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Estimation of sewer leakage to urban groundwater using depth-specific hydrochemistry

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Abstract

The contribution of sewer leakage to urban groundwater recharge remains poorly characterised. There has been a tendency to focus on estimating leakage from pipe network characteristics rather than its impact on the receiving environment. Indeed, pipeline leakage simulation models are frequently used to analyse sewage systems and optimise maintenance efforts. Here a mass balance approach employing groundwater geochemistry is presented to estimate sewer leakage rates; this is done using depth-specific groundwater quality measurements from multilevel monitoring piezometers, specially installed in the Sherwood Sandstone aquifer underlying Doncaster (UK). The results show that leakage rates from the foul sewage system are up to 10% of flow per annum (30–40% of urban recharge) and highlight the utility of groundwater quality monitoring (in particular depth-specific sampling) as an alternative means to assess sewage ingress to urban groundwater.

Introduction

Structural defects in sewer systems are a reality across all sewered cities and are a major source of capital outlay for water companies and municipalities throughout the world. Indeed, as an example, investment in sewer reparation in England and Wales costs over £250 million per year (Davis & Burn 2001). Groundwater quality studies in Europe (e.g. Eiswirth & Hoetzl 1994; Cronin *et al.* 2003; Taylor *et al.* 2006) and the Unites States (USEPA 1989; Gruenfeld 2000) have identified exfiltration from sewer systems as a major potential source of groundwater contamination. It is important that this process is studied in order to quantify the impact of this highly pollutant stream on aquifers used for drinking water production in order to allow a comparison of different effluent management options.

Current approaches to estimate sewer leakage often focus on available information on the sewer pipe system's design, age and condition (e.g. Decker & Risse 1993) though this can have significant associated uncertainties (Wolf & Hoetzl 2007). The use of qualitative tracers (i.e. components present in effluent) can help detect leaking sewage systems and include major ions (e.g. chloride or potassium in Barrett *et al.* 1999), minor elements and trace substances (e.g. boron, X-ray contrast media or hormones in Ternes 1998; Ellis 2001; Sacher *et al.* 2001; Wolf *et al.* 2004) as well as microbiological indicators (Cronin *et al.* 2006). 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 both spatially and temporally (e.g. Cronin *et al.* 2006). As natural groundwater chemical evolution is dominated by water–rock interactions, effective identification and quantification of different recharge sources requires substances or tracers that can better distinguish anthropogenic from natural sources.

This paper describes a study to estimate sewer leakage using water quality analyses of sewer and groundwater and then, using this information, to constrain modelling work based on an urban water mass flux model named urban volume and quality (UVQ). UVQ was used to track water volumes and concentrations in urban areas and linked with a pipe leakage model (PLM), both developed by CSIRO in Australia (Mitchell *et al.* 2003; Mitchell & Diaper 2003).

Methodology

Study area

Doncaster, UK (Fig. 1) has a population of about 200 000 and is dependent on the underlying Triassic Sherwood

Fig. 1. (a) Location of the city of Doncaster within the United Kingdom. (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 the multilevel piezometers

Name	Drilled depth (mbgl)	Head level (m above OD)	Easting	Northing	Hydraulic conductivity (m/day)	No. of sampling ports
Bolton Hill	51	15.153	461 230	400 704	8.2	7
Haslam Park 1	60	11.092	460 455	401 392	1.0–1.7	7
Haslam Park 2	60	10.75	460 400	401 465	1.5–2.7	7
McAuley School	61	9.621	462 597	401 786	3.5	7

Sandstone aquifer for its water supplies. The production wells of the public water supply network are located to the east and north of the study area and they are typically to depths of 140–170 mbgl and screened below about 30–60 mbgl. Further details on the geology and hydrogeology of the study area are given in Morris *et al*. (2006).

The study consisted of water quality sampling of four multilevel monitoring wells in the suburb of Bessacarr–Cantley ($6.3\,km^2$) installed in September 2003 at sites chosen to detect contamination originating from the historical and industrial centre of Doncaster and the suburb of Bessacarr–Cantley. Depths and grid references are given in Table 1 while locations are presented in Fig. 1. Between drilling and installation short pumping tests (3–10 h) were performed to better understand the average hydraulic conductivities (Table 1).

