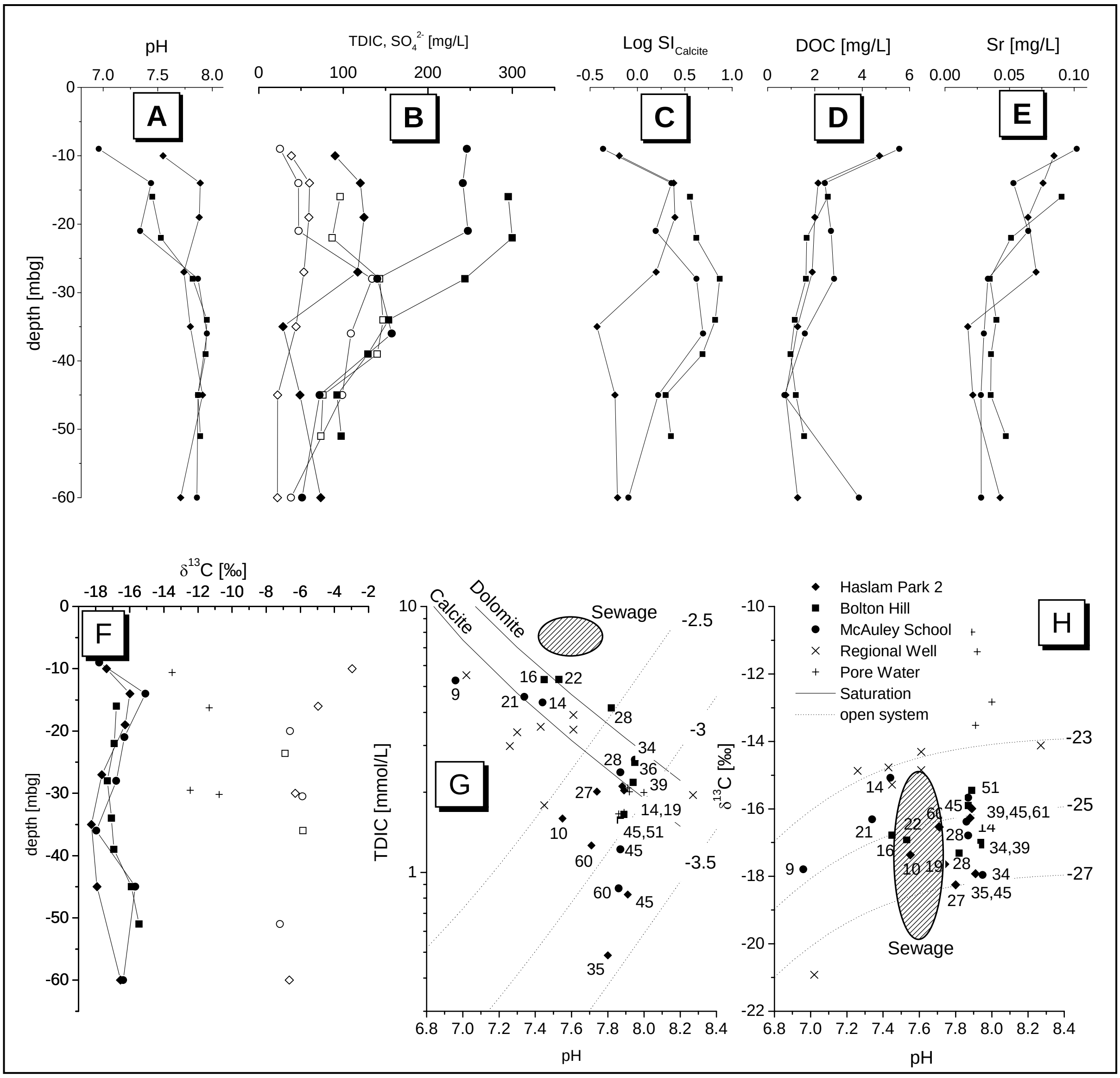


Figure 5. Depth profiles of the multilevel piezometer sites in Doncaster. Haslam Park 2 is depicted with diamonds, Bolton Hill with squares and McAuley School with circles. A) pH; B) TDIC (full symbols) and SO_4^{2-} (open symbols); C) calcite saturation index SI; D) Dissolved organic carbon; E) Sr^{2+}; and F) $\delta^{13}C$ ratios of groundwater (full symbols), rock matrix (open symbols) and pore water (crosses). G) and H) TDIC vs. pH, respectively $\delta^{13}C_{TDIC}$ vs. pH, including labelled equilibrium lines for open system evolution (dotted lines), saturation lines (full lines) and typical range of sewage samples shown with the dashed circle. Regional wells are indicated with stars.

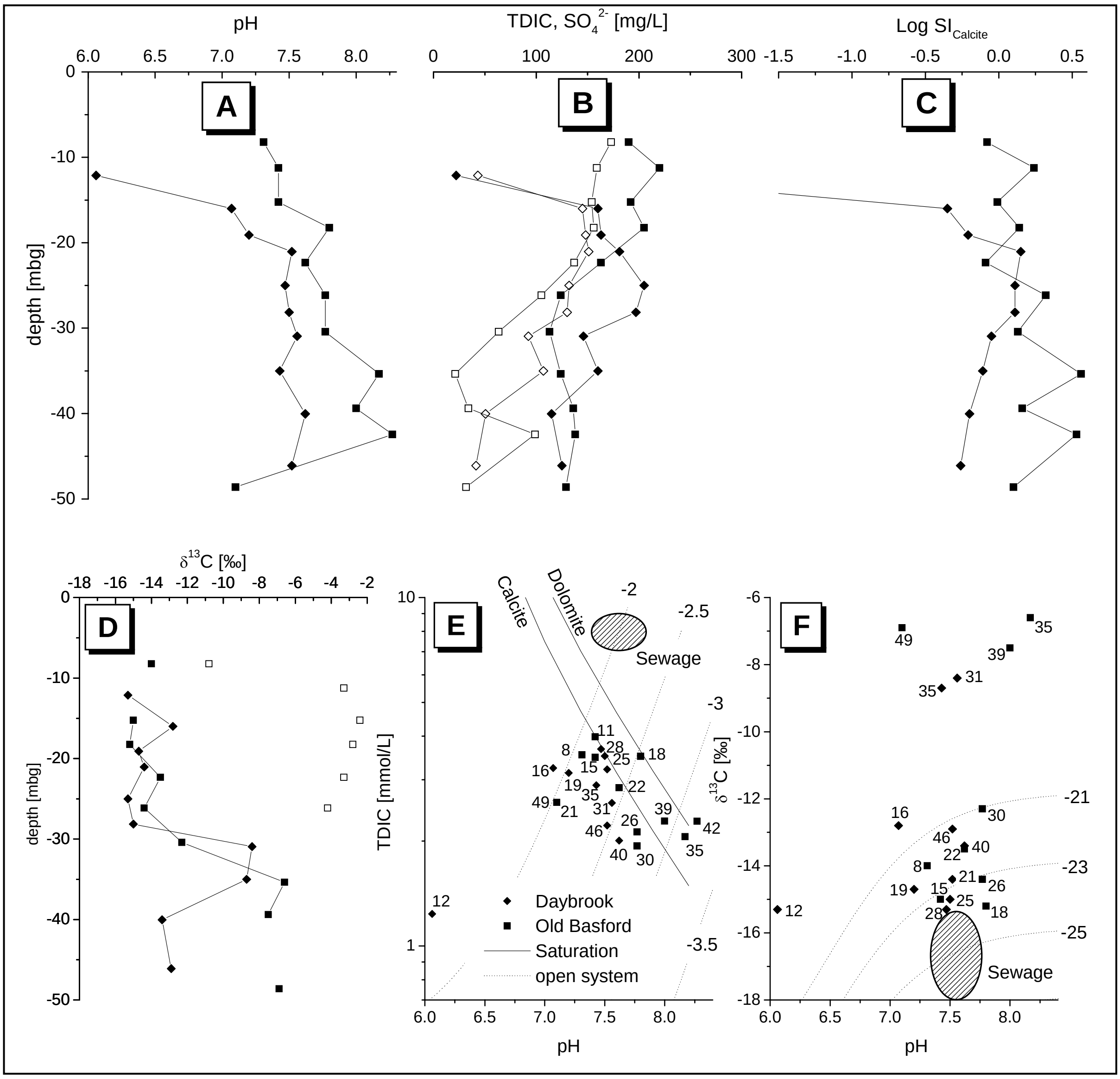


Figure 6. Depth profiles of the multilevel piezometer sites in Nottingham. Daybrook is depicted with diamonds and Old Basford with squares. A) pH; B) TDIC (full symbols) and SO_4^{2-} (open symbols); C) calcite saturation index SI and D) $\delta^{13}C$ ratios of groundwater (full symbols) and rock matrix (open symbols). E) and F) TDIC vs. pH, respectively $\delta^{13}C_{TDIC}$ vs. pH, including labelled equilibrium lines for open system evolution (dotted lines), saturation lines (full lines) and typical range of sewage samples shown with the dashed circle.

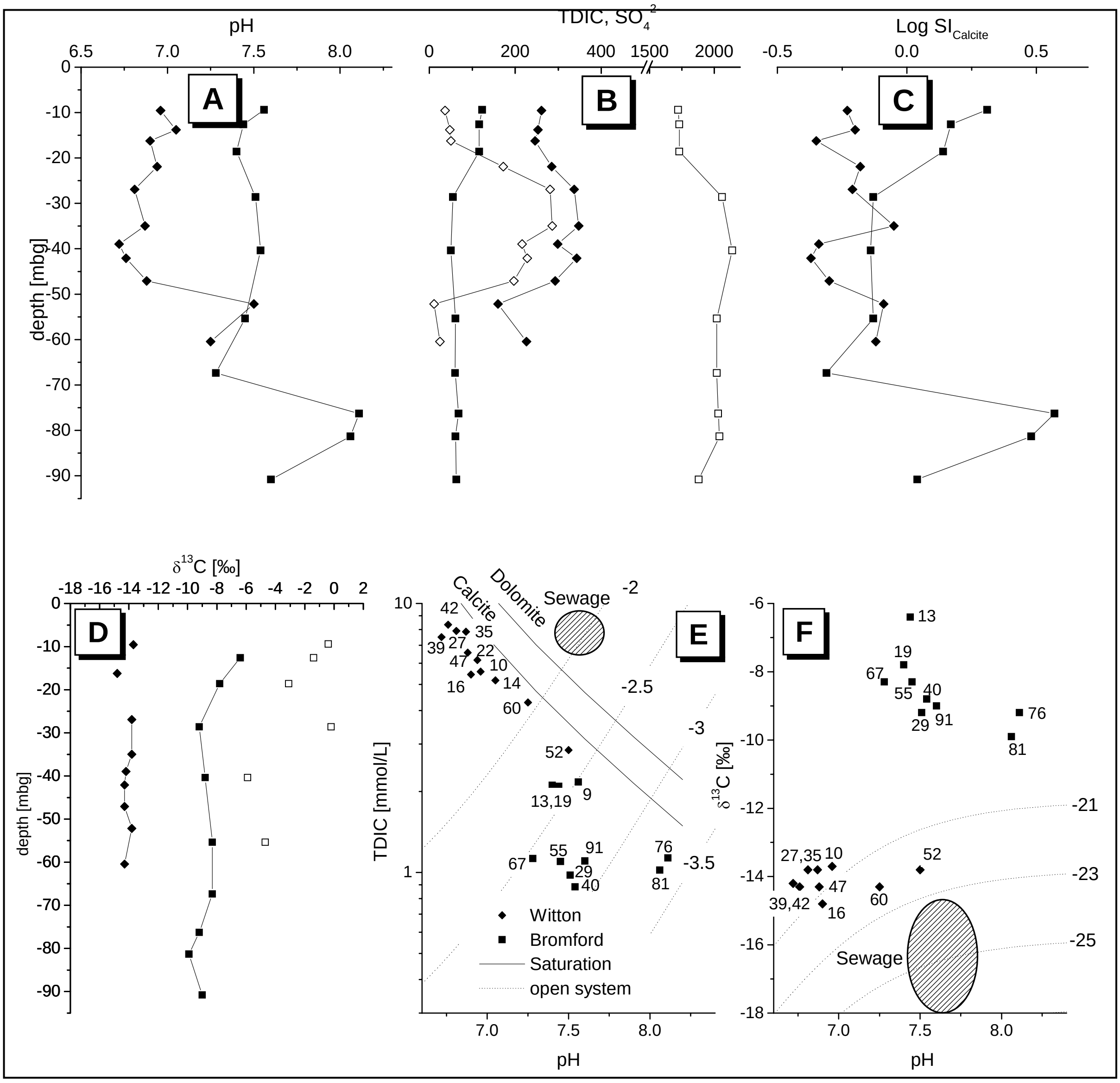
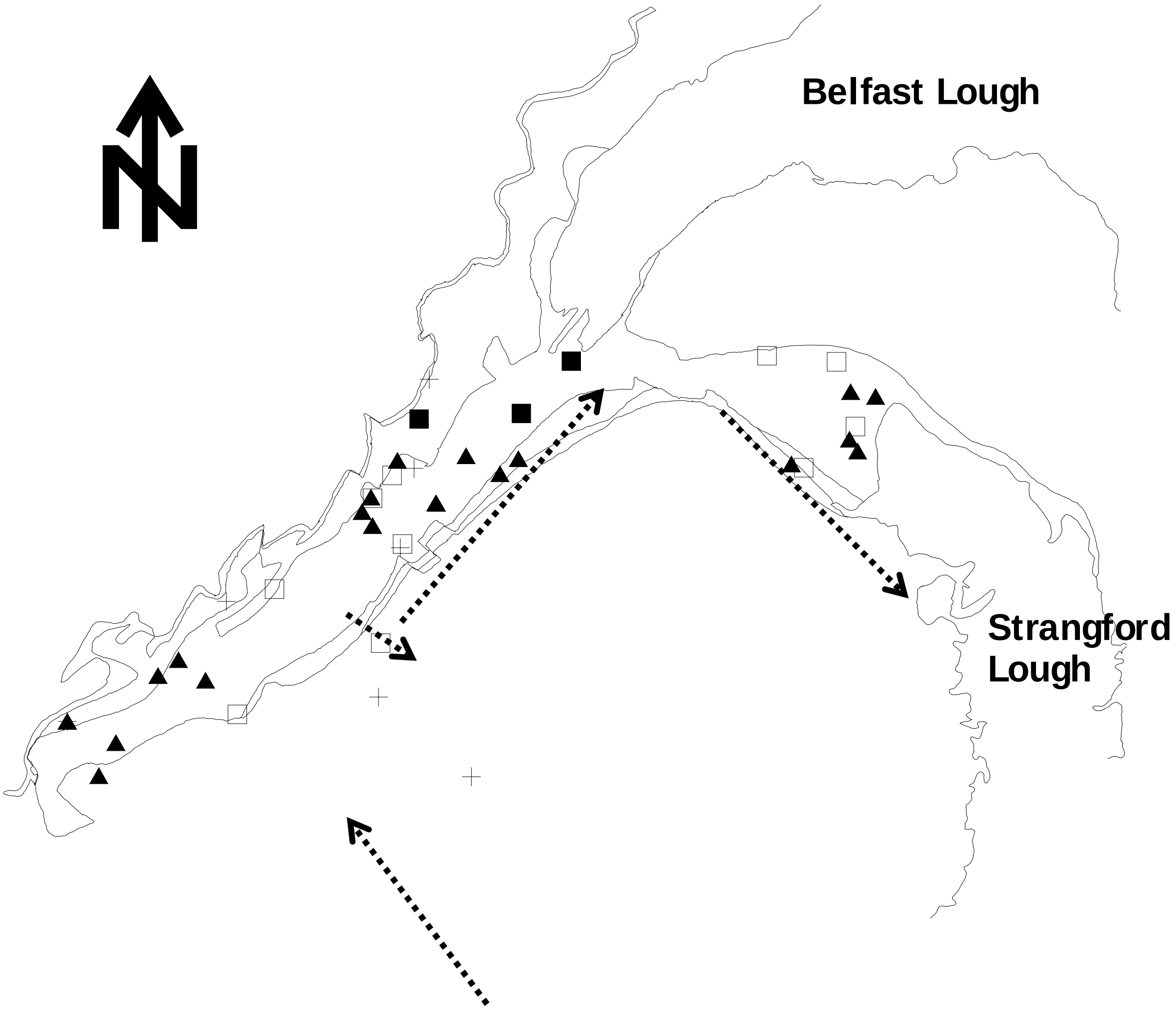


Figure 7. Depth profiles of the multilevel piezometer sites in Birmingham. Witton is depicted with diamonds and Bromford with squares. A) pH; B) TDIC (full symbols) and SO_4^{2-} (open symbols); C) calcite saturation index SI and D) $\delta^{13}C$ ratios of groundwater (full symbols) and rock matrix (open symbols). E) and F) TDIC vs. pH, respectively $\delta^{13}C_{TDIC}$ vs. pH, including labelled equilibrium lines for open system evolution (dotted lines), saturation lines (full lines) and typical range of sewage samples shown with the dashed circle.



Legend to $\delta^{13}C_{TDIC}$ values

+ = > -17‰,

□ = -17‰ to -16‰,

▲ = -16‰ to -14‰

■ = -14‰ to –8‰.

Figure 8. Distribution of the $\delta^{13}C_{TDIC}$ values (‰ w.r.t. VPDB) in the Belfast Sherwood sandstone aquifer. Dashed arrows indicate enrichment evolution lines and these agree with regional groundwater flow.

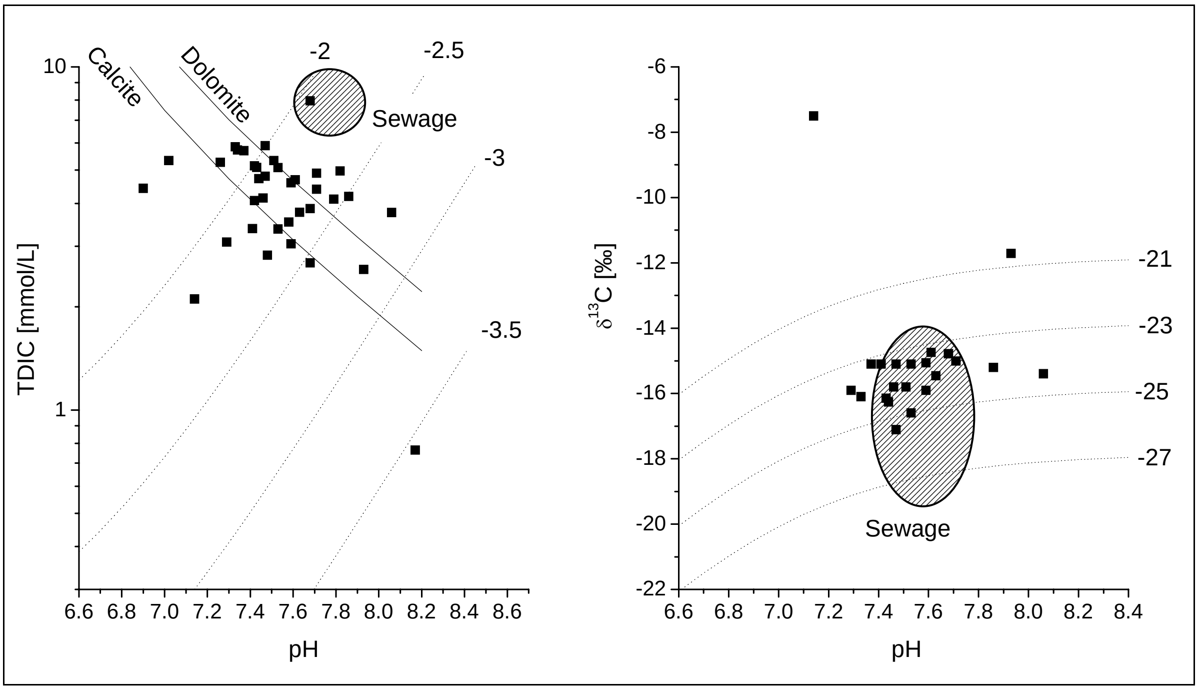


Figure 9. G) and H) TDIC vs. pH, respectively $\delta^{13}C_{TDIC}$ vs. pH of regional wells around, where labelled equilibrium lines for open system evolution (dotted lines), saturation lines (full lines) and typical range of sewage samples shown with the dashed circle.

2.3.5 Paper 5: Estimating sewer leakage using hydrochemistry sampling of multilevel piezometers (Rueedi *et al.*)

Key findings

- Current leakage rates from sewerage system were typically in the range 20-45 mm/a, corresponding to a total leakage of 7-15% of annual sewage throughput or $3.0\text{-}7.5 \cdot 10^{-5}$ L/sec/m of pipeline. This information will be used to calibrate the pipeline leakage model.
- The signature of sewage-derived recharge corresponds well with that from groundwater dating tracers, which show extensive penetration of modern (<50 years-old) recharge to similar depths.
- The method presented provides an independent validation of the usefulness of qualitative sewage indicators.
- Overall, recharge volumes in the urban area were found to be similar to natural recharge.

Introduction

Structural defects in sewer systems are a problem faced by water authorities across the world. Investment in sewer reparation in England and Wales, for example, exceed £250M per year (Davis and Burn 2001). While these costs are directly paid by the water consumer, additional socio-economic costs are borne by other sectors of society (De Silva et al. 2001). Of these social costs, public health and the environmental consequences of leaking sewers have certainly attracted the most attention. The performance standards set out in the European Union (EN 752-2) recognise the problem of groundwater contamination through leaky sewers and the EU has funded research to develop management strategies to improve the structural integrity of urban sewer systems including their water-tightness.

Groundwater studies in the UK (Cronin et al. 2003) and Germany (Eiswirth and Hoetzl 1994) have identified exfiltration from sewer systems as a source of groundwater contamination. Current approaches to estimate sewer leakage use available information on the pipe system's design and condition together with data extrapolated from single leak studies to estimate sewer leakage. However, these approaches usually lead to rather high uncertainties [Wolf 2005]. The use of passive tracers (those present naturally in effluent) to detect leaking sewage systems are discussed in numerous publications. These range from major ions (e.g. chloride or potassium in Barrett et al. 1999) to minor elements and trace substances (e.g. boron, X-ray contrast media or hormones in Makepeace et al. 1995, Ternes 1998, Sacher et al. 2001, Wolf et al. 2004) to indicator micro-organisms (Cronin et al. 2005). However, these tracers are not universally applicable to all groundwater systems because both the input (effluent) and the background (groundwater receptor) concentrations are highly variable. A few investigations have tried to independently quantify sewer leakage (Lerner et al. 1994, Eiswirth and Hoetzl 1997, Yang et al. 1999, Eiswirth et al. 2002). Eiswirth et al. (2002) and Eiswirth and Hoetzl (1997) found a leakage rate of at least 5% of dry weather flow in a study in Rastatt Germany and (Yang et al. 1999) estimated sewer leakage rates of 3 to 14% of total effluent volume in Nottingham, UK.

The European Union 5th FP project AISUWRS (Assessing and Improving Sustainability of Urban Water Resources and Systems) is addressing the issue of comparing different urban water supply and disposal scenarios by producing a Decision Support System to enable the prediction of the effects of various urban water management strategies (see www.urbanwater.de). The different scenarios will be simulated using a series of models to reproduce water flow through the urban land surface and its pipe assets, the unsaturated zone and, finally, the groundwater body. In order to evaluate both existing urban water systems and alternative strategies, the sources of contaminants, their flow paths and volumes (e.g. unaccounted-for-water, recharge from pipe leakage) and their sinks need to be identified for different urban settings. The first step of the model set-up consists of calibration to the present day conditions and the estimation of uncertainties of the different model input parameters. This paper provides information that is used to inform and better constrain the two upstream models, namely a mass flux model named Urban Volume and Quality (UVQ) and pipe leakage model (PLM), both developed by CSIRO, Australia [Mitchell et al. 2003].

Methods

STUDY AREA

The location of Doncaster is shown in Figure 1, together with an inset showing the disposition of the five multilevel monitoring wells in the suburb of Bessacarr-Cantley which is the 6.3 km^2 area of detailed study for the project.

The production wells of the public water supply network are located in an arc further east and north of the study area and they have typically been drilled to depths of 140-170 mbgl and screened below about 30-60mbgl.

The 5 multilevel piezometers were all drilled (air-flush technique) and installed in September 2003 at sites chosen to detect contamination originating from the older parts of Doncaster centre and the suburb of Bessacarr-Cantley which was constructed mainly post-1945. Planned geophysical logging had to be cancelled because the open holes were found unstable due to the poorly cemented nature of much of the uppermost 30m or so. In fact, the first hole drilled at Sandall Beat collapsed at a depth of about 16 mbgl and the multilevel piezometer had to be installed by telescoping through temporary casing. Between drilling and installation short pumping tests (3-10 hours) were performed and recovery data analysed to obtain a better idea of the average hydraulic conductivity (Table 1).

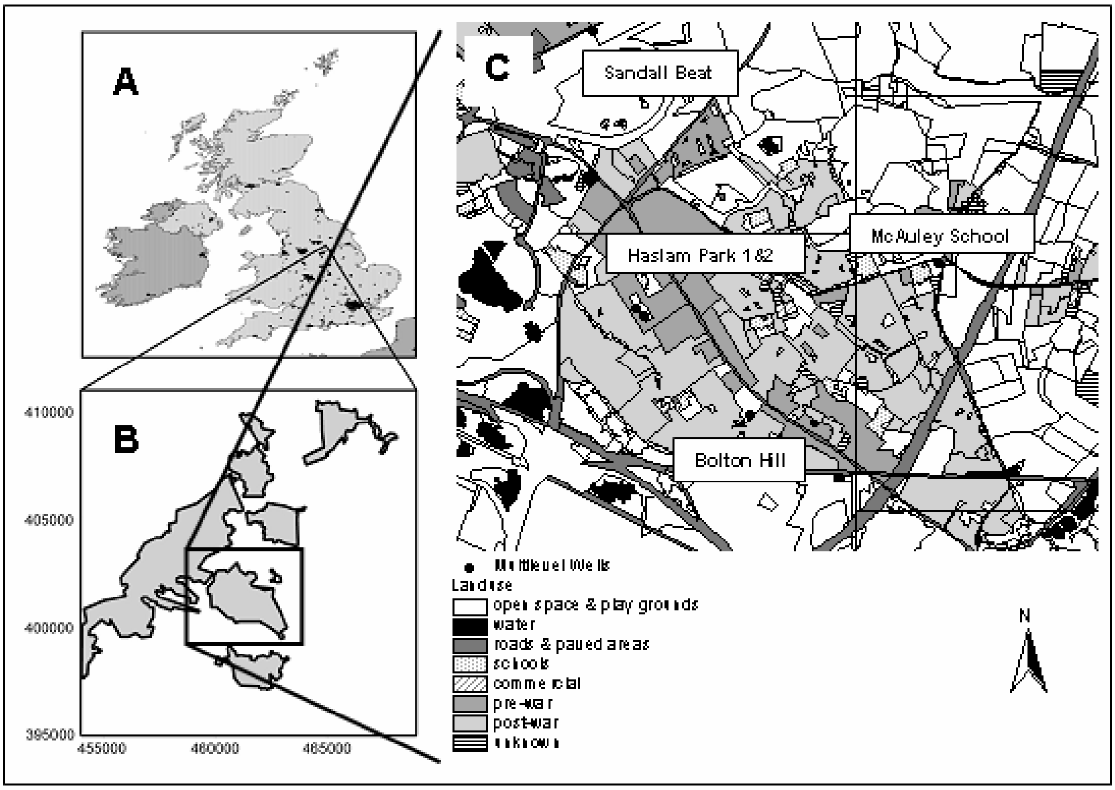


Figure 1. (A) Location of the city of Doncaster within the UK. (B) Detailed map showing the urban area of Doncaster, the study area Bessacarr-Cantley indicated with a black frame. (C) Locations of the multilevel wells underlain by the major land use types (Land use data from Doncaster Metropolitan Borough Council).

Table 1. Technical details of multilevel wells.

Name	Drilled depth [mbgl]	Head level [maOD]	Easting	Northing	Hydraulic Conductivity [m/day]	No. of sampling ports
Sandall Beat	36	8.491	460080	403457	-	5
Bolton Hill	51	15.153	461230	400704	8.2	7
Haslam Park 1	60	11.092	460455	401392	1.0-1.7	7
Haslam Park 2	60	10.75	460400	401465	1.5-2.7	7
McAuley School	61	9.621	462597	401786	3.5	7

Installation was started directly after the pumping test. Plastic pipes, arranged as shown in Figure 2a, were inserted into the borehole first. The larger size pipes are made of PVC and the smaller ones are of HDPE, tied to the centre pipe. Different sizes were used for the top and bottom-most level to enable online monitoring of water levels, temperature and specific electrical conductance SEC. The end of each pipe was sealed with a cap and a 30cm screen was constructed 20cm above the bottom of the pipe (Figure 2b).

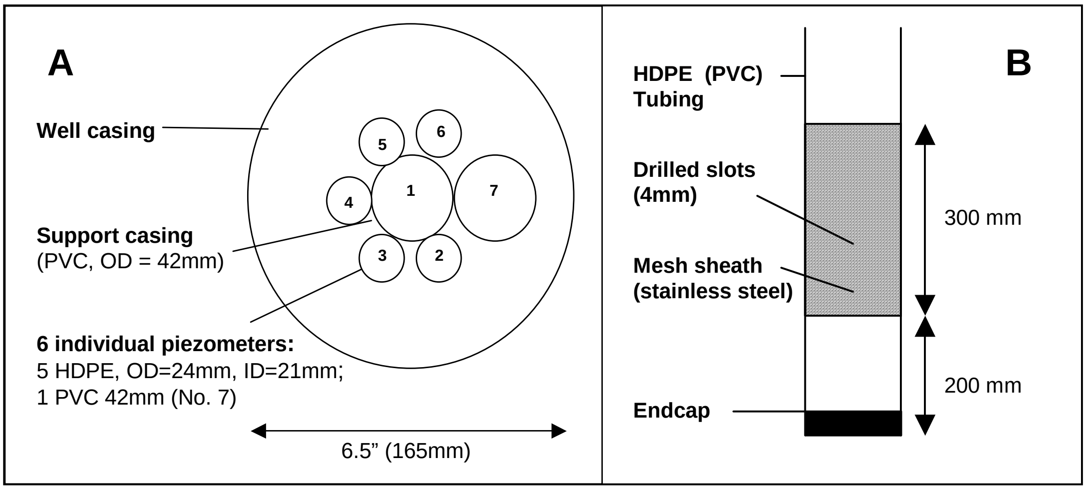


Figure 2. A) Arrangement of plastic pipes inside the open borehole before backfilling. (B) Detail of screen section showing lowest part of a HDPE (or PVC) pipe with intake section comprising holes drilled into the pipe and covered with a stainless steal mesh to prevent sand from entering the pipe during sampling.

The holes were refilled with sand around the screen and the different levels were hydraulically separated with bentonite clay plugs of 1-3 meters thickness.

Before sampling was commenced the wells were developed to remove potential construction-stage microbial contamination from the wells by introducing 50mg/L bleach solution. After leaving it for a several minutes 3-5 purge volumes were pumped out until the SEC stabilized.

Estimating sewer leakage using the urban water balance

In an unconfined aquifer situation, water leaking from sewer or water mains pipes passes through the unsaturated soil zone and, eventually, reaches underlying groundwater. Other forms of recharge such as that from precipitation, river channels or on-site drainage similarly enter the saturated zone. Therefore, groundwater recharge represents a mixture of all inputs. On a small spatial scale, the different inputs may vary significantly depending on the relative position of the observation point to pipeline leaks and on the time of observation (Wolf and Hoetzl 2005). However, on a larger scale such as the study area, these small-scale variations in volume and concentration are integrated over tens to hundreds of meters. On this larger scale it is, therefore, possible to use urban water balance calculations to estimate the different contributions to urban recharge.

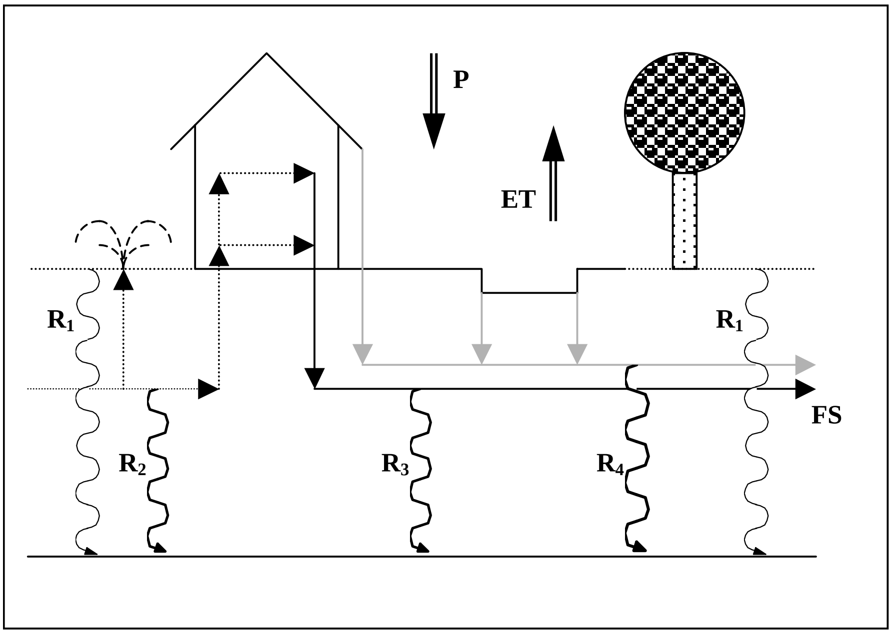


Figure 3. Schematic picture of urban water cycle. Dotted lines indicate mains water supply, full lines indicate foul water (black) and stormwater (grey) system.

The water cycle in a small urban area such as a housing district with mains water supply and wastewater connections is shown schematically in Figure 3. The urban water balance can then be calculated assuming a balance of water inputs and outputs to the area (Equation 1). The equation does not consider possible river drainage because pluvial drainage in the research area occurs mainly through the stormwater system and not through surface drainage.

$$P + DW = ET + SW + FS + GW \qquad \text{[Eq.1]}$$

P: precipitation [mm]

DW: mains water supply [mm]

ET: evapotranspiration [mm]

SW: pluvial drainage [mm]

FS: foul sewer volume [mm]

GW: recharge to groundwater [mm]

(All components expressed as depth equivalent units)

Assuming that the groundwater table is constant and no leakage occurs to or from lower aquifer units the groundwater flux GW can be split into four major fluxes $R_{1\text{-}4}$ (Equation 2).

$$GW = R_1 + R_2 + R_3 + R_4 \qquad \text{[Eq.2]}$$

R_1: natural recharge through precipitation [mm]

R_2: mains leakage [mm]

R_3: foul sewer leakage [mm]

R_4: pluvial drain leakage [mm]

Groundwater quality measured at the top of the aquifer represents the actual mixture of the various current recharge sources. Deeper in the aquifer historical recharge is observable, from other locations entering further upgradient.

The concentration of each groundwater quality component is governed by the same mass balance as given in Equations 1 and 2 but is amended to various degrees by physicochemical or microbially-mediated reactions such as decay (e.g. micro-organisms), adsorption (e.g. heavy metals) ion exchange (e.g. sodium replacing calcium and magnesium) or dissolution/precipitation (e.g. calcite and dolomite). The general mass balance for groundwater concentrations C_{GW} is therefore

$$P \cdot C_P + DW \cdot C_{DW} = ET + SW \cdot C_{SW} + FS \cdot C_{FS} + GW \cdot C_{GW} + S \qquad \text{[Eq.3]}$$

and

$$C_{GW} = \frac{R_1 \cdot C_1 + R_2 \cdot C_2 + R_3 \cdot C_3 + R_4 \cdot C_4}{GW} \qquad \text{[Eq.4]}$$

C: source concentration [mg/L]

S: sink or source [mg/m^2/y]

Equation 3 assumes that evapotranspiration loss is pure water.

Note that all parameters included in the water and contaminant balance are associated with an uncertainty that can be calculated following the general law of error propagation and a) taking into account first order errors only and b) assuming a normal distribution for all parameters (Equation 6).

$$\mu_Y = f(\mu(X_i)) \qquad \text{[Eq.5]}$$

$$\sigma_Y{}^2 = \sum_j \left(\frac{\partial f}{\partial X_j} \right)^2 \cdot \sigma_j{}^2 \qquad \text{[Eq.6]}$$

where

$$Y = f(X_i) \qquad \text{[Eq.7]}$$

μ: arithmetic mean

σ: standard deviation

Y: mixing ratio

X_i: water balance component (e.g. FS, GW, R_1, etc.)

σ_Y: uncertainty of mixing ratio

σ: uncertainty of water balance component

The above error estimation will provide mixing ratios Y and associated uncertainties σ_Y for each measured groundwater parameter g. Even in an ideal case, they will all show slightly different averages and certainly different uncertainties. However, provided these parameters are conservative, all should show the same mixing ratio. (Rueedi et al. in press) applied a solution to calculate total mixing ratios $\overline{m}$ and uncertainties $\overline{\sigma}$ using conservative hydrochemical parameters P (see Equations 8-10).

$$\overline{m} = \frac{\sum_{g=1}^{n} m_g \cdot w_g}{\sum_{g=1}^{n} w_g} \qquad \text{[Eq.8]}$$

$$\overline{\sigma} = \sqrt{\frac{1}{\sum_{g=1}^{n} \sigma_g^{-2}}} = \sqrt{\frac{1}{\sum_{g=1}^{n} w_g}} \qquad \text{[Eq.9]}$$

with

$$w_g = \frac{1}{\sigma_g^{\,2}} \qquad \text{[Eq.10]}$$

g: groundwater parameter

w: weighting factor

n: number of parameters considered

Results

URBAN WATER BALANCE

To estimate the urban water balance of the study area numerous datasets were analysed. Weekly precipitation data from the UK Meteorological Service MORECS system, from 1970 to 2004, are used to obtain an average precipitation of 692 mm/y with quite a large standard deviation of 114 mm/y. The MORECS dataset provides potential evapotranspiration data also but, as we are interested in the actual evapotranspiration, these data cannot be used directly as they would require additional assumptions on soil conductivity. Therefore, the Urban Volume and Quality model UVQ [Mitchell et al. 2003] was calibrated using these datasets. UVQ produces realistic values for stormwater runoff from impermeable areas using two field parameters: the runoff-effective paved areas and the maximum initial loss. The runoff-effective areas are estimated based on field observations of the different pavement types. Roofs and roads are assumed to direct 95% of the incoming water to the stormwater system because, even though they are fully connected with the system, there are always some leaky gutters or some pavements that drain towards green areas. The paved areas around houses, however, are very often poorly connected with the stormwater system and therefore assigned an effectiveness of only 50%. The maximum initial loss is the amount that remains on the paved areas and cannot be drained. This value is assumed to be about 2mm [Mitchell and Diaper 2003].

Using all these inputs UVQ produces an average stormwater recharge to groundwater of 118 mm/y for the period 1970-2004 and the uncertainty is estimated to be about 10%.

Mains water supply statistics for the six leakage control zones in the study area were provided by the local water company (Yorkshire Water) for 1998-2003. These were used to calculate an average water supply volume of 1.22 million m^3/y. Mains leakage was estimated based on hourly night time flow records from Yorkshire Water and corrected for domestic and commercial night-time use allowances i.e. estimated real water usage during the night such as toilet flushing. Leakage rates of the leakage control zones range from 1.1 to 5.3 L/property/h, with an average for the year 2003 of 1.85±0.39 L/property/h. This value corresponds with an average leakage rate of 9.9% of all imported water. If all available records (1997-2003) are used the average leakage rate becomes 1.82±0.53 L/property/h.

Gardens are usually watered using mains water. Therefore, the water supply records were corrected for irrigation by subtracting the water usage in winter from the observed values. This quite simple approach provides an annual water volume for garden irrigation of 7300L/property/y (corrected for leakage), corresponding to 4.9% of total supplied water. This seems like a large volume used for irrigation but it actually ranges from 2600 to 14000 L/property/y, where the lower values were found in areas with denser housing and smaller gardens and the higher ones in the older part of the suburb with large gardens.

Table 2. Key water supply and disposal parameters (left column) and key urban setting parameters (right column).

Water Balance		**Average Domestic Property Statistics**	
	[mm/y]		
Precipitation P	692±114 [1]	Area [ha]	603 [4]
Evapotranspiration ET	-	No. of domestic properties	8323 [2]
Total imported water DW	203±12 [2]	Average occupancy	2.52 [5]
Mains Leakage R_2	22±5 [2]	Road area [ha]	92 [4]
Garden Irrigation I	10±3 [2]	Paved area [ha]	39 [4]
Total Foul Sewage Volume FS (DW-R2-I)	171±7	Roof area [ha]	74 [4]
Stormwater Volume SW	118±12 [3]	Effective road area [%]	95
Ratio SW:FS	0.69±0.08	Effective paved area [%]	50
Total urban recharge	150-200 [6]	Effective roof area [%]	95
		Maximum initial loss [mm]	2

[1] from MORECS database
[2] from water supply statistics
[3] from UVQ
[4] from GIS analysis
[5] from domestic statistics
[6] from literature

A detailed analysis of the local pipe network can be found in [Cunningham et al. 2004], who used GIS techniques to analyse the pipe asset provided by the water utility. The pipe network was categorised according to age, material type, diameter and joint type. In this paper, a selection of key statistics is shown to support the results (Figure 4). It can be seen that information on pipe ages of the sewage system is limited. The pipe asset database of both sewerage and stormwater systems provide age information for only 28% of their entire length. This information would be very useful to compare estimated leakage rates with the actual age and material distribution of pipes. The information on the age distribution of the water mains is more comprehensive, showing that only 1% of the pipes are older than 50 years. The lower row of pie-charts in Figure 4 show that 99% of foul and pluvial drainage systems are built with vitreous clay and concrete pipes and that water mains are almost exclusively of ductile and cast iron materials.

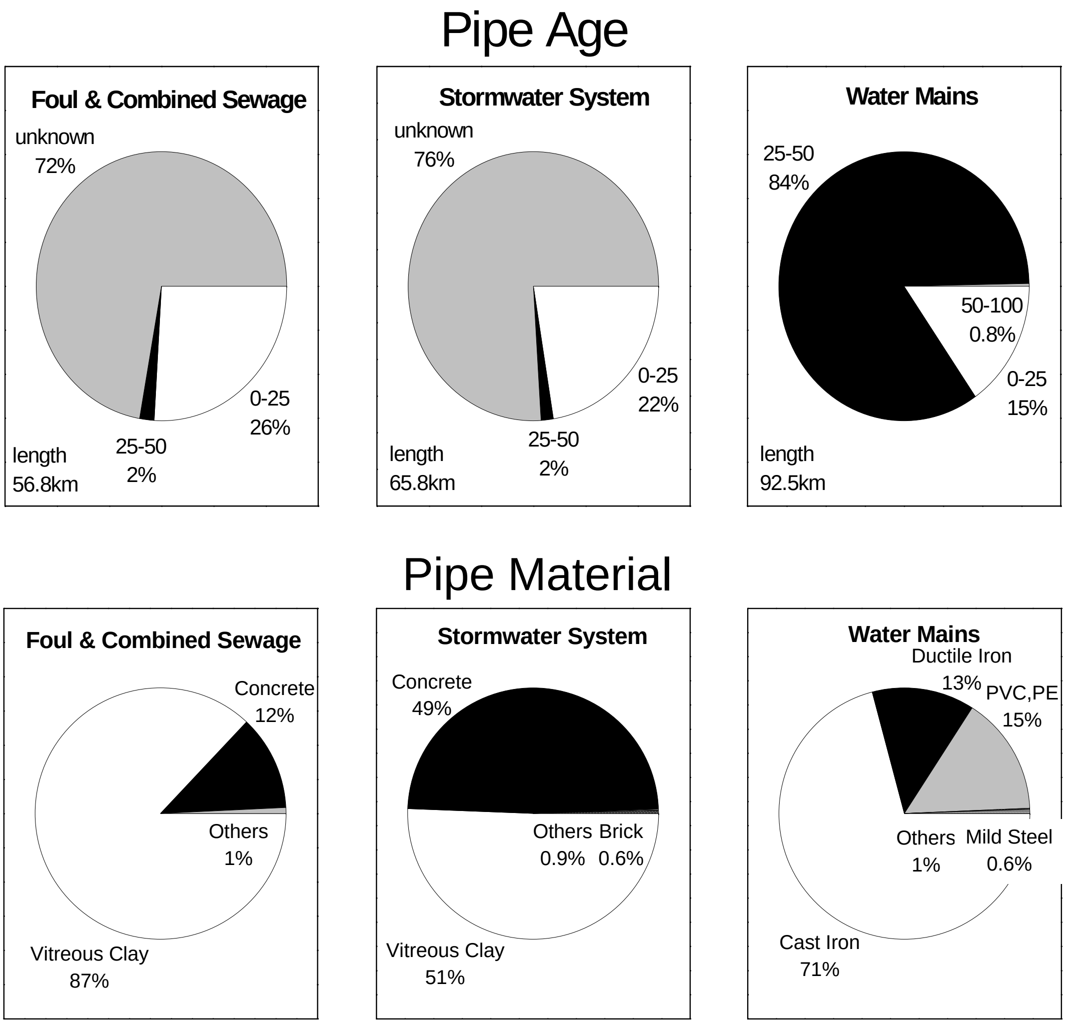


Figure 4. The top row shows age distributions (in years) of the pipe network in Bessacarr with the total length displayed in the bottom left corner. The lower row displays the percentage distribution of pipe materials in use.

An important unknown parameter, apart from the leakage rates from stormwater and foul sewage system, is the recharge from green areas and gardens. This is an output of the UVQ model too but it was found to be very sensitive to the model user's choice of different soil parameters. Therefore, literature values were used to define a range of reasonable natural recharge volumes. A natural recharge (under unpaved areas) of approximately 200mm/y was estimated by Binley et al. (2002) who sampled seasonal variations of soil moisture near Hatfield, located about 8 miles NE of Bessacarr-Cantley. A range of 150-200 mm/a also gives an acceptable calibration of water levels against observed values for the steady-state sub-regional model derived by the project from the regional transient flow model referred to below. Comparing these values with results for other urban recharge calculations in the same area, Yang et al. (1999) used a GIS approach to estimate a value of 239mm/y. for urban recharge in Nottingham while Brown and Rushton (1993) used a value of 110mm/y to calibrate the first regional Nottingham-Doncaster groundwater model.

