Assessing the impact of modern recharge on a sandstone aquifer beneath a suburb of Doncaster, UK

Brian L. Morris · W. George Darling · Aidan A. Cronin · Joerg Rueedi · Emily J. Whitehead · Daren C. Gooddy

Abstract 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, detectable concentrations of CFCs and SF_6 have been found throughout the upper 50 m of the saturated aquifer beneath a suburb of Doncaster, UK, 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 some modern recharge is travelling via fracture systems in addition to that moving down 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.

Résumé Une des principales problématiques de la qualité de l'eau en zone urbaine dont le sous-sol est occupé par un aquifère productif est l'impact de la recharge récente. Utilisant différentes chroniques dont un piezomètre multi niveaux, on a trouvé, dans les 50 m supérieurs de la zone saturée de l'aquifère situé sous le métro de Doncaster, UK, des concentrations détectables de CFC et de SF_6 qui indiquent qu'une recharge moderne (inf. à 50 ans) a pénétré au moins à cette profondeur. Cette profondeur est aussi validée par la détection de Clostridia sulfito réductives et de Streptocoques fécaux. L'aquifère supérieur est un grès peu cimenté, les marqueurs du temps de séjour suggèrent pourtant qu'une recharge récente voyage à travers un système de fracture en plus de la recharge qui se déplace par effet piston. Toutefois, du point de vue de la qualité de l'eau de l'aquifère situé sous le métro, l'impact global de 80 années d'urbanisation constante est en général limité. Ceci est attribué à une combinaison de facteurs dont l'utilisation précédente de la terre, la dilution par la recharge directe de la pluie à travers les espaces verts notamment les jardins, et un stockage important dans l'aquifère superficiel friable.

Resumen Un problema mayor de calidad de agua, en áreas urbanas subyacidas por un acuífero productivo, es el impacto de la recarga moderna. Usando una variedad de fuentes de muestreo, incluyendo pozos multi-nivel, se han encontrado concentraciones perceptibles de CFCs y SF_6 a lo largo de los 50 m superiores del acuífero saturado, bajo un suburbio de Doncaster, Reino Unido, indicando que la recarga moderna (<50 años) ha penetrado por lo menos hasta esta profundidad. Un apoyo adicional para esta penetración profunda es proporcionado por el descubrimiento de clostridia reductora de sulfitos y estreptococo fecal. A pesar que el acuífero superior es una arenisca ligeramente cementada, los indicadores de tiempo de residencia sugieren que alguna recarga moderna esta produciéndose a través de los sistemas de fractura, además de la que se produce por flujo de pistón simple. Sin embargo, el impacto global de 80 años de urbanizacion contínua, sobre la calidad de agua del acuífero ubicado bajo este suburbio ha sido limitado en general. Esto se atribuye a una combinación de factores que incluyen el uso anterior de la tierra, la dilución por la recarga directa de lluvia a través de las áreas verdes incluyendo los jardines, y localmente un almacenamiento alto en el acuífero superior friable.

Keywords Urban groundwater · CFCs and SF_6 · Microbial indicators · Permo-Triassic sandstone · Groundwater age

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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

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Fig. 1 Map of England and Wales (in central and southern UK), showing location of the cities of Birmingham, Nottingham and Doncaster on Permo-Triassic sandstone aquifer. The study area is the Doncaster suburb of Bessacarr-Cantley, which is shown in the expanded section, along with other urbanised land on the aquifer's outcrop (*darker grey areas*), motorways (M18 and A1(M)) and rivers Don and Torne

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 Doncaster (town population c. 200,000), in the South Yorkshire area of the UK, 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 provide information on 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.

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).

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 11 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).

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, County Durham (on the north east coast of England). The sandstones in the vicinity of Doncaster have an outcrop width of about 16 km and dip gently to the ENE at about 1.5°, being underlain by low permeability Permian marls 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,

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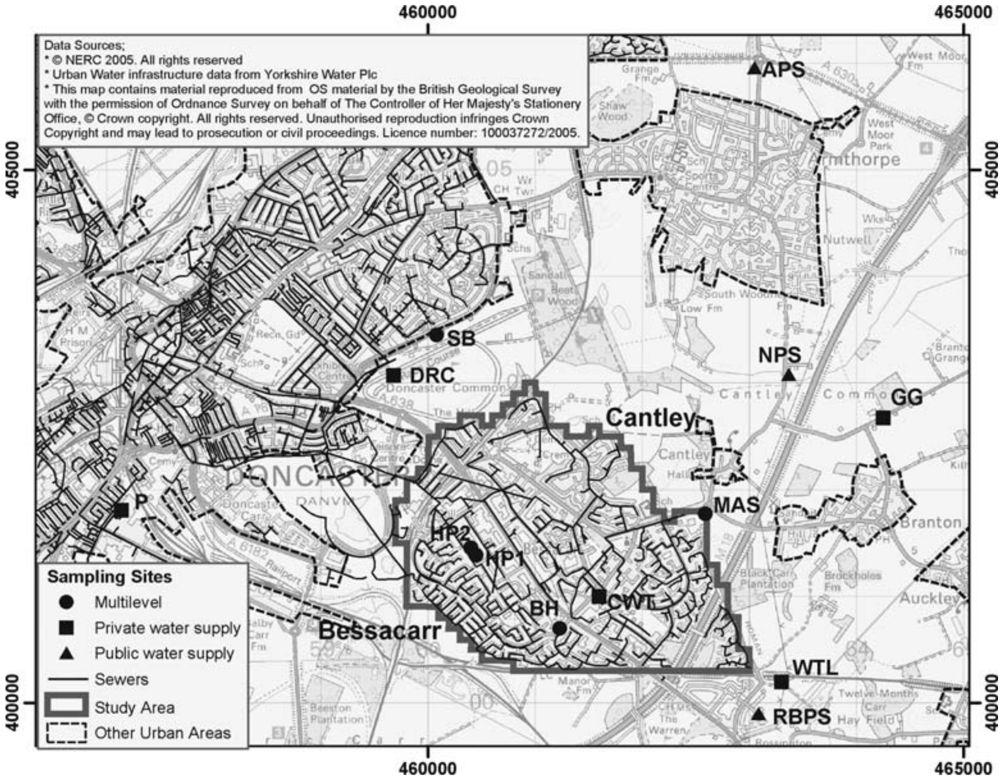

Fig. 2 Urban water infrastructure of Bessacarr-Cantley study area and sampling locations (these are named in Table 2)

flow patterns and solute/contaminant transport (Smedley and Brewerton 1997).