Urban recharge and contaminant balance

Results from the three multilevel sampling rounds in November 2003, February 2004 and May 2004 are summarized in Table 3, where arithmetic means and standard deviations are displayed for each interval of the five multilevel wells. Overall, the different sampling campaigns showed only small (seasonal) variations.

Table 3. Summary of major and some minor hydrochemical parameters (mg/L) for all intervals of the multilevel piezometers.

[mg/L]	K^+	Na^+	Mg^{2+}	Ca^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	NO_3^-	B	P
BH16	14.07 ±1.86	17.1 ±1.3	30.0 ±3.1	81.1 ±7.1	28.9 ±3.0	92.3 ±2.6	244.6 ±3.5	4.42 ±0.56	0.076 ±0.013	0.099 ±0.032
BH22	11.93 ±1.12	19.2 ±1.0	32.2 ±0.8	83.0 ±1.7	32.2 ±3.4	86.0 ±2.3	269.4 ±12.8	5.52 ±0.27	0.058 ±0.006	0.138 ±0.055
BH28	8.80 ±0.41	23.9 ±0.9	37.8 ±0.5	101.6 ±4.8	69.6 ±6.1	144.0 ±3.0	217.8 ±17.7	7.16 ±0.92	0.065 ±0.026	0.169 ±0.113
BH34	6.04 ±0.05	18.6 ±0.2	38.0 ±0.4	100.7 ±0.6	112.3 ±2.1	154.7 ±2.3	128.8 ±7.7	5.19 ±0.61	0.015 ±0.008	0.149 ±0.045
BH39	6.64 ±0.31	17.3 ±0.2	34.8 ±0.8	90.1 ±1.6	103.0 ±1.0	143.0 ±2.6	115.0 ±29.3	5.90 ±0.50	0.043 ±0.010	0.151 ±0.043
BH45	5.93 ±0.16	12.1 ±0.7	19.9 ±0.1	51.1 ±0.7	45.3 ±3.9	74.9 ±1.9	82.1 ±7.8	8.54 ±0.69	0.032 ±0.010	0.149 ±0.047
BH51	2.45 ±0.14	10.5 ±0.9	22.5 ±0.4	55.6 ±1.5	33.1 ±21.5	74.6 ±2.7	78.4 ±5.8	10.84 ±1.61	0.015 ±0.008	0.031 ±0.044
HP1 10	4.67 ±0.63	7.3 ±1.3	18.7 ±1.7	41.4 ±4.7	24.9 ±1.4	49.3 ±3.4	127.6 ±9.2	4.10 ±0.18	0.076 ±0.004	0.090 ±0.037
HP1 14	4.86 ±0.18	11.5 ±0.6	23.0 ±0.3	58.2 ±1.6	34.5 ±2.9	61.2 ±1.6	156.9 ±7.0	6.09 ±1.02	0.121 ±0.033	0.115 ±0.040
HP1 21	6.00 ±0.30	13.3 ±0.7	22.6 ±0.5	58.5 ±1.4	34.3 ±2.0	72.0 ±1.8	142.2 ±12.0	6.93 ±1.07	0.108 ±0.026	0.177 ±0.040
HP1 28	6.13 ±0.43	17.4 ±1.0	23.8 ±1.1	67.5 ±4.6	41.7 ±3.2	83.3 ±5.3	151.2 ±6.4	12.33 ±2.12	0.098 ±0.035	0.184 ±0.046
HP1 35	3.87 ±0.12	15.2 ±0.9	16.5 ±0.3	45.5 ±1.0	25.6 ±2.1	72.4 ±3.5	62.2 ±15.0	12.18 ±1.76	0.015 ±0.008	0.170 ±0.032
HP1 45	1.68 ±0.12	8.0 ±0.4	11.1 ±0.6	29.8 ±1.5	21.0 ±1.1	27.4 ±1.8	40.6 ±2.5	15.00 ±1.99	0.015 ±0.008	0.137 ±0.060
HP1 60	3.72 ±0.97	10.1 ±1.4	14.8 ±2.7	38.9 ±6.2	21.7 ±5.9	36.3 ±9.6	115.8 ±34.8	6.67 ±0.42	0.077 ±0.020	0.190 ±0.011
HP2 10	4.87 ±1.63	6.8 ±1.0	15.5 ±0.7	33.5 ±2.6	10.4 ±0.6	38.7 ±2.8	98.3 ±10.9	10.51 ±1.40	0.086 ±0.003	0.083 ±0.048
HP2 14	4.61 ±0.62	9.4 ±0.8	20.0 ±1.1	46.4 ±2.7	20.9 ±4.2	61.9 ±3.3	102.0 ±19.0	10.11 ±1.04	0.086 ±0.018	0.110 ±0.036
HP2 21	4.81 ±0.37	9.1 ±0.3	19.8 ±0.2	48.4 ±0.7	21.7 ±2.6	63.4 ±3.0	106.5 ±18.3	9.09 ±0.97	0.089 ±0.013	0.157 ±0.044
HP2 28	4.37 ±0.50	8.7 ±0.2	19.2 ±0.7	32.3 ±25.3	19.5 ±3.3	59.0 ±1.9	108.2 ±26.3	10.31 ±1.30	0.061 ±0.002	0.197 ±0.034
HP2 35	2.09 ±0.08	9.3 ±0.7	12.3 ±0.4	35.6 ±0.4	33.0 ±3.5	47.3 ±2.1	29.4 ±6.4	11.60 ±1.83	0.031 ±0.010	0.186 ±0.016
HP2 45	1.30 ±0.04	6.5 ±0.2	9.2 ±0.3	24.9 ±0.8	18.5 ±1.4	23.4 ±0.8	29.3 ±6.6	10.24 ±1.35	0.015 ±0.008	0.183 ±0.029
HP2 60	1.86 ±0.44	8.6 ±1.9	9.8 ±1.9	26.8 ±2.7	14.4 ±2.2	19.5 ±6.9	53.4 ±24.3	9.01 ±1.61	0.058 ±0.008	0.232 ±0.028
McA9	6.17	45.9	19.5	39.4	18.1	32.5	204.8	7.36	0.063	0.233

[mg/L]	K^+	Na^+	Mg^{2+}	Ca^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	NO_3^-	B	P
	±2.16	±6.3	±1.6	±4.1	±1.3	±3.9	±26.4	±1.32	±0.012	±0.058
McA14	5.68	19.2	23.5	66.1	18.7	46.5	249.9	5.17	0.059	0.174
	±0.12	±2.7	±0.9	±4.7	±1.7	±3.3	±4.4	±1.27	±0.011	±0.116
McA21	5.44	24.3	23.7	63.5	13.7	59.9	226.4	9.11	0.055	0.213
	±0.04	±3.8	±0.1	±1.2	±0.9	±3.7	±21.0	±0.94	±0.013	±0.078
McA28	4.64	12.6	23.7	80.0	15.6	147.0	147.1	12.13	0.065	0.184
	±0.65	±1.4	±0.6	±1.9	±0.6	±5.2	±3.7	±1.21	±0.002	±0.021
McA36	3.70	20.6	22.1	69.9	20.8	121.7	148.3	7.09	0.015	0.218
	±0.14	±2.1	±0.7	±3.4	±2.3	±10.4	±28.1	±0.42	±0.008	±0.016
McA45	3.05	20.0	18.0	60.7	26.4	103.1	82.5	18.93	0.015	0.199
	±0.15	±0.8	±0.3	±2.5	±1.7	±4.4	±3.5	±2.14	±0.008	±0.002
McA60	2.14	12.2	10.6	38.2	32.7	36.5	63.2	11.73	0.031	0.156
	±0.33	±2.6	±0.1	±2.3	±0.7	±0.2	±11.1	±1.12	±0.010	±0.054
SB16	2.27	32.2	41.6	112.3	77.0	127.7	261.7	13.40	0.059	0.108
	±0.08	±2.1	±0.5	±3.1	±2.9	±7.6	±9.8	±1.85	±0.011	±0.029
SB21	1.62	24.8	43.5	112.3	76.8	122.0	251.5	14.33	0.055	0.176
	±0.09	±2.5	±2.0	±6.7	±5.1	±8.0	±9.8	±3.09	±0.029	±0.042
SB26	1.85	33.3	40.9	109.3	76.3	125.3	252.4	14.10	0.059	0.129
	±0.30	±11.6	±0.7	±3.1	±3.4	±12.9	±10.4	±2.94	±0.006	±0.037
SB31	2.02	36.7	34.2	106.3	33.6	169.7	267.8	6.56	0.015	0.185
	±0.27	±9.9	±0.1	±3.5	±5.1	±5.0	±7.1	±1.23	±0.008	±0.025
SB36	2.09	28.9	33.1	99.3	45.7	118.5	251.5	7.97	0.094	0.143
	±0.29	±3.6	±6.2	±15.9	±13.7	±28.9	±28.6	±2.28	±0.030	±0.045

As these results represent the average urban recharge quality of the five locations they were used to estimate the different recharge volumes, particularly leakage from the sewage network. The volumes of each recharge source and the estimated range of total recharge is listed in Table 2. The respective water qualities are listed in Table 4. Rainwater quality and quality of supplied mains water in the research area is listed in Stuart et al. (2004). The natural background concentrations should be approximately 2 to 3 times the rainwater concentrations due to evapotranspiration [Walton 1981]. Uncertainties of rainwater quality are assumed to be ±10%. Note that the background alkalinity, observed in the deep intervals, is very low (Table 3) and waters are strongly under-saturated with respect to calcite (log SI ≈ -0.1 to –1.4) indicating a lack of calcite matrix. The recharge alkalinity is therefore assumed to be only in equilibrium with soil CO_2. The alkalinity of deep groundwaters (presumed background, unaffected by solutes in urban recharge) is about 60 mg/l so a rainwater concentration of about a third of that is assumed. Concentrations of foul sewage were analysed quarterly, with one of the four sampling rounds including more detailed sampling when 14 samples were taken at different times during a single day to gauge diurnal quality variations. Stormwater concentrations were only measured once because all other sampling rounds were during dry periods. However, the composition was similar to that observed in shallow piezometers at the Bolton Hill site, where the influence of road runoff was observed. Assumed uncertainties are 10% for all parameters except sodium and chloride where 20% is assumed in recognition of sporadic road salting during winter.

Table 4. Hydrochemical composition of different recharge sources.

[mg/L]	K^+	Na^+	Mg^{2+}	Ca^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	NO_3^-N	B	P
Rain	0.08 ±0.008	0.90 ±0.09	0.15 ±0.015	0.41 ±0.04	2.1 ±0.02	2.8 ±0.03	20 ±2	3.3* ±0.33	0.015 ±0.008	0.001 ±0.0005
DW	2.70 ±0.15	14.9 ±1.8	23.0 ±1.4	54.4 ±3.4	34.1 ±3.8	36.4 ±4.6	211.0 ±15.0	4.30 ±2.40	0.025 ±0.013	0.02 ±0.02
FS	20.50 ±2.40	100.3 ±13.2	23.8 ±0.3	52.0 ±1.7	78.6 ±12.1	87.4 ±11.1	519.0 ±62.0	47.40* ±16.50	0.42 ±0.08	11.2 ±1.9
SW	2.37 ±0.24	78.0 ±15.6	2.6 ±0.3	20.6 ±2.1	109.0 ±21.8	20.7 ±2.1	150.0 ±15.0	1.33 ±0.13	0.071 ±0.0071	0.23 ±0.023

*represents total amount of nitrogen (nitrate, nitrite and ammonium) as mg N/litre.

If Equation 4 is solved to obtain sewer leakage rates there are still two unknowns, namely the leakage from the stormwater and the foul sewage systems, even if the total recharge were known. Assuming that the percentage of leakage from these two systems is similar, a total leakage volume from the urban drainage system can be calculated for each interval of the multilevel piezometers. This was done for two rates of total annual recharge of 150 and 200 mm/y respectively (Table 5). The calculation of total sewer leakage was undertaken for K, Na, Cl, HCO_3 and B and the results averaged for each sampling interval. SO_4, Ca and Mg are not considered in the final overall averaging process because they potentially underlie congruent dissolution of gypsum [Cronin et al. 2005], making them non-conservative and therefore do not fulfil the major prerequisite of this final operation. Note that congruent dissolution of gypsum would not affect the alkalinity even though gypsum is dissolved and calcite precipitated in the process known as dedolomitization.

The resulting amounts of sewerage admixture (from foul sewer and stormwater) are listed inTable 5. The corresponding uncertainty of the estimation when using a single parameter only was of the order of 30-50%. Combining multiple parameters lead to standard deviations of about 15-25% (Table 5). The high concentrations found in the medium levels of Bolton Hill site (L28, L34 and L39) should be treated with care because the signal found there indicate an unknown additional source of chloride and sulphate, potentially due to road salting. The table shows that even the deepest levels containing the oldest groundwater seem to contain between 0 and 20mm/y of sewage, corresponding with 1-7% of total urban drainage. The shallower wells, however, contain amounts typically between 20 and 50 mm/y of sewage, corresponding with 5-12% of total urban drainage or 4-8·10-5 l/s and metre of pipe. This is more than previously assumed but it corresponds with findings from Liverpool, where sewer leakage was estimated to be of the same order as pressurized mains leakage [Howard 2001]. Overall, comparing the found rates per metre of pipeline with other studies, they are whether particularly high as found in studies in Germany, the Czech Republic or the USA nor extremely low as found in another German study. The considerable influence of sewer leakage on urban recharge in this study is a result of the considerable length of the separate sewage network in this area.

Table 5. Estimated total leakage volumes from both foul and stormwater system (i.e. both contributing with equal percentage of total flow) assuming 150 and 200 mm/y total urban recharge.

[mm/y]	**Bolton Hill**						
	L16	**L22**	**L28**	**L34**	**L39**	**L45**	**L51**
150mm	31.7±5.5	36.3±5.9	49.6±6.8	30.1±5.0	30.5±5.2	19.0±3.9	11.8±3.0
200mm	43.6±7.3	49.6±7.8	67.4±9.0	41.3±6.6	41.7±6.9	26.4±5.1	17.0±3.9
	Haslam Park 1						
	L10	**L14**	**L21**	**L28**	**L35**	**L45**	**L60**
150mm	12.5±3.3	23.2±3.8	26.4±4.3	34.2±5.1	19.2±3.8	6.4±2.2	13.6±4.1
200mm	17.7±4.3	32.0±5.0	36.5±5.6	46.9±6.7	26.9±5.0	9.8±2.9	19.2±5.4
	Haslam Park 2						
	L10	**L14**	**L21**	**L28**	**L35**	**L45**	**L60**
150mm	4.4±2.8	13.7±3.5	14.6±3.3	12.3±3.2	8.8±2.5	3.8±1.8	7.5±3.2
200mm	7.2±3.7	19.4±4.6	20.6±4.3	17.5±4.2	12.8±3.3	6.4±2.4	11.4±4.2
	McAuley School						
	L9	**L14**	**L21**	**L28**	**L36**	**L45**	**L60**
150mm	26.0±6.4	30.7±4.8	24.8±4.5	17.4±3.8	24.5±4.2	19.6±3.7	12.6±3.5
200mm	36.2±8.3	42.3±6.3	34.0±5.9	24.5±4.9	34.0±5.5	27.5±4.8	18.1±4.6
	Sandall Beat						
	L16	**L21**	**L26**	**L31**	**L36**		
150mm	21.9±3.6	13.7±2.9	18.0±4.2	17.6±4.0	20.6±4.2		
200mm	30.3±4.8	19.2±3.8	25.0±5.5	24.6±5.3	28.6±5.6		

The pipe leakage model to be applied in the AISUWRS project will compare present days renewal strategy of about 0.1 %/year (OFWAT 2004) with other possible rates (e.g. 0.5%/year) to assess potential future consequences. Leakage rates estimated in this study will be used to calibrate the model and decrease its uncertainty. Subsequently, the model suite of the AISUWRS project will assess the impact of different scenarios on local groundwater resources and potential corresponding monetary and social costs.

It was mentioned above that the estimation of average leakage rate (Equations 8-10) is based on a weighting of all parameters. Subsequently, this weighting can also be used to define the usefulness of each hydrochemical parameter involved in the calculation. This approach is valid because it uses the differences between the various sources and their uncertainties. Results for each multilevel were ranked from 1 to 8 and then averaged over all 33 samples. Sodium and potassium were found the most useful parameters in this study. Chloride and alkalinity are in a medium ranking and boron, calcium and magnesium are of low significance even though boron was found to be a useful qualitative tracer [Cronin et al. 2005]. However, its analytical detection limit is close to observed sample concentrations and this significantly decreases its value as a quantitative tracer for sewer leakage. By far the least useful tracer was sulphate due to natural sources and also potentially due to the large amount of sulphate found in urban areas as a result of waste building materials [Howard 2001].

The results presented above can be further used to analyse this problem and the observed large attenuation of phosphorous amounts to estimate nitrogen and phosphorous storage in the soil zone. Applying equation 3, it was found that 90-100% of nitrogen and phosphorous originating from leaking sewage is stored in the soil. Nitrogen is stored so efficiently because ammonium, which is the major form of nitrogen in foul sewage, is relatively immobile in soils. Leaching occurs only when bound ammonium is nitrified to nitrite and nitrate, both quite mobile in natural soils. The lack of nitrification could be due to the fact that sewage is actually released below the root zone where ammonium could be effectively nitrified and nitrate could be used by plants. Furthermore, even though the unsaturated zone is thought to be saturated with oxygen, sewer leakage could produce locally reducing conditions preventing ammonium from oxidizing to nitrite and nitrate. The results indicate storage rates of about 1000mg/m^2/y nitrogen and 200mg/m^2/y phosphate. If these amounts would become remobilized due to increasing groundwater levels nitrate and possibly phosphorous concentrations could rise remarkably. However, to allow a safer conclusion to the observations the storage of these two parameters had to be studied in more detail (e.g. hydrochemical modelling, detailed field-study of single sewer leaks).

Conclusions

An urban water budget method has been applied to estimate sewer leakage proportions for a 6 km^2 research catchment comprising a suburban area of Doncaster, UK. Using estimated total urban recharge rates of 150 and 200 mm/y, the resultant values ranged from 5 to 12% of total sewer volumes. These are in the same range as volumes found in other studies in Germany, the UK and the USA. The calculations suggest that the contributions to recharge from sewers are equivalent to 28mm/y from foul sewer, 12mm/y from stormwater and 22mm/y from mains water. The results suggest that approximately 30-40% of total recharge in this urban study area may be coming from losses from the pipe network, with the balance coming form other forms of infiltration.

The statistical approach used in this study enables an independent ranking of the value of different hydrochemical parameters used to estimate sewer leakage. This ranking found sodium and potassium, followed by chloride and alkalinity to be the best tracers. Sulphate, a tracer that was found to be useful in other studies proved to be the least useful in Doncaster to quantify sewage-derived recharge.

The results show the usefulness of multilevel piezometers to assess leakage rates from sewage systems. This is because samples represent only a very narrow depth-interval of groundwater. This, in turn, enables depth stratification to be identified and interpreted.

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2.3.6 Paper 6: Assessing the impact of modern recharge on a sandstone aquifer beneath a suburb of Doncaster, UK (Morris *et al.*)

Key findings

- Using a variety of sample sources including multi-level boreholes, the study found detectable CFCs and SF_6 throughout the upper 50 m of the saturated aquifer beneath the study area, indicating that modern (<50 year old) recharge has penetrated to at least this depth.
- The persistent detection of low counts of sulphite-reducing clostridia and faecal streptococci support the inference of deep penetration of recharge.
- Excess CFC concentrations at depths of <30 mbgl indicate local sources of enrichment
- In general the evidence suggests the mixing of groundwater rather than piston flow displacement, but the flow system in the sandstone is complex, with modern recharge travelling both via fractures and matrix
- The evidence from standard hydrochemical indicators was less conclusive. Markers such as Cl, SO_4 and NO_3 that have been successfully used as urban recharge indicators elsewhere (and so were chosen initially) showed significant variability between sites. Although most sites showed higher solute concentrations in their upper zones, there was no consistent pattern. However, other inorganic indicators such as K, Na, HCO_3 and B were typically elevated at shallow depths.
- A number of pollution indicator species show little more variation than that encountered in neighbouring rural catchments, confirming that in hydrochemical and microbiological terms, the adverse effect of urban recharge on the measured chemical quality parameters of the underlying groundwater in the Bessacarr-Cantley area has been limited.
- This is ascribed to the combined effects of a non-industrial prior land-use history, some sewer leakage and urban runoff, locally high storage capacity in the friable upper aquifer and particularly the availability of dilution from precipitation entering green space areas within the urban footprint

Introduction

A major challenge for water management in cities overlying productive aquifers lies in the complexity of the urban water balance compared with rural catchments (e.g. Lerner et al. 1990, Foster et al. 1994). Losses by leakage from the large volumes of water circulating within the pipe infrastructure (pressurised mains, foul sewers, combined sewers, pluvial drains and sometimes district heating systems), together with percolation from roof runoff/paved area soakaways, provide sources of near-surface recharge additional to those available in rural catchments. At the same time impermeabilisation of the land surface by buildings and paved areas changes the scope for local precipitation to enter the aquifer. The resultant intricate mosaic of at-surface and near-surface recharge sources complicates both the quantification of net recharge to the aquifer and the prediction of the effect such recharge may have on groundwater quality (e.g. Eiswirth and Hötzl 1994).

The application of environmental indicators of groundwater residence time has been investigated in and around a suburb of the South Yorkshire town of Doncaster (population 290,000), which is situated on and draws its water supply from a Permo-Triassic sandstone aquifer (Fig.1). The investigation formed part of a larger study that aims to provide and validate (through the medium of city case-studies) a linked array of models that can cope with the complexity of recharge to urban aquifer systems. A wide array of data are being used to inform the chemical and microbiological characterisation of urban shallow aquifer recharge needed to calibrate the numerical models that together are being used to track water (from precipitation and mains supply) and contaminants (from human activities) through the built environment, to the underlying unsaturated zone and on to the underlying aquifer. (Eiswirth et al. 2002, University of Karlsruhe 2005). Implicit in this approach was the expectation that, in comparison with the rural equivalent, recharge in the urban environment could be characterised by marker species, resulting in a recognisable groundwater ‘signature’ that would allow the extent and likely effect of urban recharge to be estimated.

Background

URBAN WATER INFRASTRUCTURE

The Sherwood Sandstone aquifer, which is the second most important in the UK after the Chalk, is part of a more extensive European Permo-Triassic Bunter and Lower Keuper red-bed sandstone sequence which also forms productive aquifers elsewhere in northwestern Europe.

Figure 1. Location of Doncaster and Bessacarr-Cantley study area on Permo-Triassic sandstone aquifer

The study area comprises the Doncaster suburb of Bessacarr-Cantley, located approximately 3 km southeast of the town centre. The district, with a population of c. 20,000 has urbanised intermittently since the early 1920s and comprises a mix of residential property and local services (schools with playing fields, retail, community buildings, green space). Town planning controls have kept the district geographically distinct, and both the urban footprint and, as Fig. 2 indicates, its associated water infrastructure of mains supply, wastewater and pluvial drains are well-defined (Morris et al. 2003).

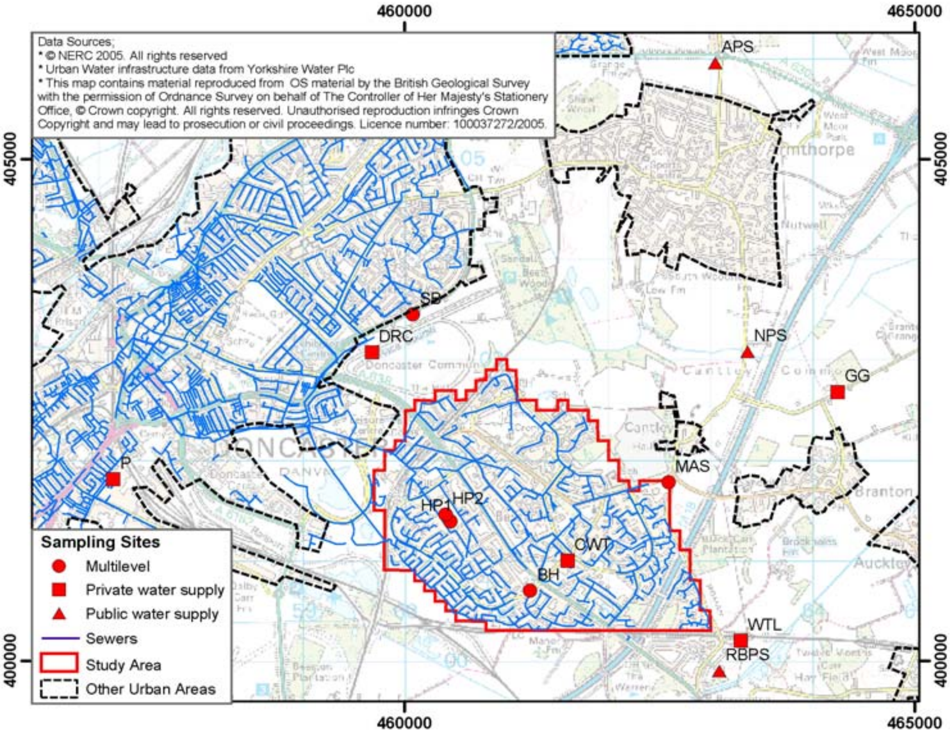
Data Sources;
* © NERC 2005. All rights reserved
* Urban Water infrastructure data from Yorkshire Water Plc
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Figure 2. Urban water infrastructure of Bessacarr-Cantley study area and sampling locations

The piped water supply for the town of Doncaster, its suburbs and surrounding rural area is supplied by the Doncaster wellfield, a linked array of eleven pumping stations extending from just to the east of the town along a 15 km arc to the northeast and southeast. The 6.3 km^2 extent of Bessacarr-Cantley is served by a total length of water bearing pipe infrastructure of almost 220 km, via iron or plastic water mains and vitreous clay or concrete foul sewers and pluvial drainage systems (Rueedi et al. 2004) (Table 1).

Table 1. Pipe infrastructure key statistics for Bessacarr-Cantley

Pipe network type	Pipe asset count	Total length (km)	Materials comments
Mains supply	1135	91.6	84 % by length cast or ductile iron, 15 % PVC/PE
Sewer-foul & combined	1205	56.9	87 % by length vitrified clay, 12 % concrete
Drain-pluvial (stormwater)	1413	71.1	47 % by length vitrified clay, 53% concrete
Totals	3753	219.6	

GEOLOGY

The Permo-Triassic Sherwood Sandstone Group of eastern and northern England outcrops in a structurally controlled arc from south of Nottingham to the North Sea at Hartlepool, Co. Durham. The sandstones in the vicinity of Doncaster have an outcrop width of about 16 km and dip gently towards the east-north-east at about 1.5°, being underlain by low permeability Permian marls on the west and overlain by Triassic mudstones to the east (Figs. 1 and 3).

In South Yorkshire the Sherwood Sandstone has little topographic expression apart from isolated and subdued ridges on its western (basal) margin. The aquifer increases in thickness from its western edge, reaching about 175 m to the east of Doncaster where the suburbs and nearby former coal mining villages are located. Quaternary superficial deposits ranging from glacial sand-and-gravel to peat and lacustrine silty clays overlie the sandstones in many places and these can exert a major control on recharge processes, flow patterns and solute/contaminant transport (Smedley and Brewerton 1997).

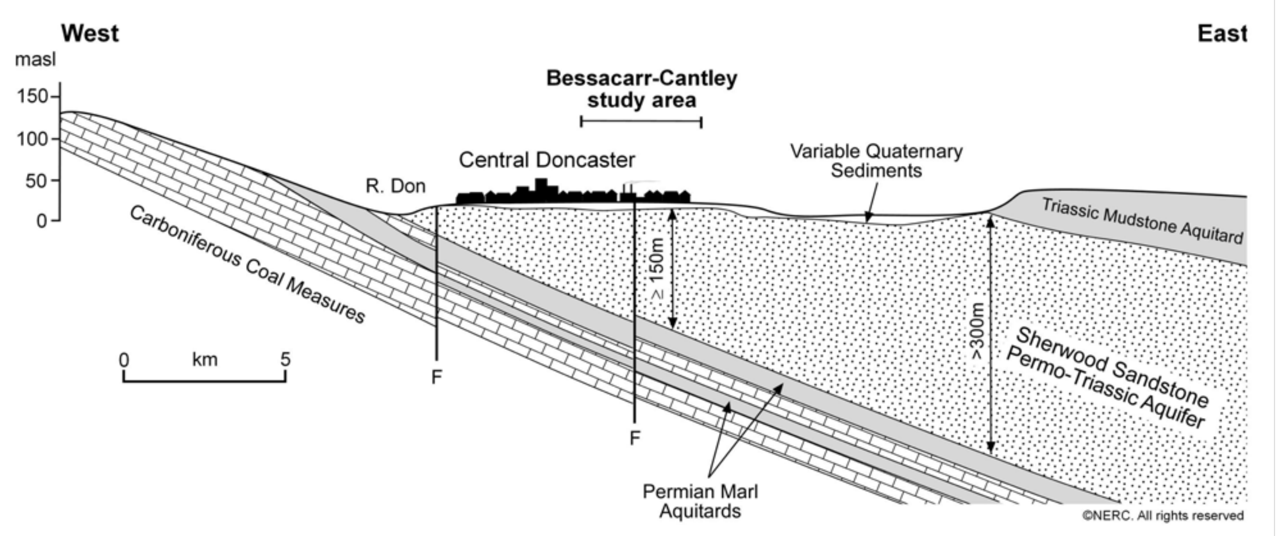

Figure 3. Sketch section across Sherwood Sandstone aquifer in vicinity of Doncaster

The fluvially deposited Sherwood Sandstone Group comprises a varied series of red and brown, friable to moderately-cemented, well to poorly-sorted and fine to medium-grained sandstones (Gaunt 1994). Thin layers and lenses of mudstone and mud-pellet conglomerates are present in the lowest 40 m of the aquifer but less common higher in the sequence. Bessacarr-Cantley is underlain either directly by the sandstone aquifer or by intervening permeable Quaternary sands and gravels up to 8 m thick (Fig. 3). The absence of low-permeability superficial deposits was one criterion in the selection of Bessacarr-Cantley as the study area.

HYDROGEOLOGY

Regional transmissivities of the unconfined Sherwood Sandstone aquifer, derived from pumping tests, lie in the range 100–700 m^2/d with a median 207 m^2/d (Allen et al. 1997). Intergranular porosity measured from core samples is typically around 30%. Specific yield values of around 0.1 are cited from both laboratory measurements and calibrated regional flow models but may be locally an underestimate for the South Yorkshire area due to the poor cementation of the upper part of the saturated aquifer. More locally, permeability tests using inflatable packers conducted by the study team in a 59 m deep borehole (CWT) in Bessacarr gave transmissivities of 76 m^2/d for the upper open-hole zone (27–41 mbgl) and 92 m^2/d for the lower zone (41–59 mbgl).

Regionally, the Sherwood Sandstone aquifer is considered to be strongly anisotropic as a result both of synsedimentary features (interbedded mud-rich horizons, presence of fining-upwards cycles, channelling) and post-diagenetic structural developments (bedding plane fractures, inclined joints, faults). Intergranular flow is believed to dominate in regional flow systems because fractures are often filled with sand. However, interconnected systems can become well-developed near boreholes as a result of prolonged pumping (Allen et al. 1997). Flow along discontinuities in such areas, especially in the top 100 m of the aquifer has been shown to play a significant role in water movement (Price et al. 1982, Jackson and Lloyd 1983, Allen et al. 1997). A recent detailed field study at sites in Nottingham and Birmingham (Taylor et al. 2003), where the Sherwood Sandstone is well-cemented, confirmed that such features exert significant control over the vertical flow component.

There is evidence that east of Doncaster much of the Sherwood Sandstone aquifer may be less indurated than equivalents elsewhere. Although the lowest 40 m forms a discontinuous series of low ridges, exposures and core recovery are poor elsewhere (Gaunt 1994) and recent drilling experience in the Bessacarr area has shown much of the upper part of the sandstone sequence to be poorly cemented (Rueedi & Cronin 2003). This local effect is likely to provide higher storage than the cited regional specific yields suggest. The subdued, near sea-level elevations of most of the Sherwood Sandstone east of Doncaster resulted in wetlands until the early 20th century. This contrasts with further south along the strike of the formation, where the sandstone outcrop is sufficiently well cemented to form relatively high ground e.g. around Nottingham.

The subdued topography around the study area makes assessment of the natural, pre-development flow system speculative. Brown and Rushton (1993) suggest that groundwater would probably have drained from high recharge areas (drift-free or with permeable drift) in the centre and south west of the area, towards the east and north. If so then Bessacarr-Cantley would historically have comprised a low eminence draining outwards to the east, north and south towards encircling wetlands underlain by a full aquifer with very shallow flow systems discharging to local watercourses.

Piezometric data over a 13 month period October 2003–November 2004 from five multilevel boreholes suggest that vertical hydraulic continuity exists throughout the upper part of the aquifer system. The measurements show that head gradients in the upper 50 m of the aquifer are very low throughout the study area, typically only a few centimetres, and that the small head differences are maintained in synchronous fashion both during periods of recharge and of recession and at depths to 60 mbgl (Fig. 4, Table 2).

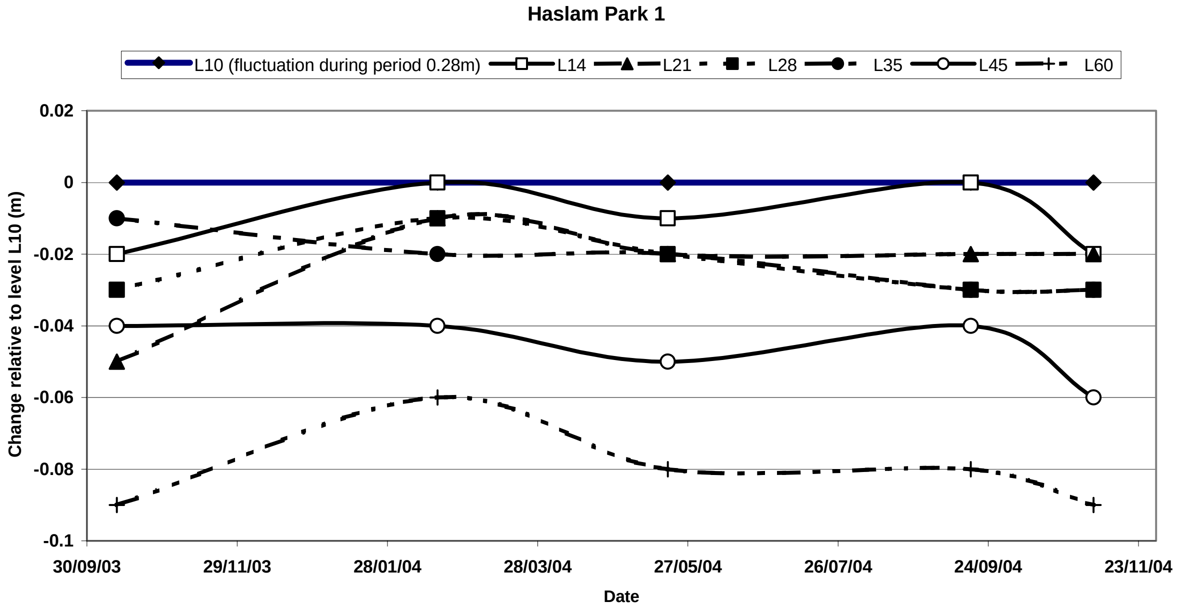

Figure 4 Piezometric variation in HP2 multilevel. L10, L14 etc refer to depth of sampling port. For comparative purposes the piezometric level in L10 (which fluctuated 0.28 m over period) has been normalised. Note synchronicity of variation at all ports down to 60 mbgl and small magnitude of relative head difference.

Table 2. Piezometric variations in five multilevels in Bessacarr-Cantley study area October 2003-November 2004

Multilevel site	Port depth (m)	23/10/03	17/02/04	19/05/04	17/09/04	05/11/04
Haslam Park 1	10	0	0	0	0	0
	14	0.01	0	–0.01	0	–0.02

Multilevel site	Port depth (m)	23/10/03	17/02/04	19/05/04	17/09/04	05/11/04
	21	0.02	–0.01	–0.02	–0.02	–0.02
	28	0.02	–0.01	–0.02	–0.03	–0.03
	35	0.02	–0.02	–0.02	–0.03	–0.03
	45	0.05	–0.04	–0.05	–0.04	–0.06
	60	0.08	–0.06	–0.08	–0.08	–0.09
	Max.head diff. (m)	*0.08*	*–0.06*	*–0.08*	*–0.08*	*–0.09*
Haslam Park 2	10	0	0	0	0	0
	14	0.01	–0.01	–0.01	0	0
	19	0.09	0	–0.09	0	–0.01
	27	0.04	–0.01	–0.04	–0.02	–0.01
	35	0.03	–0.03	–0.03	–0.02	–0.01
	45	0.03	–0.03	–0.03	–0.03	–0.03
	61	0.05	–0.04	–0.05	–0.07	–0.05
	Max.head diff. (m)	*0.09*	*–0.04*	*–0.09*	*–0.07*	*–0.05*
Bolton Hill	16	0	0	0	0	0
	22	0		0	–0.01	–0.01
	29	0		0	–0.01	–0.02
	35	0.01		–0.01	–0.01	–0.02
	40	0.01	0.02	–0.01	–0.01	–0.02
	45	0.02	0	–0.02	–0.01	–0.02
	51	0.01	0	–0.01	–0.01	–0.02
	Max.head diff. (m)	*0.02*	*0.02*	*–0.02*	*–0.01*	*–0.02*
Sandall Beat	16	0	0	0	0	0
	21	0	0	0	0.01	0
	26	0.01	0.01	–0.01	0.01	0
	31	0.01	0.01	–0.01	0.01	0
	36	0.02	0	–0.02	0	0
	Max.head diff. (m)	*0.02*	*0.01*	*–0.02*	*0.01*	*0*
McAuley School	9	0	0	0	0	0
	14	0	–0.01	0	–0.01	–0.01
	21	0	–0.01	0	–0.03	–0.02
	28	0.01	–0.01	–0.01	–0.02	–0.01
	36	0.01	–0.01	–0.01	–0.04	–0.01
	45	0.02	–0.02	–0.02	–0.04	–0.03
	61	0.02	–0.01	–0.02	–0.02	–0.03
	Max.head diff. (m)	*0.02*	*–0.02*	*–0.02*	*–0.04*	*–0.03*

ENVIRONMENTAL INDICATORS

Any study of the penetration rate of modern recharge in an urban setting demands the most sensitive possible indicators of anthropogenic activity. CFCs in particular make excellent indicators owing to the purge-and-trap preconcentration step that must be undertaken before they can be analysed at water-dating concentrations (Bullister and Weiss, 1988). However, in an urban setting, the dating of waters by CFCs can prove difficult or impossible because atmospheric input concentrations can become contaminated by underground sources of CFCs such as industrial or municipal waste disposal sites (Hohener et al. 2002). Furthermore sources releasing CFCs into the atmosphere are often found (Oster et al. 1996) in or near urban areas. Under such circumstances, another trace gas resulting from post-war industrial activity, SF_6, has been found to be useful because it is more rarely found dissolved in groundwater at above-modern concentrations (e.g. Darling et al. 2005). These atmospheric trace gases therefore potentially have a dual use in groundwater studies: as dating agents, or as indicators of contamination fronts. Further refinements in interpretation involve the use of CFC-SF_6 plots to distinguish mixing from piston flow in uncontaminated groundwater, or employing CFC ratios to identify discrete flowpaths and/or contamination sources (Busenberg and Plummer 1992).

Most major and minor inorganic species are likely to be less sensitive indicators of modern recharge, even in the urban environment. As well as often being difficult to quantify the source term, there are also the issues of concentration overlap with natural occurrence in the rock matrix and overlap with diffuse agricultural pollution indicators (Cl, N, K, SO_4) present from the period prior to urbanisation.

Sampling and analysis

LOCATIONS

Environmental indicators were sampled at 13 locations in and around the study area (Fig. 2). Five multilevel research boreholes (36.0 – 60.4 m deep) were used for depth-specific sampling, while five relatively shallow private wells (30.5 – 76.0 m deep) and three deeper public supply boreholes (147 – 168 m deep) provided depth-integrated samples from open-hole sections or long-screened intervals.

Table 3. Sampling site description including land use around wellhead

Site Ref	Site type	Site Name	Tot.depth (m)	Depth to screen top or sampling port	Land use at wellhead
1	Multilevel research b/h	Sandall Beat	36.0	15.6, 21.0, 26.0, 31.0, 36.0	Grassed public playing field downgradient of 1930s housing estate
2	Multilevel research b/h	Haslam Park 1	60.1	10.0, 14.5, 21.0, 28.0, 35.0,45.0, 60.1	Public garden surrounded by 1919-1970 substantial detached houses with gardens
3	Multilevel research b/h	Haslam Park 2	60.4	9.5,14.0,19.0, 27.0, 35.0, 45.0, 60.4	Public garden surrounded by 1919-1970 substantial detached houses with gardens
4	Multilevel research b/h	Bolton Hill	51.4	16.6, 22.2, 28.7, 34.7,39.7,45.7,51.4	Grassed public playing field surrounded by mixed detached houses with gardens
5	Multilevel research b/h	McAuley School	60.1	9.5, 21.0, 28.0, 35.0, 45.0, 60.1	School playing field surrounded by open land or school buildings
6	Private water supply b/h	Doncaster Racecourse	41.1	NA	Grass parkland
7	Private water supply well	Pegler Ltd	30.5	5.2	Brass foundry and factory in mixed industrial and 19th/20th century inner city
8	Observation b/h	Cantley Water Tower	58.9	27.2	Grassed property enclosed by detached /semi-det./terraced post-1950 housing
9	Observation b/h	Warning Tongue Lane	63.4	18.3	Paddock on rural outer edge of post-1980 housing
10	Private water supply b/h	Gatewood Grange	76.2	NA	Rural property
11	Public water supply b/h	Nutwell PS	152.4	33.0	Periurban multi-borehole complex, rural in immediate vicinity
12	Public water supply b/h	Armthorpe PS	167.6	30.5	Periurban multi-borehole complex, rural in immediate vicinity
13	Public water supply b/h	Rossington Bridge PS	147.0	28.8	Periurban multi-borehole complex, large suburban properties in immediate vicinity

The multilevel boreholes each comprise a bundled piezometer array of small-bore monitoring wells with 0.30 m long screens set at various depths, a medium-grained sand pack being placed in each interval around the centrally located screen port, and with each depth range separated from the adjacent interval by a 1 – 3 m thick bentonite clay seal. The resultant multilevel boreholes permit saturated zone sampling of up to seven different depths, typically between 10 – 60 mbgl (Rueedi & Cronin 2003) (see Fig. 5). The direction and angle of dip of the Sherwood Sandstone means that stratigraphically the five multilevel boreholes are offset in an east-north-easterly direction. Thus, after allowing for elevation differences, boreholes HP1, HP2 and SB penetrate to approximately the same stratigraphic level, but the equivalent horizon would be at 18.3 and 51.6 m depth respectively in multilevels BH and MAS.

The private wells are poorly documented but available details confirm that these abstract mixed waters from within the uppermost 70 m of saturated aquifer. The public supply boreholes overlap the depth range of both multilevels and private wells, drawing water over long screened sections from <30 to >145 mbgl. Samples were taken from dedicated raw water sampling taps in the pumping stations.

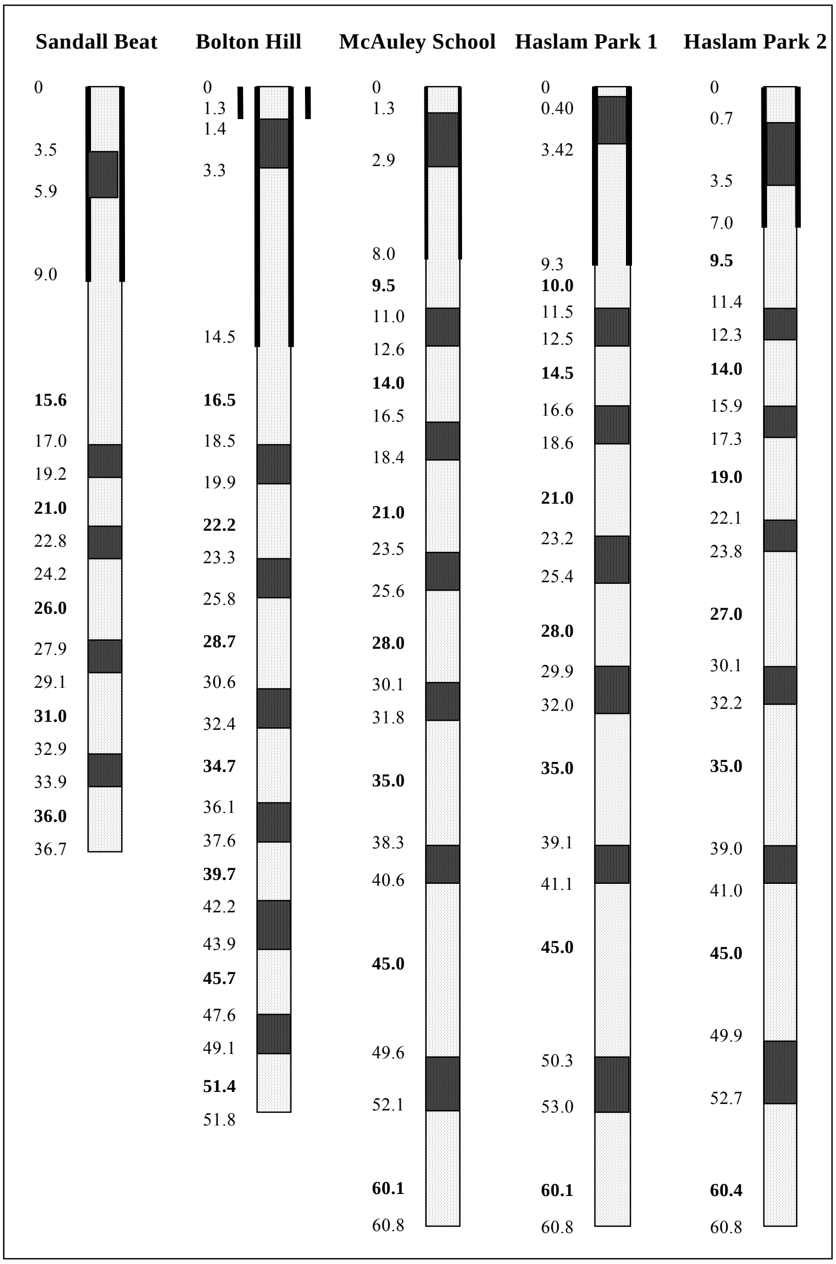

Figure 5. Design of 5 multilevel research boreholes, Bessacarr-Cantley; thick lines indicate steel casing, dark areas the location of bentonite seals, numbers in bold the depths of 0.3 m long sampling port centres (from Rueedi and Cronin 2003)

SAMPLING AND ANALYSIS

CFC, SF_6 and other hydrochemistry: CFC and SF_6 samples were collected in autumn 2004. Samples for CFC analysis were collected by the displacement method of Oster et al. (1996), without atmospheric contact in glass bottles contained within metal cans, while samples for SF_6 were similarly collected in glass bottles with conically-lined caps, according to the method of the USGS (Busenberg and Plummer, 2000). Samples for other hydrochemical analysis were taken during sampling campaigns in October 2003 and February, May, and September 2004. These were filtered through 0.45 µm cellulose nitrate membranes and collected in pairs of HDPE bottles, one being acidified to 1% with concentrated Aristar® nitric acid. Samples from the multilevels were taken using either a peristaltic pump or a small-bore inertial pump and from the other sites using either the installed pumpset or a sampling pump.