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/day with a median of 207 m^2/day (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 in a 59 m deep borehole (CWT in Fig. 2) in Bessacarr gave transmissivities of 76 m^2/day for the upper open-hole zone, 27–41 m below ground level (mbgl) and 92 m^2/day 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 upper 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

Table 1 Pipe infrastructure key statistics for Bessacarr-Cantley

Pipe network type	No. of pipe assets	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	

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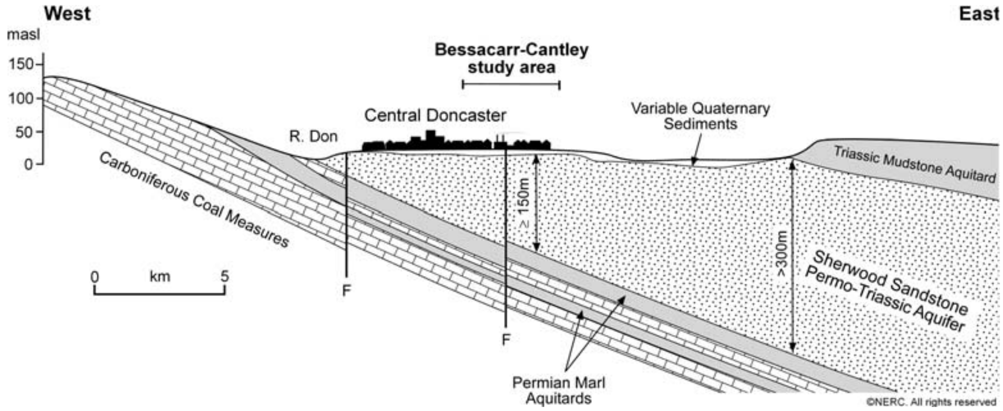

Fig. 3 Sketch section across Sherwood Sandstone aquifer in the vicinity of Doncaster. The aquifer overlies marls which divide it from the Permian Magnesian Limestone aquifer. F: fault

Sandstone is well-cemented, confirmed that such features exert significant control over the vertical flow component.

Nevertheless, 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 largely uncemented (Rueedi and 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 E, N and S 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 cm, 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).

Environmental indicators

Any study of the penetration rate of modern recharge in an urban setting demands the most sensitive possible indicators of anthropogenic activity. The atmospheric trace gases trichlorofluoromethane CCl_3F (CFC-11), dichlorodifluoromethane CCl_2F_2 (CFC-12) and SF_6 (sulphur hexafluoride) are increasingly being used as tracers of residence time (Plummer and Busenberg 1999). Large-scale production of CFC-12 began in the early 1940s, followed in the 1950s by CFC-11. These gases were used for refrigeration and air-conditioning, but inevitably leaked into the

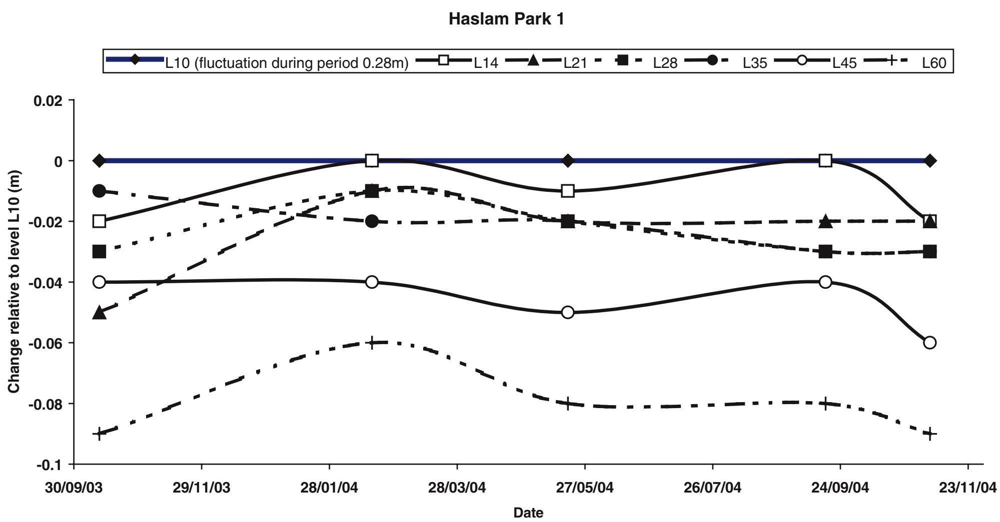

Fig. 4 Piezometric variation in HP2 multilevel at sampling point (L) at depth 10, 14 mbgl etc. For comparative purposes the piezometric level in L10 (which fluctuated 0.28 m over period) has been normalised. Note synchronicity of variation at all ports down to 60 mbgl and small magnitude of relative head difference

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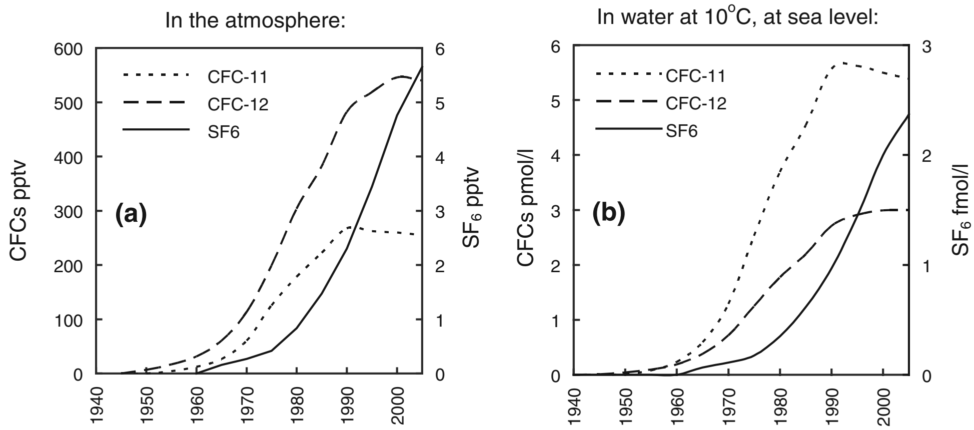

Fig. 5 (**a**) Average atmospheric mixing ratios over time of the CFCs and SF_6 in pptv (parts per trillion by volume), and (**b**) corresponding groundwater concentrations in picomoles/litre (pmol/l) and femtomoles/litre (fmol/l) respectively at a recharge temperature of 10°C. Mixing ratio data from NOAA-ESRL, 2005

environment, with atmospheric concentrations rising until the 1990s, when production was cut back to protect the ozone layer, (Fig. 5a). SF_6, another industry-derived gas, has been detectable in the atmosphere since the early 1960s and is still rising steadily in concentration (Fig. 5a). Unlike tritium, all three of these trace gases are well-mixed in the atmosphere and their input functions are better characterised. When their atmospheric mixing ratios are converted into dissolved concentrations (Fig. 5b), using Henry's Law and known solubilities (Plummer and Busenberg 1999), these can be compared with concentrations measured in groundwater samples and, assuming no contamination or degradation, the year of recharge can be inferred.

The typical routine detection limit for dissolved organics is 1 μg/l. Chlorofluorocarbons (CFCs) at datable concentrations are several orders of magnitude below this value, making them extremely sensitive indicators of post-1950 recharge. However, this also means that in an urban setting the dating of waters can prove difficult or impossible because input concentrations can be supplemented by local sources of CFCs, possibly venting to the atmosphere (Ho et al. 1998; Santella et al. 2003) or, probably more likely, contaminating the subsurface. Sewage, landfills and industry have all been implicated (Busenberg and Plummer 1992; Höhener et al. 2002; Morris et al. 2005). SF_6 is typically present in groundwaters at concentrations two orders of magnitude below those of the CFCs and can therefore only be measured at lower sensitivity. However, groundwaters are much less susceptible to SF_6 contamination (Darling et al. 2005; Morris et al. 2005).

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 (Table 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 (Fig. 2, sampling site names are given in Table 2).

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 and 60 mbgl (Rueedi and Cronin 2003) (see Fig. 6). 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 variously 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.

Sampling and analysis

CFC, SF_6 and other hydrochemistry

CFC and SF_6 sampling was conducted in autumn 2004. Samples for CFC analysis were collected by the displacement method of Oster (1994), which involves filling a glass bottle and metal can combination under water to avoid atmospheric contact. The glass bottle (100 ml) was flushed with approximately 5 l of sample water before being stoppered and sealed within the 1-l can. Samples for SF_6

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Table 2 Sampling site description including land use around wellhead[a]

Site ref	Site type	Site name	Total depth (mbgl)	Depth to screen top or sampling port (mbgl)	Land use at wellhead
HP1	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
HP2	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
BH	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
SB	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
MAS	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
DRC	Private water supply b/h	Doncaster Racecourse	41.1	NA	Grass parkland
P	Private water supply well	Pegler Ltd	30.5	5.2	Brass foundry and factory in mixed industrial and 19th/20th century inner city
CWT	Observation b/h	Cantley Water Tower	58.9	27.2	Grassed property enclosed by detached/semi-det./terraced post-1950 housing
WTL	Observation b/h	Warning Tongue Lane	63.4	18.3	Paddock on rural outer edge of post-1980 housing
GG	Private water supply b/h	Gatewood Grange	76.2	NA	Rural property
NPS	Public water supply b/h	Nutwell PS	152.4	33.0	Periurban multi-borehole complex, rural in immediate vicinity
APS	Public water supply b/h	Armthorpe PS	167.6	30.5	Periurban multi-borehole complex, rural in immediate vicinity
RBPS	Public water supply b/h	Rossington Bridge PS	147.0	28.8	Periurban multi-borehole complex, large suburban properties in immediate vicinity

b/h: borehole. PS: Pumping station

[a]The site locations are shown in Fig. 2

were similarly collected under water, but in 1-l glass bottles with conically-lined screw caps, according to the method of Busenberg and Plummer (2000). Field measurements of dissolved oxygen (DO_2), redox potential (Eh), and temperature were measured during sample collection using a flow-through cell connected directly to the wellhead sample tap.

Samples for other hydrochemical analyses 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 the British Geological Survey, Wallingford except where stated. CFCs and SF_6 were analysed by gas chromatography after pre-concentration by cryogenic methods (Bullister and Weiss 1988). Cations, P and SO_4 as S were determined on acidified sample aliquots by inductively-coupled plasma optical emission spectrometry. On the unacidified aliquots, nitrogen species and chloride were measured by automated colorimetry. Dissolved organic carbon (DOC) was measured on c. 2 ml of 0.45 μm silver membrane-filtered sample. Samples were acidified to approximately pH 3 and sparged with N gas to remove inorganic carbon prior to thermal oxidation and infra-red detection of the evolved carbon dioxide.