Measurements were made at BGS Wallingford except where stated. CFCs and SF_6 were analysed by gas chromatography after pre-concentration by cryogenic methods. Cations, P and SO_4 were determined on acidified sample aliquots by ICP-OES. On the unacidified aliquots, nitrogen species and chloride were measured by automated colorimetry.

Microbiological: Faecal coliforms, total coliforms, faecal streptococci, sulphite reducing clostridia, coliphage, and enteric virus were used as indicators of faecal contamination. These were taken during sampling campaigns in July and November 2003, then in 2004 concurrently with the hydrochemical samples. Bacterial samples were collected directly in sterile bottles and stored in an on-site refrigerator before same-day transport to the laboratory while for thermo-tolerant coliform analysis, filtration and culture was commenced on-site using a portable incubator.

Thermotolerant coliforms (TTC), faecal streptococci (FS) and sulphite reducing clostridia (SRC) were isolated from 100 ml sample volumes using membrane filtration and selectively enumerated by culture on membrane lauryl sulphate broth (TTC), Slanetz and Bartley agar (FS) and perfringens agar (SRC) respectively (Anon. 1994). The results from all analyses were recorded as colony forming units (cfu) per 100 ml (membrane filtration). Enumeration of coliphage was determined by assay of 1 ml of sample using a double agar layer technique (Adams 1959). Two methods were employed for the analysis of enteric viruses (norovirus and enteroviruses) in sample eluates. Buffalo Green Monkey (BGM) kidney cells were used for the quantification of infectious enterovirus by plaque assay, both by the confluent monolayer and suspended cell culture methods (SCA 1995). Results of coliphage and enteric viruses are given as plaque forming units (pfu) per ml. Field blanks and randomly selected duplicates were used as control procedures for all sampling rounds at all sites. All field blanks were found to be free of bacterial or viral analytes.

Results

CFCS AND SF_6

All sites. Analyses for CFCs and SF_6 are given in Table 4. Detectable concentrations of each were found in all cases except for CFC-12 in Nutwell BH2 and one port in one of the multilevel boreholes (45 m in Haslam Park 1), indicating that modern (<50 year old) recharge has penetrated to several tens of metres below ground level. The multilevel boreholes all show a broadly similar distribution of results (Table 2). For the CFCs, this means relatively high concentrations of both CFC-11 and CFC-12 immediately below the water table at 5–10 mbgl, followed by a rapid decline towards the region of 40 mbgl. Three sites (HP1, HP2, MAS) also show a slight rise again towards the bottom sampler. SF_6 concentrations on the other hand tend to show much less variation, though there is generally an overall decrease with depth.

Table 4. CFC and SF$_6$.concentrations in Sherwood Sandstone aquifer east of Doncaster

Sample location	**Type**	***Depth (m)**	**CFC-12 pmol/L**	**±**	**CFC-11 pmol/L**	**±**	**SF$_6$ fmol/L**	**±**
Haslam Pk 1	M/l	10	5.50	0.14	12.77	0.32	1.88	0.19
	M/l	14	4.86	0.12	11.90	0.30	1.39	0.14
	M/l	21	4.37	0.11	10.48	0.26	1.06	0.11
	M/l	28	2.14	0.05	5.23	0.13	1.30	0.13
	M/l	35	0.09	0.00	0.90	0.02	1.04	0.10
	M/l	45	<0.02	–	0.47	0.01	0.81	0.08
	M/l	60	2.63	0.07	6.44	0.16	1.09	0.11
Haslam Pk 2	M/l	10	3.06	0.08	8.33	0.21	2.33	0.23
	M/l	14	4.12	0.10	14.93	0.37	0.91	0.09
	M/l	19	4.20	0.11	15.42	0.39	1.62	0.16
	M/l	27	3.43	0.09	12.72	0.32	1.78	0.18
	M/l	35	0.14	0.00	0.50	0.01	0.67	0.07
	M/l	45	0.05	0.00	0.40	0.01	0.89	0.09
	M/l	60	1.02	0.03	3.66	0.09	0.81	0.08
Bolton Hill	M/l	16	5.12	0.13	13.94	0.35	1.06	0.11
	M/l	22	4.85	0.12	12.98	0.32	0.97	0.10
	M/l	28	3.58	0.09	8.69	0.22	0.61	0.06
	M/l	34	3.53	0.09	1.74	0.04	0.85	0.09
	M/l	39	3.68	0.09	2.06	0.05	0.67	0.07
	M/l	45	0.33	0.01	1.47	0.04	0.80	0.08
	M/l	51	1.57	0.04	0.77	0.02		
Sandall Beat	M/l	16	4.25	0.11	2.65	0.07	3.98	0.40
	M/l	21	5.43	0.14	2.90	0.07	3.58	0.36
	M/l	26	5.29	0.13	2.35	0.06	3.61	0.36
	M/l	31	1.32	0.03	0.34	0.01	2.00	0.20
	M/l	36	2.20	0.05	2.63	0.07	2.24	0.22
MacAuley School	M/l	9	19.20	0.48	9.83	0.25	2.61	0.26
	M/l	14	10.34	0.26	11.84	0.30	1.55	0.15
	M/l	21	12.21	0.31	12.12	0.30	2.12	0.21
	M/l	28	4.78	0.12	26.69	0.67	1.25	0.13
	M/l	36	4.19	0.10	8.43	0.21	4.28	0.43
	M/l	45	0.13	0.00	1.00	0.02	0.28	0.03
	M/l	60	0.62	0.02	1.27	0.03	0.72	0.07
Racecourse	PrW	41	4.00	0.10	3.81	0.10		
Pegler Ltd	PrW	31	0.58	0.01	1.79	0.04	1.15	0.12
Cantley WT Open hole	PrW	59	5.35	0.13	12.45	0.31	0.82	0.08
Cantley WT UZ 27-41m	PrW	41	4.76	0.12	9.03	0.23	1.19	0.12
Cantley WT LZ 41-59 m	PrW	59	4.25	0.11	7.80	0.19	1.35	0.14
Warning Tongue Lane	PrW	63	3.64	0.09	13.81	0.35	1.82	0.18
Gatewood Grange	PrW	76	1.13	0.03	0.22	0.01	0.36	0.04
Nutwell PS-BH2	PWS	152	<0.02	-	0.75	0.02	0.31	0.03
Armthorpe PS	PWS	168	1.46	0.04	6.85	0.17	1.03	0.10
Rossington Br. PS-BH 1	PWS	147	0.45	0.01	1.76	0.04	0.97	0.10

The private and public water supply boreholes obviously extract waters from a much greater thickness of aquifer than the individual multilevel ports, with the resultant mixing meaning that the high, near-surface concentrations typical of the multilevel boreholes are not generally seen in their discharge except at Cantley Water Tower, where an inflatable packer was able to isolate the zone 27–41 mbgl.

Many of the CFC data exceed the maximum concentrations possible by equilibrium with average atmospheric ratios, and therefore cannot be used to date waters in a quantitative way (Fig. 6A). These are considered further below. The other data are plotted in the form of CFC-12 vs. SF_6 concentration (Fig. 6B). Also shown in Fig. 6B is the curve showing the expected composition of recharge last in contact with the atmosphere at any given time between 1960 and 2005 (atmospheric data from http://water.usgs.gov/lab/cfc/background/air_curve.html), based on an equilibration temperature of 10°C. In theory this curve can be used to distinguish between piston flow and mixing with old (>50 years) water (Plummer et al. 2001; Darling et al. 2005). It can be seen that rather few samples fall on or near either the piston flow or mixing lines. It has been proposed that urban areas may have atmospheric trace gas excesses (Oster et al. 1996, Ho et al. 1998, Santella et al. 2003), but the present study was unable to confirm this for the Doncaster area. However, about half the sites are reasonably closely associated with the mixing line or zone between the two lines, suggesting that groundwater mixing is an important process at least in this portion of the aquifer. Samples falling well above the mixing line apparently have an SF_6 excess; while a significantly higher-than-average excess air value is possibly responsible, it seems more likely to derive from contamination since they come from adjacent intervals in a single multilevel (SB) showing above-modern SF_6 concentrations in its upper section.

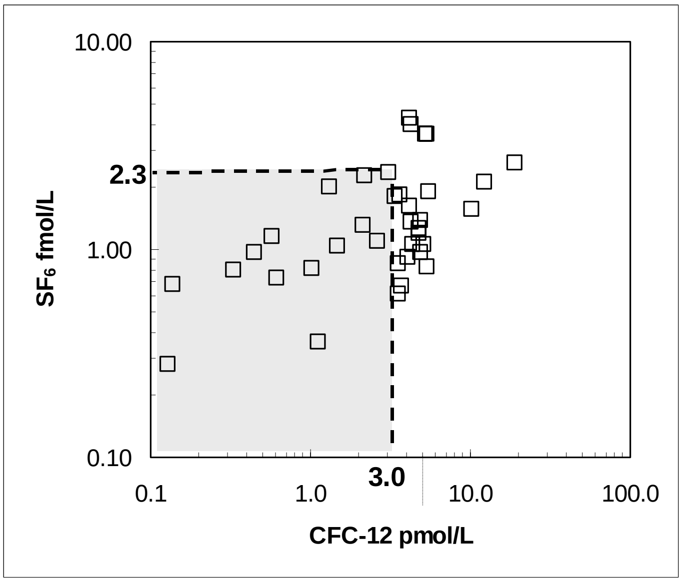

Figure 6A. CFC-12 vs. SF_6 concentrations, all samples. Dashed lines represent water in equilibrium with average 2004 atmospheric concentrations

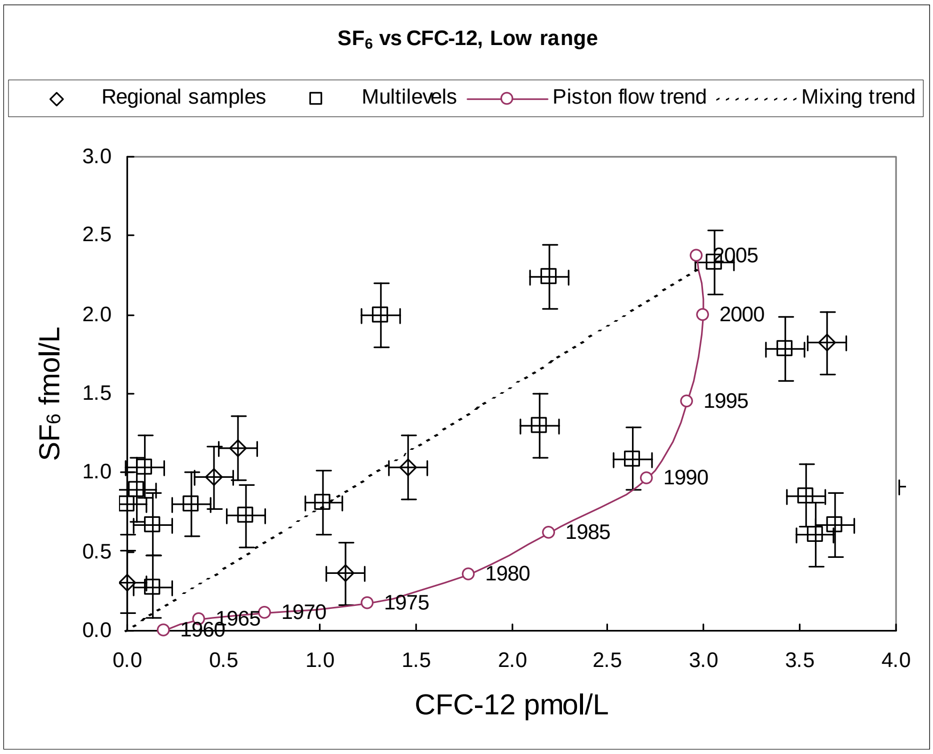

Figure 6B. CFC-12 vs SF$_6$ concentrations; subset of samples showing low or no enrichment from local CFC sources

Multilevel boreholes. On the evidence of Fig. 6, it appears that mixing between waters is more likely than the previously expected piston flow behaviour. The CFC and SF$_6$ data from these samples can therefore be converted into 'modern fraction' values, assuming the measured concentrations are the product of mixing between modern recharge and >50 year old 'dead' water (water containing no CFCs/SF$_6$). The resulting values are plotted vs. depth in Fig. 7.

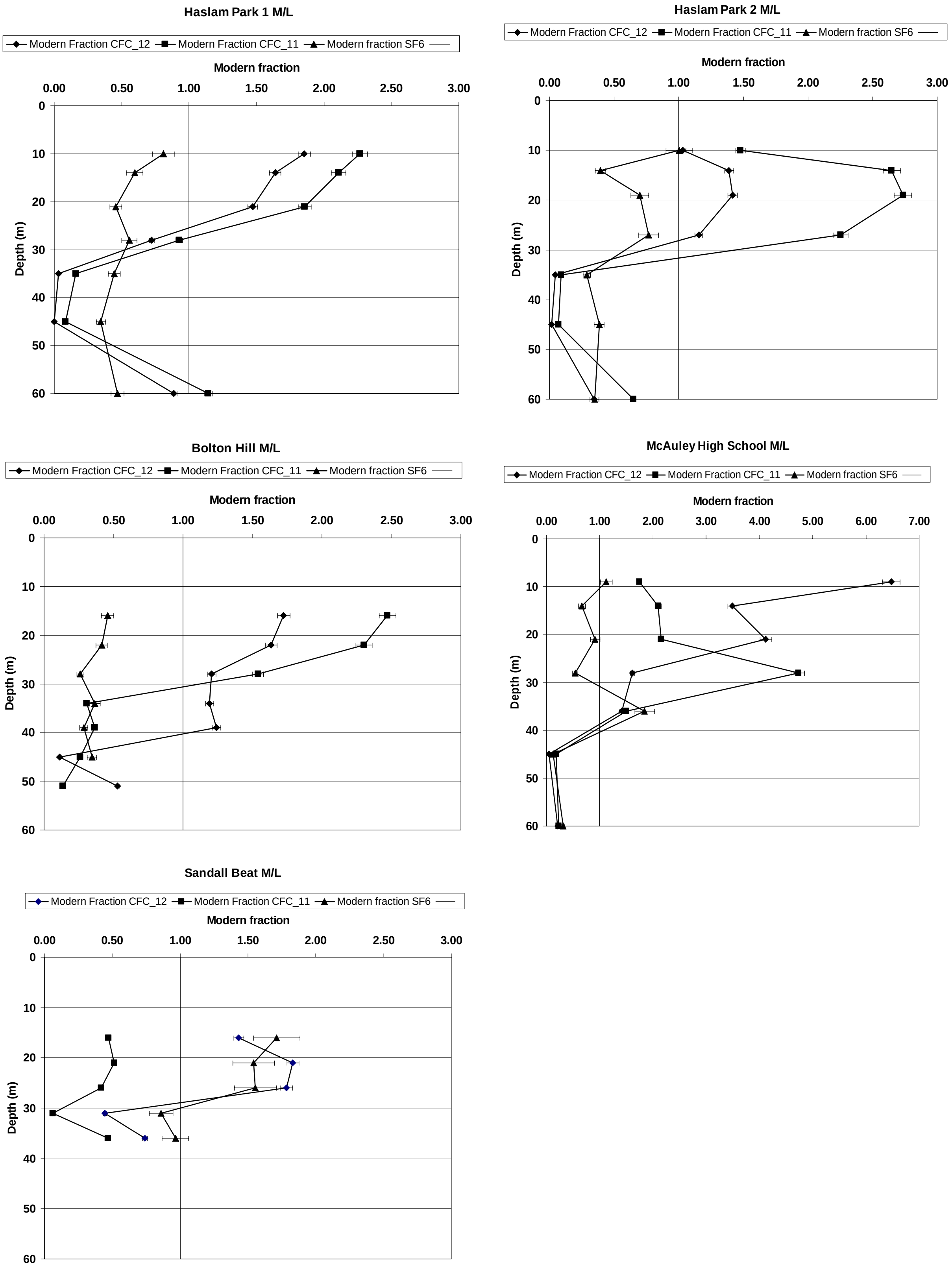

Figure 7. CFC and SF_6 depth plots in 'modern fraction' form

Four sites show above-modern CFC-11 and CFC-12 concentrations above 30 mbgl, indicating an element of enrichment from local sources (at the fifth site (Sandall Beat) CFC-12 alone shows this trend). With one exception (Sandall Beat again), the equivalent SF_6 profiles are much less affected, a feature that is also reported from other urban studies (MacDonald et al. 2003, Darling et al. 2005, Morris et al. 2005); this is attributed to its less widespread use in industry.

A clear feature of all of the profiles is the change to much lower CFC concentrations (and generally lower SF_6) below 30 m depth, usually followed by a rise in the deepest sampler. This implies the existence of preferential flow paths, an interpretation supported by the microbiological results (see section 4.3).

Ratios of CFC-11 to CFC-12 correlate well for two multilevels (HP1, HP2) and are fairly constant at a third (SB), indicating the importance of mixing/dilution, but are highly variable at the other sites (Fig. 8). A basic interpretation of this pattern is that these latter are either sites where additional sources have appeared over time, or that the catchments of each are much more heterogeneous in terms of additional sources. As the multilevel boreholes have a negligible catchment in the accepted sense (because they are not abstracting boreholes but instead are just intercepting throughflow on its way downgradient), the former interpretation seems the more likely, so that the observed concentrations are the product of several high-CFC sources interacting.

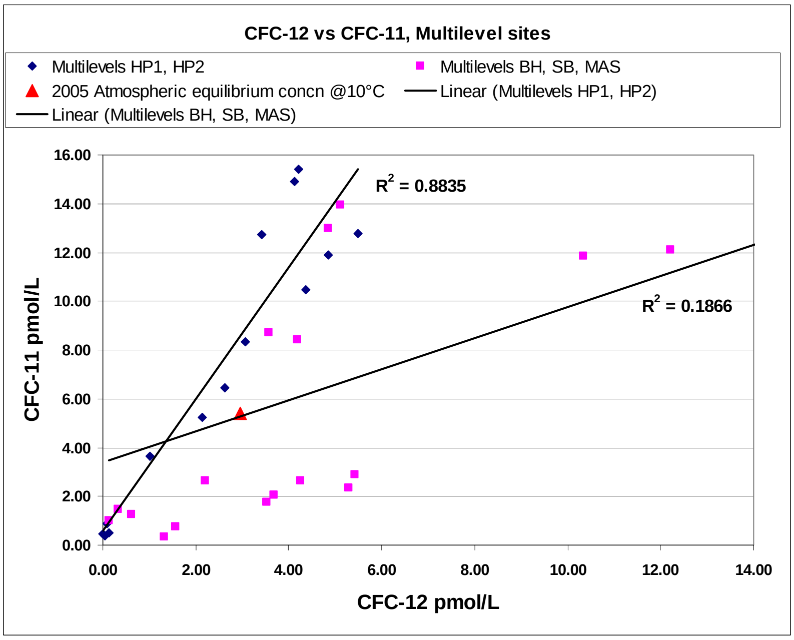

Figure 8. Crossplot of CFC-11 vs CFC-12 for multilevel sites

Supply boreholes. Table 5 shows the CFC and SF_6 data converted into modern fraction values and, alternatively, year of recharge assuming simple piston flow.

Table 5. CFC and SF_6 results expressed as modern fraction and as bulk age

Sample location	Type	*Depth (m)	CFC-12 Modern Fraction	CFC-11 Modern Fraction	SF_6 Modern fraction	CFC-12 Bulk age	CFC-11 Bulk age	SF_6 Bulk age
Doncaster Racecourse	PrW	41	1.35	0.68		>modern	1981	
Pegler Ltd	PrW	31	0.19	0.32	0.50	1969	1973	1992
Cantley WT Open hole	PrW	59	1.81	2.21	0.35	>modern	>modern	1988
Cantley WT Upper Zone	PrW	41	1.60	1.67	0.51	>modern	>modern	1993
Cantley WT Lower Zone	PrW	59	1.43	1.44	0.58	>modern	>modern	1994
Warning Tongue Lane	PrW	63	1.23	2.45	0.78	>modern	>modern	1999
Gatewood Grange Farm	PrW	76	0.38	0.04	0.15	1974	1960	1980
Nutwell PS BH2	PWS	152	0.00	0.13	0.13	<1948	1967	1979
Armthorpe PS (fm Nutwell)	PWS	168	0.49	1.21	0.44	1977	>modern	1991
Rossington Br. PS BH 1	PWS	147	0.15	0.33	0.42	1967	1972	1990

It is clear from Table 5 that, like the upper zone of the multilevels, the private boreholes are frequently ‘above-modern’ in their CFC concentrations, and therefore cannot be used as residence time indicators other than qualitatively, insofar as they indicate widespread penetration of modern water to these boreholes’ intake zones. Even the below-modern waters do not agree on grounds of either modern fraction or piston-flow age, suggesting that small amounts of CFC and/or SF_6 are being added from catchment sources. Agreement between the public supplies is rather better, although on a modern-fraction rather than bulk age basis.

Though there is as yet no evidence, the possibility cannot be wholly dicounted of locally higher-than-average atmospheric ratios, perhaps double, although the CFC ratios are not consistent between sites. This has been reported elsewhere and would have the effect of increasing groundwater ages by up to 20 years.

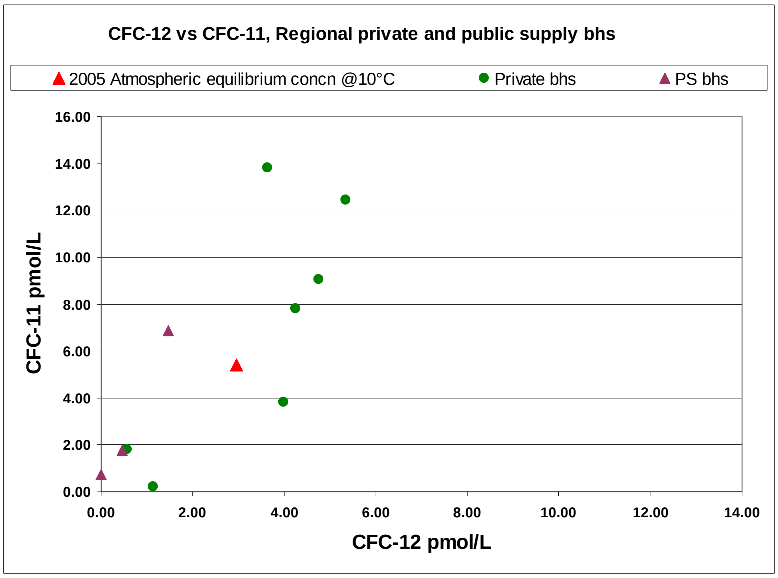

Figure 9. Crossplot of CFC-11 vs. CFC-12 for private wells and public supply boreholes in the vicinity of Bessacarr-Cantley

HYDROCHEMISTRY

Hydrochemical analyses and field measurements representing the HP2 multilevel on four occasions during an 11-month period are given in Table 6 together with selected profiles in Fig. 10. This multilevel provides the most internally consistent record of depth trends in water quality, but its features are also shared to a greater or lesser extent by the other multilevel sites.

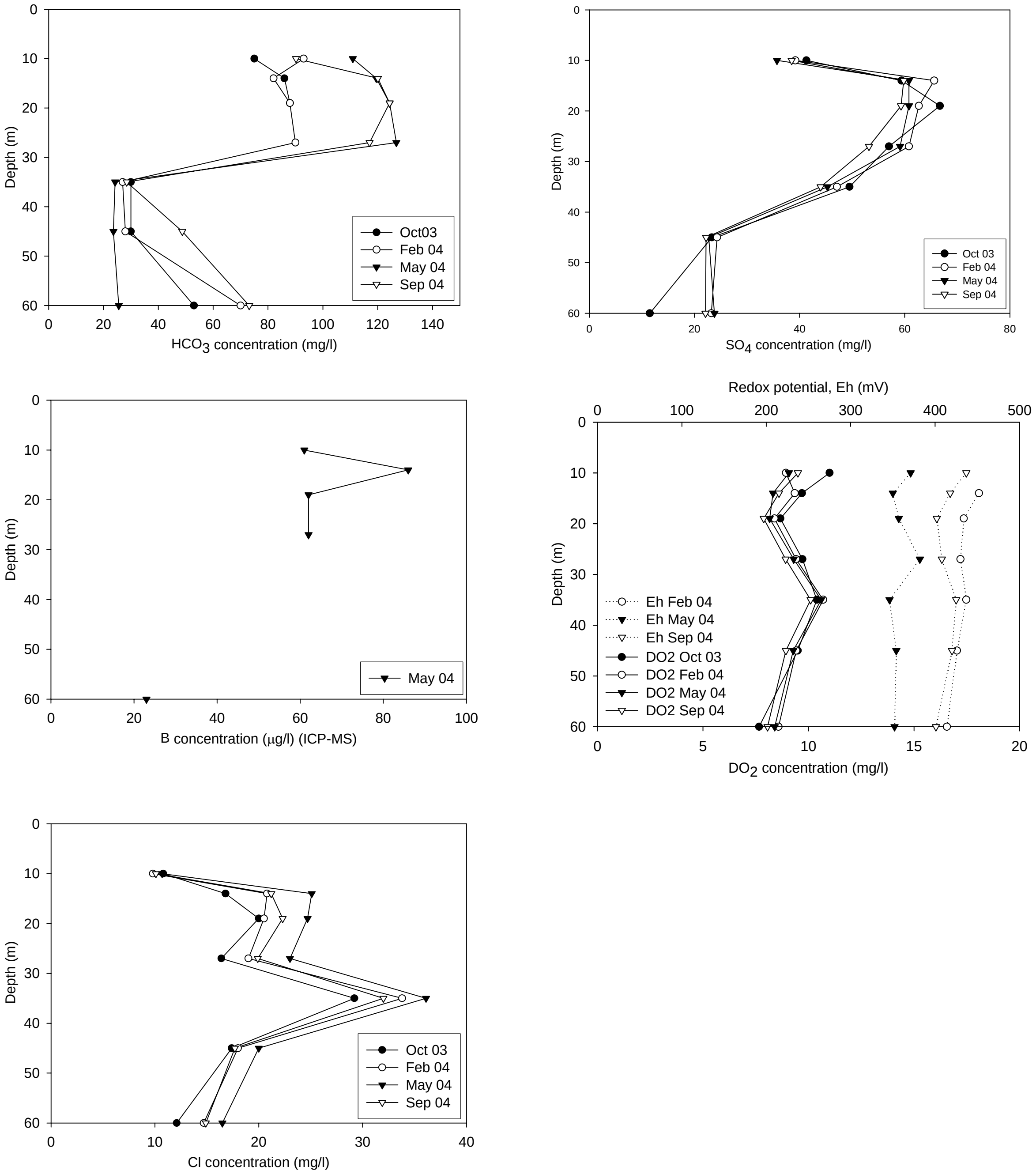

Figure 10. Urban recharge inorganic indicator profiles for HP2, October 2003-September 2004

Table 6. Hydrochemical data from HP2 multilevel, including major ion species and selected minor ion and field physicochemical measurements

		Major ions								Minor ions							Field measurements				
	Depth interval (m)	Ca	Mg	Na	K	HCO_3	Cl	SO_4	TON	Al	B [1]	B [2]	DOC	Fe	Mn	P	SEC	Temp	DO_2	Eh	pH
		mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	µg/l	µg/l	µg/l	mg/l	µg/l	µg/l	µg/l	µS/cm	˚C	mg/l	mV	
Oct 03	10	36.5	16.3	7.03	3.09	75	10.8	41.3	11	961	<80			697	33.5	110		11.6	7.08	382	6.9
	14	43.3	18.8	8.49	3.93	86	16.8	59.4	9.69	56.8	100			1.52	15.1	112		11.7		362	7.52
	19	49.1	19.9	9.49	4.44	88	20	66.7	8.67	75.2	<80			6.35	10.1	159		11.4	7	383	7.03
	27	43.1	18.5	8.51	3.78		16.4	57	9.71	178	<80			29.9	9.19	202		11.7	6.66	378	6.82
	35	35.4	11.8	10	1.98	30	29.2	49.5	10.4	703	<80			133	12.7	162		11.5	7.85	383	6.84
	45	24.4	8.95	6.72	1.29	30	17.4	23.3	9.49	1030	<80			206	6.54	172		11.5	7.69	374	7.22
	60	23.7	7.56	10.8	1.45	53	12.1	11.5	7.66	776	<80			162	10.7	184		11.2	8.25	369	7.31
Feb 04	10	32.2	15.1	7.61	6.29	93	9.8	39.2	8.93	222	84			320	18.4	119	409		5.59		7.83
	14	48.5	21	9.85	5.06	82	20.8	65.6	9.35	33.2	92			4.01	7.53	146	516	11.3	5.5	452	7.8
	19	47.7	19.5	8.94	4.92	88	20.5	62.7	8.39	33.2	80			4.8	6.53	164	507	10.9	6.02	434	7.72
	27	46.3	19.2	8.84	4.58	90	19	60.8	9.41	62.5	63			12.3	6.56	226	488	11	4.95	430	7.63
	35	36	12.5	8.77	2.12	27	33.8	47.1	10.7	77.1	<50			14	6.5	195	412	10.7	6.75	437	7.59
	45	25.8	9.53	6.42	1.34	28	18	24.3	9.42	154	<50			28	3.74	199	290	10.7	7.49	426	7.52
	60	28.7	10.7	7.54	1.91	70	14.7	23.2	8.58	294	64			67	6.6	255	312	10.1	6.26	414	8.18
May 04	10	31.8	15	5.69	5.23	111	10.7	35.7	9.07	110	88	61		44.5	22.6	28	319	10.8	8.84	371	7.81
	14	47.4	20.2	9.83	4.87	119	25.1	60.8	8.32	350	66	86		103	3.5	80	418	10.7	10.8	350	8.04
	19	48.3	19.9	8.98	5.11	124	24.7	60.8	8.16	44.5	98	62		8.36	3.3	112	500	11	8.4	357	7.91
	27	47.5	19.9	8.81	4.72	127	23	59.1	9.3	789	60	62		259	4.92	162	484	10.9	7.32	382	7.53
	35	35.3	12.6	9	2.16	24	36.1	45.3	10.6	459	31	< 20		94.6	4.21	168	403	10.8	8.89	346	7.65
	45	24.4	9.09	6.29	1.27	24	20	22.7	9.28	175	<30	< 20		35.2	2.42	150	280	11.2	9.43	354	7.7
	60	28	11.1	7.38	2.28	26	16.5	23.8	8.4	110	52	23		23.7	3.19	245	312	11.2	8	352	7.59
Oct 04	10	35.2	16	6.25	3.72	90	10.1	38.5	9.5	309	65		4.73	2520	47.7	98	387	11.3	8.1	437	7.55
	14	48.2	20.6	8.85	4.32	120	21.2	59.8	8.6	60	73		2.14	44.9	2.98	86	519	11.1	8.9	418	7.89
	19	48.9	19.8	8.81	4.48	124	22.3	59.3	7.88	42	67		2	39.4	2.38	93	515	11.3	8.9	402	7.88
	27	45	18.6	8.03	3.85	117	19.9	53.2	8.92	74	62		1.89	83.2	3.11	124	488	11	8.3	408	7.74
	35	36.1	12.8	9.14	2.27	29	32	44	10.1	108	<20		1.27	39.7	3.19	168	398	11	10.7	425	7.8
	45	24.4	9.23	6.57	1.51	49	17.7	22.2	8.93	1300	<20		0.77	307	2.24	155	288	11	11.3	420	7.91
	60	28	10.9	7.61	1.94	73	14.9	22.1	8.06	161	23		1.27	41.3	1.95	247	309	11.1	9.8	401	7.71

[1] Boron by ICP-OES, [2] Boron by ICP-MS

The concordance of most of the major ion profiles indicates consistency of results and little or no change between sampling visits. Dissolved oxygen (DO) values are consistently above 7 mg/l and often close to saturation at ambient groundwater temperatures of 10–12°C, while the redox potential (Eh) profiles, although more variable, also illustrate that the upper aquifer is aerobic throughout. Major ion profiles, although they demonstrate clear evidence of stratification, are not consistent in pattern. Several of the recognised inorganic markers of wastewater recharge (K, SO_4, HCO_3) reproduce the CFC profile pattern referred to earlier, with a change to much lower concentrations below 30 m and a small rise in the lowermost sampler interval. Similarly, boron content, while low and near detection limit at depth, is demonstrably higher in the upper 30 m. The latter has been recognised elsewhere as a potential sewer leakage indicator in residential areas due to its widespread presence in detergents (Barrett et al. 1999).

However, other commonly employed urban recharge inorganic marker species, such as Cl and total oxidised nitrogen (TON) show little evidence of significant contaminant loading compared to adjacent rural areas. This lack of contrast is in part a result of low contaminant source concentrations and high ‘natural’ background levels. This is illustrated in Table 7, which compares the concentration ranges of nine key indicators in various parts of the study area’s water infrastructure:

- Bessacarr-Cantley’s mains water supply.
- Foul sewer inspection chambers at outfalls draining the study area.
- The multilevels, with sample intervals categorised into an upper and lower zone.
- Nearby rural/periurban private boreholes/wells.
- Public supply boreholes for which Bessacarr-Cantley forms part of their catchment.

Table 7. Comparison of concentration ranges of potential sewer leakage indicators with those for other parts of urban water infrastructure in Bessacarr-Cantley

	Concentration range mg/l					
Marker species	**Bessacarr-Cantley study area**				**Rural/periurban**	
	Wastewater n=29	**Mains supply n=30-479***	**M/levels 0-30m n=75**	**M/levels 30-60m n=65**	**8 private wells n=30**	**3 public supplies n=30-410**
Cl^-	60 – 90	26 – 41	10 – 170	15 – 110	15 – 90	20–80
SO_4^{--}	60 – 100	27 – 46	30 – 140	20 – 160	20 – 350	30–80
HCO_3^-	400 – 575	180 – 240	90 – 300	35 – 275	100 – 550	100–220
K^-	17.5 – 22.5	2 – 3	1.5 – 13	1.5 – 6.5	2 – 28	2.5–3.5
B	0.15 – 0.5	Bdl (0.05)	0.04 – 0.14	0.01 – .09	0.025 – 0.1	<0.1
NH_4–N^-	25 – 75	<.02			<0.01 – 0.5	<0.02
TON	<2	0.5 – 10	2.5 – 13.5	5 – 17	<0.1 – 30	5–16
DOC	30 – 110	N/A	1 – 5	0.7 – 2	1.5 – 7	N/A
Data source	FS	YW	FS	FS	FS	FS

*****	Depending on parameter measured
Bdl	below detection level
Wastewater:	3 sites: Burnham Close, Everingham Rd, Warning Tongue Lane
Mains supply:	Nutwell combined raw (blend of Armthorpe, Boston Park, Nuwell, Thornham PS waters)
Multilevels:	5 sites: Haslam Park 1 & 2, Bolton Hill, McAuley School, Sandall Beat
Private wells:	8 sites (Beechtree Nurseries, Doncaster Racecourse, Gatewood Grange, Misson Quarry, Warning Tongue Lane, , Elmstone, Crowtree and Lings Farms)
Public supplies:	3 sites (Nutwell, Rossington Bridge and Armthorpe pumping stations, various b/hs)
FS	Data collected by AISUWRS project team Jun 2003 – Nov 2004
YW	Data from Yorkshire Water raw water quality surveillance archive Jan 1999 – Mar 2004

The concentration range of samples from the mainly rural/periurban private and public supply boreholes, is striking and at least as great as that found in the multilevel samplers within the urban study area. The reason for the high background levels in the non-urban sites is unknown, but stabilised mine spoil heaps and closed landfill sites are present throughout the area and together these are likely to have had some effect on shallow water quality additional to that which could be expected from agricultural activities. In comparison with indicator concentrations in the rural and periurban waters and incoming mains water, the wastewater loading leaving the district is relatively dilute. For example, for both chloride and sulphate, the additional load is typically only 30-50 mg/l, well within the range of variation in adjacent rural catchments.

The exceptions are the two nutrient sources nitrogen (present as ammonium in wastewater and principally as nitrate in the oxidised environment of the saturated aquifer) and carbon (measured in its organic form as dissolved organic carbon DOC). Both appear to be less mobile than other markers, possibly as a result of sequestration in microbial processes occurring in the subsoil in the biologically highly active area around sewer leaks. The presence of oxidising conditions in both saturated and unsaturated zone would provide ample opportunity for breakdown of easily degradable organic contaminants.

Microbiology

As part of the comparative microbiological study of the study area, several faecal contamination indicators were sampled (Cronin et al. 2005). Table 8 summarises the results.

Table 8. Faecal indicator sampling Jul 2003-Nov 2004 (modified from Cronin et al. 2005); results expressed as percentage positives

	Multilevel depth-specific intervals n=154	**Regional well n=45**	**Sewers n=43**
Field Thermotolerant coliforms TTC %[1]	18	11	100
E. coli %	18	16	100
Total coliforms %	34	24	100
Faecal streptococci FS. %	40	24	100
Sulphite-reducing clostridia SRC %	44	47	100
Coliphage %	1	7	100
	n=60	n=3	n=17
Enteric virus[2] %	12	0	100

[1] Analyses of thermotolerant coliforms were undertaken in the field using a portable DelAgua testing kit as well as samples being sent for laboratory filtration and confirmation (shown in the next row named *E. coli)*,

[2] Combination of results from two methods

Sulphite-reducing clostridia (SRC) has the highest number of positive detects for the regional and multilevel groundwater samples; in fact over 40% in both cases. SRC are anaerobic spore-forming non-motile bacteria exclusively of faecal origin that can survive in water for longer (months to years) than coliforms or streptococci (generally weeks to months) due to their spore-forming ability (Gleeson and Gray 1997). Faecal streptococci (FS), an indicator commonly employed both in recreational water monitoring and as a comparison for thermotolerant coliform results, are also detected in 40% of the multilevel and almost a quarter of regional well analyses.

The results indicate a high positive detects frequency of faecal indicators throughout the upper part of the aquifer. This is striking because in comparison with UK carbonate aquifers like the Chalk or the Jurassic limestones the Sherwood Sandstone is generally regarded as a high-porosity, slow-moving system. However, positive detects of enteric viruses and faecal indicator bacteria have been previously found in a similar urban setting in the Sherwood Sandstone underlying Nottingham and Birmingham (Powell et al. 2003), where they have been explained by a small but rapid flow component transporting sewer-derived leakage to depth (Cronin et al. 2003). In this study the counts in the multilevels, although frequent, were universally low. Median values of all parameters were <1 and the maximum 90% percentile values were 11 and 4 cfu/100ml for SRC and FS respectively.

These results indicate that although positive detects were frequent, the magnitude of these detects was very low, indicating that gross contamination of the groundwater is not evident.

Depth profiles of the two bacterial markers of SRC and FS are shown in Fig. 11 for the multilevel HP2. These gave positive counts in 68% and 57% of samples respectively, with positives on one occasion or another at all depth intervals except 35 m. The two faecal indicators results show broadly similar and consistent distributions. These are reminiscent of the CFC and SF_6 profiles in Fig. 7, with higher counts at the shallowest 10 m and the deepest 60 m level (in this multilevel, the 10 m port is <5 m below the water table). While there is no evidence of a quality change below 30 m, as demonstrated by the hydrochemical indicators, the intervening depth intervals consistently show low but usually positive counts.

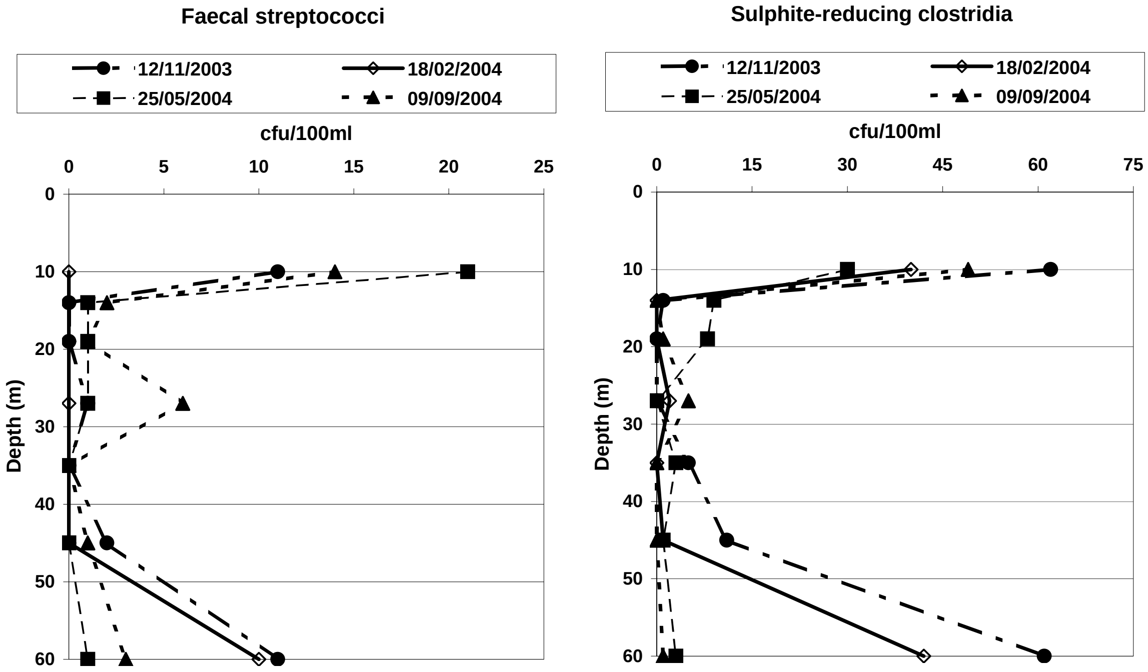

Figure 11. Microbial indicator depth profiles in HP2 multilevel Jul 2003-Sept 2004; 68% of SRC and 57% of FS samples gave positive counts

It is possible to infer a very approximate measure of possible survival time from the results. The removal rate of SRC is difficult to estimate as spores can remain viable in the subsurface for months to years, but published half lives available for FS range from 46.2 hours (Yates et al. 1985) to 72.2 hours (Keswick et al. 1982). Measured FS bacterial numbers from sewer sampling in Bessacarr (Cronin et al. 2005) are in the range 10^5 – 10^6 and this organism is not known to reproduce outside an animal host. Table 9 indicates that the time range required for effluent exiting from a sewer leak to decay to FS counts of $<10^1$ would be of the order of 25 – 50 days. As it is highly unlikely that all recharge reaching the multilevel survival ports is derived from sewer leaks, dilution effects would reduce the decay period, and the implication is that a proportion of the water sampled must be very modern. However, it is important to note that the die-off rates given in Table 9 are typical values and not maximum ones. Several researchers have found survival times for even *E. coli* cultures in excess of 100 days with some reported survival times up to 5 years in the subsurface (Van Ryneveld and Fourie 1997). Hence, the possibility of longer survival times than those calculated here, and, therefore, longer potential travel times to the sampling intervals cannot be ruled out.

Table 9. Faecal streptococci survivability comparison

	Upper range	**Lower range**
Concentration in Sewer*	10^5	10^6
Half-life (days)	1.925**	3.008***
Time to decay to $<10^1$ (days)	27.0	51.1

* Sewer sampling July 2003 – November 2004, 6 sampling campaigns

** from Yates et al. 1985

*** from Keswick et al. 1982

Discussion

CONCEPTUAL GROUNDWATER FLOW MODEL

In the light of the evidence for large-scale water mixing provided by the environmental indicators used in this study, any groundwater flow model has to explain how recharge is penetrating so rapidly and deeply into a rather poorly consolidated part of the Sherwood Sandstone under conditions of low vertical head difference and negligible local pumping.

Elsewhere the Sherwood Sandstone is observed to behave as a layered aquifer (Jackson and Lloyd 1983, Allen et al. 1997), and leakage between strata would occur either via cross-layer fractures or require significant head differences in order to counteract the effects of anisotropy (Buckley 2003). Both the drilling/coring and the seasonal water level response in the multilevel boreholes suggest that bedding plane fractures and other features of a well-cemented sandstone sequence are infrequent throughout much of the uppermost 30 m of saturated aquifer, so intergranular flow would seem likely to predominate.