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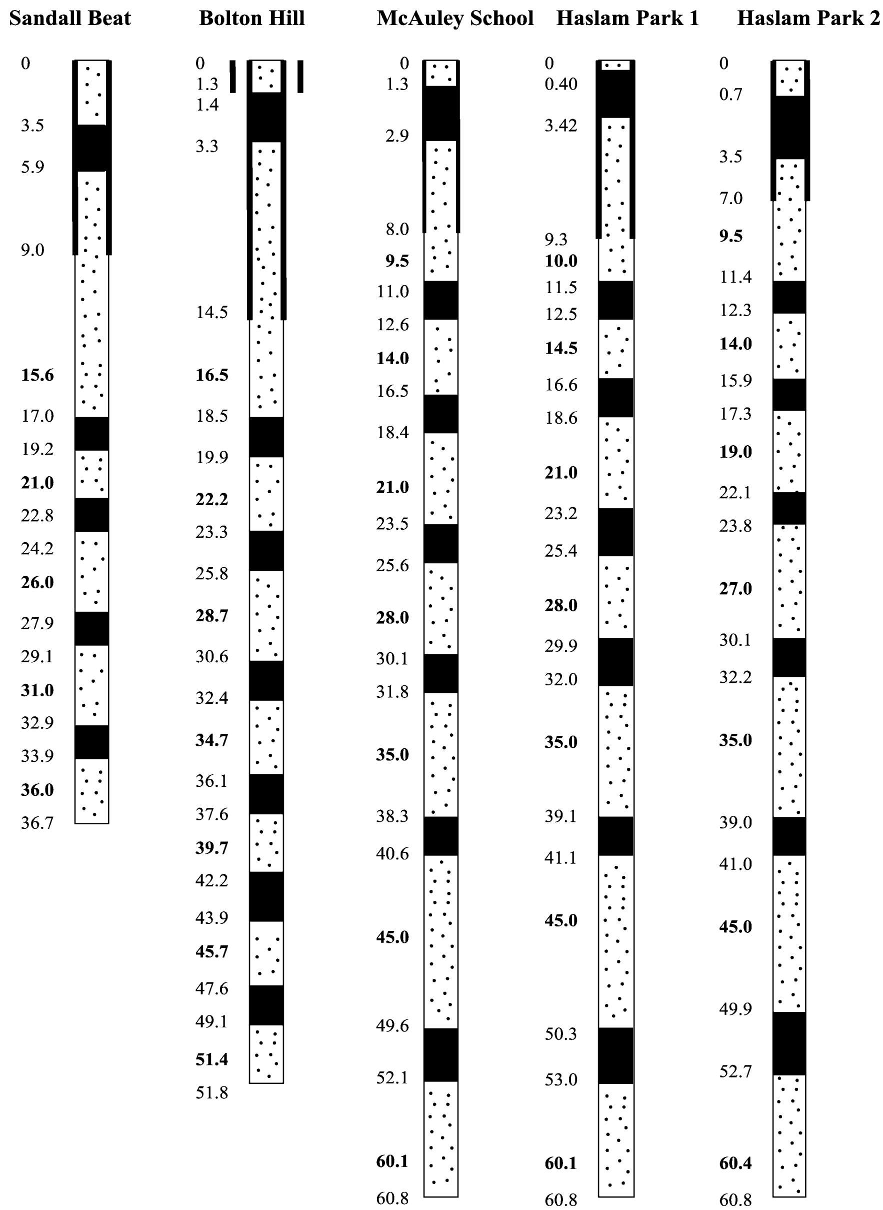

Fig. 6 Design of five multilevel research boreholes, Bessacarr-Cantley; *thick lines* indicate steel casing; *dark areas* show the location of bentonite seals; numbers in *bold* indicate the depths of the centre of the 0.3 m long sampling ports (from Rueedi and Cronin 2003)

Microbiological

Faecal coliforms, total coliforms, faecal streptococci (FS), sulphite reducing clostridia (SRC), 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 and sulphite-reducing clostridia were isolated from 100 ml sample volumes using membrane filtration and selectively enumerated by culture on membrane lauryl sulphate broth (for TTC), Slanetz and Bartley agar (for FS) and perfringens agar (for 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.

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Table 3 CFC and SF_6 concentrations in Sherwood Sandstone aquifer east of Doncaster

Sample location	Type	Depth[a] (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
Doncaster 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 27–41 m	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

M/l: multilevel, PrW: Private well, PWS: public water supply, UZ, LZ: Upper, Lower Zone CFC and SF_6 concentrations are in picomoles/litre (pmol/l)and femtomoles/litre (fmol/l), respectively. A blanket correction factor of 0.75 has been applied to measured SF_6 concentrations to account for the presence of 'excess air', based on N_2/Ar measurements made by Wilson et al. (1994) in the Sherwood Sandstone north of Nottingham

[a]Total depth of well or depth interval of multilevel

Results

CFCs and SF_6

All sites

Analyses for CFCs and SF_6 are given in Table 3. 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 3). 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.

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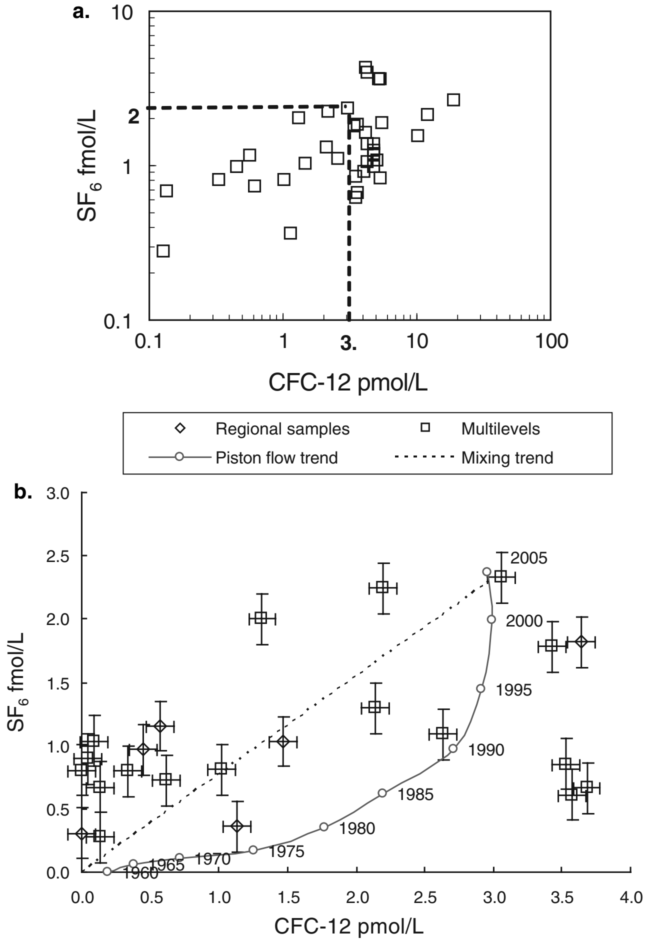

Fig. 7 **a** CFC-12 vs SF_6 concentrations—all samples. *Dashed lines* represent water in equilibrium with average 2004 atmospheric concentrations. **b** CFC-11 vs. SF_6 concentrations—subset of samples, with measurement error bars, showing low or no enrichment from local CFC sources

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 at 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. 7a). These are considered further below. The other data are plotted in the form of CFC-12 vs. SF_6 concentration (Fig. 7b). Also shown in Fig. 7b 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 NOAA-ESRL 2005), 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 localised contamination because they come from adjacent intervals in a single multilevel (SB) showing above-modern SF_6 concentrations in its upper section.

Multilevel boreholes

On the evidence of Fig. 7, 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. 8.

Four sites show 'over-modern' CFC-11 and CFC-12 fractions (i.e. >1) above 30 mbgl, indicating an element of enrichment from local sources, possibly via sewer leakage (at Sandall Beat, CFC-12 alone shows this trend; see Fig. 8e). With one exception (also Sandall Beat), 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 and the residential environment.

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.

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. 9). A basic interpretation of this pattern is that these other sites 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 (that additional sources have appeared over time) seems the more likely, so that the observed concentrations are the product of several high-CFC sources interacting.

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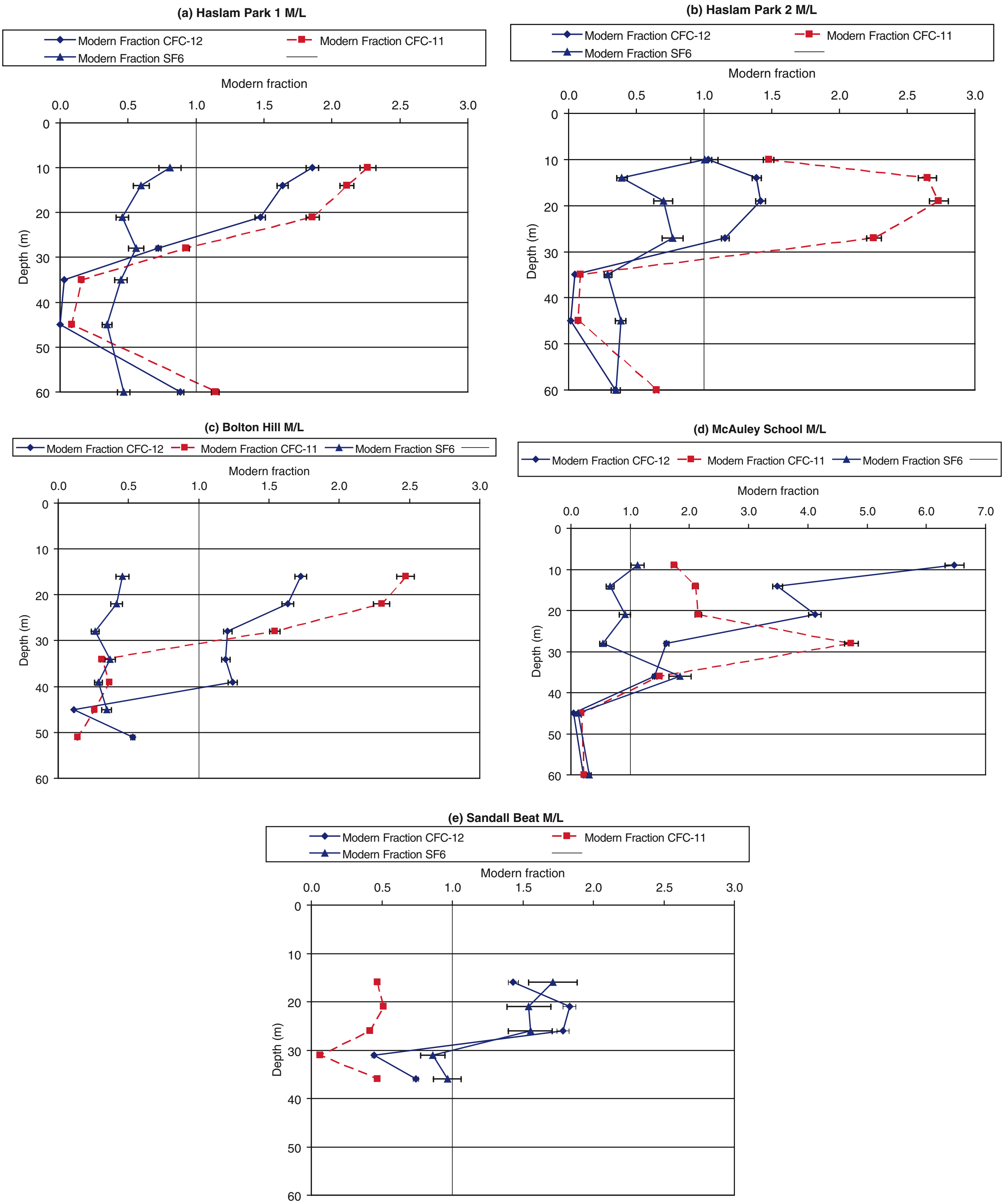