A typical intergranular flow rate can be estimated from the mean of the hydraulic conductivity from the packer tests at Cantley Water Tower (5.25 m/d), the local water table gradient of 0.0033 (derived from the sub-regional model representation; Neumann and Hughes 2003) and an effective porosity of 0.1. This gives a rate of ~63 m/a. If the upper aquifer were isotropic, a flow rate of this order could quite feasibly allow recharge to penetrate to the 60 m depths implied by the detection of CFCs and SF_6, and a mixing-with-modern-water interpretation could explain the concentrations encountered. However it could not explain the microbiological results; intergranular flow rates would just be too slow to displace modern recharge to the depths encountered.

Therefore it appears that some mixing must be occurring with modern water moving via fractures in more highly cemented sandstone bands interspersed with the less indurated members. Using the same groundwater gradient and effective porosity values cited above, fracture horizons with a hydraulic conductivity of 30 – 40 m/d would theoretically permit a contaminant to pass to the 50 – 60 m depths at which the faecal indicators were encountered in 50 days. These are well within the observed range for fissures in the Sherwood Sandstone: Allen et al. 1997 report high transmissivities south of the study area in Nottinghamshire of the order of 1500 m^2/d of which only 300 m^2/d may be accounted for by intergranular permeability.

Fractures in the more competent horizons alternating with the less competent sandy strata could provide limited access for modern recharge to penetrate under low head gradient to significant depths. Fig. 12 illustrates the conceptual model, in which the upper 30 m or so is composed mainly of poorly cemented strata interspersed with harder fractured horizons.

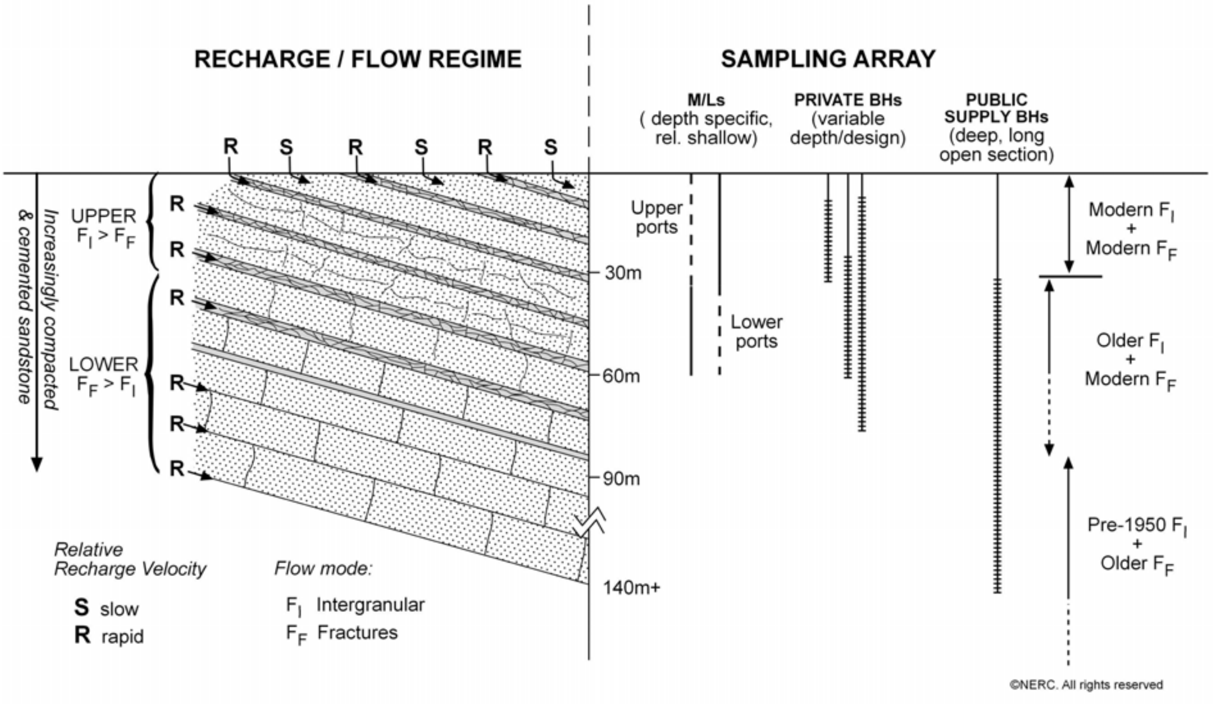

Figure 12. Conceptual model of flow system in Sherwood Sandstone in general vicinity of Bessacarr-Cantley suburb of Doncaster

At depths below 30 m or so, the scope for mixing with more modern recharge brought down in linked fracture systems is easier to visualise, as depth of burial and increasing cementation produces harder, more structurally competent sandstones in which fracturing can occur more widely. Then the potential for more rapid flow in linked fracture systems is limited mainly by aperture and the extent of infill of the fractures by sand. In the Doncaster area such structurally-produced discontinuities in the deeper sandstone horizons are likely to be increased by subsidence effects as a result of extensive coal extraction from the underlying Coal Measures (U. Carboniferous). Downhole logging has confirmed the importance of linked fracture systems on flow patterns in the deeper aquifer in water supply boreholes within the Doncaster wellfield, on the margins of which the study area lies (Buckley 2003). It seems reasonable to infer therefore that fracture flow below 30 m depth is increasingly important, but the effect on apparent age will not be the same throughout: at medium (30 – 60 m) depths there is still scope for matrix water to mix with relatively modern water from the surface, whereas with greater depth the fracture-borne water is itself also becoming progressively older and so the bulk age signature increases.

If this conceptualisation is correct, the pattern of the hydrochemical depth profiles does not necessarily imply the slow passage vertically downward of a ‘front’ of urban recharge. Instead, a given profile could be the product of a complex series of mixing ‘cells’, slowly evolving as water moves generally downdip (and occasionally cross-dip along discontinuities) into the deeper aquifer. Stratification effects would be the consequence of variable contaminant loadings at the land surface and relative speed of flow.

WATER QUALITY IMPLICATIONS

The implications for final water quality in the urban aquifer based on the results presented here are somewhat paradoxical. On the one hand, the CFC and SF_6 environmental tracers and the bacterial indicators show that a component of relatively fast-moving water is entering the aquifer and penetrating to depths of at least 60 m under modest local pumping influence (from the Doncaster wellfield several kilometres down-gradient). On the other hand, comparison with shallow groundwater from nearby rural/periurban catchments and from deep public supply boreholes shows that to date the impact of Bessacarr-Cantley's urbanisation over the last 80 years on both hydrochemical and microbiological water quality appears to be slight, at least for the range of parameters examined in this study. Likely reasons for this are:

(i) A groundwater-benign urbanisation history; Bessacarr-Cantley's development as a residential district with suburban population densities directly replaced a rural landuse. Unlike the more central areas of many cities, there is no 'brown field' legacy of an intervening period of industry or high-density 19th century housing to leave its contaminant footprint.

(ii) Light contaminant loadings; as Table 7 shows, the predominantly residential land-use, which is relatively low density (<35 persons/ha) is providing a relatively dilute sewage effluent, at least for the inorganic parameters analysed for in this study. This reduces the potential contaminant load from sewer leaks, which are also, given the relatively young age of the housing stock, likely to be less frequent compared with an older inner-city area.

(iii) High aquifer storage capacity; the frequency of consolidated but practically uncemented horizons in the upper part of the saturated aquifer would tend to maximise available storage, providing high dilution potential for recent recharge from water stored in the matrix.

(iv) Availability of dilution from precipitation; the suburban nature of the catchment provides more than 80% of total area as some form of green space (domestic gardens, public parks, school playing fields, verges) and a high proportion of this is able to accept direct recharge from rainfall. An additional indirect contribution would come from those properties where roof runoff is directed to on-site soakaways.

(v) Further dilution of contamination at the public water supply wells sited downdip: these are usually drilled to at least 125m and often have long screened intervals below about 30 mbgl. This means that urban contamination, currently observed mainly in the top 30 m of the aquifer, is significantly diluted by older, uncontaminated water at depth.

The result seems to be an urban recharge system that is relatively resilient in terms of adverse water quality impact, at least in terms of the contaminant indicators used in this study.

Conclusions

Groundwater has been characterised in a regionally important Permo-Triassic sandstone aquifer beneath a suburb of Doncaster, eastern England in order to assess the nature and effect of urban recharge. A variety of environmental indicators were used to infer the flow regimes of shallow groundwater. The main observations are:

- The anthropogenic compounds CFC-12, CFC-11 and SF_6 have been found throughout the upper 50–55 m of saturated aquifer, indicating that modern (<50 year old) recharge has penetrated to many tens of metres below ground level. Excess CFC concentrations at depths of <30 mbgl indicate local sources of enrichment. In general the evidence suggests the mixing of groundwater rather than piston flow displacement.
- The distribution of microbiological marker species (faecal streptococci and sulphite-reducing clostridia) provides support for this interpretation, in that small positive counts were also consistently detected down to 60 mbgl.
- The evidence from standard hydrochemical indicators is less conclusive. While indicators such as K, Na, HCO_3 and B were typically elevated at shallow depths, other markers that have been successfully used as urban recharge indicators elsewhere, such as Cl, SO_4 and NO_3 showed

significant variability between sites. Although most sites showed higher solute concentrations in their upper zones, there was no consistent pattern.

- In hydrochemical and microbiological terms, the adverse effect of urban recharge on underlying groundwater quality has been limited, at least in terms of the parameters measured in this study. A number of pollution indicator species show little more variation than that encountered in neighbouring rural catchments. This is ascribed to the combined effects of a non-industrial prior land-use history, light contaminant loadings from sewer leakage and urban runoff, locally high storage capacity in the friable upper aquifer and the availability of dilution from precipitation entering green space areas within the urban footprint.

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2.4 APPLICATION OF NEW TECHNIQUES USING NEWLY ACQUIRED FIELD DATA

This section includes 2 papers that combine available data acquired from stakeholders (Yorkshire Water, Doncaster Metropolitan Borough Council, the Environment Agency) with new data collected during the field investigations, generally to complete tasks relating to the setting up or calibration of models in the AISUWRS model array.

2.4.1 Paper 7: Geographic Information System Analysis in pipe infrastructure modelling: making the most of available data (Cunningham *et al.*)

Key findings

- GIS can be used to provide information about the role of the pipe infrastructure in urban recharge.
- Successful application of GIS to the problem of assessing urban recharge sources has to take into account three main problems when using available data provided by stakeholders: (i) Understanding the technical terms from another discipline (ii) assessing resolution/scale issues (iii) comparing different dataset timeframes. Resolving these problems needed significant effort and time before the models could be populated with meaningful data
- Pipe network characterisation using the Yorkshire Water pipe asset database was possible, but the type curve approach of the PLM needs to be used because CCTV survey coverage is very limited; this is the norm for UK sewer systems.
- Division of the sewer system into hydraulic entities is a vital precursor to UVQ neighbourhood selection. However, significant manipulation of the data will generally be required, as in practice most urban sewer systems include pumped reaches (pressure flow, not gravity) or combined sewers (with storm overflow measures), both of which change the branching-tree pre-requisite of the UVQ and PLM models.
- Even small study areas have extensive pipe asset inventories; the 7 km^2 district of Bessacarr-Cantley has a mains, sewer and pluvial drainage network comprising 220 km of pipe infrastructure and over 3750 pipe assets. These are large datasets to manipulate if manual intervention is required.
- Municipal GIS datasets of the type employed for town planning purposes provide excellent detailed urban land use information for the UVQ model; significant category simplification can be achieved if statistics on water use and occupancy levels can be incorporated to transform and group together land use categories into neighbourhoods of similar broad water use.

Introduction

The Geographic Information Systems (GIS) analysis detailed here is contributing to an ongoing EC 5th Framework European research project that aims to assist the sustainable management of water in groundwater-dependent cities. The AISUWRS project (Assessing and Improving Sustainability of Urban Water Resources and Systems) is developing and applying an array of linked urban water models and a decision support system, using urban areas in England (Doncaster), Germany (Rastatt), Slovenia (Ljubljana) and Australia (Mt Gambier) as city case studies. The British Geological Survey and the Robens Centre for Public and Environmental Health of the University of Surrey are the research agencies concerned with the UK component. This ambitious project involves development of new models, the application, improvement and linking together of existing codes and a significant field investigation component in each urban study area (1,2). One of the challenges is the manipulation of the many different datasets that the models demand, both those derived from existing sources and the supplementary data being generated by the project's own field investigations. GIS analysis is an essential element of this process.

The importance of groundwater as a source of drinking water in England and Wales should not be underestimated - it accounts for almost a third of all water put into public supply and over half of private water consumption (3). Doncaster is groundwater-dependent, like a number of other towns in the East Midlands, and the city, nearby former mining villages and a large rural hinterland are supplied from a network of boreholes. These pump from the Permo-Triassic Sherwood Sandstone aquifer at 11 sites downgradient of the metropolitan area.

Doncaster was chosen as case study for the UK for its high local groundwater dependence. Like many industrial cities in the UK, it has experienced a shift from heavy manufacturing and engineering and with the decline in nearby coal mining, the area's economy, water demand and usage patterns have changed significantly over the past few decades.

Use of GIS in the AISUWRS programme

The six models that are employed in the project (Figure 1) track the quality and quantity of water derived both from precipitation and from mains supply as it moves through the urban system (residential, commercial, industrial, public/community properties) and its associated pipe infrastructure (mains, foul/combined sewer and pluvial drainage), down through the unsaturated zone and into the underlying aquifer.

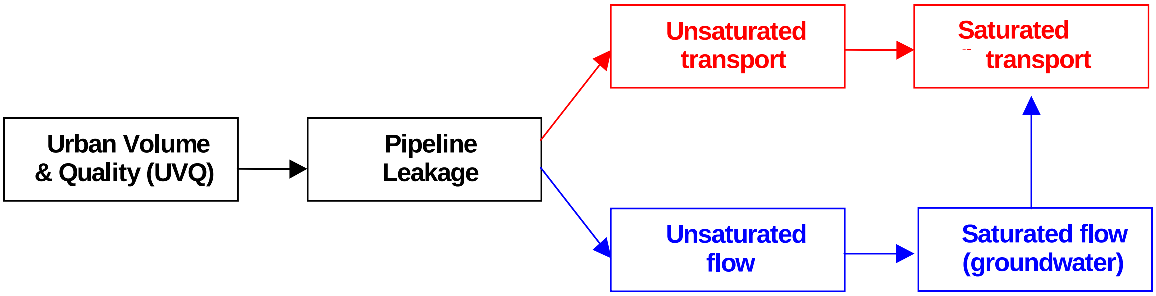


Figure 1. AISUWRS suite of urban water models for a groundwater-dependent city (4)

The resultant model suite therefore covers numerous parameters in diverse fields, from those dealing with land categorisation and the design, condition and distribution of the pipe networks through soil-related characteristics to aquifer parameters. The GIS, associated with a comprehensive database organisation, is able to combine numerous datasets in a flexible and dynamic mapping environment for data display and analysis. It is indispensable for three reasons:

- to analyse available data obtained from stakeholder organisations (water utility, regulatory agency, municipality) and other sources in order to populate the models,
- to permit comparison of the spatial datasets that the two-dimensional and three-dimensional models require, and
- to help model linkage by permitting spatial outputs from one model to cascade into subsequent model inputs.

Available data

Any research project of this complexity is highly dependent on existing data that have been generated by other organisations for different purposes. The six AISUWRS models together have >300 input fields (4) and cover a number of engineering and earth science disciplines (Figure 2). So the more that generic, already available data can be employed the more likely it is that the models (and the decision support system that acts as a user interface) will be adopted in the future as a practical urban water management tool.

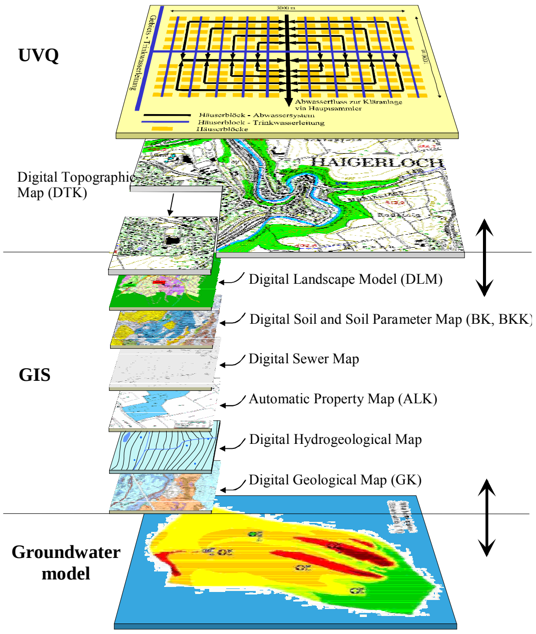


Figure 2. AISUWRS Data and models (from Eiswirth, 2002)

However, an inevitable consequence of employing available data is that the original purposes for which they were collected and the way that the data are used by the supplier organisation (or others) are quite different from AISUWRS project requirements (Table 1).

Table 1. Available data, acquired from a diverse range of organisations

Data Type	**Source**
Topography (1:10 000 Colour raster, 1:50 000 Colour raster & Meridian)	Ordnance Survey
Bedrock Geology (1:50 000 & 1:250 000)	British Geological Survey
Superficial Geology (1:50 000)	British Geological Survey
Soil Data	National Soil Resources Institute (NSRI)
Land Use	Doncaster Metropolitan Borough Council
Aquifer Vulnerability (1: 100 000)	Environment Agency
Source Protection Zones (SPZ's)	Environment Agency
Sewer infrastructure (Pipe infrastructure, soakaways, Overflows, Outfalls, Waste Water Treatment Works, CCTV)	Yorkshire Water
Mains Pipe infrastructure Pipe infrastructure Public Supply wells	Yorkshire Water
Digital Terrain Models	-Centre for Ecology & Hydrology (Wallingford) -NextMap©; 5 m resolution Intermap Technologies Inc
Water Levels	Environment Agency
Multilevel monitoring database	Project Specific Data

In practice we encountered three principal problems using available data:

(i) *Understanding the technical terms from another discipline.* Database field names or spreadsheet column headings inevitably use shorthand or abbreviated terms that may be ambiguous to users not familiar with the original context of the dataset and could inadvertently lead to a misinterpretation or mis-combination of datasets. Thus for instance the term 'rising main' means one thing to a hydrogeologist and another to a wastewater design engineer.

(ii) *Assessing resolution/scale issues.* This can greatly influence the time required to transform an available dataset into one that is usable as a model input. If the data are very detailed and part of a very large and complex array, several stages of extraction may be required in order to end up with a dataset of manageable proportions. An example of this might be a water utility's pipe asset database. Conversely some accommodation might be needed to make a more general dataset usable as a particular model input. As an example of this the dearth of potential evaporation values in published climatic data for the UK.

(iii) *Comparing different dataset timeframes.* While the GIS system greatly facilitates comparisons of spatial datasets, these may internally be derived from time-series data that will inevitably cover a wide range of time periods. Thus, unaccounted-for water statistics for mains supply may be held as a well-maintained and up-to-date dataset because the figures are required for operational and regulatory purposes. In contrast flow and water quality for the area's corresponding sewer network may only be available from historic records of sporadic short-term field tests prompted by infrequent sewer improvement design studies.

This paper demonstrates two examples [4] of how already available data, produced for quite different purposes, were transformed by interpretation and manipulation into inputs suitable for the urban water flow and transport models. The examples, (structured to show how the GIS investigated data sources, analysed the results and produced a product required to progress the modelling phase in the case-study area) are:

- the urban pipe infrastructure analysis and resulting sewer gain/loss map
- land-use analysis for the UVQ (Urban Volume and Quality model).

Pipe infrastructure analysis

The pipe infrastructure analysis used Yorkshire Water's asset database of mains, foul/combined sewer and surface water drainage, to characterise the pipe network of the study area, the 6.3 km^2 Doncaster residential district of Bessacarr-Cantley. As well as helping achieve a better visualisation of the components of the water infrastructure of the Bessacarr-Cantley (mains pipe network, sewer/pluvial pipe networks, flow directions, sewage sampling points, sewage pumping stations and outfalls) the pipe asset information was used in several ways to inform and facilitate the urban water models:

(iv) *Network characterisation* of the sewer and pluvial drainage networks in terms of pipe size, material, age and local soil condition so as to compute pipe leakage rates in the Pipeline Leakage Model (PLM)

(v) To produce a *sewer gain/loss map,* again for the PLM, showing the mutually exclusive sewered areas of the network where wastewater could leak out and, more importantly for the water company, groundwater could leak in.

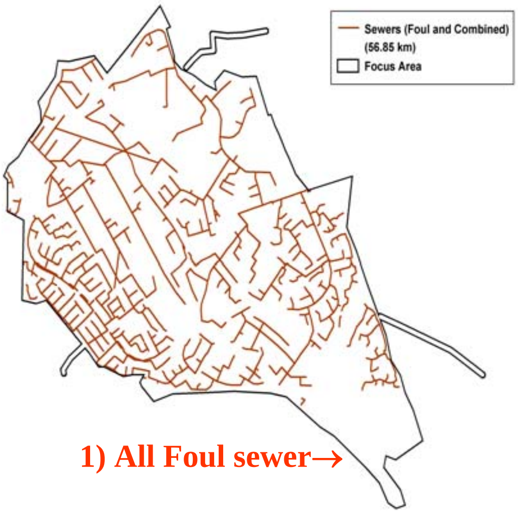
(vi) To *subdivide the study area into hydraulic entities* within the UVQ to enable the apportionment of flows and contaminant loads at house/neighbourhood scale.

NETWORK CHARACTERISATION

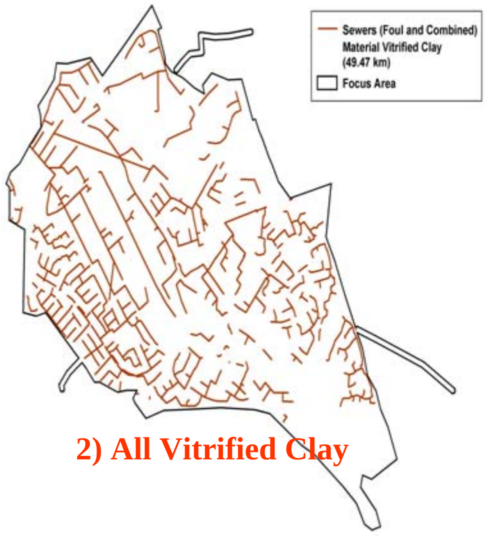
The geographic data analysis for the pipe infrastructure used the functionality of ESRI's ArcGIS © to calculate the length of the entire pipe network and subsequently the length of the parts of the network according to a breakdown by attributes. The latter involved querying the attributes of sewer, pluvial drain and pressurised mains assets according to specific characteristics used as criteria in the PLM to assign pipe leakage rates. At each stage the length of the assets within that group were recalculated to provide a hierarchical breakdown of that part of the pipe infrastructure. The geo-processing tools within the GIS enabled the pipe network to be separated into the appropriate sub categories.

The first processing involved dividing the network according to the material type of the pipe itself (for example cast iron, vitrified clay etc.). Then each subset of data (from within the pipe material type) was classified depending upon the pipe diameter, the asset length was again calculated and the data mapped. The same procedure was then used to subdivide the data according to the age of the pipe. The mapping enabled the results of the breakdown to be viewed at each stage. A separate soils dataset was reclassified by the AISUWRS project team into two basic types ('reactive' or 'non-reactive') in terms of their effects on the pipes when they pass through the soil) and then mapped with the subsets of the pipe data to establish the length of asset that passed through each soil type.

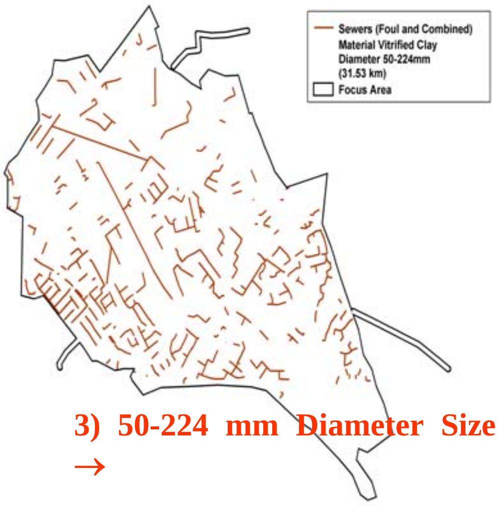
The results of this process were maps and quantitative asset statistics on length and percentage of the total pipe infrastructure. These characteristics are employed in the analysis of defects and the estimation of leakage rates per asset. Figure 3 demonstrates the stages of this process and breakdown of data using a foul sewer example.



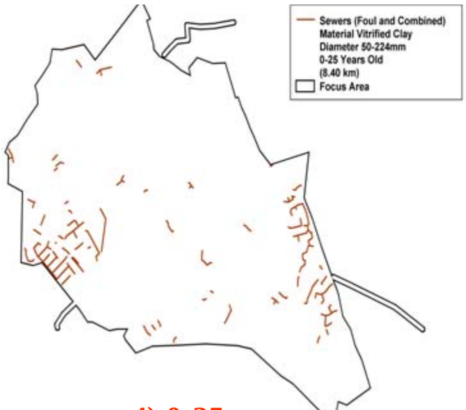
1) All Foul sewer→



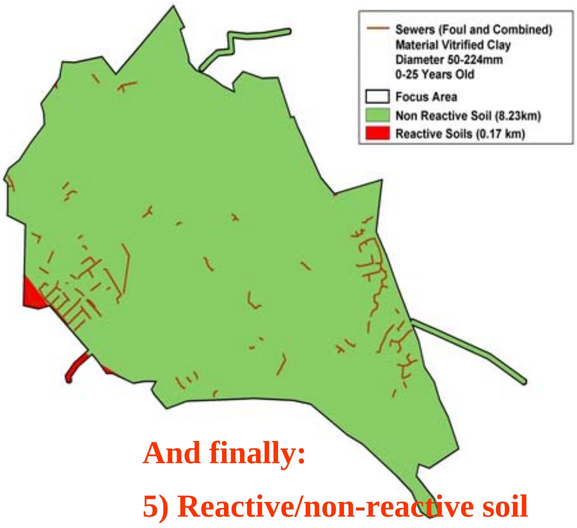
2) All Vitrified Clay



3) 50-224 mm Diameter Size →

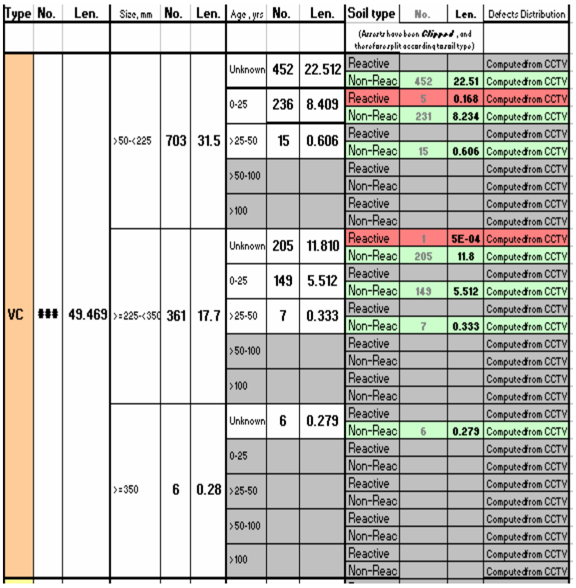


4) 0-25 years age



And finally:

5) Reactive/non-reactive soil



Type	No.	Len.	Size, mm	No.	Len.	Age , yrs	No.	Len.	Soil type	No.	Len.	Defects Distribution
									(Assets have been ***Clipped*** , and therefore split according to soil type)			
VC	###	49.469	>50-<225	703	31.5	Unknown	452	22.512	Reactive			Computed from CCTV
									Non-Reac	452	22.51	Computed from CCTV
						0-25	236	8.409	Reactive	5	0.168	Computed from CCTV
									Non-Reac	231	8.234	Computed from CCTV
						>25-50	15	0.606	Reactive			Computed from CCTV
									Non-Reac	15	0.606	Computed from CCTV
						>50-100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
						>100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
			>=225-<350	361	17.7	Unknown	205	11.810	Reactive	1	5E-04	Computed from CCTV
									Non-Reac	205	11.8	Computed from CCTV
						0-25	149	5.512	Reactive			Computed from CCTV
									Non-Reac	149	5.512	Computed from CCTV
						>25-50	7	0.333	Reactive			Computed from CCTV
									Non-Reac	7	0.333	Computed from CCTV
						>50-100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
						>100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
			>=350	6	0.28	Unknown	6	0.279	Reactive			Computed from CCTV
									Non-Reac	6	0.279	Computed from CCTV
						0-25			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
						>25-50			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
						>50-100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV
						>100			Reactive			Computed from CCTV
									Non-Reac			Computed from CCTV

Summary of data

Figure 3. Example of pipe infrastructure analysis; All foul sewers/All Vitrified Clay/All 50-224 mm in Size/ All 0-25 Years Old/ Reactive;Non-reactive

SEWER GAIN/LOSS MAP

There are no studies extant on sewer leakage/ gain in the Doncaster area, a situation common to most urban areas in the UK. Yet groundwater ingress can be an important design factor when sizing a sewer system, especially in low-lying areas where gradients are low. For this project, each circumstance has consequences both in terms of volume and solute flux. It was thus necessary to assess where sewer assets might lie below the water table i.e. where groundwater ingress could occur as opposed to wastewater egress.

The main datasets used were ground surface elevation, the sewer invert elevation and groundwater levels. The ground elevation was initially developed using Ordnance Survey Land Form Profile © data, but for some areas its accuracy seemed questionable, and instead a recently-available NextMap © digital terrain dataset was substituted, resulting in better lateral and vertical resolution of the ground surface (Figure 4). The second dataset required was the elevation of the sewer invert for each pipe asset (measured below ground at surface access points SAPs, usually manholes). The depths of the sewers were interpolated into a continuous surface across Bessacarr-Cantley (Figure 5). The third dataset was the water table surface, compiled from the project's own research boreholes supplemented with Environment Agency regional observations. A groundwater surface was interpolated from the point water levels by kriging, a statistical method commonly employed in the interpolation of hydrogeological data to produce a continuous raster surface across a modelled area (Figure 6).

The raster data analysis calculated the sewer invert depth surface below SAP and the ground surface to establish the sewer depth MAOD rather than MBGL (its original format). Subsequently the sewer invert depth was related to the water table to produce a new surface showing where the sewer level was below water table and groundwater gain could occur. Figure 7 shows this surface combined with OS topographic data and the sewer line work to show how vulnerable reaches of the sewer network can easily be identified. As this analysis uses the same area as the groundwater flow model grid it could also be used more indirectly to provide water gain/loss values to a given node. Interestingly, the identification of sewer gain areas on the south side of the study area for February 2004 coincides with recent reports of localised groundwater flooding following abstraction regime reductions introduced several years ago as part of a wetland management plan.

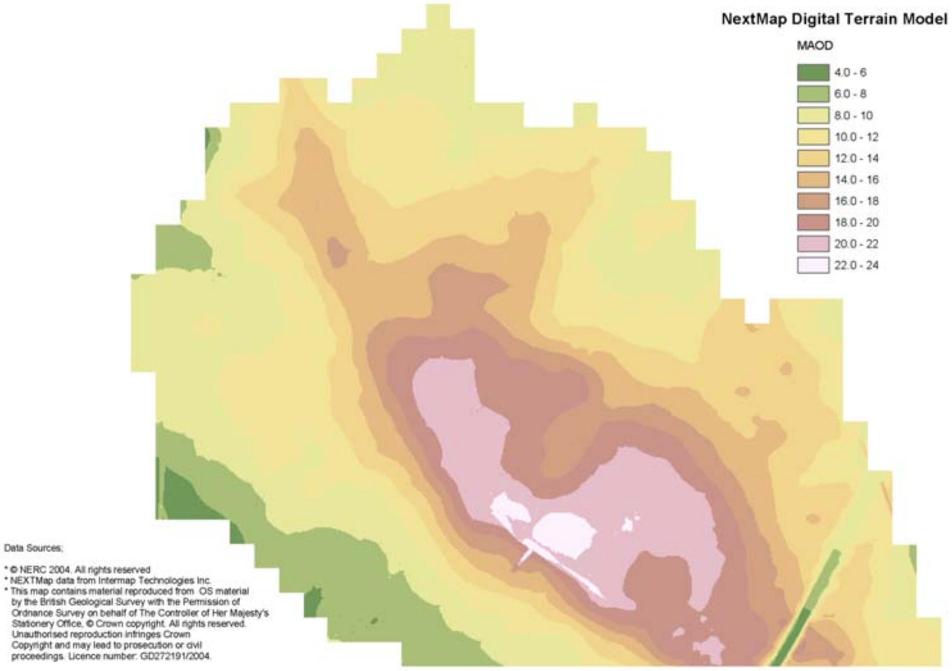


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Figure 4. Digital Surface Model from NextMap©

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* Water level data from the EA and multilevel piezometers installed for this research

Figure 5. Sewer Pipes Surface (Invert Depth)

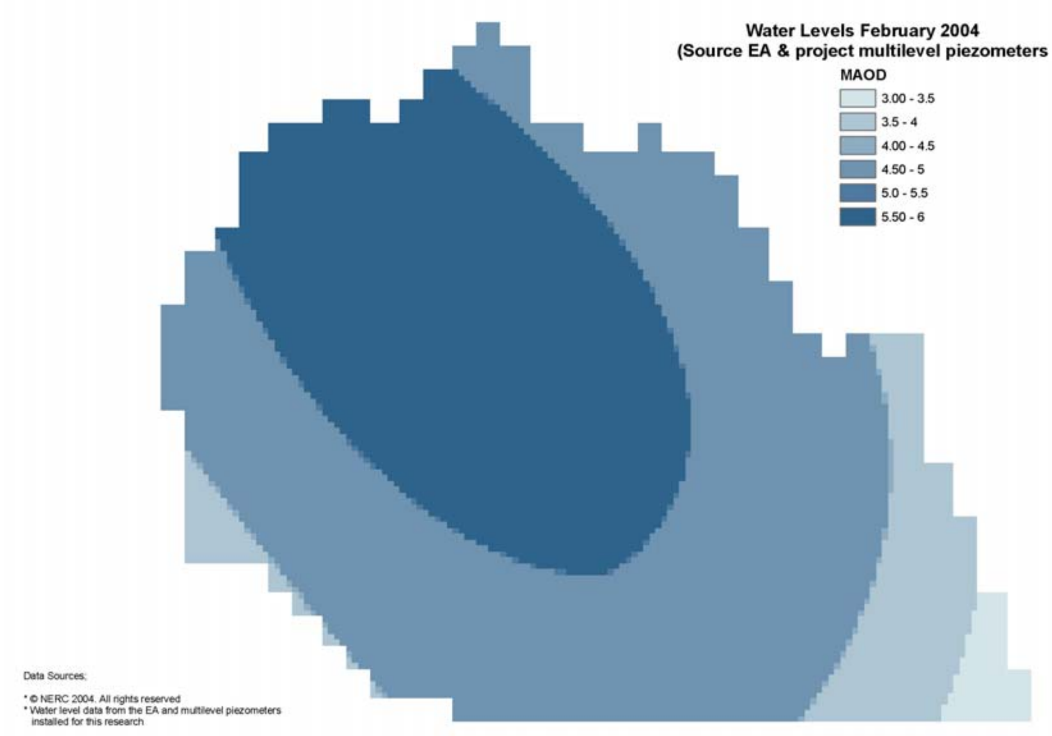

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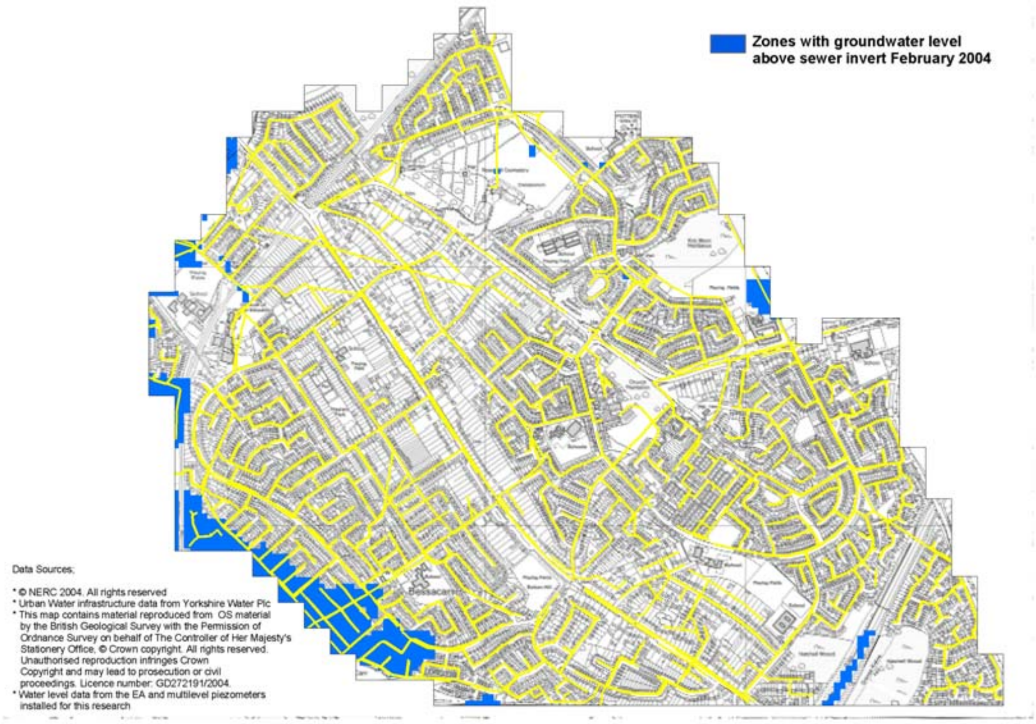
Figure 6. Water table surface



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- Water level data from the EA and multilevel piezometers installed for this research

Figure 7. Example of the final Sewer gain/loss map for water levels from February 2004, including reaches of sewer networks prone to groundwater gain.

Having developed a robust methodology we believe this innovative analysis tool could be used in other urban areas for the production of sewer gain/loss maps.

SUBDIVISION INTO HYDRAULIC ENTITIES

The UVQ model processes water flux and contamination load through the urban system using neighbourhoods as the unit of water use (see next section). Water enters a district either as precipitation or through the pipe and channel system. To track either the receipt of or the transmission to an adjacent district and to compute the changes within the district during a daily time step, the urban land surface needs to be subdivided into hydraulic entities. Using a process analogous to drainage net analysis for a natural catchment, the GI analysis enabled the subdivision of the study area into 9 initial regions in which there is only one sewage and storm water outflow per region, a necessary precursor to neighbourhood selection. An additional, rather more demanding requirement is that both pressurised (mains supply) and gravity (sewer/pluvial drainage) networks need to coincide.

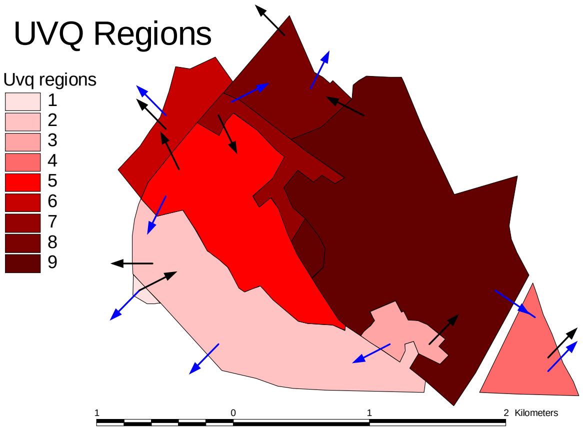

Figure 8. The 9 hydraulic entities based on the restriction that there is only one sewage and storm water outflow from each unit. Black pointers indicate sewage water outflow, blue pointers storm water outflow.

Land-use analysis

The **UVQ land-use analysis** interpreted a complex land-use dataset provided by Doncaster Metropolitan Borough Council. This land-use cover, designed for town planning purposes unrelated to water supply, wastewater disposal or drainage, was used to produce area subdivisions that are the building blocks of the UVQ model. The detailed, digital land use map in combination with the OS topography data facilitated comparison of different housing types and densities.

In addition to identifying hydraulically separate entities for water-in-pipe budgeting, the UVQ model needs to group properties with similar *water use* characteristics in order to produce a *neighbourhood,* which is the basic unit of water accounting. Within each neighbourhood, infiltration characteristics of the land cover are assigned. For instance, rainwater landing on paved areas or rooftops might be directed into pluvial drains, whereas water falling on garden or park areas would soak into the ground, resulting in direct infiltration to the subsurface.

The OS 1:10 000 digital maps were used along with the town planning map to classify the area of interest according to whether the area was mainly paved, mainly roofed, or mainly unbuilt, and by extension whether precipitation infiltrated within the neighbourhood boundary or entered the pluvial drainage pipe system. Figure 8 shows the extent to which aggregation was required, as the original land use classification contained 85 different categories at its most detailed level. The range of colours in this small area reflects the complexity of the dataset. Two stages were required.

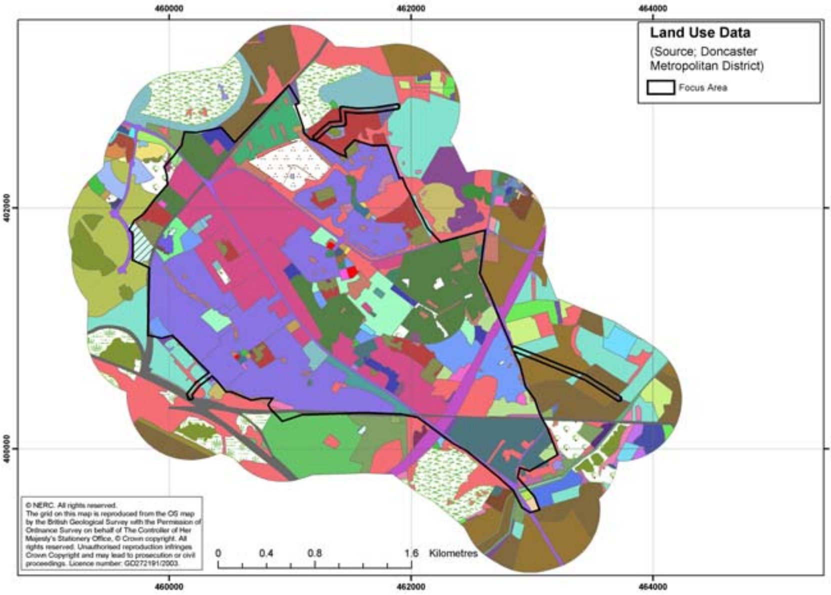

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Figure 9. Full land use classification provided by Doncaster Metropolitan Borough Council

Firstly the GIS enabled polygons of land that were not housing to be re-attributed according to a first-pass tripartite group; mainly roofed, mainly paved and mainly unbuilt. Thus within a school property for example, the school building was 'mainly roofed' (runoff to soakaway) the playgrounds, forecourt and car parking 'mainly paved' (runoff to pluvial drains and the playing fields 'mainly unbuilt' (runoff to subsurface via unsaturated zone). See Figure 10.

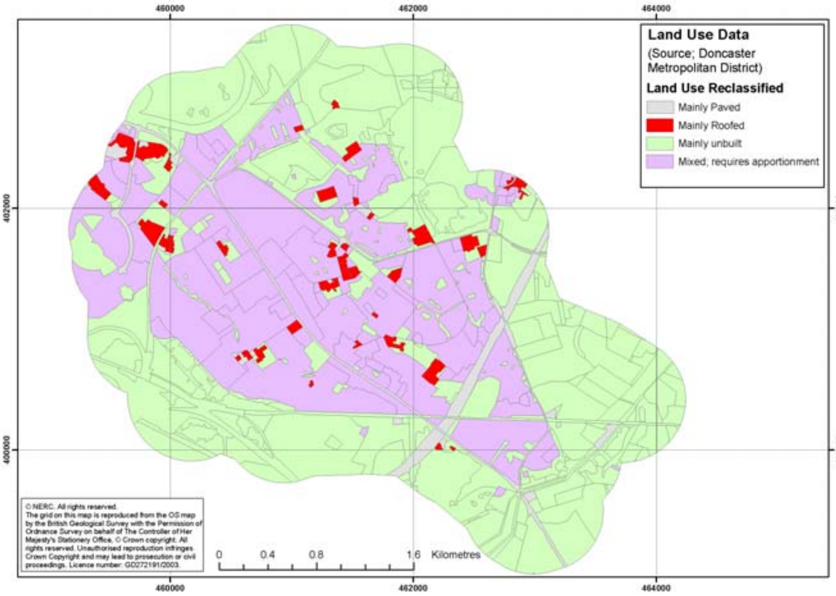

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Figure 10. First-pass reclassification of land use categories to infer infiltration characteristics

Secondly, for areas of housing, alternative sources of data (aerial photographs) and field surveys were used to aid the interpretation of number of households, extent of road/pavement etc. Classification was carried out to distinguish between different housing types and densities, interpreted by the project team. A spreadsheet analysis based on number of gardens per m^2 of polygon area permitted the 15 housing types used for town planning purposes to be simplified to two water use types.

The result of the land use classification was combined with the hydraulic entity analysis to propose 'neighbourhoods' for the UVQ model $^{(5)}$.

Discussion

The AISUWRS project undoubtedly has ambitious targets; calculating the urban water balance for a groundwater-dependent city and also tracking the fate of key contaminants as they pass through the urban pipe infrastructure down into the subsurface requires the simulation of various physical processes in both the manmade and the natural environment. Once the models have been developed, validation through the medium of case-studies means they have to be applied to a geographical area (in the UK case the Doncaster district of Bessacarr-Cantley), as spatial datasets. Given the very large number of input parameters needed to populate the suite of models, and the fact that the two upstream models require analysis of linear features (the urban pipe infrastructure), it is difficult, if not impossible, to conceive how these could be applied in the absence of a GIS and associated database to organize, manipulate and analyse the datasets.