Fig. 8 CFC, SF_6 depth plots in 'modern fraction' form for all study area multilevel (M/L) boreholes; a value of >1 signifies a local source additional to average atmospheric ratios

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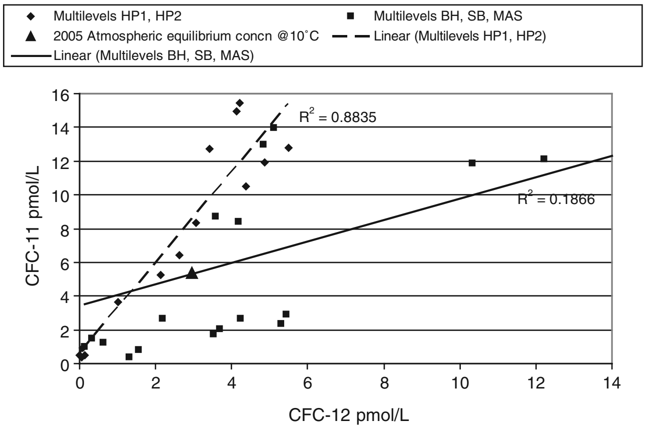

Fig. 9 Crossplot of CFC-11 vs. CFC-12 for multilevel sites. R^2 = Square of the Pearson product moment correlation coefficient

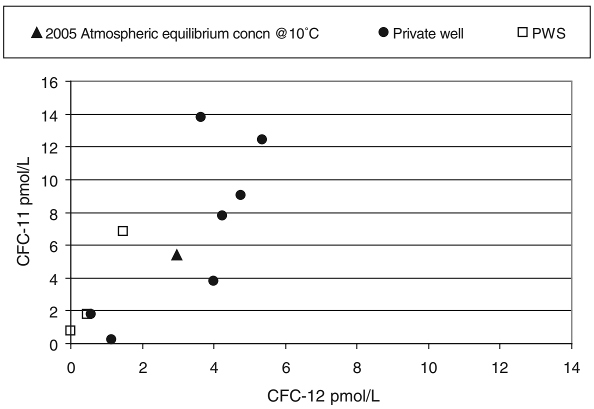

Fig. 10. Crossplot of CFC-11 vs. CFC-12 for private wells and public supply boreholes in the vicinity of Bessacarr-Cantley. PWS = public water supply

Supply boreholes

Table 4 shows the CFC and SF_6 data converted into modern fraction values and, alternatively, year of recharge assuming simple piston flow. It is clear from Table 4 and Fig. 10 that, like the upper zone of the multilevels, the private wells are frequently over-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 well/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.

Locally higher-than-average atmospheric ratios, perhaps double, may also be occurring, although the CFC ratios are not consistent between sites. While there is as yet no evidence for higher atmospheric concentrations, the possibility cannot be wholly discounted as it has been reported elsewhere (e.g. Oster et al. 1996). This would have the effect of increasing groundwater ages by up to 20 years.

Hydrochemistry

Hydrochemical analyses and field measurements representing the sampling of the HP2 multilevel on four occasions during an 11-month period are given in Table 5 together with selected profiles in Fig. 11. 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.

The concordance of most of the major ion profiles indicates consistency of results and little or no change between sampling visits. DO_2 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 6, 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.

The concentration range of these indicators in samples from the mainly rural/periurban private and public supply boreholes, is large, 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 Cl and SO_4, the additional

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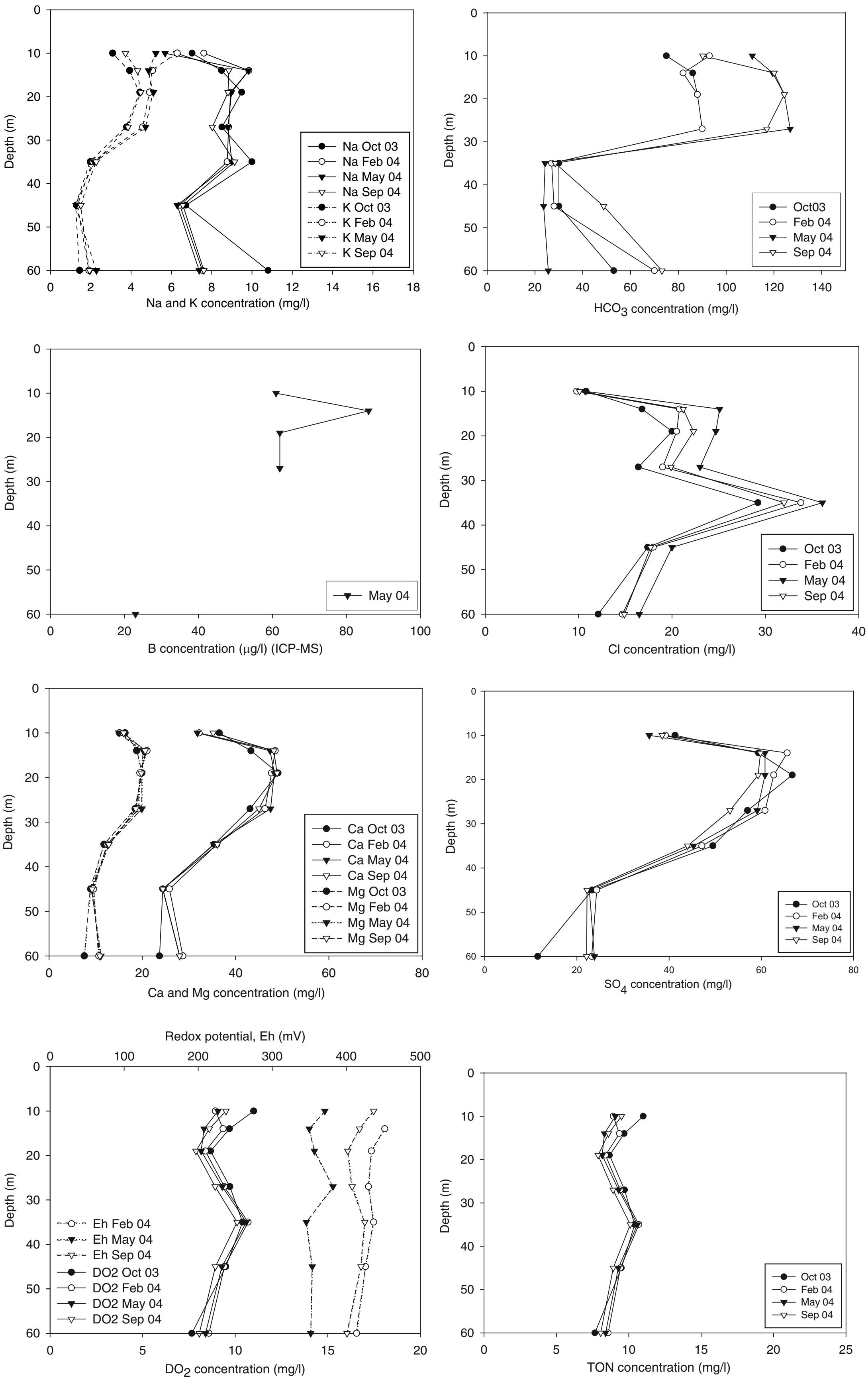