At the time of writing this paper, the 3-year AISUWRS project is approximately half-way through, with the tasks of populating the unsaturated zone models and linking the component models of the suite still to be undertaken. Nevertheless, even at the current incomplete stage of the AISUWRS project, GIS analysis has already provided useful products that could be applied outside the remit of this particular research programme;

(vii) *Sewer gain/loss map*: this project product was required to implement the sewer and pluvial drainage parts of the pipeline leakage model for Bessacarr-Cantley, insofar as each node of the model needs to be set in losing-to- or gaining-from-subsurface mode. However, in the wider context, there are a number of other UK cities where rising groundwater, partly the result of an excess urban recharge, has become an issue. This problem is already reported from parts of London, Birmingham and Liverpool (6,7). The sewer infrastructure plays an as-yet unquantified but possibly important role in groundwater level rebound, in early stages potentially providing a source of relatively low-quality recharge but in later stages, as groundwater levels rise, the same network may act as *de facto* drains. Sewer gain/loss maps could help quantify the real contribution of sewer leakage to the rising groundwater problem that is starting to trouble a number of old-established groundwater-using cities.

(viii) *Pipe infrastructure analysis*: this product informs the project's pipeline leakage model that itself draws on detailed defects analysis of different pipe materials, diameters and configurations from previous urban engineering studies. It predicts likely network leakage losses both for CCTV-surveyed areas, and for sectors where on-site defect surveys have not been undertaken. In the UK, this is the default condition for a large proportion of foul sewer, combined and pluvial drainage urban pipe networks. Where these overlie an aquifer, used for sensitive purposes such as potable supply, wider application of the pipeline leakage module could provide a first-pass environmental impact assessment tool, permitting water utilities to gauge where the condition of their water disposal networks might be prejudicing their water supply function.

Conclusions

1. An innovative application of GIS has enabled characterisation of the sewer network and groundwater levels of a district of Doncaster UK, permitting production of a map showing where sewers can gain from or lose to shallow groundwater.
2. A further application of GIS-based network analysis is facilitating application of a pipeline leakage model to the mains, sewer and pluvial drainage network of this7 km^2 district, which comprises 220 km^2 of pipe infrastructure and over 3750 pipe assets.

3. Transformation of an existing town planning-based land use classification has similarly permitted its use in an urban water cycle model.
4. These GIS exercises in applied GIS analysis comprise part of an EC shared-cost research project that aims to characterise the urban water balance, contaminant indicator fluxes and the impact on the underlying aquifer of a groundwater dependent city. This project is developing a suite of 6 models and testing the system in 4 urban areas in Europe and Australia.
5. The examples demonstrate how available data can be utilised to populate, inform and enable a demanding suite of models in a way that respects the complexity of the system the array is attempting to simulate.

Acknowledgements

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2.4.2 Paper 8: Groundwater surcharging of sewers: example from Doncaster, England of a technique for identifying its extent (Morris *et al.*)

Key findings

- Gravity sewer and pluvial drainage systems needs to be zoned to indicate where the network may be above the water table (leakage-out condition) and where it may be below the water table (gain from groundwater condition)
- 3 GIS datasets can provide this information: (i) the ground surface elevation (ii) the pipe network, providing elevation of pipe inverts via corresponding surface access points (manholes) (iii) the groundwater level elevation
- Elevation precision is required for ground elevation; after experimentation the project used the NextMap Digital Terrain Model; this provided satisfactory precision and good ground control.
- Pipe asset information from Yorkshire Water provided sewer pipe elevations after referencing sewer invert depths to trigonometrically-surveyed manhole ground elevations. This permitted raster data analysis to generate a continuous surface referenced to sea-level.
- A water-level surface can similarly be generated from local piezometer measurements, using kriging to interpolate the hydrogeological data. Subsequently groundwater levels from the groundwater flow model have become available and these have been employed so that scenarios can be tested.
- These techniques have been successfully applied to the study area in order to identify potential sewer gain and potential loss zones. This map informs both the Pipeline Leakage Model and the Groundwater Flow Model.
- The technique has much wider application potential, especially for cities experiencing groundwater rebound problems

Introduction

One of the aims of the AISUWRS project (Assessing and Improving the Sustainability of Urban Water Resources and Systems) (Eiswirth et al, 2002) is to provide and validate, through the medium of city case-studies, a linked array of models that can cope with the complexity of recharge to urban aquifer systems (Fig 1).

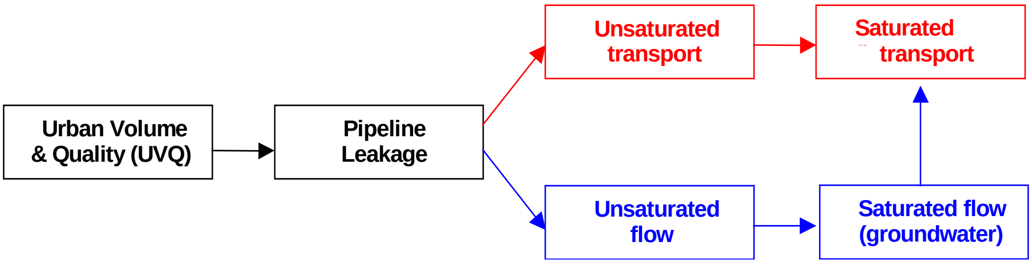

Figure 1. The AISUWRS array of urban water models; these are linked as a functioning array by a decision support system

When compared to rural catchments, one of several additional sources of water available to recharge an aquifer with an urbanized catchment is leakage from wastewater and pluvial drainage infrastructure, and the Pipeline Leakage Model of the AISUWRS array permits losses from these networks to be quantified. This paper describes a stage in the modelling process that was initially a GIS task undertaken to condition data to be used at the pipe leakage assessment stage. However, when combined with a saturated flow groundwater model the technique can have a much wider application as a means of assessing the potential for sewer surcharging by groundwater. In circumstances where drainage networks have to accommodate unforeseen extra baseflow, for example where local pumping has kept groundwater levels low for many years, unplanned surcharging through old or defective sewers and stormwater drains can exacerbate storm event urban flooding problems.

Technique Development For Case Study

The technique described was developed as part of the AISUWRS project's UK case study in the Doncaster suburb of Bessacarr-Cantley, which has a population of c.20,000.

CASE STUDY SETTING

Bessacarr-Cantley is a predominantly residential district of Doncaster, situated on Permo-Triassic sandstones known regionally as the Sherwood Sandstone. This nationally important aquifer, is approximately 100-150m thick beneath the study area and in the vicinity of Doncaster has an outcrop width of about 16 km (Gaunt 1994). The sandstones dip gently towards the east-north-east at about 1½°, being underlain by poorly permeable Permian marls and overlain by Triassic mudstones that together form the lower (western) and upper (eastern) outcrop boundaries (Fig. 2). Locally, the aquifer is unconfined, being present either at outcrop or below thin permeable fluvial or glacial sands and gravels.

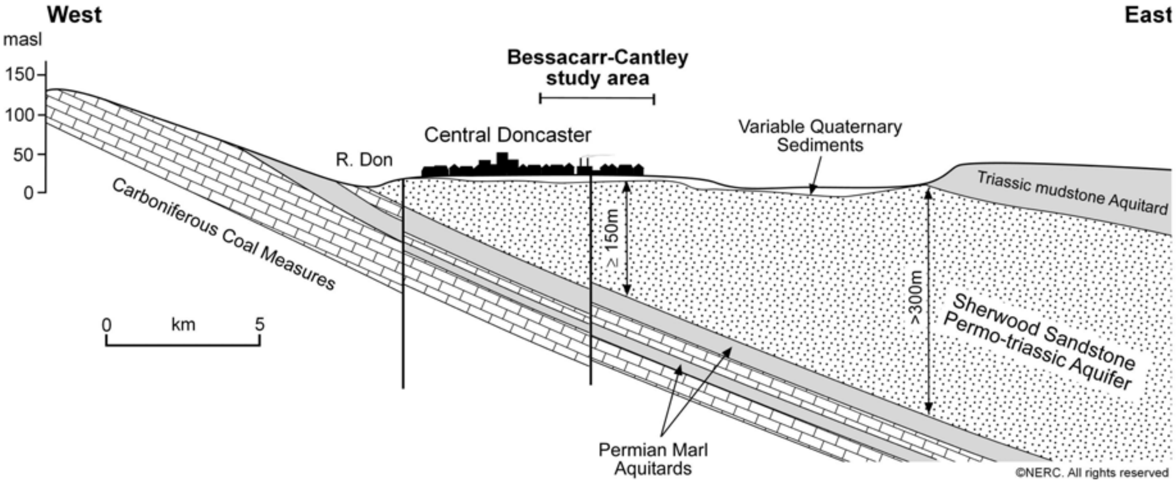

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Figure 2. Sketch section across Sherwood Sandstone aquifer in vicinity of Doncaster

The piped water supply for the town of Doncaster, its suburbs and surrounding rural hinterland is supplied by the Doncaster wellfield, a linked array of eleven pumping stations extending from just to the east of the town along a 15km arc to the northeast and southeast. The combined output from four of these wellsites provides the mains water supply to Bessacarr-Cantley from a single groundwater treatment works (Morris et al 2003). There is other minor pumping from the aquifer but in the general area of Doncaster the decline in manufacturing industry has meant that abstraction for mains water supply constitutes over 90% of the total (Brown and Rushton, 1993).

STUDY AREA DESCRIPTION

Bessacarr-Cantley has c.8300 properties with service connections in an area of 6.3 km^2. The suburb is served by a 220 km urban pipe infrastructure of sewers, pluvial drains and mains water distribution comprising over 3,750 pipe assets (Rueedi et al 2004). The land use is predominantly residential with local services (schools with playing fields, retail, community buildings, green-space). The district lies on both sides of a subdued northwest-southeast trending 15-20 m high ridge, with the low-lying margins adjoining present or former wetland areas. Town planning controls have kept the district geographically distinct so far, and both the urban footprint and its associated water infrastructure of mains supply, wastewater and pluvial drains are well-defined (Morris et al 2003)(Fig.3).

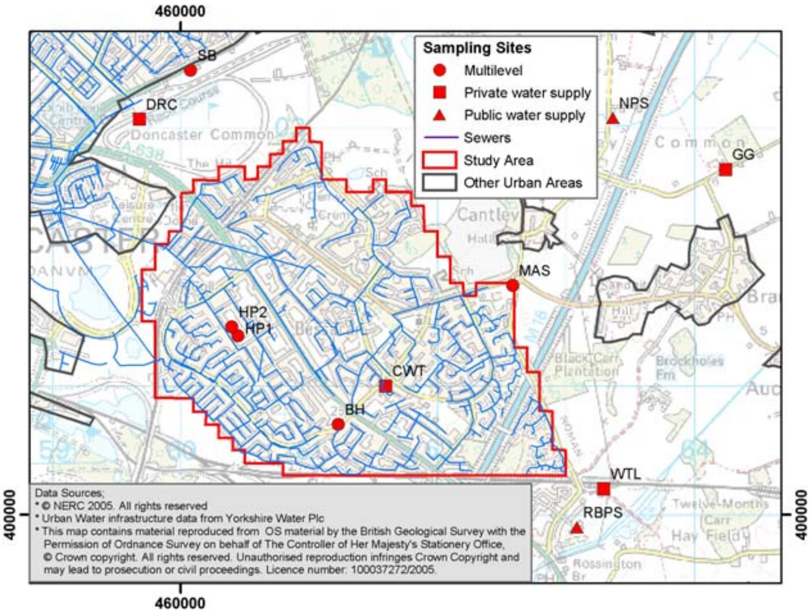

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Figure 3. Bessacarr-Cantley study area; note well-defined urban boundary

DESCRIPTION OF PROBLEM

In the AISUWRS model array, the Pipeline Leakage Model (PLM) permits the calculation of exfiltration from gravity sewer or pluvial drainage pipelines using wastewater flow volume, pipe design and condition (construction material, joint type, diameter and asset age) and either CCTV records of defect distribution or generic curves developed from other case studies if CCTV cover is incomplete or absent (Burn et al 2004). The PLM therefore has an interface upstream with the Urban Volume and Quality Model UVQ (an urban mass balance model for all water inputs and outputs) from which it receives the pluvial and wastewater flows leaving that model's neighbourhood-scale land use blocks. It is a key requirement of the PLM that the relative level of the water table be known, so that each pipe asset can be assigned, via the GIS, a potentially losing or gaining condition.

This information was not previously available and a GIS-based technique had to be developed that could employ already available data to assign the appropriate exfiltration or infiltration condition to each pipe asset in the 128 km long sewer and pluvial drain networks.

TECHNIQUE DEVELOPED

Three surfaces, referenced to mean sea level, were needed to obtain the final map:

- The ground surface elevation, derived from digital terrain mapping cover;
- The sewer pipe elevation; approximated from the sewer invert level;
- The groundwater level elevation; measured from water levels in observation wells

Ground elevation surface. The nature of the exercise required good precision (ideally to ±0.5 m or better). The sewer network was referenced to c.2300 Surface Access Points (SAPs). Approximately two-thirds of these have surface elevation values and provide good local ground control, being tied to Ordnance Survey trigonometric points/bench marks that would have been employed to survey in the sewer network. Although it was hoped that the SAPs themselves could be used to develop a surface elevation model, incomplete spatial coverage of OD levels for this dataset within the study area meant that a supplemental source of ground elevation had to be sought.

The NextMap© Digital Terrain Model (DTM) was used due to its good lateral (5.0 m) and vertical (1.5 m or less) resolution. NextMap was derived from IFSAR (Interferometric Synthetic Aperture Radar) sensor technology and the DTM has been adjusted to remove artefacts such as vegetation and buildings in order to provide a representation of the ground surface. The DTM interpolated ground elevation where the SAPs had no ground surface elevation with an OD value. Complementary use of the two datasets enabled a new surface elevation to be produced using as much of the original reference point data as possible from the surveyed-in SAPs, and the result is considered to have good ground control at the required precision. (Map A in Fig. 4)

Sewer pipe elevation. Pipe infrastructure information was obtained from the water utility Yorkshire Water. This provided a sewer invert depth, referenced to depth below surface access points (SAPs); these are usually manholes that have been trigonometrically surveyed. After developing the ground elevation surface, raster data analysis calculated the sewer invert depth as a continuous surface below SAPs and the ground surface to establish the sewer depth in relation to sea level (Map B in Fig 4).

Water table surface. This was compiled from the project's 5 dedicated research boreholes located in and near the study area and supplemented by available Environment Agency regional observations.

For the initial map, the groundwater surface was interpolated from the point water level elevations by kriging to produce a continuous raster surface across the study area (Map C in Fig 4).

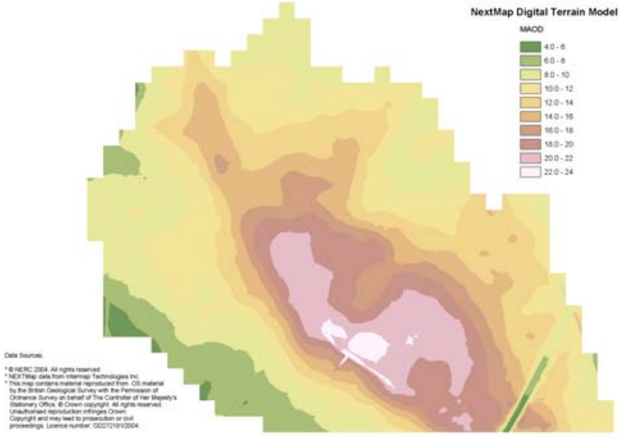
Subsequently the sewer invert depth was related to the water table to produce a new surface showing where the sewer level was below water table and gain from groundwater could occur. That surface has been combined with OS topographic data and the sewer line work in Map D of Fig 4 to show how vulnerable reaches of the sewer network can easily be identified.

EXTENSION OF TECHNIQUE TO PERMIT SCENARIO MODELLING

The procedure described above was improvised to meet the specific data input requirement of the AISUWRS project’s pipeline leakage model, but its usefulness was subsequently extended by incorporating the saturated zone groundwater flow model into the procedure in order to perform scenario modeling. The technique can then be used predictively, either to inform new sewer design studies or for ongoing assessment of the likely effects of changes in shallow water table on existing sewer/drain networks. These can occur for example in response to adjustments in abstraction from nearby wellfields or to longer-term climate change trends. The same layer could also, if required, be used as an input field to improve calibration of the grid-based saturated groundwater flow model by enabling groundwater intercepted and drained out of nodes by the sewer system to be accounted for.

SCENARIO MODELING EXAMPLE

Simple scenario modeling of this nature was undertaken as an exercise for Bessacarr-Cantley, in part because the identification of sewer gain areas on the south side of the study area for February 2004 coincided with emerging reports of localised groundwater flooding. Much of the metropolitan area of Doncaster is low-lying, with ground-level only a few metres above



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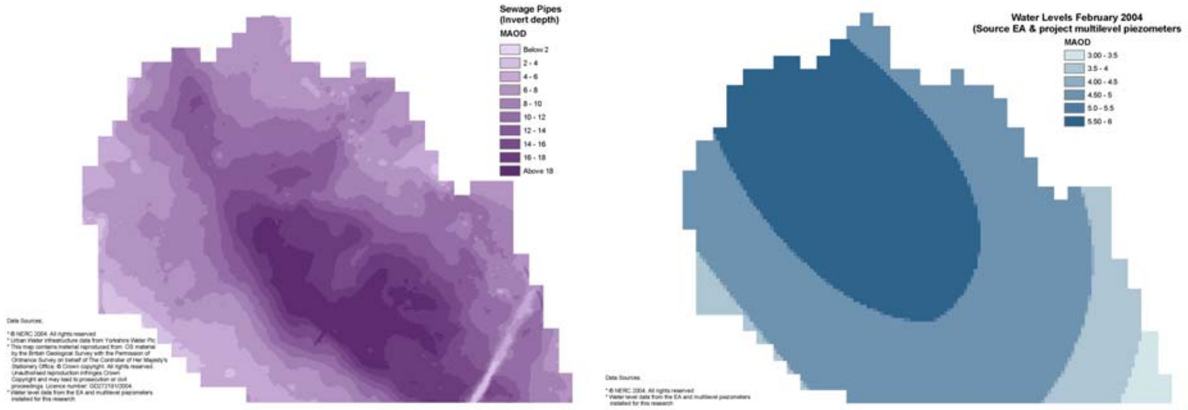
A. Digital Surface Model from NextMap©



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- * Water level data from the EA and multilevel piezometers installed for this research

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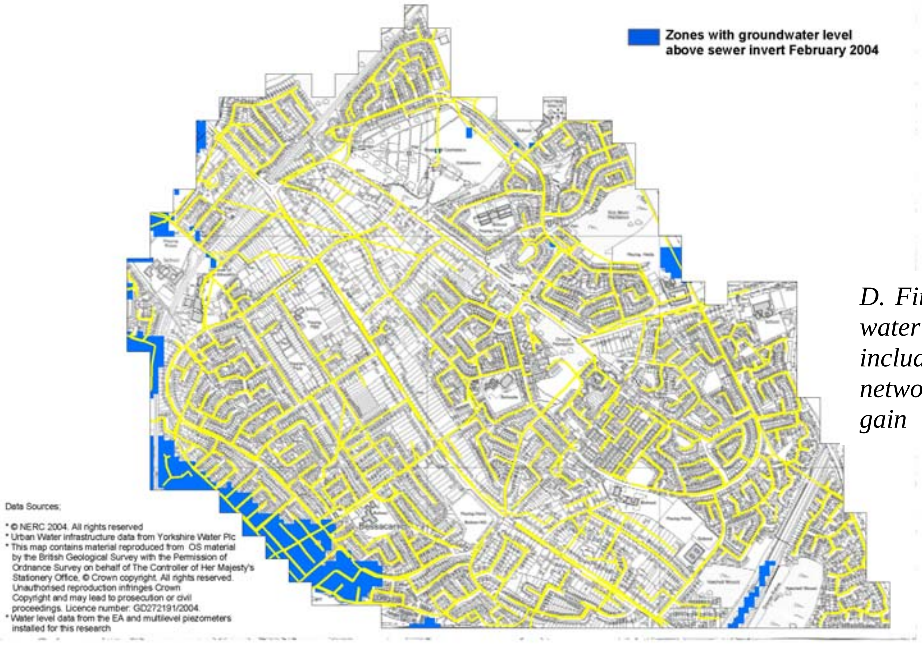
- * Water level data from the EA and multilevel piezometers installed for this research

B. Sewer Pipes Surface (Invert Depth).

C. Water table surface



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- * Water level data from the EA and multilevel piezometers installed for this research

D. Final Sewer gain/loss map for water levels from February 2004, including reaches of sewer networks prone to groundwater gain

Figure 4. Development of sewer gain/loss map for Bessacarr-Cantley using digital terrain, sewer invert level and groundwater table surface.

sea-level. In some of these areas sewer design by the water utility and its predecessors has long recognized that gain from groundwater needs to be taken into account. However east of the town, abstraction by the Doncaster wellfield has been substantial for many years and grew steadily until the late 1980's (Fig 5).

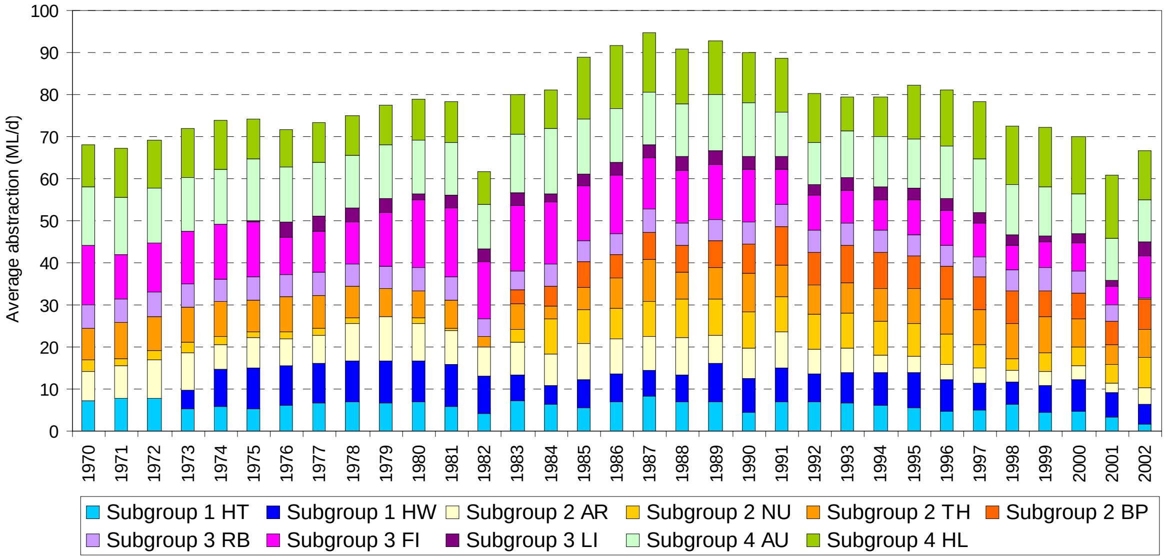

Figure 5. Pumping trends in Doncaster wellfield 1970-2002

Pumping kept water levels depressed, and during the 1990s, concern over wetland derogation just to the east of the wellfield, led to a programme of pumping regime changes mutually agreed with the environmental regulator. The changes had reduced annual abstraction to about 70% of the 1987-89 totals by 2002 (Fig. 5).

Coincidentally, the recovery of water levels in Bessacarr by 2003 may have been accentuated by local changes in abstraction. Pumping from the Rossington Bridge Pumping Station (in the catchment of which part of the study area lies) ceased from April 2001 to January 2003 while renovation works were carried out, and subsequently continued at about one third capacity. This would have resulted in a period of historically low pumping radius influence in the study area during 2003 and 2004 during the AISUWRS project field investigations.

The groundwater model deployed in the AISUWRS model chain was used in a predictive mode to run scenarios. The regional-scale groundwater model developed for the Environment Agency and Yorkshire Water was written in FORTRAN in 1993 (Brown and Rushton, 1993), extended in 1997 by Shepley (Shepley, 2000) and is regarded as a well-calibrated regional flow model that adequately represents the aquifer conditions in the Doncaster area. This original model, was first translated into the MODFLOW code and then slightly modified in order to use it to best effect in the field study area. The two-dimensional areal model is 265 km² in size. The cell size increases from 100 m x 100 m in the Bessacarr-Cantley study area to 1 x 1 km at its boundaries. This grid-refinement allowed more detailed simulation of local groundwater conditions.

The model was run in a steady-state mode for the scenario modelling. Several scenarios were run by altering pumping rates in the public supply boreholes in closest proximity (i.e. Rossington Bridge, Nutwell and Armthorpe pumping stations) in order to estimate their effect on groundwater levels. These model runs confirmed that changes in the pumping regime have strong effects on the groundwater level in Bessacarr-Cantley. Generally, elevated pumping rates keep the groundwater table several metres below the ground surface throughout the district. The gradual reduction in abstraction rates, such as has occurred since the late 1980s, results in water level recovery.

Figure 6 illustrates two of the scenarios. Map A uses pumping volumes from the 1987 peak year for abstraction. Troublesome shallow groundwater is confined to the extreme southwest margin adjacent to the Potteric Carr wetland. The scenario in Map B reduces pumping rates at Rossington Bridge, Nutwell and Armthorpe to 35% of the 1987 peak. This pumping regime is an operationally reasonable scenario. Groundwater-gain conditions extend to more low-lying parts of the district, including the road cutting of the M18 motorway on its southeastern edge. The scenario modelling illustrates that groundwater levels in the study area are currently finely balanced through pumping. Further significant reductions in local abstraction could see substantial segments of the sewer network in the study area below the water table, which may have implications for continued effective drainage and groundwater flooding hazard in Bessacarr-Cantley's more low-lying areas.

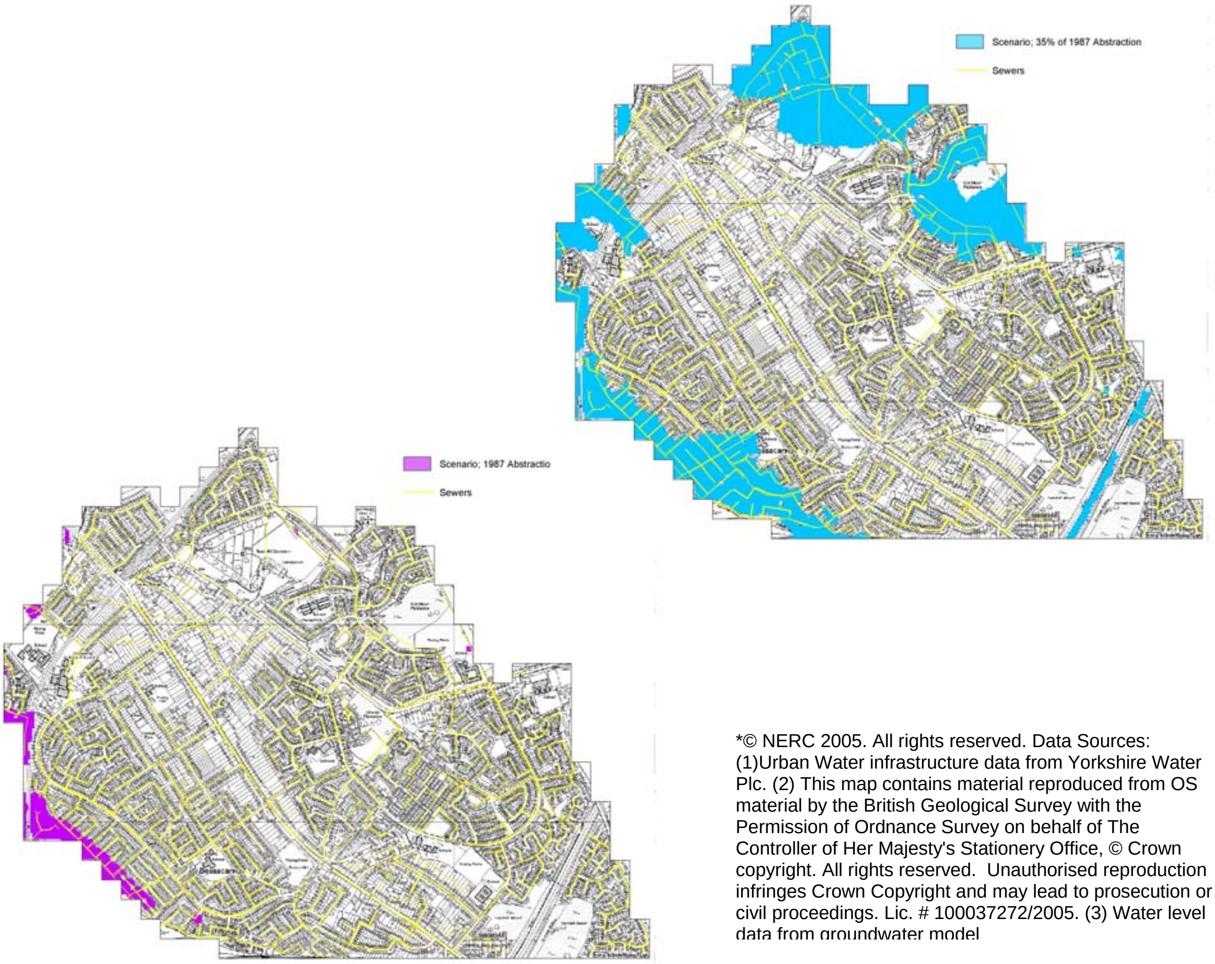

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Figure 6. Scenarios illustrating change in areas of potential groundwater surcharge to sewers under different public water supply pumping regimes.

Conclusions

- A GIS-based technique has been developed which can use existing data to assess whether and where sewer surcharging by shallow groundwater is becoming a problem. The technique uses digital terrain mapping, pipe network databases from the water utility and local groundwater surface observations to derive OD-referenced surfaces, and these can be overlain on the mapped sewer network to assess which assets may be gaining groundwater from the underlying aquifer.
- The technique is sensitive to inaccuracies in the ground elevation surface, and confidence in digital terrain mapping tools is much increased by combining this surface with ground control from pipe network reference locations tied in by surveying to trigonometric points.
- If the water level surface is generated by a calibrated groundwater flow model, scenario modeling can indicate the possible effects of local changes in groundwater recharge and discharge on sewer networks, such as those brought about by increased urban recharge or local decreases in wellfield pumping regimes. A smaller cell-size than is generally employed in regional flow models would be required if sufficiently small segments of the pipe network were to be mapped.
- A worked example of the technique demonstrated for the Doncaster suburb of Bessacarr-Cantley shows the importance of considering the urban drainage dimension when managing nearby wellfield abstraction regimes to meet environmental as well as water supply objectives.

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3 Outcomes and Conclusions

3.1 OUTCOMES

In this section the outcomes are listed again but placed in the context of the specific objectives of the work package as listed in Section 1.2.

3.1.1 Assess relative magnitude of sources of urban groundwater recharge and their effects on the quality and availability of water for public and private supply.

- Detailed analysis of water supply records (daily and hourly) has provided much information about night-time leakage volumes, average per capita water usage and the application of irrigation in the study area. Irrigation values range between 6 and 22% of total water supply volume, depending on the leakage control district. Overall, leakage from pressurised mains was found to be about 10% of the supplied volume. This does not include losses beyond the household connection, i.e. pipeline leaks on properties, wastage inside the house due to faulty ball cocks etc.
- Recharge volumes in the urban study area were found to be similar to natural (pre-urbanisation) recharge. The deficit of natural recharge due to the extent of impermeable area is replaced by leakage from pressurised mains and from the sewage and pluvial drainage system.
- Current leakage rates from the sewerage system were deduced to be in the range of 20 to 45 mm/a corresponding to a total leakage of 7-15% of annual sewage throughput or 3.0 to $7.5 \cdot 10^{-5}$ l/sec/m of pipeline. Such values are not dissimilar (given the magnitudes of errors involved) to previously-reported sewer volume losses of typically 3 to 5% (Ellis et al., 2004).
- A number of pollution indicator species show little more variation than that encountered in neighbouring rural catchments, confirming that in hydrochemical and microbiological terms, the adverse effect of urban recharge on the measured quality parameters of the underlying groundwater in the Bessacarr-Cantley area has been limited so far. This is ascribed to the combined effects of a non-industrial prior land-use history, locally high storage capacity in the friable upper aquifer and particularly the availability of dilution from precipitation entering urban green space areas. Also it should be noted that rural wells are not free of anthropogenic influences. However, shallow wells in and near the urban areas indicate a trend towards increasing influence of sewage-derived recharge. Private boreholes, usually withdrawing groundwater from shallow depths are influenced by urban recharge in and near the urban areas. Public supply wells, located farther outside the urban area and withdrawing water from depths below 30 mbgl are less influenced.
- An increase in sewer leakage rates would exacerbate the currently observed effects.

3.1.2 Detail the distribution and persistence of standard sewage indicators and sewage derived viruses and their seasonal fluctuations.

- Concentrations of microbial sewage indicators (bacterial as well as viral indicators) and pathogenic viruses in sewage were found to vary over several orders of magnitude, both on a daily and an annual time-scale. This has to be considered when using them to assess quantitatively the influence of sewage on microbial groundwater quality. Chemical parameters were found to vary significantly on a daily time-scale in the sewer but the quarterly sampling suggests the variation may be less on an annual time-scale.
- No significant seasonal variations were detected for either the chemical or the microbial parameters measured in the groundwater.

- Monitoring of the specially installed multilevels provided detailed depth profiles of various groundwater parameters. Recharge with the chemical and microbiological characteristics of sewage were typically detected to depths of about 35 m. below ground level.
- Nitrate levels were not found to be significantly or consistently enhanced in the upper aquifer, and ammonium concentrations were universally low (which is consistent with the aerobic condition of the aquifer throughout). This suggest that either nitrate is being stored in the unsaturated zone or that it is being converted to gaseous nitrogen (and lost to the system) in complex bacterially-mediated nitrification/denitrification reactions via nitrite in the vicinity of leaks.
- Faecal indicator micro-organisms, sulphite reducing clostridia (SRC) as well as faecal streptococci were detected to depths of 60 mbgl. This suggests deep penetration of modern (<50 year old) water. SRC spores are known to be long-lived and therefore they are expected to survive in groundwater over a timescale from months to years. However, faecal streptococci, with a half-life of typically 10s to 100s of days, are less persistent. Their occurrence is therefore more surprising. The conceptual picture of groundwater flow in this area needs to recognise the presence of very recent recharge at significant depths. It is concluded that flow is likely to be a product of slower (matrix–flow) and faster (fracture-flow) components. This conceptualisation has risk implications for sewage-derived contaminants such as viruses because it implies a more extensive penetration of pathogens than would otherwise be suspected from the evidence of groundwater dating tracers and intergranular hydraulic flow calculations.

3.1.3 Describe vertical variations in lithology, structure and vertical hydraulic gradients in the aquifer.

- The local sandstone aquifer is documented as a homogeneous fine-grained sandstone with pebble layers and mudstone/mud pellet conglomerate horizons. In other regions the sandstone was found to be relatively well-cemented but the drilling works undertaken during this project confirmed anecdotal reports that much of the upper few tens of metres of the formation in this part of South Yorkshire are unstable and poorly cemented.
- The sandstone aquifer is locally overlain by complex sequences of Quaternary sediments of fluvial and glacial origin.
- Vertical hydraulic gradients in the multilevel wells were found to be small ranging from 0 to 0.0018. This could be a consequence of the small topographic differences in the research area where the highest points are only a few tens of meters above sea level.
- Surprisingly, seasonal changes in water level were well-propagated through the aquifer, with deep access ports not only responding to rainfall recharge events but also showing negligible time-delay compared with ports located close to the water table. It is concluded that the upper 50m of saturated aquifer shows good hydraulic continuity in the study area, probably as a results of linked fracture systems in the harder more well-cemented horizons.
- It is assumed that the vertical hydraulic gradients strongly depend on the long-term pumping regime at the public water supply wells located down-stream of the focus study area. Overall, the downward gradients at the multilevel wells indicate that the focus study area is located in the recharge area of the aquifer.

3.1.4 Provide key information to design and calibrate models and to assist the quantification of recharge sources.

- **Urban Volume and Quality model (UVQ):** the physical settings were assembled from pipeline information, land use maps and air photographs to assess the degree of surface sealing, drainage direction, etc. This information was fed into a GIS to facilitate the distribution of supply and calibration information. Water inflow information was mostly available or could be deduced (e.g. water supply volumes, precipitation, garden irrigation) although some fields need to be supplemented by generic data (e.g. split of water use types).

  To calibrate UVQ some existing information on sewage flow was used but additional sewage flow and concentration measurements were taken during the field campaigns. This information has proved to be crucial because the loadings of different use types (e.g. toilet, kitchen, etc.) were not analysed and they are reported to be highly variable.

- Additional and more detailed measurements of sewage flow and concentration in different neighbourhoods would have been helpful to better constrain the model but no such level of detail was anticipated.

- Doubt remains about the actual recharge under open areas (e.g. playing fields, gardens, road verges etc). Assessing spatial recharge from precipitation is still one of the biggest problems in groundwater management. Assumptions can be made and documented values from similar settings can be added but the uncertainty connected with this input volume needs to be considered in the subsequent models.

- **Pipe Leakage Model (PLM):** The detailed pipe network information (pipe length, number of connections, flow direction etc) provided the data for the input files for the model. Leakage estimates generated from the model can then be calibrated, using recharge fluxes inferred from samples from shallow levels of the newly installed multilevel piezometers

- **Unsaturated Flow and Transport Models:** the physical setup of these models will be done using approximations from available soil classification datasets which are based on texture, thickness and ascribed physical characteristics. Calibration of these models (e.g. flow rates, inactivation rates, adsorption rates, etc.) will not be possible directly because no direct measurements are available.

- **Groundwater Flow and Transport Models:** Much effort has been invested to provide a detailed database to setup and calibrate the groundwater models because groundwater is the final receptor of all inputs and flows that have been modelled.

3.2 CONCLUSIONS

- The collection, collation and re-interpretation of new information, not presented earlier, had to continue throughout the project in order to obtain the necessary datasets for the interpretation and modelling tasks. Background information on drinking water supply volumes, mains leakage and, particularly, the pipeline infrastructure are essential for the success of the overall project tasks and hence the extra effort spent in obtaining this information was justified.

- GIS analysis techniques were used to assess the data and redistribute them onto the defined UVQ modelling areas (neighbourhoods), which are the basic building blocks of the modelling suite. A very useful outcome of this task was that it led to the development of a new technique to determine areas of groundwater ingress into the sewage system. This information is of more general application because it would allow, for instance, a water utility to study different groundwater abstraction regimes that partly control the water table beneath the urban area and hence design for infiltration or exfiltration conditions in affected parts of the pipe network.

- The sampling strategy applied in Doncaster has proved to be successful. Five sampling campaigns from June 2003 to November 2004 have provided a wealth of water quality and hydraulic head information. These data from both the 11 chosen regional wells and the 5 specially-installed multilevel monitoring wells have provided information about the local flow system and the impacts of urbanisation at different spatial and temporal timescales.
- The regional wells have provided the wider framework for the focus study area of Bessacarr/Cantley and confirmed that groundwater flow is driven by the public supply wellfield located to the east of Doncaster. Local variations in groundwater chemistry are apparent; groundwater in the south is calcite dominated and almost saturated with oxygen while that to the north tends to be subject to reducing conditions and usually contains higher concentrations of iron and manganese. These variations are probably controlled by the permeability of the overlying superficial deposits.
- New information from both drilling logs and cores from the 5 dedicated multilevel piezometers and from detailed geophysical logging of the existing EA water level monitoring well Cantley Tower borehole have shown that the local sandstone aquifer is weakly cemented to depths up to 30m. This contrasts with other regions where the Triassic sandstone is known to be moderately to well-cemented (e.g. Birmingham, Nottingham), and fracture-flow acknowledged to be important.
- Packer testing of the Cantley Water Tower observation well showed little difference in the bulk permeability of the near watertable zone and that between 40-60m depth, both giving values of about 5 m/d.
- The sewerage system was regularly sampled over the same period (June 2003 to Nov. 2004) as the groundwater wells. Measurements taken during this project have confirmed the general sanitary engineering observation that daily variations in sewage volumes in a suburban area are significant. The observed trend follows a strong peak in the morning with another increase in the afternoon, whereas sewage volumes during the night are low. Chemical, isotopic and microbial results have shown large daily variations in sewage indicator concentrations. One of the major drivers for these variations is toilet waste as it correlates with the largest variations in measured flows and contains large amounts of nutrients (nitrate and phosphate). In fact, detailed temporal measurements of micro-organisms (e.g. total coliforms, faecal coliforms, sulphite reducing clostridia (SRC), enteric viruses) have proven to be a useful method to infer the differences in domestic wastewater input required by UVQ (toilet, bathroom, laundry, kitchen), the first model in the modelling suite of AISUWRS. Attempts were made to assess daily source loadings of different contaminants because it was found that values in the literature were wide-ranging and led to increased uncertainties in the model output.
- The multilevel piezometers have shown that vertical hydraulic gradients are quite small in this area (0-9cm over 60m depth), partly because of the subdued topography. Furthermore, vertical trends in hydrochemical and isotopic ($\delta^{13}C$) composition of groundwater have enabled a distinction between natural background recharge and sewer-derived recharge. Recharge from leaking sewers in Doncaster is estimated to be up 20-45 mm/yr. (i.e. ~10-20% of total recharge), significantly more than previously assumed. This information will be crucial to calibrate and better constrain the pipe leakage model, which is the second model in the ASIUWRS modelling suite.
- The urban tracers proving most useful to quantitatively assess urban recharge were potassium, sodium, boron and SF_6. The most useful qualitative tracers were the alkalinity, $\delta^{13}C$ ratios, the faecal indicators sulfite reducing clostridia (SRC) and faecal streptococci and CFCs.

- The largest frequency of positive detects of the bacterial faecal indicators, and to a lesser extent the enteric viruses were found in the shallow multilevel intervals (0-30m) of the multilevel wells where the largest sewage contributions to recharge were estimated from the major and minor ion mass balance calculations. It should be noted though that the magnitude of these microbial faecal indicators was low and the groundwater is not grossly contaminated. Interestingly, indicator micro-organisms as well as enteric viruses were also found at depths of up to 60mbgl. Similar findings resulted from several depth-specific groundwater sampling campaigns in Nottingham and Birmingham, where indicator micro-organisms and pathogens were detected at depths up to 60mbgl. The occurrence of faecal indicators in Doncaster corresponds with the profiles of the groundwater dating tracers CFC-11, CFC-12 and SF_6.
- Groundwater dating tracers have proved useful in estimating horizontal and vertical transport rates of groundwater. The combination of dating tracers and microbial indicators has proved invaluable in providing a new conceptual picture of groundwater flow in this area of the Permo-Triassic sandstone aquifer. This conceptualisation suggests predominantly Darcian flow in the uppermost section of the aquifer with an increasing contribution of preferential flow through higher conductivity zones such as fractures or pebbly horizons with depth. In fact, both features are often observed and well documented in the Triassic sandstone. The results will be used to adapt and calibrate the final groundwater flow and transport models.

References not separately cited with papers

Eiswirth, M 2002. AUSUWRS Description of Work. Project Document EVK-2001-00229. 68pp. University of Karlsruhe, Karlsruhe GermanyMicrobiological data summary

Appendix 1: Final report– Microbiological data

Bolton Hill

Depth	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
mbgl		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
16	14/11/2003	19		200	<1	3	120	<1	<1	-	-			
16	19/02/2004	<1		<1	<1	2	<1	6	<1	-	-			
16	25/05/2004	<1		<1	<1	<1	<1 (<1)	<1 (2)	<1	-	-			
16	07/09/2004	2		1	1	<1	<1	1						
22	14/11/2003	<1		1	<1	<1	15	<1	<1	-	-			
22	19/02/2004	<1		<1	<1	<1	<1	<1	<1					
22	25/05/2004	<1		<1	<1	<1	<1	<1	<1					
22	07/09/2004			<1	<1	<1	<1	<1						
22	07/09/2004	<1		<1	<1	<1	<1	<1						
28	14/11/2003	1		<1	<1	<1	<1	<1						
28	19/02/2004	<1		<1	<1	<1	<1	<1						
28	25/05/2004	<1		<1	<1	<1	<1	<1	<1					
28	07/09/2004	<1		<1	<1	<1	<1	<1						
34	14/11/2003	<1		<1	<1	<1	<1	<1						
34	19/02/2004	<1		<1	<1	<1	<1	1						
34	19/02/2004	<1		<1	<1	<1	<1	<1						
34	25/05/2004	<1		<1	<1	<1	<1	<1						
34	25/05/2004	<1		<1	<1	<1	<1	<1	<1	-	-			
34	07/09/2004	<1		<1	<1	<1	<1	<1						
39	14/11/2003	<1		<1	<1	<1	<1	<1						
39	19/02/2004	<1		<1	<1	1	<1	<1						
39	25/05/2004	<1		<1	<1	<1	<1	<1						
39	07/09/2004	<1		<1	<1	<1	<1	<1	<1					
39	07/09/2004			<1	<1	<1	<1	<1						
45	14/11/2003	<1		<1	<1	<1	<1	<1						
45	19/02/2004	<1		<1	<1	<1	<1	<1						
45	25/05/2004	<1		<1	<1	<1	<1	<1						
45	07/09/2004	<1		<1	<1	<1	<1	<1	<1	-	+			
51	14/11/2003	<1		<1	<1	<1	<1	<1						
51	19/02/2004	<1		<1	<1	<1	<1	<1					>100000	460
51	25/05/2004	<1		<1	<1	<1	<1	<1						
51	07/09/2004	<1		<1	<1	<1	<1	<1						

Haslam Park 1

Depth	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
mbgl		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
10	11/11/2003	<1		1	1	4	1	<1	<1	-	-			
10	17/02/2004	<1		10	10	10	<1	3						
10	26/05/2004	<1		<1	<1	<1	<1	<1	<1	-	-			
10	08/09/2004	<1		1	<1	2	<1	2						
10	04/11/2004	<1		1	<1	<1	3	<1					4960	>100000
14	11/11/2003	<1		<1	<1	15	<1	<1	<1	-	-			
14	11/11/2003	<1		<1	<1	<1	<1	<1						
14	17/02/2004	1		10	10	10	<1	10						
14	26/05/2004	<1		<1	<1	<1	<1 (<1)	5 (2)						
14	08/09/2004	<1		<1	<1	<1	<1	<1	<1	+	-			
14	04/11/2004	<1		1	<1	<1	3	<1					640	2160
21	11/11/2003	<1		<1	<1	107	18	<1						
21	17/02/2004	2		40	40	900	<1	2	<1	-	-			
21	26/05/2004	<1		<1	<1	<1	<1	9	<1					
21	08/09/2004	<1		<1	<1	1	<1	5	<1					
28	11/11/2003	<1		1	1	600	26	<1						
28	17/02/2004	<1		10	10	10	<1	<1	<1	-	-			
28	26/05/2004	<1		<1	<1	<1	<1	1						
28	08/09/2004	<1		<1	<1	<1	<1	3						
35	11/11/2003	1		<1	<1	<1	40	<1						
35	17/02/2004	<1		10	10	10	<1	1						
35	17/02/2004	<1		10	10	10	<1	<1						
35	26/05/2004	<1		<1	<1	<1	<1	1	<1					
35	08/09/2004	<1		<1	1	1	<1	<1					7200	420
45	11/11/2003	1		<1	<1	22	<1	<1						
45	17/02/2004	<1		10	10	10	<1	<1						
45	26/05/2004	<1		9	<1	<1	<1	<1						
45	26/05/2004	<1		6	<1	2	<1	<2						
45	08/09/2004	<1		<1	<1	1	<1	4						
45	04/11/2004	40		24	14	20	<1	<1						
60	11/11/2003	<1		<1	<1	<1	<1	<1						
60	17/02/2004	<1		10	10	10	<1	6					>100000	6720
60	26/05/2004	<1		<1	<1	<1	<1	5	<1	-	-			
60	08/09/2004	<1		<1	<1	<1	<1	<1						
60	04/11/2004	<1	13	<1	<1	2	<1	<1					270	4480

Haslam Park 2

Depth	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
mbgl		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
10	12/11/2003	3		4	2	11	54	<1	<1	-	-			
10	12/11/2003	3		2	2	10	62	<1						
10	18/02/2004	7		2	<1	<1	<1	40	<1	-	-			
10	25/05/2004	18		2	<1	21	<1	30	<1	-	-			
10	09/09/2004	6		1	1	14	<1	49	<1					
10	04/11/2004	<1		<1	<1	<1	<1	<1						
14	12/11/2003	<1		<1	<1	<1	1	<1	<1	-	-			
14	18/02/2004	<1		<1	<1	<1	<1	<1						
14	25/05/2004	<1		1	1	1	<1	9						
14	09/09/2004	<1		<1	<1	2	<1	<1						
19	12/11/2003	<1		<1	<1	<1	<1	<1						
19	18/02/2004	<1		2	<1	<1	<1	<1						
19	26/05/2004	<1		4	4	1	<1	8						
19	09/09/2004	2		3	<1	1	<1	1	<1					
19	04/11/2004	<1		<1	<1	2	1	<1						
27	12/11/2003	<1		<1	<1	1	<1	<1						
27	18/02/2004	<1		5	<1	<1	<1	2						
27	26/05/2004	<1		<1	<1	1	<1	<1	<1					
27	09/09/2004	<1		<1	<1	6	<1	5						
35	12/11/2003	3		9	<1	<1	5	<1						
35	18/02/2004	<1		2	<1	<1	<1	<1						
35	26/05/2004	1		2	<1	<1	<1	3						
35	09/09/2004	<1		<1	<1	<1	<1	<1						
45	12/11/2003	<1		1	<1	2	11	<1						
45	18/02/2004	<1		2	<1	<1	<1	1	<1	-	+			
45	26/05/2004	<1		<1	<1	<1	<1 (<1)	<1 (1)	<1					
45	09/09/2004	<1		<1	<1	1	<1	<2						
45	04/11/2004	<1	2	<1	<1	<1	<1	<1						
60	12/11/2003	2		7	1	11	61	<1						
60	17/02/2004	<1		10	10	10	<1	42					>100000	4800
60	26/05/2004	<1		<1	<1	1	<1	3	<1	-	-			
60	09/09/2004	<1		<1	<1	3	<1	1	<1	-	+			
60	04/11/2004	<1		2	<1	<1	<1	<1						

McAuley School

Depth	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
mbgl		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
9	13/11/2003	9		<1	<1	<1	1	<1	<1	-	-			
9	20/02/2004	9		<1	2	<1	<1	1	<1	-	-			
9	27/05/2004	9		<1	<1	<1	<1	1	<1	-	-			
9	10/09/2004	9		2	<1	3	<1	<1						
9	05/11/2004	9		<1	<1	<1	<1	<1						
14	13/11/2003	14		<1	<1	<1	9	<1	<1	-	-			
14	20/02/2004	14		<1	2	<1	<1	2	<1					
14	27/05/2004	14		<1	<1	<1	<1	<1						
14	10/09/2004	14		<1	<1	3	<1	<1	<1	+	-			
14	05/11/2004	14		<1	<1	<1	<1	<1						
21	13/11/2003	21		<1	<1	<1	<1	<1						
21	20/02/2004	21		<1	2	4	<1	1						
21	27/05/2004	21		<1	<1	<1	<1	<2						
21	10/09/2004	21		<1	<1	4	<1	<1						
21	05/11/2004	21		<1	<1	<1	<1	<1						
28	13/11/2003	28		<1	<1	<1	<1	<1	2	-	-			
28	13/11/2003	28		<1	<1	<1	<1	<1						
28	20/02/2004	28		<1	<1	<1	<1	<1	<1					
28	27/05/2004	28		1	<1	<1	<1	<1						
28	10/09/2004	28		1	<1	<1	1	<1						
28	10/09/2004	28		<1	<1	2								
36	13/11/2003	36		<1	<1	<1	<1	<1						
36	20/02/2004	36		<1	2	<1	<1	7						
36	20/02/2004	36		<1	2	4	<1	7						
36	27/05/2004	36		14	<1	1	<1	<1	<1	-	-			
36	10/09/2004	36		<1	<1	<1	<1	<1						
36	10/09/2004	36		<1	<1	2	<1	5						6720
45	13/11/2003	45		<1	<1	<1	1	<1						
45	20/02/2004	45		<1	2	<1	<1	3						
45	27/05/2004	45		<1	<1	<1	<1	<1	<1					
45	10/09/2004	45		<1	<1	1	<1	<1	<1					
60	13/11/2003	60		<1	<1	1	13	<1						
60	20/02/2004	60		<1	2	<1	<1	<1	<1					>100000
60	27/05/2004	60		<1	<1	<1	<1	<1						
60	27/05/2004	60		<1	<1	<1	<1	<1						
60	10/09/2004	60		<1	<1	4	<1	1						
60	05/11/2004	60		<1	<1	<1	<1	<1						

Sandall Beat

Depth	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
mbgl		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
16	10/11/2003	<1		<1	<1	3	14	<1	<1	-	-			
16	16/02/2004	<1		10	<10	<10	<1	7	<1	-	-			
16	24/05/2004	<1		1	<1	<1	<1	1	<1	-	-			
16	06/09/2004	<1		<1	<1	14	<1	<1	<1	-	+			
16	06/11/2004	<1	<1	3	<1	<1	14	<1					230	4000
21	13/11/2003	1		<1	<1	<1	<1	<1	<1	-	-			
21	16/02/2004	222		70	40	860	2	19	<1	-	-			
21	24/05/2004	<1		2	<1	1	<1	9	<1	-	-			
21	06/09/2004	2		<1	<1	<1	<1	2	<1					
21	06/11/2004	<1	<1	<1	<1	<1	<1	<1					20	1040
26	10/11/2003	2		<10	<10	<10	<1	<1						
26	10/11/2003	6		<10	<10	<10	<1	<1						
26	16/02/2004	3		10	10	10	<1	<1	<1	-	-			
26	16/02/2004	2		10	10	10	<1	<1						
26	24/05/2004	<1		<1	<1	<1	<1	<1	<1					
26	06/09/2004	<1		<1	<1	<1	<1	<1						
31	10/11/2003	<1		<1	<1	<1	<1	<1						
31	16/02/2004	<1		10	<10	<10	<1	<1	<1	-	-			
31	24/05/2004	<1		<1	<1	<1	<1	<1						
31	06/09/2004	<1		<1	<1	<1	<1	<1						
36	10/11/2003	<1		<1	<1	1	1	<1						
36	16/02/2004	<1		10	<10	<10	<1	1					>100000	2480
36	24/05/2004	<1		<1	<1	<1	<1	4	<1					
36	24/05/2004	<1		<1	<1	<1	<1	2						
36	06/09/2004	<1		<1	<1	9	<1	<1						
36	06/09/2004			<1	<1	7	<1	<1						
36	06/11/2004	<1		6	4	<1	<1	<1					20	8320

Shallow Piezometers

Name	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
		cfu/100ml	cfu/10L	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
Piezo 1	14/11/2003	<1		<1	<1	1	53	<1						
Piezo 1	19/02/2004	<1		<1	<1	<1	<1	<1						
Piezo 1	25/05/2004	<1		<1	<1	2	<1(<1)	<1(1)						
Piezo 2	14/11/2003	<1		6	< 2	22	45	<1						
Piezo 2	18/02/2004	<1		<1	<1	<1	<1	<1						
Piezo 2	25/05/2004	<1		1	<1	<1	<1	<1						
Piezo 2	25/05/2004	7		420	<1	128	<1	<1						

Regional Wells

Name	Date	Field TTC		Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type	Total Viable Counts	
		cfu/100ml	cfu/10L	cfu/100ml	cfu/100 ml	cfu/100ml	cfu/100 ml	pfu/ml	pfu/10L	+/-	+/-		cfu/ml @22C	cfu/ml @37C
Allotments	03/06/2003	<1		14900	1	2	6000	9						
Armthorpe	10/09/2004	<1		<1	<1	<1	<1	<1					6230	1088
Beech Tree Nursery	04/06/2003	<1		<1	<1	<1	<1	<1	<1	-	-			
Beech Tree Nursery	11/11/2003	<1		<1	<1	<1	<1	<1						
Beetch Tree Nursery	18/02/2004	<1		<1	<1	<1	<1	<1						
Cantley Water Tower	12/11/2003	<1		1	<1	1	<1	<1						
Cantley Water Tower	17/02/2004	<1		<1	2	<1	<1	<1					800	640
Cantley Water Tower	26/05/2004	<1		<1	<1	<1	<1	<1						
Cantley Water Tower	08/09/2004	<1		3	<1	1	<1	<1						
Crow Tree Farm	03/06/2003	<1		<1	<1	<1	1	<1						
Crow Tree Farm	11/11/2003	<1		<1	<1	<1	<1	<1						
Crow Tree Farm	18/02/2004	<1		<1	<1	<1	<1	<1						
Elmstone Farm	03/06/2003	1		18	<1	<1	<1	<1						
Elmstone Farm	11/11/2003	<1		<1	<1	<1	<1	<1						
Elmstone Farm	17/02/2004	<1		<1	2	<1	<1	<1						
Gatewood Grange	04/06/2003	<1		4	<1	<1	1	<1						
Gatewood Grange	13/11/2003	<1		<1	<1	<1	3	<1						
Gatewood Grange	17/02/2004	<1		10	10	10	<1	22						
Gatewood Grange	26/05/2004	<1		2	<1	<1	<1	<1						
Gatewood Grange	08/09/2004	2		<1	<1	<1	8	17						
Lings Farm	04/06/2003	<1		<1	<1	2	<1	<1						
Lings Farm	11/11/2003	<1		<1	<1	<1	2	<1						
Lings Farm	17/02/2004	59		20	20	10	<1	1						
Misson Quarry	03/06/2003	<1		<1	<1	1	7	<1						
Misson Quarry	13/11/2003	<1		<1	<1	<1	2	<1						
Misson Quarry	18/02/2004	<1		<1	<1	<1	<1	<1						
Nutwell BH2	10/09/2004	<1		<1	<1	<1	<1	<1					1280	360
Peglars	04/06/2003	<1		<1	<1	<1	<1	<1	<1	-	-			
Peglars	30/07/2003	<1		<1	<1	<1	2	<1						
Peglers	12/11/2003	<1		<1	<1	1	2	<1						
Peglers	18/02/2004	<1		<1	<1	<1	<1	1					>100000	>100000
Peglers	25/05/2004	<1		<1	<1	<1	<1	8						
Peglers	07/09/2004	<1		<1	<1	<1	<1	<1						
Race course	02/06/2003	<1		<1	<1	<1	1	<1	<1	-	-			
Race course	10/11/2003	<1		<1	<1	<1	<1	<1						
Race Course	16/02/2004	<1		2	<1	<1	<1	1					400	190
Race Course	24/05/2004	<1		<1	<1	<1	<1	<1						
Race Course	06/09/2004	5		<1	<1	<1	<1	<1						
Rossington Bridge	04/11/2004			<1	<1	<1	<1	<1						
Sandall Common Farm	13/11/2003	<1		<1	<1	<1	1	<1						
Sandall Common Farm	18/02/2004	<1		<1	<1	<1	<1	1						
Warning Tongue Lane	04/06/2003	<1		<1	<1	3	2	1						
Warning Tongue Lane	13/11/2003	<1		18	2	11	8	<1						
Warning Tongue Lane	17/02/2004	<1		10	10	10	<1	6						
Warning Tongue Lane	26/05/2004	<1		<1	<1	<1	<1	<1						
Warning Tongue Lane	09/09/2004	<1		<1	<1	<1	<1	<1						

Sewerage

Name	Date	time	Field TTC	Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type
			cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-	
Burnham Close	29/07/2003		3.73E+06	4.80E+07	1.00E+07	5.00E+06	1.40E+06	524	250	+	+	Cox B1,B2, Noro Type 1,2
Burnham Close	14/11/2003		3.77E+06	6.00E+06	1.90E+06	3.20E+05	>30000	>1000	2000	-	+	Cox B2
Burnham Close	19/02/2004		3.90E+06	2.00E+06	4.30E+05	2.10E+05	700	>30000	<1	+	+	
Burnham Close	27/05/2004			1.20E+07	2.00E+06	4.00E+06	>300	>30000	35556	+	-	Cox B2 x 3, Cox B4 x 9
Burnham Close	10/09/2004		9.70E+06	4.18E+07	6.80E+06	7.00E+05	400	500000	4000	+	+	Cox B3 x 3; Unknown x 1
Everingham Road	29/07/2003		2.32E+06	2.25E+08	5.10E+07	3.00E+06	1.54E+06	736	2600	+	+	Cox B1,B12, Noro Type 1
Everingham Road	14/11/2003		6.00E+06	7.00E+06	4.00E+06	7.00E+05	4000	>1000	2700	+	+	Cox B1, Cox B3, Polio 3
Everingham Road	19/02/2004		9.20E+06	3.60E+06	1.60E+06	3.20E+05	1180	>30000				
Everingham Road	27/05/2004			1.80E+07	6.00E+06	2.50E+06	>300	>30000	55000	+	-	Cox B4 x 17
Everingham Road	10/09/2004		8.00E+06	2.85E+07	1.22E+07	1.60E+06	1280	820000	<1	+	+	N/A
Everingham Road	04/11/2004	12:15		1.22E+07	1.65E+06	8.00E+05		1990				
Everingham Road	04/11/2004	13:00		1.23E+07	2.80E+06	5.50E+05		1315				
Everingham Road	04/11/2004	13:45		1.28E+07	2.65E+06	2.40E+06		1810				
Everingham Road	04/11/2004	15:20		1.38E+07	4.80E+06	8.00E+05	8.40E+04	2830				
Everingham Road	04/11/2004	16:05		1.45E+07	5.70E+06	1.00E+05	1.52E+05	3110				
Everingham Road	04/11/2004	17:05		6.80E+06	1.80E+06	2.60E+05	7.20E+04	2475				
Everingham Road	04/11/2004	18:55		9.60E+06	3.40E+06	5.00E+05	6.80E+04	2720				
Everingham Road	04/11/2004	20:20		8.70E+06	2.30E+06	8.00E+05	8.80E+04	5800				
Everingham Road	04/11/2004	22:45		9.90E+06	3.00E+06	4.00E+05	4.40E+04	12240				
Everingham Road	05/11/2004	0:30		1.52E+07	6.90E+06	1.10E+06	7.60E+04	2710				
Everingham Road	05/11/2004	6:40		6.40E+06	1.80E+06	2.00E+05	1.44E+05	3020				
Everingham Road	05/11/2004	9:10		7.10E+06	2.20E+06	2.70E+06	5.20E+05	63600	11111	+	+	Cox B2 x 7, Cox B5 x 1
Everingham Road	05/11/2004	9:55		1.80E+07	4.10E+06	3.40E+06	9.30E+05	27200				
Everingham Road	05/11/2004	10:40		1.22E+07	2.80E+06	2.30E+06	9.00E+04	5360				
Warning Tongue Lane	29/07/2003		1.74E+06	2.20E+07	3.00E+06	2.80E+05	4.20E+04	79	111	+	+	Cox B1, Noro Type 1,2
Warning Tongue Lane	14/11/2003		2.06E+06	4.00E+06	1.75E+06	3.10E+05	>30000	146	<1	-	-	
Warning Tongue Lane	19/02/2004		2.50E+06	1.20E+07	4.00E+05	4.00E+04	940	>30000	<1	+	+	
Warning Tongue Lane	27/05/2004			2.00E+07	4.00E+06	3.00E+06	15	>30000	40000	+	-	Cox B4 x 7
Warning Tongue Lane	08/09/2004		8.00E+06	5.20E+07	9.20E+06	1.80E+06	820	70000	7000	+	+	Cox B2 x 1, Cox B4 x 6
Warning Tongue Lane	04/11/2004	11:40		1.13E+07	1.20E+06	1.05E+06		63	1000	+	+	Cox B3 x 1
Warning Tongue Lane	04/11/2004	12:45		8.10E+06	1.55E+06	1.00E+06		150.5				
Warning Tongue Lane	04/11/2004	13:20		3.45E+06	6.80E+04	2.60E+06		588				
Warning Tongue Lane	04/11/2004	15:05		2.24E+07	4.60E+06	6.00E+05	1.21E+05	3960				
Warning Tongue Lane	04/11/2004	15:40		5.70E+06	2.40E+06	4.00E+05	1.33E+05	4700	2000	+	+	Cox B3 x 2
Warning Tongue Lane	04/11/2004	16:55		1.10E+07	2.60E+06	1.00E+05	2.00E+04	5740				
Warning Tongue Lane	04/11/2004	18:10		2.39E+07	4.40E+06	1.00E+05	1.48E+05	890				
Warning Tongue Lane	04/11/2004	20:05		3.30E+06	1.40E+05	1.00E+05	4.80E+04	1650				
Warning Tongue Lane	04/11/2004	22:10		1.42E+07	8.00E+05	4.00E+05	2.40E+04	160				
Warning Tongue Lane	05/11/2004	0:00		5.80E+06	9.00E+05	9.00E+05	8.00E+04	260				
Warning Tongue Lane	05/11/2004	6:05		5.50E+06	3.00E+05	1.00E+05	1000	3660				
Warning Tongue Lane	05/11/2004	8:45		9.70E+06	2.20E+06	1.90E+06	3.70E+04	870	5000	+	-	Cox B3 x 5
Warning Tongue Lane	05/11/2004	9:30		1.82E+07	6.10E+06	2.30E+06	3.30E+04	18160				
Warning Tongue Lane	05/11/2004	10:15		1.02E+07	2.30E+06	8.40E+06	7.20E+04	5300				

Stormwater

Name	Date	Time	Field TTC	Total Coliforms	Faecal Coliforms	Faecal Streptococci	SRC	Coliphage	Enterovirus Plaque Assay	Enterovirus PCR	Norovirus PCR	Virus Type
			cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	cfu/100ml	pfu/ml	pfu/10L	+/-	+/-	
Ditch	19/02/2004		TNTC	350	90	100	48	>300				
Howden Close Drain	19/02/2004		24000	3900	1100	200	20	1300	<1	-	-	-

Appendix 2 – Hydrochemistry data

Multilevels

Bolton Hill

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L
S03-01694	16.5	Oct-03	0.148	< 0.02	0.088	0.0506	< 0.001	73.5	0.00049			27.7	< 0.0009	0.00262	< 0.002	7.99		256.85	0.071	199	241	16.2	< 0.0004	0.0151
S04-00212	16.5	Feb-04	0.0352	0.02	0.062	0.058	< 0.0008	82.1	0.00099			26.7	< 0.0006	0.002	0.0031	6.36		432	0.00256	249	284	13.2	< 0.0007	
S04-00517	16.5	May-04	0.0091	< 0.01	0.077	0.061	< 0.0004	87.6	< 0.0004			32.3	< 0.0006	0.0022	< 0.001	10.73		386	0.00396	243	302	12.8	< 0.0007	0.0118
S04-01068	16.5	Sep-04	< 0.01	< 0.01	0.093	0.0649	< 0.0004	91.7	< 0.0004			31.4	< 0.0005	< 0.002	0.0039	8	2.55	250	0.0559	295		12.4	< 0.0008	0.0111
S03-01696	22.2	Oct-03	0.0917	< 0.02	0.065	0.0449	< 0.001	81.3	< 0.0004			29.5	< 0.0009	0.00146	< 0.002	7.59		256.51	0.0149	211	288	13.2	< 0.0004	0.0127
S04-00214	22.2	Feb-04	0.0285	0.0109	0.056	0.0518	< 0.0003	83.1	0.00053			31	< 0.0005	< 0.001	0.0017	7.21		333	0.00118	283		11.5	< 0.0007	0.0136
S04-00519	22.2	May-04	0.0146	< 0.01	0.054	0.0536	< 0.0004	84.6	< 0.0004			36	< 0.0006	0.0017	< 0.001	8.44		307	0.00308	268	304	11.1	< 0.0007	0.0107
S04-01069	22.2	Sep-04	< 0.01	< 0.01	0.068	0.0597	< 0.0004	87.6	< 0.0004	12.98	4.85	33.6	< 0.0005	< 0.002	0.0028	8.6	1.65		0.00108	299.87		11.2	< 0.0008	0.00964
S03-01695	28.7	Oct-03	0.071	< 0.02	0.063	0.086	< 0.001	97.9	0.00074			63.4	< 0.0009	0.00066	< 0.002	6.64		256.74	0.0141	167	240	8.71	< 0.0004	0.0102
S04-00218	28.7	Feb-04	0.0329	< 0.008	0.092	0.0916	< 0.0003	107	0.00077			69.8	< 0.0005	< 0.001	0.0052	6.17		359	0.0017	212	242	9.25	< 0.0007	0.0154
S04-00520	28.7	May-04	0.0158	< 0.01	0.04	0.0893	0.00043	99.8	< 0.0004			75.5	< 0.0006	0.0013	< 0.001	10.31		306	0.00103	238		8.46	< 0.0007	0.0116
S04-01070	28.7	Sep-04	< 0.01	< 0.01	0.049	0.0944	< 0.0004	101	< 0.0004	8.69	3.58	69.3	< 0.0005	< 0.002	0.0028	7.7	1.62		0.0008	243.8		8.6	< 0.0008	0.00994
S03-01693	34.7	Oct-03	0.117	< 0.02	0.066	0.122	< 0.001	101	0.0008			113	< 0.0009	0.00084	< 0.002	5.66		256.87	0.0365	102	151	6.03	< 0.0004	0.0062
S04-00217	34.7	Feb-04	0.028	< 0.008	< 0.05	0.145	< 0.0003	101	0.00071			110	0.00063	< 0.001	0.0016	5.06		349	0.00109	124		6.03	< 0.0007	0.0094
S04-00518	34.7	May-04	0.0158	< 0.01	< 0.03	0.144	< 0.0004	100	< 0.0004			114	< 0.0006	0.0013	< 0.001	8.42		337	0.00115	138		6.09	< 0.0007	0.00819
S04-01071	34.7	Sep-04	< 0.01	< 0.01	0.028	0.151	< 0.0004	105	< 0.0004	1.74	3.53	106	< 0.0005	< 0.002	0.0025	7	1.15	250	0.00808	153.59		6.11	< 0.0008	0.00662
S03-01690	39.7	Oct-03	0.0894	< 0.02	< 0.08	0.13	< 0.0008	91.6	< 0.0005			103	< 0.0009	0.00061	< 0.002	6.26		256.75	0.00262	79	129	7.03	< 0.0006	0.0054
S04-00215	39.7	Feb-04	0.0325	< 0.008	< 0.05	0.132	< 0.0003	90.1	0.00056			102	< 0.0005	< 0.001	< 0.001	5.77		343	0.00254	100		6.49	< 0.0007	0.0084
S04-00516	39.7	May-04	< 0.005	< 0.01	0.043	0.141	< 0.0004	88.5	< 0.0004			104	< 0.0006	0.0012	< 0.001	7.31		267.6	0.00493	149	132	6.43	< 0.0007	0.00481
S04-01072	39.7	Sep-04	< 0.01	< 0.01	< 0.02	0.148	< 0.0004	91.7	< 0.0004			98.6	< 0.0005	< 0.002	0.0025	7.6	0.97	250	0.00085	129.21		6.32	< 0.0008	0.00347
S03-01691	45.7	Oct-03	0.0866	< 0.02	< 0.08	0.0672	< 0.0008	51.9	< 0.0005			41.7	0.00119	0.00099	< 0.002	6.24		256.8	0.00578	72	86	6.08	< 0.0006	0.002
S04-00213	45.7	Feb-04	0.0334	< 0.008	< 0.05	0.104	< 0.0003	50.8	0.0002			44.8	0.00048	< 0.001	< 0.001	7.05		248	0.00284	73		5.89	< 0.0007	0.0059
S04-00514	45.7	May-04	0.0324	< 0.01	0.032	0.15	< 0.0004	50.6	< 0.0004			49.4	< 0.0006	0.0012	< 0.001	10.2		375	0.00914	85		5.79	< 0.0007	0.00496
S04-01073	45.7	Sep-04	< 0.01	< 0.01	0.021	0.17	< 0.0004	55	< 0.0004	1.47	0.33	48.4	< 0.0005	< 0.002	0.0021	9.3	1.2	250	0.0136	92.64		6.39	< 0.0008	0.00403
S03-01692	51.4	Oct-03	0.111	< 0.02	0.086	0.0842	< 0.001	56.6	0.00049			45.8	< 0.0009	0.00068	< 0.002	6.24		256.92	0.0464	59	94	2.6	< 0.0004	0.0042
S04-00216	51.4	Feb-04	0.0251	0.0081	< 0.05	0.096	< 0.0003	56.4	0.00052			45.2	< 0.0005	< 0.001	0.0013	7.7		338	0.23	80		2.42	< 0.0007	< 0.001
S04-00515	51.4	May-04	0.0283	< 0.01	< 0.03	0.0949	< 0.0004	53.9	< 0.0004			48.2	< 0.0006	< 0.001	< 0.001	11.2		390	0.00586	83		2.32	< 0.0007	0.00265
S04-01074	51.4	Sep-04	< 0.01	< 0.01	< 0.02	0.101	< 0.0004	56.7	< 0.0004			43	< 0.0005	< 0.002	0.0016	9.8	1.55	251	0.00197	97.52		2.42	< 0.0008	< 0.0005

(continued on next page)

Bolton Hill continued

Sample ID	Depth m	Sampling date	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01694	16.5	Oct-03	26.6	0.00698	< 0.002	18.4	< 0.007	0.0043	0.0093	0.128	0.0112	7.85	0.000353			6.81	90.9	0.0995	12.5		3.96	0.00208	< 0.0002	0.0092
S04-00212	16.5	Feb-04	30.8	0.00808	< 0.0008	16.9	< 0.006	0.00618	0.0042	0.13	< 0.004	7.53	0.00031	786		7.74	90.6	0.0779	10.6		4.27	0.00131	0.00018	0.0176
S04-00517	16.5	May-04	32.7	0.00428	< 0.0009	15.9	< 0.03	0.00576	0.00085	0.066	< 0.004	7.55	0.000361	715		7.73	95.3	0.0769	10.5	5.04	4.07	0.00163	< 0.0001	0.018
S04-01068	16.5	Sep-04	33.6	0.00584	< 0.001	16.1	< 0.01	0.0067	0.00213	0.075	< 0.004	7.45	0.0004	767		7.75	96.3	0.0903	10.5	4.08	3.63	< 0.001	0.00029	0.02
S03-01696	22.2	Oct-03	31.3	0.00769	< 0.002	20	< 0.007	0.0046	0.0072	0.189	0.0138	7.51	0.000397			6.59	88.4	0.0646	12.6		5.38	0.00202	< 0.0002	0.009
S04-00214	22.2	Feb-04	32.9	0.00527	< 0.0008	19.6	< 0.006	0.00616	0.0027	0.115	< 0.002	7.48	0.000303	814		7.05	85.8	0.0535	10.5		5.34	0.00118	< 0.0001	0.0144
S04-00519	22.2	May-04	32.4	0.00403	< 0.0009	18.1	< 0.03	0.00568	0.00945	0.095	< 0.004	7.46	0.000376	722		6.89	83.8	0.0535	10.8	5.83	4.91	0.0024	< 0.0001	0.0168
S04-01069	22.2	Sep-04	33.7	0.00253	< 0.001	18.8	< 0.01	0.0067	0.0187	0.099	< 0.004	7.53	0.00033	843	0.97	7.07	86.7	0.0511	11.8	4.78	4.5	< 0.001	< 0.0003	0.0182
S03-01695	28.7	Oct-03	37.5	0.0071	< 0.002	23.9	< 0.007	0.0056	0.0024	0.264	0.0148	7.74	0.000385			5.69	144	0.034	12.6		6.7	0.00142	< 0.0002	0.01
S04-00218	28.7	Feb-04	38.4	0.00206	< 0.0008	24.7	< 0.006	0.00665	0.0041	0.107	< 0.002	7.85	0.00026	977		5.73	147	0.0345	10.8		6.56	0.00185	< 0.0001	0.017
S04-00520	28.7	May-04	37.5	0.00105	< 0.0009	23	< 0.03	0.00626	0.00105	0.098	< 0.004	7.88	0.000409	829		5.7	141	0.0334	10.4	8.22	6.37	0.00125	< 0.0001	0.0174
S04-01070	28.7	Sep-04	38.5	0.00069	< 0.001	23.9	0.272	0.0067	0.0238	0.106	< 0.004	7.82	0.0003	984	0.61	5.66	143	0.0348	11.1	6.07	5.77	< 0.001	< 0.0003	0.0192
S03-01693	34.7	Oct-03	38.3	0.00418	< 0.002	18.7	< 0.007	0.0066	0.0021	0.18	0.0156	7.87	0.000391			4.92	156	0.0371	12.6		4.82	0.00195	< 0.0002	0.0104
S04-00217	34.7	Feb-04	38.2	0.00229	< 0.0008	18.7	< 0.006	0.00596	0.0035	0.132	< 0.002	7.92	0.000252	990		5.06	156	0.0367	10.9		4.85	0.00219	< 0.0001	0.018
S04-00518	34.7	May-04	37.5	0.00143	< 0.0009	18.3	< 0.03	0.00614	0.0098	0.117	< 0.004	8.01	0.000368	869		4.99	152	0.0355	10.6	5.89	4.82	0.00205	< 0.0001	0.0174
S04-01071	34.7	Sep-04	36.9	0.00889	< 0.001	19.8	< 0.01	0.0068	0.00316	0.116	< 0.004	7.95	0.00035	988	0.85	4.85	147	0.0399	11.1	4.96	4.73	< 0.001	< 0.0003	0.0194
S03-01690	39.7	Oct-03	35.7	0.00826	< 0.002	17.5	< 0.007	0.00635	< 0.001	0.168	0.0106	7.75	0.000443			4.75	142	0.0404	12.6		5.61	0.00226	< 0.0001	0.0078
S04-00215	39.7	Feb-04	34.7	0.00286	< 0.0008	17.2	< 0.006	0.0062	0.0037	0.13	< 0.002	7.91	0.00029	915		4.8	141	0.0356	9.9		5.61	0.0025	< 0.0001	0.0188
S04-00516	39.7	May-04	34.1	0.0031	< 0.0009	17.1	< 0.03	0.00618	0.00075	0.122	< 0.004	7.94	0.000405	791		4.73	146	0.0358	11.7	6.48	5.54	0.00138	< 0.0001	0.0165
S04-01072	39.7	Sep-04	35.5	0.00068	< 0.001	16.5	0.109	0.0071	0.00878	0.109	< 0.004	7.94	0.00033	908		4.72	140	0.0357	11	6.05	5.39	< 0.001	< 0.0003	0.0183
S03-01691	45.7	Oct-03	19.8	0.00704	< 0.002	12.8	< 0.007	0.004	0.0022	0.166	0.0084	7.8	0.000289			3.85	77	0.0537	12.8		8.17	0.00187	< 0.0001	0.0063
S04-00213	45.7	Feb-04	19.8	0.00508	< 0.0008	11.5	< 0.006	0.00412	0.0015	0.136	< 0.002	7.9	0.000186	564		3.79	74.2	0.0393	10.7		8.11	0.00275	< 0.0001	0.0128
S04-00514	45.7	May-04	20	0.00378	< 0.0009	12.1	< 0.03	0.00355	0.00375	0.107	< 0.004	8.04	0.00027	497		3.82	73.4	0.0341	10.7	9.33	7.93	0.00193	0.00014	0.0122
S04-01073	45.7	Sep-04	21	0.00318	< 0.001	12.3	< 0.01	0.0057	0.00344	0.102	< 0.004	7.87	0.00029	571	0.80	3.97	75.8	0.0354	10.8	7.64	7.5	< 0.001	< 0.0003	0.0149
S03-01692	51.4	Oct-03	22.7	0.00296	< 0.002	11.5	< 0.007	0.0037	0.0036	0.093	0.013	7.92	0.000339			4.38	76.3	0.0482	12.6		9.86	0.00145	0.00017	0.0082
S04-00216	51.4	Feb-04	22.7	0.00245	< 0.0008	10.1	< 0.006	0.00437	0.0023	0.073	< 0.002	7.87	0.000217	586		4.47	76	0.0477	10.8		9.97	0.00144	< 0.0001	0.012
S04-00515	51.4	May-04	22	0.00121	< 0.0009	9.84	< 0.03	0.00382	0.00245	0.062	< 0.004	8	0.000322	501		4.33	71.5	0.0456	11.2	12.7	9.89	< 0.0008	< 0.0001	0.015
S04-01074	51.4	Sep-04	22.8	0.00081	< 0.001	10.1	0.234	0.0064	0.0232	0.066	< 0.004	7.89	0.00022	574		4.43	73.4	0.0471	10	9.99	9.85	< 0.001	< 0.0003	0.0149

P1

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01697	Oct-03	0.189	< 0.02	0.103	0.0644	< 0.001	67.6	0.00056	95.7	0.00102	0.00065	< 0.002			0.102	161	221	5.24	< 0.0004	< 0.002	25.5
S04-00190	Feb-04	0.0278	< 0.01	0.095	0.072	< 0.0008	78.2	0.00087	177	< 0.0006	< 0.001	0.004	8.93	423	< 0.0008	134	169	5.47	< 0.0007		30.1
S04-00542	May-04	0.0214	< 0.01	0.112	0.0714	< 0.0004	76.8	< 0.0004	142	< 0.0006	< 0.001	< 0.001	10.5	388.8	0.0234	140		5.21	< 0.0007	< 0.0006	28.9

Sample ID	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01697	Oct-03	0.00267	< 0.002	76	< 0.007	0.0053	0.002	0.043	0.0126	6.84	0.000409		0.863	64.5	0.138	13.7		13.9	0.00099	0.00119	0.0123
S04-00190	Feb-04	0.00107	< 0.0008	74.9	< 0.006	0.00514	< 0.001	0.0618	< 0.004	6.99	0.0003	1116	0.542	56.2	0.16	8.8		9.4	< 0.001	0.00123	0.0188
S04-00542	May-04	0.0167	< 0.0009	66.7	< 0.06	0.00592	< 0.001	< 0.02	< 0.004	7.01	0.000385	914	0.793	89.8	0.161	10.5	12.6	10	< 0.0008	0.00115	0.0161

P2

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01698	Oct-03	0.277	< 0.02	0.091	0.184	< 0.001	135	0.00155	184	0.00666	0.00045	< 0.002				6.2	200	272	9.7	0.00046	< 0.002	17.7
S04-00191	Feb-04	0.0337	< 0.01	0.065	0.165	< 0.0008	108	0.00155	164	0.00229	< 0.001	0.0025	8.72		314	4.2	252		8.17	< 0.0007		14.5
S04-00543	May-04	0.0121	< 0.01	0.088	0.113	< 0.0004	74.6	< 0.0004	131	0.00098	< 0.001	< 0.001	9.22		144	0.903	222	268	8.19	< 0.0007	< 0.0006	9.48
S04-01067	Sep-04	< 0.01	< 0.01	0.086	0.145	< 0.0004	86.6	< 0.0004	166	0.00155	< 0.002	0.0027	8.9	4.73	246	1.4	265.74		9.24	< 0.0008	< 0.0005	11.1

Sample ID	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01698	Oct-03	1.29	< 0.002	81.7	0.371	0.0078	0.0038	0.06	0.0148	6.85	0.000503		3.83	84.4	0.349	12.2		2.65	0.00205	0.00076	0.0156
S04-00191	Feb-04	0.821	< 0.0008	101	0.28	0.00775	0.0023	0.0511	< 0.004	7.1	0.00033	1231	2.89	83.2	0.294	8.4		2.77	< 0.001	0.00044	0.0256
S04-00543	May-04	0.163	< 0.0009	117	< 0.06	0.00605	0.0057	< 0.02	< 0.004	7.26	0.000407	937	2.2	53.3	0.206	13	6.62	4.93	< 0.0008	0.00017	0.0178
S04-01067	Sep-04	0.344	< 0.001	131	0.454	0.0065	0.00401	< 0.01	< 0.004	7.02	0.00041	1196	3.01	65.1	0.242	16.5	5.84	4.61	< 0.001	< 0.0003	0.0207

Haslam Park 1

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01668	10.0	Oct-03	0.117	< 0.02	< 0.1	0.0477	< 0.0004	46.3	< 0.0004			26.4	< 0.001	0.0013	< 0.002	9		258.08	0.649	112	130	4.74	< 0.0003	0.0019	20.6
S04-00200	10.0	Feb-04	0.306	< 0.01	0.073	0.0585	< 0.0008	40.9	0.00063			24.8	< 0.0006	< 0.001	< 0.002	4.71		406	0.0825	128		5.28	< 0.0007		18.2
S04-00524	10.0	May-04	0.0225	< 0.01	0.079	0.0345	< 0.0004	36.9	< 0.0004			23.6	< 0.0006	< 0.001	< 0.001	5.44		243	0.0444	118		4.02	< 0.0007	< 0.0006	17.4
S04-01053	10.0	Sep-04	0.062	< 0.01	0.064	0.038	< 0.0004	38.6	< 0.0004	12.77	5.50	22.8	< 0.0005	< 0.002	0.0013	6.3	2.49	250	1.82	117.02		3.65	0.00109	< 0.0005	18
S03-01674	14.5	Oct-03	0.074	< 0.02	0.123	0.073	< 0.0008	56.6	< 0.0005			32.5	< 0.0009	0.00046	< 0.002	5.67		258.11	0.00301	132	169	4.66	< 0.0006	0.0014	22.8
S04-00203	14.5	Feb-04	0.316	0.02	0.153	0.0834	< 0.0008	58.2	0.00081			33.2	< 0.0006	< 0.001	0.007	4.89		414	0.072	149		5.06	< 0.0007		22.8
S04-00525	14.5	May-04	0.0659	< 0.01	0.087	0.0783	< 0.0004	59.7	< 0.0004			37.9	< 0.0006	< 0.001	0.0071	5.27		314	0.0174	161		4.81	< 0.0007	0.00561	23.4
S04-01054	14.5	Sep-04	0.018	< 0.01	0.074	0.0797	< 0.0004	59.1	< 0.0004	11.90	4.86	33.3	< 0.0005	< 0.002	0.003	5.7	2.38	250	0.00875	170.66		4.67	< 0.0008	0.00302	23.2
S03-01673	21.0	Oct-03	0.0937	< 0.02	0.137	0.116	< 0.0008	57	< 0.0005			32.5	< 0.0009	0.00048	< 0.002	5.57		257.89	0.00725	128		5.73	< 0.0006	0.0018	22.4
S04-00197	21.0	Feb-04	0.158	0.013	0.09	0.172	< 0.0008	58.7	0.00062			33.9	< 0.0006	< 0.001	0.0039	4.73		425	0.0317	135		6.29	< 0.0007		22.2
S04-00523	21.0	May-04	0.186	< 0.01	0.096	0.159	< 0.0004	59.7	< 0.0004			36.4	< 0.0006	0.0012	0.0016	6.74		420	0.0411	135	165	6.01	< 0.0007	0.00598	23.2
S04-01055	21.0	Sep-04	< 0.01	< 0.01	0.071	0.142	< 0.0004	58.4	< 0.0004	10.48	4.37	32.7	< 0.0005	< 0.002	0.0028	5.9	1.29	250	0.0278	168.22		5.91	< 0.0008	0.0057	22.9
S03-01672	28.0	Oct-03	0.0745	< 0.02	0.137	0.194	< 0.0008	63	< 0.0005			38.4	< 0.0009	0.00083	< 0.002	5.88		257.86	0.00201	118		5.91	< 0.0006	0.0062	22.7
S04-00204	28.0	Feb-04	0.0305	0.018	0.089	0.315	< 0.0008	72.2	0.00078			42	< 0.0006	< 0.001	< 0.002	4.72		417	0.00189	154		6.62	< 0.0007		24.8
S04-00526	28.0	May-04	0.0132	< 0.01	0.069	0.269	< 0.0004	67.4	< 0.0004			44.8	< 0.0006	< 0.001	< 0.001	6.5		354	0.00207	156		5.87	< 0.0007	0.00567	23.9
S04-01056	28.0	Sep-04	0.02	< 0.01	0.039	0.275	< 0.0004	68.3	< 0.0004	5.23	2.14	41.6	< 0.0005	< 0.002	0.0024	8.3	1.16	251	0.00948	156.03		5.82	< 0.0008	0.0032	24.2
S03-01671	35.0	Oct-03	0.138	< 0.02	< 0.1	0.0676	< 0.0004	46.7	< 0.0004			24.3	< 0.001	< 0.001	< 0.002	5.92		257.86	0.0222	68		3.96	< 0.0003	0.0036	17
S04-00201	35.0	Feb-04	0.212	0.015	< 0.05	0.136	< 0.0008	45.1	0.00049			24.7	< 0.0006	< 0.001	0.0027	5.25		409	0.0522	51	75	3.74	< 0.0007		16.4
S04-00202	35.0	Feb-04	0.237	0.019	< 0.05	0.146	< 0.0008	45.8	0.00071			24.7	< 0.0006	< 0.001	< 0.002	5.25		409	0.0473	51	74	3.94	< 0.0007		16.4
S04-00528	35.0	May-04	0.526	< 0.01	< 0.03	0.153	< 0.0004	44.4	< 0.0004			28.7	< 0.0006	0.0013	0.0021	7.16		419	0.115	63		3.79	< 0.0007	0.00582	16.2
S04-01057	35.0	Sep-04	0.051	< 0.01	< 0.02	0.172	< 0.0004	43.7	< 0.0004	0.90	0.09	25.1	< 0.0005	< 0.002	0.003	8.8	0.84	250	0.0224	75.58		3.55	< 0.0008	0.00353	15.8
S03-01670	45.0	Oct-03	0.238	< 0.02	< 0.1	0.378	< 0.0004	31.5	< 0.0004			20.2	< 0.001	0.001	< 0.002	7.95		257.93	0.0879	35		1.78	< 0.0003	< 0.002	11.7
S04-00199	45.0	Feb-04	0.921	< 0.01	< 0.05	0.485	< 0.0008	29	< 0.0003			20.5	< 0.0006	0.0016	< 0.002	7.51		415	0.179	41		1.69	< 0.0007		10.7
S04-00529	45.0	May-04	0.0215	< 0.01	< 0.03	0.465	< 0.0004	28.9	< 0.0004			22.2	< 0.0006	< 0.001	< 0.001	8.31		444	0.00527	38		1.56	< 0.0007	< 0.0006	10.8
S04-01058	45.0	Sep-04	0.749	< 0.01	0.041	0.533	< 0.0004	29.9	< 0.0004	0.47	0.00	20.2	< 0.0005	< 0.002	0.0024	10.5	1	250	0.166	38.4		1.84	0.00146	< 0.0005	11.3
S03-01669	60.1	Oct-03	0.199	< 0.02	< 0.1	0.0616	< 0.0004	39.5	< 0.0004			19.2	< 0.001	0.0012	< 0.002	7.05		257.72	0.52	86	109	3.34	< 0.0003	< 0.002	14.8
S04-00198	60.1	Feb-04	0.527	< 0.01	< 0.05	0.064	< 0.0008	32.4	0.00051			17.4	< 0.0006	< 0.001	< 0.002	6.73		406	0.117	88		3.03	< 0.0007		12.1
S04-00527	60.1	May-04	0.0698	< 0.01	0.077	0.0861	< 0.0004	44.7	< 0.0004			28.4	< 0.0006	< 0.001	< 0.001	8.17		355	0.0127	155		4.83	< 0.0007	0.0006	17.5
S04-01059	60.1	Sep-04	0.402	< 0.01	0.068	0.0846	< 0.0004	40.8	< 0.0004	6.44	2.63	22.9	< 0.0005	< 0.002	0.0032	8.9	1.51	250	0.099	120.68		4.19	< 0.0008	< 0.0005	16.1