Fig. 11 Urban recharge inorganic indicator profiles for HP2, October 2003–September 2004

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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 N (present as ammonium in wastewater and principally as nitrate in the oxidised environment of the saturated aquifer) and organic C (measured as 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 or of denitrification to nitrogen gas via nitrite or nitrous oxide. 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 evaluation of the study area, several faecal contamination indicators were sampled (Cronin et al. 2005). Table 7 summarises the results.

Sulphite-reducing clostridia (SRC) has the highest number of positive detections 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). SRC spores are often found in aerobic systems; while originating in an anaerobic environment, the spores are environmentally hardy and can remain viable in the subsurface for months or longer. The SRC counts include both the non-motile bacteria and the spore stages. 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 detection 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 detections of enteric viruses and faecal indicator bacteria have been previously found in similar urban settings 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/100 ml for SRC and FS, respectively. These results indicate that although positive detections were frequent, the magnitude of these detections 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. 12 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.

A temporal evolution (1–2 orders of magnitude) was found in sewage samples from three sampling sites in the project area where maxima were observed in autumn and minima in spring. However, the temporal variations of microbial tracers found in the groundwater samples are quite small and they do not show a consistent pattern. While a much longer sampling period would be required to draw more detailed conclusions, the consistent presence of low counts of different microbial indicators to several tens of metre depth firmly suggests an element of rapid flow along fracture horizons even under the low vertical head gradients noted earlier.

Table 4 CFC and SF_6 results expressed as modern fraction and as bulk age; a modern fraction value of >1 signifies a local source additional to average atmospheric ratios

Sample location	Type	Depth[a] (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 UZ	PrW	41	1.60	1.67	0.51	>Modern	>Modern	1993
Cantley WT LZ	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	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	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

[a]Recorded depth of borehole, or in the case of Cantley WT, to the base of packer test zone. UZ, LZ: Upper, Lower Zone

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Table 5 Hydrochemical data from HP2 multilevel borehole, including major and selected minor ions and field physicochemical measurements

	Major ions								Minor ions							Field measurements				
Depth interval (m)	Ca (mg/l)	Mg (mg/l)	Na (mg/l)	K (mg/l)	HCO_3 (mg/l)	Cl (mg/l)	SO_4 (mg/l)	TON (mg/l)	Al (µg/l)	B[a] (µg/l)	B[b] (µg/l)	DOC (mg/l)	Fe (µg/l)	Mn (µg/l)	P (µg/l)	SEC (µs/cm)	Temp (°C)	DO_2 (mg/l)	Eh (mV)	pH
October 2003																				
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
February 2004																				
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 2004																				
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
October 2004																				
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

SEC, Specific electrical conductance; TON, expressed as N

[a]Boron by ICP-OES

[b]Boron by ICP-MS

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Table 6 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					
	Bessacarr-Cantley study area				Rural/periurban	
Marker species	Wastewater ($n=29$)	Mains supply ($n=30$–479)[a]	M/levels (0–30 m, n = 75)	M/levels (30–60 m, $n=65$)	Eight private wells ($n=30$)	Three public supplies ($n=30$–410)[a]
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	0.01–0.09	0.025–0.1	<0.1
NH_4–N^-	25–75	<0.02			<0.01–0.5	<0.02
NO_2–N^-	0.01–2.5	0.003–0.02	<0.001–0.03	<0.001 –.0.03	<0.001–0.14	<0.001–0.08
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

[a]Depending on parameter measured

Bdl, below detection level; Wastewater, 3 sites: Burnham Close, Everingham Rd, Warning Tongue Lane; Mains supply: Nutwell PS combined raw (blend of waters from Armthorpe, Nutwell, and the nearby Boston Park* and Thornham PS*); 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, three sites (Nutwell, Rossington Bridge and Armthorpe pumping stations, various b/hs); FS, Data collected for this project June 2003–November 2004; YW, Data from Yorkshire Water raw water quality surveillance archive January 1999–March 2004 (unpublished data, 2004); TON, Total oxidised nitrogen as nitrogen, a laboratory measurement representing sample nitrate and nitrite concentrations; nitrate NO_3-N is the main constituent because all of the groundwaters were aerobic and nitrite concentrations were generally <0.1 mg/l NO_2-N; DOC, Dissolved organic carbon. ($*$) Sampling sites not shown in Fig. 2 but within 12 km of study area boundary

Table 7 Faecal indicator sampling July 2003–November 2004 (modified from Cronin et al. 2005); results expressed as percentage positive detections

	Multilevel depth-specific intervals ($n=154$)	Regional well ($n=45$)	Sewers ($n=43$)
Field Thermotolerant coliforms TTC %[a]	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[b] %	12	0	100

[a]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*)

[b]Combination of results from two methods

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 h (Yates et al. 1985) to 72.2 h (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 8 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 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 8 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.