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01668	10.0	Oct-03	0.0185	< 0.003	8.54	< 0.007	0.0034	0.002	0.11	< 0.007	8.08		0.000315			5.6	52.8	0.057	11.2		4.26	0.00149	0.0002	0.0098
S04-00200	10.0	Feb-04	0.0254	< 0.0008	7.56	< 0.006	0.00378	0.0061	0.121	< 0.004	8.28		0.00019	470		5.91	49	0.0574	10.2		4.13	0.00134	< 0.0001	0.0141
S04-00524	10.0	May-04	0.0298	< 0.0009	5.89	< 0.03	0.00293	0.001	0.049	< 0.004	7.76		0.000286	390		5	46.1	0.0492	11.8	3.9	2.91	< 0.0008	< 0.0001	0.0109
S04-01053	10.0	Sep-04	0.0311	< 0.001	6.39	< 0.006	0.0044	0.00357	0.053	< 0.004	7.69		0.00016	378	1.84	5.44	45	0.0527	11	3.84	3.47	< 0.001	0.00035	0.0125
S03-01674	14.5	Oct-03	0.00398	< 0.002	10.9	< 0.007	0.00284	0.0044	0.132	0.007	8.11		0.000333			5.53	59.7	0.0516	11.4		5.47	0.00136	< 0.0001	0.006
S04-00203	14.5	Feb-04	0.00405	< 0.0008	11.9	< 0.006	0.00525	0.004	0.162	< 0.004	8.51		0.00024	590		6.48	62.9	0.0542	10.4		5.54	0.00157	0.00017	0.0219
S04-00525	14.5	May-04	0.00196	< 0.0009	11.8	< 0.03	0.0047	0.0031	0.085	< 0.004	8.13		0.000403	568		5.79	61	0.0526	11.2	7.27	5.5	0.00121	< 0.0001	0.019
S04-01054	14.5	Sep-04	0.00184	< 0.001	11.2	< 0.006	0.0056	0.00871	0.075	< 0.004	8.14		0.00023	548	1.33	5.73	60.1	0.0539	11	5.68	5.35	< 0.001	< 0.0003	0.0157
S03-01673	21.0	Oct-03	0.00292	< 0.002	12.5	< 0.007	0.0034	0.0052	0.211	0.0078	7.89		0.000311			4.89	70.8	0.0359	11		5.94	0.00126	< 0.0001	0.0064
S04-00197	21.0	Feb-04	0.00165	< 0.0008	13.8	0.0125	0.00451	0.0024	0.202	< 0.004	8.41		0.00026	571		5.13	74.1	0.0274	10.1		6.78	0.00106	< 0.0001	0.021
S04-00523	21.0	May-04	0.00144	< 0.0009	13.5	< 0.03	0.00424	0.00265	0.131	< 0.004	8.09		0.000312	533		5.27	71.1	0.036	10.7	8.06	6.14	0.0017	< 0.0001	0.0193
S04-01055	21.0	Sep-04	0.00283	< 0.001	13.5	< 0.006	0.0056	0.00175	0.123	< 0.004	8.06		0.00021	603	1.04	5.07	68.1	0.037	10.6	5.89	5.95	< 0.001	< 0.0003	0.0152
S03-01672	28.0	Oct-03	0.00313	< 0.002	16.6	< 0.007	0.00402	0.0036	0.191	0.0065	7.86		0.000357			4.29	80.7	0.0295	11.2		10.4	0.00198	< 0.0001	0.0066
S04-00204	28.0	Feb-04	0.00175	< 0.0008	18.6	< 0.006	0.00553	0.0035	0.223	< 0.004	8.34		0.00022	719		4.97	89.4	0.0288	10.5		12	0.00108	< 0.0001	0.0185
S04-00526	28.0	May-04	0.00121	< 0.0009	17.1	< 0.03	0.00449	0.0123	0.133	< 0.004	7.98		0.000397	688		4.64	79.8	0.0292	10.7	14.6	11.4	< 0.0008	< 0.0001	0.0162
S04-01056	28.0	Sep-04	0.00092	< 0.001	17	< 0.006	0.006	0.0124	0.128	< 0.004	7.99		0.00024	700	1.27	4.82	79.5	0.0341	10.2	11.5	11	< 0.001	< 0.0003	0.0167
S03-01671	35.0	Oct-03	0.00287	0.003	16.6	< 0.007	0.0045	0.0086	0.208	< 0.007	7.86		0.000303			4.25	76.7	0.0341	11.2		11	0.00165	< 0.0001	0.0059
S04-00201	35.0	Feb-04	0.00256	< 0.0008	14.8	< 0.006	0.00431	0.0023	0.193	< 0.004	8.45	7.67	0.00022	503		4.94	71.2	0.0335	10.4		11.5	0.00197	< 0.0001	0.0158
S04-00202	35.0	Feb-04	0.00149	< 0.0008	14.9	< 0.006	0.00417	0.0024	0.163	< 0.004	8.45	7.39	0.00022	503		4.97	73.3	0.0338	10.4		11.4	0.00126	< 0.0001	0.0168
S04-00528	35.0	May-04	0.00108	< 0.0009	14.5	< 0.03	0.00328	0.0038	0.129	< 0.004	7.54		0.000348	497		4.98	68.3	0.0312	10.8	14.8	11.8	0.00186	< 0.0001	0.0124
S04-01057	35.0	Sep-04	0.00062	< 0.001	14	0.123	0.0049	0.0271	0.107	< 0.004	6.4		0.00017	494	1.02	4.2	66	0.0319	10.7	12.5	11.6	0.0013	< 0.0003	0.0139
S03-01670	45.0	Oct-03	0.00265	< 0.003	8.16	< 0.007	0.0031	0.0016	0.158	< 0.007	7.93		0.000333			4.16	29.4	0.0299	11.1		13.9	0.00143	< 0.0001	0.0045
S04-00199	45.0	Feb-04	0.00144	< 0.0008	8.19	0.0368	0.00352	0.0128	0.129	< 0.004	8.46		0.00022	346		5.34	27.1	0.0283	9.6		13.8	0.00309	< 0.0001	0.0118
S04-00529	45.0	May-04	0.00031	< 0.0009	7.46	< 0.03	0.00191	0.00553	0.08	< 0.004	8.05		0.000171	337		3.68	25.8	0.027	10.6	17.3	13.8	0.00105	< 0.0001	0.00875
S04-01058	45.0	Sep-04	0.00166	< 0.001	7.99	0.143	0.0039	0.022	0.084	< 0.004	8.29		0.00025	352	0.79	5.17	26.9	0.0294	11.1	13	13.4	0.0023	< 0.0003	0.0121
S03-01669	60.1	Oct-03	0.00819	< 0.003	11.3	< 0.007	0.0033	0.0036	0.212	< 0.007	7.72		0.000233			4.53	36.7	0.0372	11.3		6.5	0.00158	0.00015	0.0057
S04-00198	60.1	Feb-04	0.00793	< 0.0008	8.63	0.0166	0.00384	0.0098	0.192	< 0.004	8.16		0.00018	146		5.03	26.6	0.0292	10.5		6.36	0.00256	< 0.0001	0.0167
S04-00527	60.1	May-04	0.00291	< 0.0009	10.5	< 0.03	0.00321	0.0024	0.179	< 0.004	7.65		0.00031	483		4.63	45.7	0.0345	11.4	7.14	5.53	0.00123	< 0.0001	0.0262
S04-01059	60.1	Sep-04	0.00306	< 0.001	9.89	0.0916	0.0045	0.0235	0.203	< 0.004	6.5		0.00027	445	1.07	5.2	38.9	0.0337	10.8	6.15	5.91	< 0.001	< 0.0003	0.0139

Haslam Park 2

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01678	9.5	Oct-03	0.961	< 0.02	< 0.08	0.0343	< 0.0008	36.5	< 0.0005			10.8	0.00093	0.00258	< 0.002	7.08		255.9	0.697	75	94	3.09	< 0.0006	< 0.001	16.3
S04-00206	9.5	Feb-04	0.222	< 0.01	0.084	0.0384	< 0.0008	32.2	0.00052			9.8	< 0.0006	0.0014	0.0058	5.59			0.32	93		6.29	< 0.0007		15.1
S04-00522	9.5	May-04	0.11	< 0.01	0.088	0.0349	0.00059	31.8	< 0.0004			10.7	0.00072	< 0.001	0.0022	8.84		371	0.0445	111	91.3	5.23	< 0.0007	< 0.0006	15
S04-01060	9.5	Sep-04	0.309	< 0.01	0.065	0.0565	0.00318	35.2	0.00056	8.33	3.06	10.1	0.00076	0.0021	0.0041	8.1	4.73	250	2.52	90.21		3.72	0.00211	< 0.0005	16
S03-01675	14.0	Oct-03	0.0568	< 0.02	0.1	0.0344	< 0.0008	43.3	< 0.0005			16.8	< 0.0009	0.0006	< 0.002			256.52	0.00152	86	108	3.93	< 0.0006	< 0.001	18.8
S04-00207	14.0	Feb-04	0.0332	< 0.01	0.092	0.0461	< 0.0008	48.5	0.00053			20.8	< 0.0006	< 0.001	0.0035	5.5		452	0.00401	82	121	5.06	< 0.0007		21
S04-00521	14.0	May-04	0.35	< 0.01	0.066	0.043	< 0.0004	47.4	< 0.0004			25.1	< 0.0006	0.0014	< 0.001	10.8		350	0.103	119		4.87	< 0.0007	< 0.0006	20.2
S04-01061	14.0	Sep-04	0.06	< 0.01	0.073	0.0517	< 0.0004	48.2	< 0.0004	14.93	4.12	21.2	< 0.0005	< 0.002	0.0059	8.9	2.14	250	0.0449	120.07		4.32	< 0.0008	< 0.0005	20.6
S03-01681	19.0	Oct-03	0.0752	< 0.02	< 0.08	0.0352	< 0.0008	49.1	< 0.0005			20	< 0.0009	0.00054	< 0.002	7		257.03	0.00635	88	118	4.44	< 0.0006	0.0035	19.9
S04-00208	19.0	Feb-04	0.0332	0.017	0.08	0.0452	< 0.0008	47.7	0.00048			20.5	< 0.0006	< 0.001	< 0.002	6.02		434	0.0048	88	123	4.92	< 0.0007		19.5
S04-00531	19.0	May-04	0.0445	< 0.01	0.098	0.0463	< 0.0004	48.3	< 0.0004			24.7	< 0.0006	< 0.001	0.0024	8.4		357	0.00836	124		5.11	< 0.0007	< 0.0006	19.9
S04-01062	19.0	Sep-04	0.042	< 0.01	0.067	0.0555	< 0.0004	48.9	< 0.0004	15.42	4.20	22.3	< 0.0005	< 0.002	0.002	8.9	2	250	0.0394	124.34		4.48	< 0.0008	< 0.0005	19.8
S03-01680	27.0	Oct-03	0.178	< 0.02	< 0.08	0.0233	< 0.0008	43.1	< 0.0005			16.4	0.00132	0.00061	< 0.002	6.66		255.82	0.0299		107	3.78	< 0.0006	< 0.001	18.5
S04-00211	27.0	Feb-04	0.0625	0.024	0.063	0.0395	< 0.0008	46.3	0.00059			19	< 0.0006	< 0.001	< 0.002	4.95		430	0.0123	90	110	4.58	< 0.0007		19.2
S04-00530	27.0	May-04	0.789	< 0.01	0.06	0.0535	< 0.0004	47.5	< 0.0004			23	< 0.0006	0.002	0.0019	7.32		382	0.259	127		4.72	< 0.0007	< 0.0006	19.9
S04-01063	27.0	Sep-04	0.074	< 0.01	0.062	0.0414	< 0.0004	45	< 0.0004	12.72	3.43	19.9	< 0.0005	< 0.002	0.0029	8.3	1.89	250	0.0832	117.02		3.85	< 0.0008	< 0.0005	18.6
S03-01679	35.0	Oct-03	0.703	< 0.02	< 0.08	0.0487	< 0.0008	35.4	< 0.0005			29.2	< 0.0009	0.00164	< 0.002	7.85		255.84	0.133	30	36	1.98	< 0.0006	< 0.001	11.8
S04-00209	35.0	Feb-04	0.0771	0.018	< 0.05	0.0976	< 0.0008	36	0.00038			33.8	< 0.0006	< 0.001	< 0.002	6.75		437	0.014	27		2.12	< 0.0007		12.5
S04-00533	35.0	May-04	0.459	< 0.01	0.031	0.126	< 0.0004	35.3	< 0.0004			36.1	< 0.0006	0.0016	< 0.001	8.89		346	0.0946	24	33	2.16	< 0.0007	< 0.0006	12.6
S04-01064	35.0	Sep-04	0.108	0.016	< 0.02	0.162	0.00046	36.1	< 0.0004	0.50	0.14	32	< 0.0005	< 0.002	0.0019	10.7	1.27	#REF!	0.0397	28.5		2.27	< 0.0008	< 0.0005	12.8
S03-01677	45.0	Oct-03	1.03	< 0.02	< 0.08	0.136	< 0.0008	24.4	< 0.0005			17.4	< 0.0009	0.00186	< 0.002	7.69		256.22	0.206	30	36	1.29	< 0.0006	< 0.001	8.95
S04-00210	45.0	Feb-04	0.154	0.013	< 0.05	0.209	< 0.0008	25.8	0.00036			18	< 0.0006	< 0.001	< 0.002	7.49		426	0.028	28	36	1.34	< 0.0007		9.53
S04-00534	45.0	May-04	0.175	0.011	< 0.03	0.242	< 0.0004	24.4	< 0.0004			20	< 0.0006	0.0013	< 0.001	9.43		354	0.0352	24	35	1.27	< 0.0007	< 0.0006	9.09
S04-01065	45.0	Sep-04	1.3	0.014	< 0.02	0.299	< 0.0004	24.4	< 0.0004	0.40	0.05	17.7	< 0.0005	0.0027	0.0016	11.3	0.77	250	0.307	48.76		1.51	0.00118	< 0.0005	9.23
S03-01676	60.4	Oct-03	0.776	< 0.02	< 0.08	0.0562	< 0.0008	23.7	< 0.0005			12.1	< 0.0009	0.00169	< 0.002	8.25		257.31	0.162	53	69	1.45	< 0.0006	< 0.001	7.56
S04-00205	60.4	Feb-04	0.294	0.019	0.064	0.0637	< 0.0008	28.7	0.00047			14.7	< 0.0006	< 0.001	< 0.002	6.26		414	0.067	70	70	1.91	< 0.0007		10.7
S04-00532	60.4	May-04	0.11	< 0.01	0.052	0.0513	< 0.0004	28	< 0.0004			16.5	< 0.0006	< 0.001	< 0.001	8		352	0.0237	26	68	2.28	< 0.0007	< 0.0006	11.1
S04-01066	60.4	Sep-04	0.161	< 0.01	0.023	0.0561	< 0.0004	28	< 0.0004	3.66	1.02	14.9	< 0.0005	< 0.002	0.0016	9.8	1.27	250	0.0413	73.14		1.94	0.00122	< 0.0005	10.9

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01678	9.5	Oct-03	0.0335	0.0028	7.03	0.0441	0.00507	0.0141	0.11	0.0078	6.9		0.000416			5.85	41.3	0.0706	11.6		11	0.00295	0.00094	0.0075
S04-00206	9.5	Feb-04	0.0184	< 0.0008	7.61	< 0.006	0.00441	0.0134	0.119	< 0.004	7.83		0.00017	409		4.75	39.2	0.0773			8.93	0.00134	0.00038	0.0185
S04-00522	9.5	May-04	0.0226	< 0.0009	5.69	< 0.03	0.00356	0.0046	0.028	< 0.004	7.81		0.000249	319		4.22	35.7	0.0731	10.8	11.6	9.07	0.00087	< 0.0001	0.0129
S04-01060	9.5	Sep-04	0.0477	< 0.001	6.25	0.0113	0.0066	0.015	0.098	< 0.004	7.55		0.00026	387	2.34	4.82	38.5	0.0844	11.3	8.46	9.5	< 0.001	0.00255	0.0323
S03-01675	14.0	Oct-03	0.0151	< 0.002	8.49	< 0.007	0.00341	0.0104	0.112	0.0064	7.52		0.000288			4.62	59.4	0.0652	11.7		9.69	0.00098	< 0.0001	0.0065
S04-00207	14.0	Feb-04	0.00753	< 0.0008	9.85	0.059	0.00466	0.0056	0.146	< 0.004	7.8	7.9	0.0002	516		5.47	65.6	0.0797	11.3		9.35	< 0.001	< 0.0001	0.017
S04-00521	14.0	May-04	0.0035	< 0.0009	9.83	< 0.03	0.00394	0.00355	0.08	< 0.004	8.04		0.0003	418		5.67	60.8	0.0719	10.7	11.3	8.32	0.00196	0.00021	0.0145
S04-01061	14.0	Sep-04	0.00298	< 0.001	8.85	0.0684	0.0047	0.0056	0.086	< 0.004	7.89		0.00024	519	0.88	5.2	59.8	0.0759	11.1	8.49	8.6	< 0.001	< 0.0003	0.0152
S03-01681	19.0	Oct-03	0.0101	< 0.002	9.49	< 0.007	0.0035	0.0081	0.159	0.0096	7.03		0.000364			4.59	66.7	0.0601	11.4		8.67	0.00143	< 0.0001	0.0076
S04-00208	19.0	Feb-04	0.00653	< 0.0008	8.94	< 0.006	0.00462	0.0029	0.164	< 0.004	7.72	7.89	0.00021	507		5.12	62.7	0.0661	10.9		8.39	0.00128	< 0.0001	0.0134
S04-00531	19.0	May-04	0.0033	< 0.0009	8.98	< 0.03	0.00362	0.0028	0.112	< 0.004	7.91		0.000306	500		4.91	60.8	0.0655	11	10.2	8.16	0.00127	< 0.0001	0.0137
S04-01062	19.0	Sep-04	0.00238	< 0.001	8.81	< 0.006	0.0043	0.00414	0.093	< 0.004	7.88		0.00022	515	1.56	4.98	59.3	0.0644	11.3	7.46	7.88	< 0.001	< 0.0003	0.0144
S03-01680	27.0	Oct-03	0.00919	< 0.002	8.51	< 0.007	0.00416	0.0136	0.202	0.0083	6.82		0.00035			4.44	57	0.061	11.7		9.71	0.00218	< 0.0001	0.013
S04-00211	27.0	Feb-04	0.00656	< 0.0008	8.84	< 0.006	0.00436	0.0026	0.226	< 0.004	7.63	7.78	0.00027	488		4.93	60.8	0.0698	11		9.41	0.00119	< 0.0001	0.0127
S04-00530	27.0	May-04	0.00492	< 0.0009	8.81	< 0.03	0.00386	0.0027	0.162	< 0.004	7.53		0.000417	484		6.01	59.1	0.0701	10.9	11.8	9.3	0.00242	0.00039	0.0136
S04-01063	27.0	Sep-04	0.00311	< 0.001	8.03	< 0.006	0.0045	0.00797	0.124	< 0.004	7.74		0.00025	488	1.71	4.67	53.2	0.0705	11	8.98	8.92	0.0014	< 0.0003	0.0142
S03-01679	35.0	Oct-03	0.0127	< 0.002	10	< 0.007	0.00445	0.0039	0.162	0.0094	6.84		0.000326			4.73	49.5	0.0395	11.5		10.4	0.00199	0.00021	0.013
S04-00209	35.0	Feb-04	0.0065	< 0.0008	8.77	< 0.006	0.00341	0.004	0.195	< 0.004	7.59		0.00018	412		4.37	47.1	0.0235	10.7		10.7	0.00124	< 0.0001	0.0115
S04-00533	35.0	May-04	0.00421	< 0.0009	9	< 0.03	0.00287	0.0008	0.168	< 0.004	7.65		0.000344	403		4.84	45.3	0.0186	10.8	13.7	10.6	0.00156	< 0.0001	0.0115
S04-01064	35.0	Sep-04	0.00319	< 0.001	9.14	< 0.006	0.0036	0.0229	0.168	< 0.004	7.8		0.00025	398	0.65	4.31	44	0.0176	11	11.5	10.1	< 0.001	< 0.0003	0.0135
S03-01677	45.0	Oct-03	0.00654	< 0.002	6.72	< 0.007	0.00316	0.0024	0.172	0.0051	7.22		0.000314			5.52	23.3	0.0302	11.5		9.49	0.00278	0.00014	0.0052
S04-00210	45.0	Feb-04	0.00374	< 0.0008	6.42	0.0111	0.00266	0.0066	0.199	< 0.004	7.52	7.39	0.00015	290		4.68	24.3	0.0257	10.7		9.42	0.00182	< 0.0001	0.0094
S04-00534	45.0	May-04	0.00242	< 0.0009	6.29	< 0.03	0.00157	0.0019	0.15	< 0.004	7.7		0.000194	280		4.33	22.7	0.0223	11.2	11.8	9.28	0.00146	< 0.0001	0.00868
S04-01065	45.0	Sep-04	0.00224	< 0.001	6.57	0.0213	0.0033	0.0235	0.155	< 0.004	7.91		0.00033	288	0.86	6.36	22.2	0.0215	11	8.33	8.93	0.0035	0.00029	0.0117
S03-01676	60.4	Oct-03	0.0107	< 0.002	10.8	< 0.007	0.00212	0.0071	0.184	0.0072	7.31		0.000246			4.94	11.5	0.0371	11.2		7.66	0.0023	< 0.0001	0.0049
S04-00205	60.4	Feb-04	0.0066	< 0.0008	7.54	0.0247	0.00312	0.0123	0.255	< 0.004	8.18		0.00021	312		4.56	23.2	0.0456	10.1		8.58	0.00172	0.00024	0.0151
S04-00532	60.4	May-04	0.00319	< 0.0009	7.38	0.031	0.00201	0.0013	0.245	< 0.004	7.59		0.000242	312		4.07	23.8	0.0461	11.2	10.8	8.4	0.00137	< 0.0001	0.00942
S04-01066	60.4	Sep-04	0.00195	< 0.001	7.61	0.0087	0.003	0.014	0.247	< 0.004	7.71		0.00023	309	0.78	4.17	22.1	0.0427	11.1	7.91	8.06	< 0.001	< 0.0003	0.0108

McAuley High School

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01685	9.5	Oct-03	0.114	< 0.01	0.1	0.0691	< 0.0007	40.5	< 0.0004			19.6	< 0.0009	0.00162	< 0.002	9.76		256.57	0.0241	155	228	4.65	< 0.0006	0.0051	17.7
S04-00220	9.5	Feb-04	0.0912	< 0.008	0.071	0.0915	< 0.0003	42.8	0.00034			17.6	< 0.0005	< 0.001	0.0028	9.1		409	0.0404	190	246	8.65	< 0.0007	0.0035	20.3
S04-00538	9.5	May-04	0.849	< 0.01	0.054	0.117	< 0.0004	34.9	< 0.0004			17.2	< 0.0006	0.003	< 0.001	7.31		161	0.706	235		5.17	0.0007	0.00422	20.5
S04-01075	9.5	Sep-04	0.013	< 0.01	0.042	0.137	< 0.0004	38.9	< 0.0004	9.83	19.20	11.8	< 0.0005	< 0.002	0.004	12.2	5.57	249	0.0155	246.24		2.65	0.00105	< 0.0005	22.2
S03-01687	14.0	Oct-03	0.399	< 0.01	< 0.08	0.0608	< 0.0007	70.4	< 0.0004			18.7	< 0.0009	0.00162	0.0024	8.37		256.66	0.26	208	259	5.75	< 0.0006	0.005	24.5
S04-00224	14.0	Feb-04	0.196	< 0.008	0.051	0.0795	< 0.0003	66.9	0.00044			17	< 0.0005	< 0.001	0.0036	7.07		387	0.0425	245		5.67	< 0.0007	0.0066	23.3
S04-00536	14.0	May-04	0.0187	< 0.01	0.067	0.0848	< 0.0004	61.1	< 0.0004			20.4	< 0.0006	0.0019	0.0017	7.11		194.9	0.00423	251		5.56	< 0.0007	0.00738	22.7
S04-01076	14.0	Sep-04	0.087	< 0.01	0.041	0.115	< 0.0004	68.6	< 0.0004	11.84	10.34	21	< 0.0005	< 0.002	0.0028	11.4	2.42	250	0.029	241.36		5.07	< 0.0008	0.00469	23.4
S03-01688	21.0	Oct-03	0.0806	< 0.01	< 0.08	0.0545	< 0.0007	64.3	< 0.0004			13.5	< 0.0009	0.00122	< 0.002	7.8		256.5	0.00923	167	231	5.42	< 0.0006	0.0048	23.7
S04-00221	21.0	Feb-04	0.0625	< 0.008	0.064	0.0687	< 0.0003	64.2	0.0003			13	< 0.0005	< 0.001	0.002	7.33		414	0.00994	230		5.48	< 0.0007	0.0073	23.8
S04-00540	21.0	May-04	0.291	< 0.01	0.045	0.0851	< 0.0004	62.1	< 0.0004			14.7	< 0.0006	0.0023	< 0.001	6.81		126	0.0675	245		5.45	< 0.0007	0.00706	23.7
S04-01077	21.0	Sep-04	< 0.01	< 0.01	0.04	0.0994	< 0.0004	57.8	< 0.0004			13	< 0.0005	< 0.002	0.0026	11.8	2.68	250	0.00648	247.46		4.03	< 0.0008	0.00308	23
S03-01686	28.0	Oct-03	0.0792	< 0.01	< 0.08	0.048	< 0.0007	80.4	< 0.0004			15	< 0.0009	0.00103	< 0.002	7.3		256.53	0.00166	118	137	3.93	< 0.0006	< 0.001	23.3
S04-00219	28.0	Feb-04	0.0257	< 0.008	0.063	0.0562	< 0.0003	81.7	0.00054			16	< 0.0005	< 0.001	0.0017	6.4		408	0.00058	151	141	5.14	< 0.0007	0.0072	24.3
S04-00541	28.0	May-04	0.0188	< 0.01	0.066	0.052	< 0.0004	78	< 0.0004			15.9	< 0.0006	0.002	< 0.001	7.1		95	0.00334	146		4.87	< 0.0007	0.00545	23.4
S04-01078	28.0	Sep-04	0.037	< 0.01	0.058	0.0675	< 0.0004	79.1	< 0.0004			14.2	< 0.0005	< 0.002	0.003	9.8	2.81	250	0.0252	140.19		4.81	0.00144	0.00287	23.5
S03-01684	36.0	Oct-03	0.0893	< 0.02	< 0.08	0.0352	< 0.0008	72.9	< 0.0005			19	< 0.0009	0.0007	< 0.002	6.7		256.62	0.00918	123	164	3.69	< 0.0006	0.0048	22.6
S04-00225	36.0	Feb-04	0.0869	< 0.008	< 0.05	0.0368	< 0.0003	70.5	0.00043			20	< 0.0005	< 0.001	0.0019	6.47		376	0.0153	119	165	3.84	< 0.0007	0.0058	22.5
S04-00535	36.0	May-04	0.054	< 0.01	< 0.03	0.0559	< 0.0004	66.2	< 0.0004			23.3	< 0.0006	< 0.001	< 0.001	6.9		188.1	0.00952	176		3.57	< 0.0007	0.00432	21.3
S04-01079	36.0	Sep-04	0.046	< 0.01	0.038	0.0491	< 0.0004	69.2	< 0.0004	8.43	4.19	23.6	< 0.0005	< 0.002	0.0032	9	1.58	249	0.0298	157.25		3.86	< 0.0008	0.00134	22
S03-01682	45.0	Oct-03	0.124	< 0.02	0.095	0.0428	< 0.0008	63.1	< 0.0005			25.3	< 0.0009	0.00104	< 0.002	7.87		257.57	0.0108	66	77	2.92	< 0.0006	0.006	18.2
S04-00222	45.0	Feb-04	0.257	< 0.008	< 0.05	0.0464	< 0.0003	60.8	0.00036			25.6	< 0.0005	< 0.001	0.0022	7.02		395	0.0447	87		3.2	< 0.0007	0.0064	18.1
S04-00539	45.0	May-04	0.317	< 0.01	< 0.03	0.0527	< 0.0004	58.1	< 0.0004			28.3	< 0.0006	0.0017	< 0.001	7		142.3	0.0973	80		3.04	< 0.0007	0.00572	17.7
S04-01080	45.0	Sep-04	0.019	< 0.01	0.029	0.0623	< 0.0004	59.4	< 0.0004	1.00	0.13	25.7	< 0.0005	< 0.002	0.0017	10	0.71	249	0.00865	71.92		2.93	< 0.0008	0.00458	17.9
S03-01683	60.1	Oct-03	0.292	< 0.02	< 0.08	0.0706	< 0.0008	40.4	< 0.0005			33.3	< 0.0009	0.001	< 0.002	8.9		257.94	0.0458	42	48	1.89	< 0.0006	< 0.001	10.5
S04-00223	60.1	Feb-04	0.142	0.0083	< 0.05	0.0925	< 0.0003	38.5	0.00041			31.9	< 0.0005	< 0.001	0.0021	7.97		382	0.0227	73	47	2	< 0.0007	0.007	10.7
S04-00537	60.1	May-04	0.0141	< 0.01	0.031	0.105	< 0.0004	35.8	< 0.0004			32.8	< 0.0006	< 0.001	< 0.001	8.05		169.9	0.00312	65		2.51	< 0.0007	0.00524	10.5
S04-01088	60.1	Sep-04	0.048	< 0.01	0.041	0.163	< 0.0004	38.4	< 0.0004			31.7	< 0.0005	< 0.002	0.0014	10.8	3.86	249	0.0111	51.2		2.18	< 0.0008	0.00228	11

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01685	9.5	Oct-03	0.00884	< 0.002	52.1	< 0.007	0.00452	0.0248	0.258	0.0083	7.57		0.00023			3.14	35.5	0.0833	11.8		6.89	0.00208	0.00023	0.0074
S04-00220	9.5	Feb-04	0.00518	< 0.0008	39.6	0.0072	0.00422	0.0033	0.199	< 0.002	7.44	8.15	0.00026	616		3.31	33.9	0.0919	10.2		6.33	0.00204	0.00022	0.0119
S04-00538	9.5	May-04	0.0108	< 0.0009	45.9	< 0.06	0.00446	0.0047	0.198	0.0048	7.26		0.000337	516		4.14	28.1	0.109	12.4	8.85	6.6	0.00275	0.00136	0.0127
S04-01075	9.5	Sep-04	0.00567	< 0.001	44.3	0.301	0.0061	0.0356	0.214	< 0.004	6.96		0.00023	619	2.59	2.22	25	0.102	12	7.23	6.85	< 0.001	0.0004	0.0129
S03-01687	14.0	Oct-03	0.014	< 0.002	20.9	< 0.007	0.00697	0.0166	0.251	0.0092	7.66		0.000328			5.88	50.1	0.0632	11.6		4.46	0.00233	0.00073	0.0119
S04-00224	14.0	Feb-04	0.00494	< 0.0008	16	0.0105	0.00538	0.0038	0.147	< 0.002	7.62		0.000272	636		5.49	45.7	0.054	10.2		4.41	0.00218	0.00012	0.0144
S04-00536	14.0	May-04	0.00276	< 0.0009	20.6	< 0.03	0.00494	0.00255	0.072	< 0.004	7.46		0.000326	619		4.8	43.7	0.0554	11.1	6.63	4.86	0.00104	< 0.0001	0.0162
S04-01076	14.0	Sep-04	0.00164	< 0.001	16.6	< 0.01	0.0072	0.00268	0.1	< 0.004	7.44		0.00026	645	1.54	5.3	46.8	0.053	11	4.94	4.31	< 0.001	0.00036	0.0185
S03-01688	21.0	Oct-03	0.012	< 0.002	28.7	< 0.007	0.00634	0.0212	0.251	0.0086	7.5		0.000296			5.05	62.9	0.058	11.5		8.63	0.00156	0.00016	0.0102
S04-00221	21.0	Feb-04	0.00596	< 0.0008	22.2	0.0079	0.00582	0.0022	0.188	< 0.002	7.64		0.00031	652		5	61	0.0527	10.7		8.51	0.00219	0.00013	0.0145
S04-00540	21.0	May-04	0.0039	< 0.0009	22.1	< 0.06	0.00554	0.0179	0.148	< 0.004	7.47		0.000341	618		5.26	55.7	0.0564	11.5	10.2	7.69	0.00159	0.00015	0.0158
S04-01077	21.0	Sep-04	0.00302	< 0.001	27.5	< 0.01	0.0074	0.00317	0.124	< 0.004	7.34		0.00021	640		4.27	47.1	0.0645	11.2	6.86	6.54	< 0.001	< 0.0003	0.0152
S03-01686	28.0	Oct-03	0.0188	< 0.002	14.1	< 0.007	0.0072	0.0064	0.19	0.0084	7.53		0.000339			4.42	150	0.0786	11.5		11.7	0.0013	< 0.0001	0.0107
S04-00219	28.0	Feb-04	0.00897	< 0.0008	12.3	< 0.006	0.00589	0.004	0.157	< 0.002	7.7		0.000307	720		4.95	150	0.0404	10.5		11.2	0.00168	< 0.0001	0.016
S04-00541	28.0	May-04	0.00531	< 0.0009	11.3	< 0.06	0.0059	0.0011	0.193	< 0.004	7.76		0.000356	683		4.7	141	0.0333	11.5	13.5	10.5	0.00159	< 0.0001	0.0166
S04-01078	28.0	Sep-04	0.00288	< 0.001	11.2	0.286	0.0063	0.0279	0.177	< 0.004	7.87		0.00033	668		4.97	134	0.0333	11.4	9.4	9.19	< 0.001	< 0.0003	0.0181
S03-01684	36.0	Oct-03	0.0123	< 0.002	22.7	0.0076	0.00484	0.0056	0.216	0.0074	7.62		0.00043			4.12	130	0.0845	11.8		7.32	0.00218	0.00013	0.0094
S04-00225	36.0	Feb-04	0.00572	< 0.0008	20.5	< 0.006	0.00527	< 0.001	0.23	< 0.002	7.7	8.19	0.0002	680		4.76	125	0.0378	10.9		6.61	0.00222	0.00013	0.0143
S04-00535	36.0	May-04	0.00348	< 0.0009	18.6	< 0.03	0.00435	0.0012	0.223	< 0.004	7.85		0.000304	696		4.64	110	0.0296	12.1	7.35	5.67	0.00153	< 0.0001	0.0174
S04-01079	36.0	Sep-04	0.0023	< 0.001	19.8	0.407	0.0065	0.0269	0.209	< 0.004	7.95		0.00023	649	4.20	4.92	109	0.0302	11.6	5.46	4.62	0.0016	0.00038	0.017
S03-01682	45.0	Oct-03	0.0156	< 0.002	20.9	< 0.007	0.00472	< 0.001	0.188	0.0091	7.57		0.000332			4.32	107	0.0603	11.2		17.7	0.00173	0.00013	0.0072
S04-00222	45.0	Feb-04	0.00875	< 0.0008	19.4	< 0.006	0.00482	0.0022	0.196	< 0.002	7.55		0.00033	628		5.04	104	0.0326	10.3		17.7	0.00209	< 0.0001	0.0152
S04-00539	45.0	May-04	0.00558	< 0.0009	19.7	< 0.06	0.00466	0.0028	0.196	< 0.004	7.71		0.000375	610		5.12	98.4	0.0284	10.8	21.4	17	0.00178	0.00015	0.0145
S04-01080	45.0	Sep-04	0.00318	< 0.001	20.6	0.044	0.006	0.0157	0.199	< 0.004	7.87		0.00019	609	0.34	4.68	98.7	0.0279	11.6	17.7	16.7	< 0.001	< 0.0003	0.0157
S03-01683	60.1	Oct-03	0.00768	< 0.002	11.1	< 0.007	0.00331	< 0.001	0.122	0.0065	7.94		0.00029			4.86	36.4	0.0395	11.1		11.3	0.00173	< 0.0001	0.0071
S04-00223	60.1	Feb-04	0.00558	< 0.0008	10.3	< 0.006	0.00314	0.0038	0.164	< 0.002	7.82		0.000228	404		4.97	36.4	0.0298	11		10.9	0.00139	< 0.0001	0.0194
S04-00537	60.1	May-04	0.00417	< 0.0009	15.2	< 0.06	0.00313	0.0041	0.207	< 0.004	7.65		0.000236	405		4.51	36.8	0.0283	11.4	13	10.1	0.00089	< 0.0001	0.0104
S04-01088	60.1	Sep-04	0.00334	< 0.001	12.8	< 0.01	0.0035	0.0121	0.184	< 0.004	7.86		0.00025	402		4.68	38	0.028	13.1	11	10.2	0.0014	< 0.0003	0.0123

Sandall Beat

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-01665	15.6	Oct-03	0.0929	< 0.02	< 0.1	0.117	< 0.0004	109	< 0.0004			75.5	< 0.001	< 0.001	< 0.002	5.5		360.9	0.0077	210	276	2.25	0.00264	0.0166	41.2
S04-00195	15.6	Feb-04	0.0415	0.022	0.066	0.115	< 0.0008	113	0.00109			75.1	< 0.0006	< 0.001	0.0026	3.86		431	0.0245	256		2.36	< 0.0007		41.4
S04-00513	15.6	May-04	0.255	< 0.01	0.051	0.112	< 0.0004	115	< 0.0004			80.3	< 0.0006	0.0016	< 0.001	3.47		44.7	0.364	273		2.26	< 0.0007	0.0172	42.2
S04-01048	15.6	Sep-04	0.013	< 0.01	0.056	0.104	< 0.0004	117	< 0.0004	2.65	4.25	75.4	< 0.0005	< 0.002	0.0045	4.2	2.62	249	0.005	300.48		2.19	< 0.0008	0.0135	41.2
S03-01689	21	Oct-03	0.0864	< 0.01	< 0.08	0.105	< 0.0007	120	< 0.0004			73.8	< 0.0009	< 0.0002	< 0.002			257.29	0.00106	199	280	1.69	< 0.0006	0.0174	45.6
S04-00194	21	Feb-04	0.0318	0.02	0.075	0.0999	< 0.0008	108	0.00101			73.9	< 0.0006	< 0.001	0.0028	3.11		430	0.00897	250		1.52	< 0.0007		43.3
S04-00510	21	May-04	0.016	< 0.01	0.034	0.105	< 0.0004	109	< 0.0004			82.6	< 0.0006	< 0.001	< 0.001	3.2		111	0.0158	262		1.64	< 0.0007	0.0167	41.7
S04-01049	21	Sep-04	< 0.01	0.022	0.044	0.106	< 0.0004	112	< 0.0004	2.90	5.43	75.6	< 0.0005	< 0.002	0.0027	3.5	0.87	249	0.00149	280.37		1.63	< 0.0008	0.0147	41.4
S03-01667	26	Oct-03	0.104	< 0.02	< 0.1	0.0839	< 0.0004	112	0.00059			73.8	< 0.001	< 0.001	< 0.002	5.2		641	0.0237	216	286	2.22	0.002	0.0134	40.4
S04-00196	26	Feb-04	0.0448	0.018	0.054	0.0753	< 0.0008	106	0.00089			74.9	< 0.0006	< 0.001	0.0028	2.96		429	0.0142	243		1.65	< 0.0007		40.7
S04-00512	26	May-04	0.0437	< 0.01	0.063	0.0826	< 0.0004	110	< 0.0004			80.1	< 0.0006	0.0012	< 0.001	3		89.5	0.051	251	287	1.71	< 0.0007	0.0159	41.7
S04-01050	26	Sep-04	< 0.01	< 0.01	0.039	0.096	< 0.0004	107	< 0.0004			75	< 0.0005	< 0.002	0.0028	5.9	1.12	249	0.00272	280.37		1.68	< 0.0008	0.016	41.3
S03-01666	31	Oct-03	0.0922	< 0.02	< 0.1	0.0605	0.00047	110	< 0.0004			39.5	< 0.001	< 0.001	< 0.002	4.4		338.6	0.0063	222	302	2.34	0.00243	0.0124	34.2
S04-00192	31	Feb-04	0.0284	< 0.01	< 0.05	0.0513	< 0.0008	106	0.00116			30.5	< 0.0006	< 0.001	0.0048			428	0.00814	260		2	< 0.0007		34.2
S04-00509	31	May-04	< 0.005	0.011	< 0.03	0.0527	< 0.0004	103	< 0.0004			30.7	< 0.0006	< 0.001	< 0.001	1.89		220.5	0.00152	273		1.77	< 0.0007	0.012	34.1
S04-01051	31	Sep-04	< 0.01	< 0.01	< 0.02	0.0481	< 0.0004	108	< 0.0004	0.34	1.32	27.5	< 0.0005	< 0.002	0.0036	1	1.93	249	0.004	273.06		1.72	< 0.0008	0.00961	34.3
S03-01664	36	Oct-03	0.0968	< 0.02	< 0.1	0.0891	0.00047	116	< 0.0004			61.4	< 0.001	< 0.001	< 0.002	1.9		159.5	0.0302	225	297	2.45	0.00257	0.013	39.7
S04-00193	36	Feb-04	0.135	0.015	0.094	0.0619	< 0.0008	97.6	0.0009			39.2	< 0.0006	< 0.001	0.0029	2.78		421	0.127	219	286	2.06	< 0.0007		32
S04-00511	36	May-04	0.0233	< 0.01	< 0.03	0.0476	< 0.0004	84.4	< 0.0004			36.4	< 0.0006	< 0.001	< 0.001	6.51		86.4	0.0395	261		1.82	< 0.0007	0.0108	27.5
S04-01052	36	Sep-04	< 0.01	< 0.01	0.025	0.0709	< 0.0004	100	< 0.0004	2.63	2.20	48.5	< 0.0005	< 0.002	0.0028	2.3	1.07	249	0.0142	285.25		1.95	< 0.0008	0.0126	34.6

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01665	15.6	Oct-03	0.686	< 0.003	34.3	0.015	0.0075	0.145	0.111	< 0.007	7.22		0.000449			7.99	131	0.106	11.2		13.3	0.00106	< 0.0001	0.0129
S04-00195	15.6	Feb-04	0.138	< 0.0008	32	0.0148	0.00774	0.121	0.144	< 0.004	7.1		0.00032	796		8.31	133	0.114	10.1		11.6	0.00234	< 0.0001	0.0245
S04-00513	15.6	May-04	0.0909	< 0.0009	30.2	< 0.03	0.00719	0.0158	0.083	< 0.004	7.19		0.000392	1053		8.88	119	0.107	11.4	15.3	12.3	0.00098	0.00035	0.0212
S04-01048	15.6	Sep-04	0.0668	< 0.001	31.8	0.152	0.0073	0.018	0.065	< 0.004	7.31		0.00036	1093	3.95	8.73	112	0.101	12.6	12.8	12.3	< 0.001	< 0.0003	0.023
S03-01689	21	Oct-03	0.493	< 0.002	25.7	< 0.007	0.00767	0.0202	0.19	0.0115	7.29		0.000421			10.1	130	0.0639	11.1		12.7	0.00222	< 0.0001	0.017
S04-00194	21	Feb-04	0.0187	< 0.0008	22	0.0154	0.00642	0.0089	0.198	< 0.004	7.2		0.00031	1035		9.8	122	0.0568	10.2		12.4	0.00186	< 0.0001	0.0239
S04-00510	21	May-04	0.0404	< 0.0009	26.7	< 0.03	0.00591	0.0028	0.127	< 0.004	7.14		0.00038	1019		9.83	114	0.0589	12.1	17.9	13.7	0.00151	< 0.0001	0.0188
S04-01049	21	Sep-04	0.0236	< 0.001	26.4	< 0.006	0.0069	0.00146	0.121	< 0.004	7.4		0.0003	1055	3.56	9.82	110	0.0566	12.4	14	13.5	0.0013	< 0.0003	0.0235
S03-01667	26	Oct-03	0.469	0.0038	46.7	< 0.007	0.0075	0.0274	0.135	< 0.007	7.18		0.000469			7.84	140	0.0997	11		12.4	0.00166	< 0.0001	0.0099
S04-00196	26	Feb-04	0.00804	< 0.0008	27.1	< 0.006	0.00721	0.0116	0.17	< 0.004	7.18		0.0003	1038		8.41	120	0.0716	10.1		12.4	0.00189	< 0.0001	0.021
S04-00512	26	May-04	0.0158	< 0.0009	26	< 0.03	0.00627	0.0014	0.116	< 0.004	7.17		0.000404	940		9.09	116	0.0613	10.5	17.5	13.1	0.00135	< 0.0001	0.02
S04-01050	26	Sep-04	0.003	< 0.001	27.5	< 0.006	0.0071	0.00142	0.118	< 0.004	7.42		0.00032	1060		9.46	112	0.0593	11.5	14	13.4	0.0019	< 0.0003	0.0209
S03-01666	31	Oct-03	0.288	< 0.003	47.7	< 0.007	0.0069	0.0216	0.162	< 0.007	7.26		0.000427			7.43	165	0.0894	11		7.84	0.00175	< 0.0001	0.01
S04-00192	31	Feb-04	0.00787	< 0.0008	33.8	0.0071	0.00645	0.0043	0.198	< 0.004	7.23		0.00024	946		7.51	175	0.0751	10.5		5.38	0.00186	< 0.0001	0.0328
S04-00509	31	May-04	0.034	< 0.0009	28.6	< 0.03	0.00621	0.0134	0.156	< 0.004	7.33		0.000368	801		7.82	169	0.056	11.6	6.45	5.38	0.00212	< 0.0001	0.0186
S04-01051	31	Sep-04	0.0843	< 0.001	28.3	< 0.006	0.0067	0.0105	0.154	< 0.004	7.56		0.00036	918	2.01	8.07	163	0.0502	11.9	5.46	5.25	0.0018	< 0.0003	0.024
S03-01664	36	Oct-03	0.299	< 0.003	32.9	< 0.007	0.0083	0.156	0.121	< 0.007	7.13		0.000455			7.68	149	0.115	11.2		10.6	0.00172	0.00021	0.0118
S04-00193	36	Feb-04	0.026	< 0.0008	27.6	0.0161	0.00728	0.0233	0.19	< 0.004	7.26	8.09	0.00025	875		7.62	115	0.0756	10.8		6.82	0.00183	0.00018	0.0268
S04-00511	36	May-04	0.0147	< 0.0009	26.1	< 0.03	0.00556	0.00335	0.14	< 0.004	7.44		0.000372	755		7.25	91.6	0.0503	12.4	6.49	5.25	0.00151	0.00017	0.0174
S04-01052	36	Sep-04	0.00361	< 0.001	28.8	< 0.006	0.007	0.003	0.11	< 0.004	7.2		0.00031	865	2.23	7.95	109	0.0718	12.9	9.2	8.81	0.0016	< 0.0003	0.0217

Regional sites

Allotments

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00883	Shallow	Jun-03	0.0682	< 0.01	0.166	0.108	< 0.0004	0.443	218	0.00068	174	< 0.0006	0.00083	0.0111	-17.58	-6.53	-43.4	2.14	38.4	-5.7	0.328	0.135	289		106	< 0.0004	0.0069	65.5