Discussion

Conceptual groundwater flow model

In 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

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Table 8 Faecal streptococci survivability comparison

	Upper range	Lower range
Concentration in sewer[a]	10^5	10^6
Half-life (days)	1.925[b]	3.008[c]
Time to decay to $<10^1$ (days)	27.0	51.1

[a]Sewer sampling July 2003–November 2004, six sampling campaigns

[b]From Yates et al. (1985)

[c]From Keswick et al. (1982)

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) being generally porous but well-cemented with a variety of minerals including calcite, dolomite, anhydrite and iron oxide. In these sandstones, 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). In the study area, 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 the Cantley Water Tower borehole CWT (5.25 m/day), 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

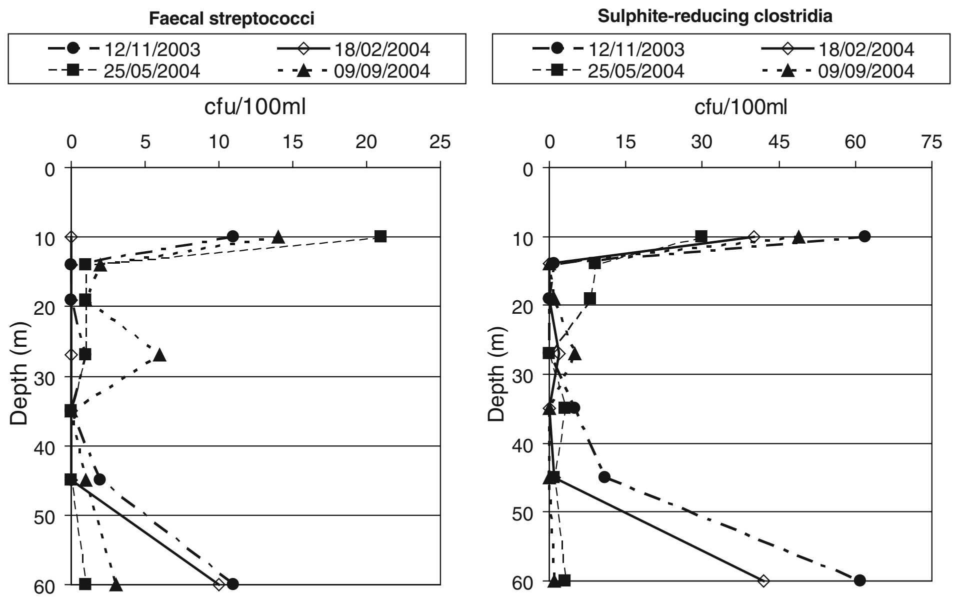

Fig. 12 Microbial indicator depth profiles in HP2 multilevel Jul 2003–September 2004; 57% of faecal streptococci and 68% of sulphite-reducing clostridia samples gave positive counts. cfu: colony-forming units

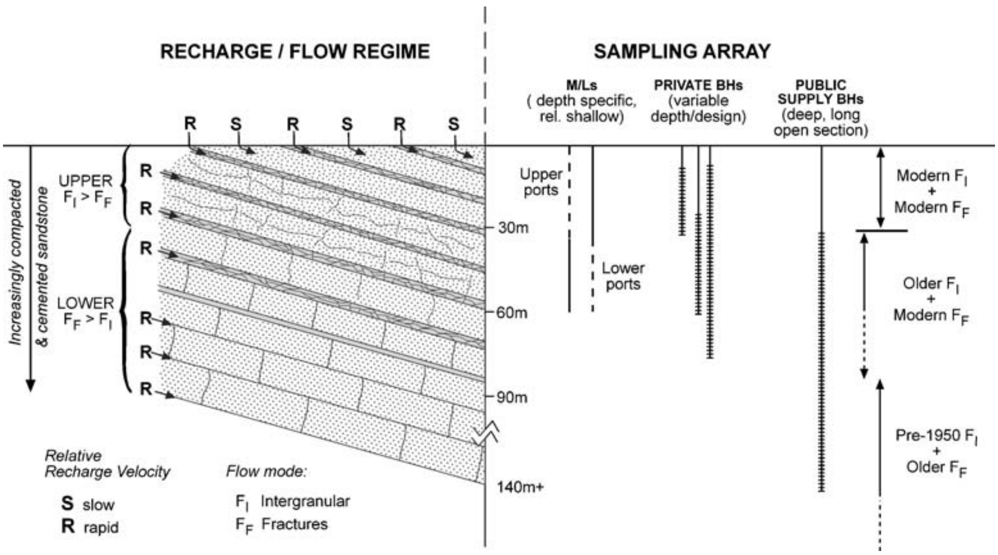

Fig. 13 Conceptual model of flow system in Sherwood Sandstone in general vicinity of Bessacarr-Cantley suburb of Doncaster. M/Ls: multilevel boreholes

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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 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/day 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/day of which only 300 m^2/day 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. Figure 13 illustrates the conceptual model, in which the upper 30 m or so is composed mainly of poorly-cemented strata interspersed with harder fractured horizons.

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 would be 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 (Upper 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 km 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 sewered residential district with suburban population densities directly replaced a rural landuse. Unlike the more central areas of many cities, there is no 'brownfield' legacy of an intervening period of industry or high-density 19th century housing to leave its contaminant footprint.

II. Light contaminant loadings; as Table 6 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 greenspace (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 125 m 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,

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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, 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, 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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