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00883	Shallow	Jun-03	0.148	< 0.002	58.8	0.0424	0.0166	0.0766	21.1	0.161	0.028	7.77	< 0.3	0.000471	1970	4.45	436	0.353	13.6	19.6	0.00145	< 0.0002	0.0345

Armthorpe PWS

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S04-01087		Sep-04	< 0.01	< 0.01	0.043	0.0797	< 0.0004		49.6	< 0.0004	6.85	1.46	37.7	< 0.0005	< 0.002	0.0028	6.76	1.03		0.0032	99.35	101	2.51	< 0.0008	0.00662	20.8

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-01087		Sep-04	0.00211	< 0.001	20.2	< 0.01	0.004	0.00048	0.019	< 0.004	7.45	7.93	0.0002	572	1.04	5.56	66.4	0.055		12.6	12.3	< 0.001	< 0.0003	0.0145

Beech Tree Nurseries

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00878	30.5	Jun-03	0.0387	< 0.01	< 0.05	0.376	< 0.0004	0.754	157	0.00273	34.3	0.0057	0.00046	< 0.002	-16.61	-7.95	-54.2	1.29	4.95	-155.2	0.13	22.5	229	2.09	0.00074	0.0048	20.6
S03-01657	30.5	Oct-03	0.102	< 0.02	< 0.1	0.327	< 0.0004		143	0.0029	34	0.0043	< 0.001	< 0.002				0.75		258.07		21.1	247	2.22	0.0025	0.0087	18.8
S04-00181	30.5	Feb-04	0.0408	0.06	< 0.05	0.325	< 0.0008		154	0.00416	35.6	0.00511	< 0.001	0.0044				0.55		100		22.6	256	2.01	< 0.0007		18.8

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00878	30.5	Jun-03	0.305	< 0.002	14.2	0.567	0.0219	0.0297	0.109	0.0452	7.01	< 0.3	0.000304	981	9.03	244	0.103	8.9	0.095	0.00494	< 0.0002	0.315
S03-01657	30.5	Oct-03	0.225	< 0.003	12.6	0.478	0.0158	0.006	0.053	0.0259	7.07		0.00032		8.02	224	0.0971	8.5	< 0.06	0.00469	< 0.0001	0.202
S04-00181	30.5	Feb-04	0.252	< 0.0008	13.2	0.505	0.0173	0.0139	0.054	0.0115		6.92	0.00022	8.52	8.41	233	0.112	10.3	< 0.07	0.00461	< 0.0001	0.234

Cantley Water Tower

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L
S03-01113	65.5	Aug-03	0.0425	< 0.04	< 0.07	0.0342	< 0.001	< 0.2	26.4	0.00082			31.5	< 0.001	0.00049	< 0.003	-19.32	-8.15	-54.3	6.95	0.93	106	0.092	0.0079	18.3	20.7	5.94
S03-01653	65.5	Oct-03	0.0584	< 0.02	< 0.1	0.0565	< 0.0004		46.9	< 0.0004			59.5	< 0.001	< 0.001	0.0069				8.65		256.41		0.047	92		6.53
S04-00182	65.5	Feb-04	0.0581	0.023	< 0.05	0.1	< 0.0008		67.7	0.00096			61	< 0.0006	< 0.001	0.0028				8.31		120		0.02	149	148	7.32
S04-00550	65.5	May-04	0.0105	< 0.01	0.061	0.0948	< 0.0004		68.3	< 0.0004			61.2	< 0.0006	0.0012	< 0.001				7.01		87		0.00287	123		7.17
S04-01083	65.5	Sep-04	< 0.01	< 0.01	0.045	0.0989	< 0.0004		69.4	< 0.0004	12.45	5.35	65	< 0.0005	< 0.002	0.0036				8.5	1.94	250		0.0123	157.25		7.48

Sample ID	Depth m	Sampling date	La mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	PO4 mg/l	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01113	65.5	Aug-03	< 0.0006	0.0087	9.87	0.0208	< 0.002	21.5	< 0.005	0.0102	< 0.0006	0.122	0.0136	5.79	6.07	0.145	0.000184	362		5.74	64.8	0.0436	11.5		4.92	< 0.0004	0.000415	0.035
S03-01653	65.5	Oct-03	< 0.0003	0.0093	14.7	0.0579	< 0.003	27.4	< 0.007	0.0116	< 0.001	0.14	0.009	6.41			0.000275			6.71	46.7	0.0405	10.7		5.47	< 0.0006	0.00026	0.0212
S04-00182	65.5	Feb-04	< 0.0007		21.2	0.00902	< 0.0008	23.4	< 0.006	0.00594	< 0.001	0.127	< 0.004		7.39		0.00029	624		6.78	72.5	0.0354	10.8		9.13	< 0.001	< 0.0001	0.0225
S04-00550	65.5	May-04	< 0.0007	0.0078	21.7	0.00567	< 0.0009	23.9	< 0.06	0.00596	0.0025	0.078	< 0.004	6.88			0.00033	650		6.71	74.8	0.0322	10.9	11.5	8.94	< 0.0008	< 0.0001	0.0197
S04-01083	65.5	Sep-04	< 0.0008	0.00653	21.7	0.00754	< 0.001	24.5	< 0.01	0.0067	< 0.0003	0.07	< 0.004	6.96			0.00026	394	0.78	6.75	69.1	0.0351	10.9	9.01	8.52	< 0.001	< 0.0003	0.0258

Crowtree Farm

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00874		Jun-03	0.127	< 0.01	0.32	0.19	< 0.0004	< 0.05	62.8	< 0.0004	14.9	< 0.0006	0.00051	< 0.002	-18.71	-7.48	-50.2	< 0.1	10.3	-294	0.322	0.473	330		28.3	< 0.0004	0.0251	31.1
S03-01656		Oct-03	0.0763	< 0.02	< 0.1	0.0993	< 0.0004		82.2	0.00082	208	< 0.001	< 0.001	< 0.002				< 0.1		259.2		0.886	254		4.92	< 0.0003	0.0261	54.8
S04-00180		Feb-04	0.0483	0.034	< 0.05	0.0642	< 0.0008		252	0.00169	204	0.00096	< 0.001	0.0092								0.987		543	5.26	< 0.0007		81.5

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00874		Jun-03	0.0326	0.016	38.7	0.0032	0.0071	< 0.0003	0.115	0.0192	8.22	< 0.3	0.000332	800	4.22	74.7	0.417	11.1	< 0.09	< 0.0008	< 0.0002	0.0106
S03-01656		Oct-03	0.0433	0.014	91.7	0.282	0.0053	< 0.001	0.068	0.0085	8.2		0.00034		1.96	108	0.966	10.2	< 0.06	0.00158	< 0.0001	0.0087
S04-00180		Feb-04	0.111	< 0.0008	69.4	0.411	0.0154	< 0.001	0.0857	< 0.004		7.74	0.00034	1721	5.71	324	2.43		0.335	0.00204	< 0.0001	0.0312

Doncaster Infirmary

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	DOC mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S04-01092	21	Sep-04	< 0.01	< 0.01	0.033	0.129	< 0.0004	54.2	< 0.0004	39.9	< 0.0005	0.0126	0.0058	2.06	0.0541	191	2.46	< 0.0008	0.00987	28.5
S04-01093	26	Sep-04	< 0.01	0.013	0.031	0.108	< 0.0004	96.8	< 0.0004	55.2	< 0.0005	< 0.002	0.0051	2	0.00203	242	2.63	< 0.0008	0.0102	38.9
S04-01094	30	Sep-04	< 0.01	< 0.01	0.029	0.1	< 0.0004	101	< 0.0004	56.7	< 0.0005	< 0.002	0.0042	1.53	0.00248	247	2.65	< 0.0008	0.0109	39.9
S04-01095	34	Sep-04	< 0.01	< 0.01	0.028	0.103	< 0.0004	104	< 0.0004	60.6	< 0.0005	< 0.002	0.0033	4.14	0.00257	248	2.71	< 0.0008	0.0104	41.4
S04-01096	41	Sep-04	< 0.01	< 0.01	< 0.02	0.197	< 0.0004	62.6	< 0.0004	35	< 0.0005	< 0.002	0.0019	1.17	0.00118	199	2.11	< 0.0008	0.00923	27.2
S04-01097	46	Sep-04	< 0.01	0.014	< 0.02	0.243	< 0.0004	51.8	< 0.0004	29.8	< 0.0005	< 0.002	0.0021	3.86	0.00167	179	1.98	< 0.0008	0.00983	23.9
S04-01098	bulk sample	Sep-04	< 0.01	< 0.01	0.052	0.106	< 0.0004	99.6	< 0.0004	55.8	< 0.0005	< 0.002	0.0036	2.07	0.00454	244	2.71	< 0.0008	0.00926	39.7

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-lab	Sc mg/L	Si mg/L	SO4 mg/L	Sr mg/L	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-01092	21	Sep-04	0.0421	< 0.001	15.6	0.084	0.0083	0.13	< 0.01	< 0.004	7.87	0.00019	3.82	52.2	0.0761	8.56	6.06	< 0.001	< 0.0003	2.54
S04-01093	26	Sep-04	0.00877	< 0.001	18.3	< 0.01	0.0145	0.00316	< 0.01	< 0.004	7.95	0.00026	6.74	108	0.0879	17.1	15.9	< 0.001	< 0.0003	0.116
S04-01094	30	Sep-04	0.00116	< 0.001	18.7	< 0.01	0.0068	0.00115	< 0.01	< 0.004	7.95	0.00022	6.86	115	0.0867	17.5	16.6	< 0.001	< 0.0003	0.0516
S04-01095	34	Sep-04	0.00085	< 0.001	18.9	< 0.01	0.0072	0.00054	< 0.01	< 0.004	7.97	0.0003	6.84	123	0.0849	18.2	17.5	0.0014	< 0.0003	0.0377
S04-01096	41	Sep-04	0.00045	< 0.001	13.1	< 0.01	0.0047	0.00033	< 0.01	< 0.004	7.91	0.00024	5.99	43.9	0.0785	11.2	10.5	< 0.001	< 0.0003	0.0256
S04-01097	46	Sep-04	0.00052	< 0.001	11.8	< 0.01	0.0046	0.00045	< 0.01	< 0.004	7.96	0.0003	5.74	30.2	0.0804	9.7	8.9	< 0.001	< 0.0003	0.0202
S04-01098	bulk sample	Sep-04	0.00111	< 0.001	18.6	< 0.01	0.0085	0.00056	0.02	< 0.004	7.98	0.00026	6.87	113	0.0852	17.1	16.3	< 0.001	< 0.0003	0.0327

Doncaster Racecourse

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00875	41.1	Jun-03	0.363	< 0.01	< 0.05	0.434	< 0.0004	< 0.05	40.8	< 0.0004	16.3	< 0.0006	0.00034	0.0025	-15.68	-7.92	-51.4	6.68	2	9.1	0.307	0.634	160	1.61	< 0.0004	0.0057	14.5
S03-01660	41.1	Oct-03	0.054	< 0.02	< 0.08	0.63	< 0.0008		42.3	< 0.0005	16.8	< 0.0009	< 0.0003	< 0.002				8.28		257.68		0.139	166	1.46	< 0.0006	0.0037	14.7
S04-00174	41.1	Feb-04	0.028	0.0148	< 0.05	0.282	< 0.0008		44.7	0.00056	18.1	< 0.0006	< 0.0007	0.0036						316	0.26		180	1.57	< 0.0003	0.0062	16
S04-00547	41.1	May-04	0.0119	0.014	0.032	0.509	< 0.0004		39.5	< 0.0004	18.5	< 0.0006	< 0.001	< 0.001				2.45		260.2		0.169	163	1.49	< 0.0007	0.004	14.1
S04-01081	41.1	Sep-04	< 0.01	0.015	0.024	0.269	< 0.0004		42.5	< 0.0004	17.9	< 0.0005	< 0.002	0.003				7.57	0.65	249		0.344	158.06	1.81	< 0.0008	0.00583	15.8

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T oC	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00875	41.1	Jun-03	0.0403	< 0.002	11.2	0.024	0.0052	0.0076	0.061	0.0117	7.79	< 0.3	0.000255	379	6.17	14.3	0.0343	10.8		0.316	0.0009	< 0.0002	0.0102
S03-01660	41.1	Oct-03	0.118	0.0033	10.8	< 0.007	0.00227	< 0.001	0.031	< 0.004	7.68		0.000286		5.84	16.6	0.0327	10.8		< 0.06	0.00086	< 0.0001	0.0061
S04-00174	41.1	Feb-04	0.249	< 0.0008	10.7	0.0127	0.00377	< 0.001	0.0434	< 0.004	7.55		0.000211	365	5.54	20.7	0.0342	9.6		< 0.07	0.00082	< 0.0001	0.0181
S04-00547	41.1	May-04	0.148	< 0.0009	10.1	< 0.02	0.0034	< 0.003	< 0.02	< 0.004	7.37		0.000263	339	5.48	16.1	0.035	10.6	0.13	< 0.08	< 0.0008	< 0.0001	0.0119
S04-01081	41.1	Sep-04	0.182	< 0.001	11.4	0.025	0.0049	0.00035	< 0.01	< 0.004	7.26		0.00023	416	5.7	24.3	0.0359	11.6	1.2	0.7	< 0.001	< 0.0003	0.0198

Elmstone Farm

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00881	50	Jun-03	0.077	< 0.01	0.053	0.0242	< 0.0004	0.166	185	0.00075	49.7	< 0.0006	0.00075	< 0.002	-16.93	-7.17	-47.4	< 0.1	3.27	-94.1	0.067	0.82	538	3.22	< 0.0004	0.0289	84.6
S03-01659	50	Oct-03	0.112	< 0.02	< 0.1	0.0183	< 0.0004		186	< 0.0004	42.7	< 0.001	0.001	< 0.002				< 0.1		257.61		0.67	546	3.22	0.00259	0.031	86.1
S04-00184	50	Feb-04	0.0277	0.021	0.127	0.0245	< 0.0008		182	0.0015	48.4	< 0.0006	< 0.001	0.0038				< 0.1		136		0.666	555	3.38	< 0.0007		83.4

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00881	50	Jun-03	1.27	< 0.002	37.1	0.297	0.0115	0.0011	0.12	0.0275	6.88	< 0.3	0.000458	1463	7.49	306	0.672	10.7	< 0.09	0.00169	< 0.0002	0.0148
S03-01659	50	Oct-03	1.33	< 0.003	37.4	0.252	0.0069	< 0.001	0.055	0.0102	6.61		0.000493		6.91	345	0.693	10.3	< 0.06	0.00208	< 0.0001	0.0112
S04-00184	50	Feb-04	1.28	< 0.0008	35.2	0.247	0.00794	< 0.001	0.0885	< 0.004		7.29	0.00035	1336	7.14	320	0.68		< 0.07	0.00171	< 0.0001	0.0258

Gatewood Grange Farm

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00882	76.2	Jun-03	0.0357	< 0.01	< 0.05	0.23	< 0.0004	< 0.05	90	< 0.0004	47.3	< 0.0006	0.00037	< 0.002	-14.37	-8.05	-55.3	< 0.1	2.89	96.1	0.187	0.107	201	3.16	0.00107	0.0098	25.5
S03-01650	76.2	Oct-03	0.0772	< 0.02	< 0.1	0.291	< 0.0004		80.4	0.0005	45	< 0.001	< 0.001	< 0.002				< 0.1		258.83		0.0044	204	1.26	< 0.0003	0.0101	22.5
S04-00175	76.2	Feb-04	0.0417	0.0129	< 0.05	0.266	< 0.0008		84.7	0.00069	47.3	< 0.0006	< 0.0007	0.0032				< 0.1		277		0.00252	205	1.32	< 0.0003	0.0103	23.5
S04-00549	76.2	May-04	0.0115	< 0.01	< 0.03	0.272	< 0.0004		81.1	< 0.0004	48.9	< 0.0006	< 0.001	0.0017				0		-44.2		0.00271	191	1.31	< 0.0007	0.00985	22.7
S04-01085	76.2	Sep-04	0.013	< 0.01	< 0.02	0.191	< 0.0004		88	< 0.0004	54.1	< 0.0005	< 0.002	0.0041				0	1.92	249		0.00166	195.04	1.49	< 0.0008	0.00714	26.7

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00882	76.2	Jun-03	0.328	< 0.002	19.6	0.0046	0.0087	0.147	0.092	0.0228	7.42	< 0.3	0.000296	696	6.17	79.4	0.0415	11.5		8.94	< 0.0008	< 0.0002	0.015
S03-01650	76.2	Oct-03	0.347	< 0.003	17.9	< 0.007	0.0049	0.13	0.042	0.0095	7.83		0.000391		5.84	65.3	0.0322	9.8		8.58	0.0009	< 0.0001	0.0102
S04-00175	76.2	Feb-04	0.375	< 0.0008	18.7	< 0.006	0.00513	0.14	0.0622	< 0.004	7.75		0.00028	648	6.28	71.6	0.0333	9.1		8.88	< 0.0007	< 0.0001	0.0213
S04-00549	76.2	May-04	0.362	< 0.0009	17.8	< 0.06	0.00538	0.135	< 0.02	< 0.004	7.52		0.000411	729	5.95	67	0.0331	10.9	10.6	8.4	< 0.0008	< 0.0001	0.0174
S04-01085	76.2	Sep-04	0.391	< 0.001	22.1	< 0.01	0.0056	0.162	< 0.01	< 0.004	7.43		0.00025	793	6.08	86.5	0.0329	11.9	12.6	12	< 0.001	< 0.0003	0.0252

Lings Farm

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L
S03-00879	12	Jun-03	0.0229	< 0.01	0.056	0.0745	< 0.0004	0.092	64	< 0.0004	39.7	< 0.0006	0.00107	0.003	-15.4	-8.82	-57.9	6.48	2.56	124.6	0.163	0.0825	185	225	31.1
S03-01658	12	Oct-03	0.058	< 0.02	0.091	0.0538	< 0.0004	0.1	63.1	0.00043	45.4	< 0.001	0.00092	< 0.002				9.35		258.54	0.1	0.0055	197	194	31.6
S04-00176	12	Feb-04	0.0256	0.021	0.096	0.0615	< 0.0008		62.8	0.00112	45.3	< 0.0006	< 0.001	0.0068				8.47		216		0.0134	197		28.2

Sample ID	Depth m	Sampling date	La mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00879	12	Jun-03	< 0.0004	0.0058	58.9	0.0327	< 0.002	13.3	0.0057	0.0081	0.00085	29.8	0.116	0.0184	7.59	< 0.3	0.000352	902	6.59	101	0.214	12.5	29.6	0.00152	< 0.0002	0.0295
S03-01658	12	Oct-03	< 0.0003	0.00573	56.3	0.00182	< 0.002	12.8	0.254	0.0048	< 0.001	33.7	0.082	0.0059	7.54		0.000242		6.02	109	0.218	10.1	32.8	0.00108	< 0.0001	0.0099
S04-00176	12	Feb-04	< 0.0007		56.9	0.0165	< 0.0008	14.9	< 0.006	0.00514	< 0.001		0.0975	< 0.004	7.78		0.00023	855	6.09	107	0.216	10.3	30.7	< 0.001	< 0.0001	0.0215

Misson Quarry

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S03-00877	76.2	Jun-03	0.0252	< 0.01	0.054	0.345	< 0.0004	0.553	128	0.00044	80.3	< 0.0006	0.00039	< 0.002	-17.26	-7.03	-46.3	1.21	7.2	-26	0.075	0.0202	401	5.15	0.0009	0.0147	63.1
S03-01651	76.2	Oct-03	0.0985	< 0.02	< 0.1	0.321	< 0.0004		117	< 0.0004	83.2	< 0.001	0.0012	0.0052				1.02		258.45		0.03	393	5.09	< 0.0003	0.0158	58.7
S04-00183	76.2	Feb-04	0.0294	0.013	0.133	0.361	< 0.0008		127	0.00111	86.8	< 0.0006	< 0.001	0.0067				1.28		144		0.118	405	5.2	< 0.0007		61.6

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00877	76.2	Jun-03	0.371	< 0.002	31.8	0.0046	0.0096	0.0587	10.2	0.116	0.0249	7.27	< 0.3	0.000439	1147	5.9	116	0.0863	10.4	9.45	0.0016	< 0.0002	0.0229
S03-01651	76.2	Oct-03	0.402	0.0029	31.1	< 0.007	0.008	0.0466		0.053	0.0097	7.45		0.0004		5.44	112	0.0854	10.1	11.8	0.00188	< 0.0001	0.0213
S04-00183	76.2	Feb-04	0.407	< 0.0008	29.1	< 0.006	0.00713	0.071		0.0823	< 0.004	6.67		0.00027	1108	5.76	112	0.0944	9.8	13.1	0.00185	< 0.0001	0.0293

Nutwell Bh 2

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S04-01086		Sep-04	< 0.01	< 0.01	< 0.02	0.323	< 0.0004	54.9	< 0.0004	0.75	0.00	21.4	< 0.0005	< 0.002	0.0075	0	1.81	246	0.00275	223.69	2.65	< 0.0008	0.017	22.2

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-01086		Sep-04	0.0633	< 0.001	10.9	0.028	0.0051	0.0759	< 0.01	< 0.004	7.61	0.00027	517	4.48	27.9	0.13	17	1.02	0.57	< 0.001	< 0.0003	0.0532

Pegler Ltd

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L
S03-00876	30.5	Jun-03	0.0257	< 0.01	0.059	0.045	< 0.0004	0.094	91	< 0.0004			39.9	< 0.0006	0.00167	0.004	-14.85	-8.11	-54.5	0.78	1.33	153.6	0.159	0.0145	207		4.74
S03-01077	30.5	Jul-03	0.0843	< 0.02	< 0.1	0.0457	< 0.0004	0.058	86.8	< 0.0004			43.7	< 0.001	0.0017	0.0055	-10.6	-8.3	-56.8	0.85	1.59	46.1	0.098	0.0088	200	222	4.67
S03-01654	30.5	Oct-03	0.0778	< 0.02	< 0.1	0.0317	< 0.0004		87.4	< 0.0004			40.2	< 0.001	0.0021	0.0085				0.68		254.51		0.0229	214		4.83
S04-00178	30.5	Feb-04	0.0359	0.024	0.079	0.0341	< 0.0008		85.2	0.00121			38.3	0.00094	0.0052	0.058				0.5		254		0.288	212		5
S04-00551	30.5	May-04	0.0106	0.013	0.07	0.0239	< 0.0004		79.9	< 0.0004			40.9	< 0.0006	0.0028	0.0083				0.49		52.2		0.0189	192		4.61
S04-01082	30.5	Sep-04	< 0.01	< 0.01	0.066	0.0256	< 0.0004		72.4	< 0.0004	1.79	0.58	38.3	< 0.0005	< 0.002	0.0151				0.04	2.31	246		0.0122	196.87		4.82

Sample ID	Depth m	Sampling date	La mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00876	30.5	Jun-03	0.00089	0.0161	37.7	0.00737	< 0.002	16.9	< 0.002	0.009	0.0044	12.6	0.091	0.023	7.4	< 0.3	0.000404	801		5.93	118	0.229	12.1		11.8	0.00098	< 0.0002	0.0158
S03-01077	30.5	Jul-03	< 0.0003	0.0151	35.6	0.00685	< 0.003	16.6	< 0.006	0.0053	0.0029	12.7	0.051	0.0073	7.355	< 0.3	0.000322	872		5.62	107	0.183	16.7		13.5	0.00119	< 0.0001	0.0197
S03-01654	30.5	Oct-03	< 0.0003	0.0181	36.4	0.00449	< 0.003	16.5	< 0.007	0.0059	< 0.001		0.044	0.0076	7.51		0.000408	5.3		5.3	119	0.238	15.8		12.4	0.00114	< 0.0001	0.0297
S04-00178	30.5	Feb-04	< 0.0007		36.8	0.137	< 0.0008	17.3	< 0.006	0.0136	0.0045		0.0867	< 0.004	7.45		0.00033	738		5.29	122	0.245	17.3		10.7	0.00156	< 0.0001	0.0772
S04-00551	30.5	May-04	< 0.0007	0.0173	34.2	0.0152	< 0.0009	15.9	< 0.06	0.00915	0.0481		< 0.02	< 0.004	7.46		0.000316	796		5.08	113	0.224	17.4	13.3	10.6	0.00096	< 0.0001	0.0464
S04-01082	30.5	Sep-04	< 0.0008	0.0172	34.1	0.0486	< 0.001	16.7	< 0.01	0.0078	0.0139		< 0.01	< 0.004	7.61		0.00021	739	1.15	4.31	108	0.209	18.2	7.59	7.33	< 0.001	< 0.0003	0.0468

Rossington Bridge PWS

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	dO2 mg/L	DOC mg/L	Eh mV	Fe mg/L	HCO3-fld mg/L	K mg/L	La mg/L	Li mg/L	Mg mg/L
S04-01143		Nov-04	< 0.03	< 0.005	0.098	0.191	< 0.0008	74.1	< 0.001	1.76	0.45	73.4	< 0.0006	< 0.001	0.0324	4.17	2.42	143	< 0.04	182.2	3.8	0.00132	0.0106	32.4

Sample ID	Depth m	Sampling date	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-fld pH	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-01143		Nov-04	0.0117	< 0.001	29.1	0.0303	0.0058	0.0044	< 0.04	< 0.003	7.12	< 0.0005	823	5.74	87.4	0.0982	11.2	9.55	8.87	0.0015	< 0.0002	0.0322

Sandall Common Farm

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L
S03-01114	63.4	Aug-03	0.112	< 0.04	< 0.07	0.0464	< 0.001	0.667	236	0.0006	637	< 0.001	0.00181	< 0.003	-14.5	-8.07	-56.4	6.48	1.6	113	< 0.02	0.005	216	217	10.9
S03-01652	63.4	Oct-03	0.114	< 0.02	0.13	0.0444	< 0.0004		190	0.00113	440	< 0.001	0.0011	< 0.002				8.52		258.66		0.0078	186		8.69
S04-00179	63.4	Feb-04	0.0401	0.025	0.055	0.0422	< 0.0008		168	0.00145	321	< 0.0006	< 0.001	0.0201				7.25		259		0.0285	155	160	7.84

Sample ID	Depth m	Sampling date	La mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01114	63.4	Aug-03	< 0.0006	0.0156	100	0.00086	< 0.002	169	< 0.005	0.0083	< 0.0006	17.4	0.101	0.0347	7.4	7.64	0.000521	2930	7.75	203	0.141	11.4	16.3	0.0017	< 0.00006	0.0082
S03-01652	63.4	Oct-03	< 0.0003	0.0118	79.2	0.00488	< 0.003	125	< 0.007	0.0067	< 0.001		0.089	0.0097	7.66		0.000495		7.11	207	0.099	10.4	21.2	0.00196	< 0.0001	0.0166
S04-00179	63.4	Feb-04	< 0.0007		69	0.0131	< 0.0008	93.5	< 0.006	0.00802	0.0014		0.12	< 0.004		7.68	0.00028	1749	6.78	197	0.0891	10.7	23	0.0017	< 0.0001	0.0369

Warning Tongue Borehole

Sample ID	Depth m	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/l	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC_12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	dO2 mg/L	DOC mg/L	Eh mV	F mg/l	Fe mg/L	HCO3-fld mg/L	HCO3-lab mg/L	K mg/L
S03-00880	63.4	Jun-03	0.0207	< 0.01	< 0.05	0.0398	< 0.0004	< 0.05	19.1	< 0.0004			10.6	< 0.0006	0.00054	0.0055	-17.06	-7.79	-53.6	5.59	2.9	118	0.092	0.038	29.6	34	4.57
S03-01655	63.4	Oct-03	0.0939	< 0.02	< 0.1	0.07	< 0.0004		76.1	< 0.0004			49.1	< 0.001	0.0015	0.0044				8.67		259.3		0.038	124		6.17
S04-00177	63.4	Feb-04	0.0415	0.016	0.098	0.0608	< 0.0008		66.3	0.0008			39.1	< 0.0006	0.0016	0.0051				8.22		186		0.0319	109	115	6.39
S04-00548	63.4	May-04	0.0096	< 0.01	0.084	0.0555	< 0.0004		74.2	< 0.0004			52.5	< 0.0006	0.0015	< 0.001				6.25		106.3		0.00157	106		6.05
S04-01084	63.4	Sep-04	< 0.01	< 0.01	0.073	0.0571	< 0.0004		74.8	< 0.0004	13.81	3.64	50.5	< 0.0005	< 0.002	0.003				8.02	1.59	250		0.00144	118.24		5.95

Sample ID	Depth m	Sampling date	La mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	PO4 mg/l	Sc mg/L	SEC uS/cm	SF6 fmol/L	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-00880	63.4	Jun-03	< 0.0004	< 0.002	8.85	0.0165	< 0.002	5.08	0.0044	0.0035	0.0015	4.1	0.177	0.0072	6.76	0.37	0.000147	198		1.99	28	0.0322	8.8		3.94	< 0.0008	< 0.0002	0.84
S03-01655	63.4	Oct-03	< 0.0003	0.0095	35.2	0.0393	< 0.003	16.4	< 0.007	0.0066	< 0.001		0.129	0.0089	8.3		0.000379			5.3	154	0.063	10.2		9.54	0.00161	< 0.0001	0.0138
S04-00177	63.4	Feb-04	< 0.0007		32.7	0.00444	< 0.0008	15.4	< 0.006	0.00458	< 0.001		0.111	< 0.004	8.29		0.00024	565		5.44	140	0.0734	9.9		8.32	0.00145	< 0.0001	0.0224
S04-00548	63.4	May-04	< 0.0007	0.00756	33.5	0.00022	< 0.0009	16	< 0.06	0.00518	0.0083		0.099	< 0.004	8.12		0.000349	796		5.3	153	0.0623	10.7	12.1	9.46	< 0.0008	< 0.0001	0.0156
S04-01084	63.4	Sep-04	< 0.0008	0.00626	33.7	0.00043	< 0.001	16.2	< 0.01	0.0052	0.00035		0.106	< 0.004	8.27		0.00016	757	1.81	5.23	148	0.0623	11	9.23	9.08	< 0.001	< 0.0003	0.0187

WasteWaters

Burnham Close Sewer

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	DOC mg/L	F mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L
S03-01075	Jul-03	0.0856	< 0.02	0.4	0.144	< 0.0004	< 0.05	57.4	< 0.0004			66.6	< 0.001	0.0015	0.0191	-19.39	-7.8	-53.2	38.6	0.103	0.0419	382	13.9	< 0.0003	0.0138
S03-01663	Oct-03	0.747	< 0.02	0.61	0.17	< 0.0004		53.4	< 0.0004			61.6	< 0.001	0.0017	0.0372						0.11	472	16.8	< 0.0003	0.0174
S04-00187	Feb-04	0.0574	< 0.01	0.32	0.148	< 0.0008		52	0.00071			109	< 0.0006	< 0.001	0.0283						0.0953	515	19.7	< 0.0007	
S04-00544	May-04	0.0635	< 0.01	0.384	0.131	< 0.0004		52.7	< 0.0004			83.7	< 0.0006	0.002	0.0266						0.0384	582	21	< 0.0007	0.00976
S04-01091	Sep-04	0.111	0.015	0.321	0.157	< 0.0004		52	< 0.0004			70	< 0.0005	< 0.002	0.0234				46		0.0733	442	19.1	< 0.0008	0.0119

Sample ID	Sampling date	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01075	Jul-03	23.2	0.0127	0.0058	79.7	24.5	0.0052	0.0053	< 0.3	7.26	< 0.007	6	7.69	0.000358	840	6.51	90.8	0.0925	18.1		< 0.05	0.00089	< 0.0001	0.0465
S03-01663	Oct-03	23.7	0.0172	< 0.003	87.5	32.6	0.0057	0.0122		9.17	< 0.007	5		0.0003		7.35	80.5	0.103			< 0.06	0.0012	< 0.0001	0.0761
S04-00187	Feb-04	23.6	0.0133	< 0.0008	109	44.4	0.00539	2.25		16.4	< 0.004		7.79	0.00025		6.63	68.4	0.0969			2.37	0.00143	< 0.0001	0.0619
S04-00544	May-04	23.9	0.0259	< 0.0009	92.7	72.1	0.00713	2.55		11.8	< 0.004		7.61	0.000304	1180	6.34	90.7	0.087		76.6	2.82	0.00113	< 0.0001	0.0592
S04-01091	Sep-04	23.4	0.0155	< 0.001	75	36.7	0.0063	0.00698		10.5	< 0.004		7.33	0.00025		6.24	60.1	0.108		34.8	< 0.2	< 0.001	< 0.0003	0.0526

Storm Drain Stoven’s Plantation

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L	La mg/L
S04-00186	Feb-04	0.0385	0.014	0.075	0.0446	< 0.0008	39.7	0.00066	119	< 0.0006	< 0.001	0.0073	0.0239	108	3.01	< 0.0007

Sample ID	Sampling date	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-lab	Sc mg/L	Si mg/L	SO4 mg/L	Sr mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-00186	Feb-04	10.1	0.0518	< 0.0008	81.8	0.244	0.00375	0.0623	0.181	< 0.004	7.61	0.00019	2.52	46.3	0.098	2.22	0.00228	< 0.0001	0.0387

Everingham Road Sewer

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/L	Ca mg/L	Cd mg/L	CFC-11 pmol/L	CFC-12 pmol/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	DOC mg/L	F mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L
S03-01076	Jul-03	0.104	< 0.02	0.49	0.169	0.00047	0.082	53.2	< 0.0004			104	< 0.001	0.0018	0.0365	-20.05	-8	-53.8	68.8	0.149	0.0375	456	20.2
S03-01661	Oct-03	0.0828	< 0.02	0.47	0.124	< 0.0004		50.1	< 0.0004			83.5	< 0.001	0.0015	0.0389						0.0263	666	25.1
S04-00189	Feb-04	0.531	0.013	0.377	0.115	< 0.0008		51.2	0.0008			77.9	< 0.0006	0.0017	0.0412						0.0845	496	19
S04-00545	May-04	0.076	< 0.01	0.459	0.119	< 0.0004		50.3	< 0.0004			86.7	< 0.0006	0.0021	0.0356						0.0298	579	21.5
S04-01089	Sep-04	0.088	< 0.01	0.333	0.113	< 0.0004		52.5	< 0.0004			79.5	< 0.0005	0.0019	0.0363				34.9		0.056	499	19
S04-01145	Nov-04	0.17	< 0.005	0.353	0.102	< 0.0008		50.9	< 0.001			76.5	0.0007	0.0016	0.0366				83.9		0.087	482	18.2
S04-01147	Nov-04	0.07	< 0.005	0.42	0.116	< 0.0008		50.1	< 0.001			74.3	< 0.0006	0.0015	0.0326				46.6		0.053	454	18
S04-01149	Nov-04	0.124	< 0.005	0.313	0.143	< 0.0008		52.6	< 0.001			94.7	< 0.0006	0.0019	0.0408				84.9		0.071	503	31
S04-01151	Nov-04	0.039	< 0.005	0.251	0.149	< 0.0008		51.2	< 0.001			85.7	< 0.0006	0.0012	0.0324				51.8		0.043	465	21.9
S04-01153	Nov-04	0.064	< 0.005	0.339	0.161	< 0.0008		52	< 0.001			83.2	< 0.0006	< 0.001	0.0278				106		0.039	523	22.6
S04-01155	Nov-04	< 0.03	< 0.005	0.175	0.163	< 0.0008		55	< 0.001			84.3	< 0.0006	< 0.001	0.0124				26.3		0.136	537	20.5
S04-00651	Nov-04	0.097	< 0.005	0.35	0.116	< 0.0008		49.3	< 0.001			77.9	< 0.0006	0.0016	0.0319				132		0.066	584	21.9

Sample ID	Sampling date	x mg/L	Li mg/L	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01076	Jul-03	< 0.0003	0.0146	24.5	0.0161	< 0.003	116	34.5	0.007	0.0078	< 0.3	10.2	0.0096	6	7.49	0.000265	1000	6.69	88.7	0.109	17.8		< 0.05	0.00118	0.00015	0.095
S03-01661	Oct-03	< 0.0003	0.0159	23.6	0.0147	0.0031	91.5	79.6	0.0055	0.0046		10.9	< 0.007	5		0.000276		5.93	84.4	0.103			< 0.06	0.0011	< 0.0001	0.0691
S04-00189	Feb-04	< 0.0007		24.1	0.0156	< 0.0008	127	32.6	0.00594	0.0016		11.6	< 0.004		7.41	0.00021		7.67	107	0.0942			< 0.07	0.00177	< 0.0001	0.117
S04-00545	May-04	< 0.0007	0.00932	23.8	0.0125	< 0.0009	109	63.9	0.00661	2.41		10.5	< 0.004		7.58	0.000325	1090	6.53	100	0.0889		67.5	2.42	0.00133	< 0.0001	0.0691
S04-01089	Sep-04	< 0.0008	0.00998	22.8	0.0183	< 0.001	114	38.5	0.0062	0.0508		9.53	< 0.004		7.37	0.00027	900	6.75	86.7	0.106		39	< 0.2	< 0.001	< 0.0003	0.0686
S04-01145	Nov-04	< 0.0008	0.0124	23.7	0.0512	< 0.001	131	36.1	0.0061	0.0123		11.6	< 0.003		7.65	< 0.0005		8.25	104	0.0942			< 0.06	< 0.001	< 0.0002	0.0796
S04-01147	Nov-04	< 0.0008	0.0112	23.3	0.0614	< 0.001	108	31.7	0.0062	0.0124		9.03	< 0.003		7.47	< 0.0005		7.23	79.6	0.0954			< 0.06	< 0.001	< 0.0002	0.0703
S04-01149	Nov-04	< 0.0008	0.0119	23.9	0.0717	< 0.001	116	45	0.0063	0.0205		9.61	< 0.003		7.17	< 0.0005		7.14	83.1	0.104			< 0.06	< 0.001	< 0.0002	0.0817
S04-01151	Nov-04	< 0.0008	0.0113	22.8	0.0386	< 0.001	88.8	47	0.0058	0.036		7.14	< 0.003		7.42	< 0.0005		6.16	70.6	0.103			< 0.06	0.0014	< 0.0002	0.0703
S04-01153	Nov-04	< 0.0008	0.0104	23.4	0.0625	< 0.001	82.3	60.4	0.0058	0.0076		9	< 0.003		7.5	< 0.0005		6.1	67.5	0.105			< 0.06	< 0.001	< 0.0002	0.0543
S04-01155	Nov-04	< 0.0008	0.0091	23.8	0.237	< 0.001	67.4	66.6	0.0074	0.134		8.56	< 0.003		8.2	< 0.0005		5.53	59.9	0.115			1.16	< 0.001	< 0.0002	0.0295
S04-00651	Nov-04	< 0.0008	0.0129	23.2	0.0347	< 0.001	103	74.2	0.0065	0.667		10.5	< 0.003		7.58	< 0.0005		6.62	86.4	0.0965			0.648	< 0.001	< 0.0002	0.0589

Howden Close Drain

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L	La mg/L
S04-00185	Feb-04	0.0397	< 0.01	0.071	0.0193	< 0.0008	20.6	0.00044	109	< 0.0006	< 0.001	0.0094	0.0346	61	2.37	< 0.0007

Sample ID	Sampling date	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	P mg/L	Pb mg/L	pH-lab	Sc mg/L	Si mg/L	SO4 mg/L	Sr mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S04-00185	Feb-04	2.55	0.0305	< 0.0008	78	0.285	0.00223	0.0685	0.231	< 0.004	7.59	< 0.0001	2.07	20.7	0.048	1.33	0.00292	< 0.0001	0.0337

Warning Tongue Lane Sewer

Sample ID	Sampling date	Al mg/L	As mg/L	B mg/L	Ba mg/L	Be mg/L	Br mg/L	Ca mg/L	Cd mg/L	Cl mg/L	Co mg/L	Cr mg/L	Cu mg/L	d13C per mille	d18O per mille	d2H per mille	DOC mg/L	F mg/L	Fe mg/L	HCO3-lab mg/L	K mg/L	La mg/L	Li mg/L
S03-01074	Jul-03	0.145	< 0.02	0.52	0.127	0.00053	0.096	51.3	0.0007	74.5	< 0.001	0.0021	0.0737	-14.3	-7.69	-51.4	106	0.155	0.0285	566	22.3	< 0.0003	0.0211
S03-01662	Oct-03	0.266	< 0.02	0.44	0.112	< 0.0004		49.3	0.0007	69.9	< 0.001	0.0024	0.057						0.0673	517	18.7	< 0.0003	0.0187
S04-00188	Feb-04	0.314	< 0.01	0.463	0.151	< 0.0008		55.6	0.00088	65.3	< 0.0006	0.0017	0.0495						0.038	442	19	< 0.0007	
S04-00546	May-04	0.0837	< 0.01	0.429	0.161	< 0.0004		51.5	< 0.0004	76.4	< 0.0006	0.002	0.0601						0.0223	484	17.6	< 0.0007	0.0123
S04-01090	Sep-04	0.094	< 0.01	0.178	0.145	< 0.0004		53.2	< 0.0004	66.7	0.0006	< 0.002	0.0626				32.2		0.0307	416	18.5	< 0.0008	0.0191
S04-01144	Nov-04	3.17	< 0.005	0.622	0.153	< 0.0008		58.1	< 0.001	69.4	< 0.0006	0.0018	0.0558				69.9		0.062	492	15.6	< 0.0008	0.0109
S04-01152	Nov-04	< 0.03	< 0.005	0.24	0.158	< 0.0008		52	< 0.001	73.4	< 0.0006	0.0011	0.0334				48.3		< 0.04	544	21.2	< 0.0008	0.011
S04-01154	Nov-04	< 0.03	< 0.005	0.102	0.14	< 0.0008		65.1	< 0.001	47.1	< 0.0006	0.0016	0.0113				21.6		< 0.04	524	62.9	< 0.0008	0.0178
S04-01156	Nov-04	0.189	< 0.005	0.454	0.126	< 0.0008		51	< 0.001	69.2	< 0.0006	0.0036	0.0615				158		< 0.04	512	21.6	< 0.0008	0.0124
S04-01146	Nov-04	0.109	< 0.005	0.445	0.144	< 0.0008		55.5	< 0.001	70.9	< 0.0006	0.0013	0.0509				92.3		< 0.04	398	18.3	< 0.0008	0.0141
S04-01148	Nov-04	0.065	< 0.005	0.439	0.134	< 0.0008		52	< 0.001	68.1	< 0.0006	0.0013	0.0358				66.6		0.051	422	16.7	< 0.0008	0.0143
S04-01150	Nov-04	0.061	< 0.005	0.139	0.132	< 0.0008		49.9	< 0.001	87.3	< 0.0006	0.0018	0.0429				48.1		0.039	464	24	< 0.0008	0.0106

Sample ID	Sampling date	Mg mg/L	Mn mg/L	Mo mg/L	Na mg/L	NH4-N mg/L	Ni mg/L	NO2-N mg/L	NO3-N mg/L	P mg/L	Pb mg/L	pH-fld	pH-lab	Sc mg/L	SEC uS/cm	Si mg/L	SO4 mg/L	Sr mg/L	T ˚C	TDN mg/L	TON mg/L	V mg/L	Y mg/L	Zn mg/L
S03-01074	Jul-03	23.7	0.00622	0.0036	128	44.4	0.0053	0.0107	< 0.3	9.04	0.0097	6	7.79	0.000369	940	8.55	94.5	0.106	19.8		< 0.05	0.00144	< 0.0001	0.0783
S03-01662	Oct-03	23.4	0.0149	< 0.003	119	40.4	0.0045	0.0072		10.4	< 0.007	5		0.000321		8.6	102	0.0988			< 0.06	0.00143	< 0.0001	0.0988
S04-00188	Feb-04	24.6	0.0133	< 0.0008	82.6	26.4	0.0058	0.0051		8.75	< 0.004		7.51	0.00024		6.34	90.3	0.113			< 0.07	0.00108	< 0.0001	0.0783
S04-00546	May-04	23.3	0.00952	< 0.0009	91.1	36.2	0.00639	0.0176		12.3	< 0.004		7.575	0.000311	702	6.48	82.1	0.0907		37.8	< 0.1	0.0011	< 0.0001	0.0782
S04-01090	Sep-04	22.9	0.0108	< 0.001	78.8	31.6	0.0067	0.849		10.3	< 0.004		7.36	0.00022	920	6.51	69	0.123		30.3	0.64	< 0.001	< 0.0003	0.077
S04-01144	Nov-04	24.7	0.0183	< 0.001	133	21.9	0.0065	0.0106		12.7	< 0.003		7.44	< 0.0005		12.6	98.6	0.126			< 0.06	0.0018	< 0.0002	0.0822
S04-01152	Nov-04	22.8	0.0096	< 0.001	66.4	68.3	0.0052	0.0058		10.7	< 0.003		8.18	< 0.0005		5.96	63.9	0.128			< 0.06	0.0017	< 0.0002	0.0518
S04-01154	Nov-04	25.6	0.014	< 0.001	47.4	61.3	0.0058	0.56		7.18	< 0.003		8.23	< 0.0005		5.79	76.5	0.173			0.725	0.0022	< 0.0002	0.0315
S04-01156	Nov-04	24.2	0.0125	< 0.001	111	53.9	0.0051	0.0112		19.6	< 0.003		7.37	< 0.0005		7.51	88.3	0.104			< 0.06	0.0019	< 0.0002	0.0901
S04-01146	Nov-04	24.1	0.0101	< 0.001	88.8	30.9	0.0061	0.281		7.06	< 0.003		7.71	< 0.0005		6.9	83.8	0.125			0.348	0.0015	< 0.0002	0.0687
S04-01148	Nov-04	22.4	0.0156	< 0.001	63.4	31.3	0.0056	0.0104		8.46	< 0.003		7.4	< 0.0005		5.92	64	0.119			< 0.06	0.0021	< 0.0002	0.0788
S04-01150	Nov-04	24	0.0129	< 0.001	98.9	44	0.0049	0.389		15.4	< 0.003		7.55	< 0.0005		8.47	78.3	0.115			0.416	0.0018	< 0.0002	0.0534