Installation of the multilevels incorporated central riser PVC pipes with bundled smaller diameter HDPE tubing (Fig. 2a). This design facilitated real time monitoring of water levels, temperature and specific electrical conductance (SEC) via down-hole data loggers. The pipes were installed right after the pumping test to prevent cross flow

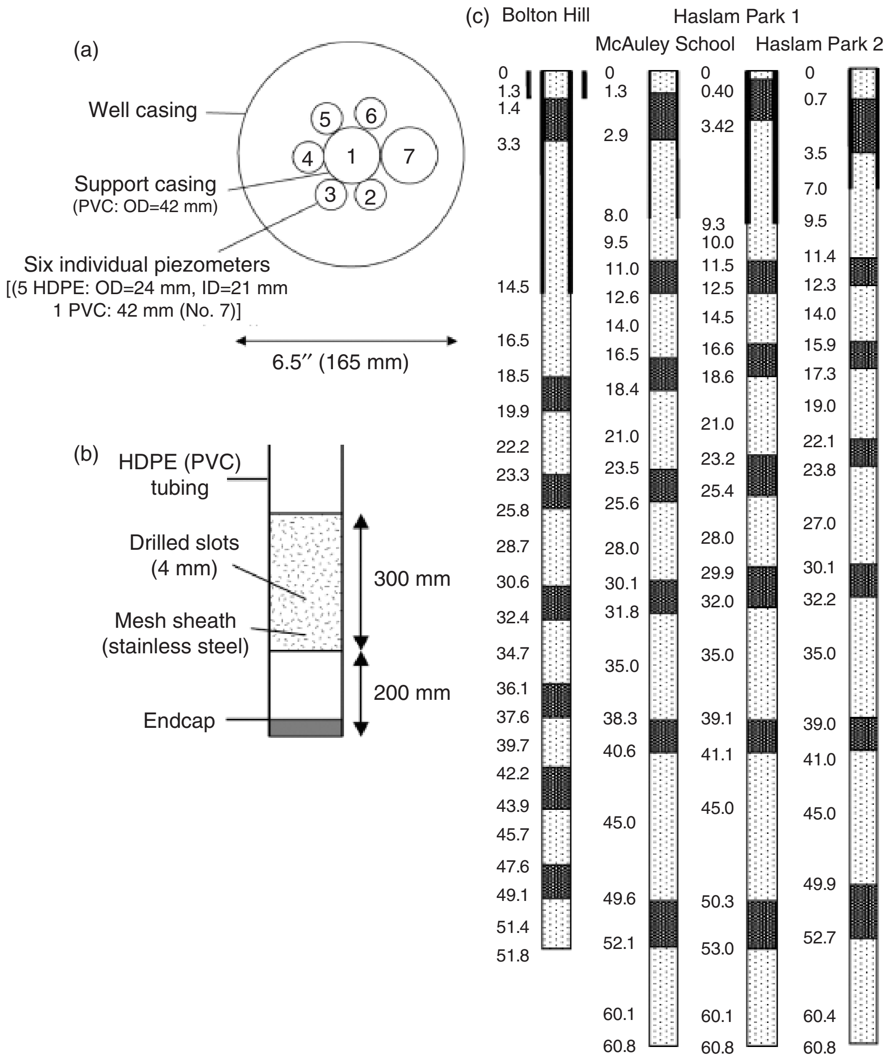

Fig. 2. (a) Arrangement of plastic pipes inside the open borehole before backfilling. (b) Detail of screen section showing lowest part of a HDPE (or PVC) pipe with intake section comprising holes drilled into the pipe and covered with a stainless-steal mesh to prevent sand from entering the pipe during sampling. (c) Depths (in metres below ground level) of the seal locations and sampling intervals of each of the four multilevels.

within the open hole. The HDPE pipes were attached to the centre PVC pipe using cable ties and caps were added at the end of each pipe to prevent water entering from the bottom. At the bottom of the centre tube centralisers were added to guarantee the pipe to stay in the centre of the open hole. The end of each pipe was capped and a 30 cm screen was constructed 20 cm above the bottom of the pipe (Fig. 2b). Screen intakes for each piezometer were hydraulically isolated using bentonite clay seals of between 1 and 3 m in thickness.

With the pipes installed in the hole, coarse quartz sand (grain size ~1 mm) and bentonite (Mikolit 300®, MGS Ltd., UK) was added in small portions through a funnel, and depths were continuously monitored to enable a tight control over the sand and clay levels. Above the uppermost level sand was added to 3 m below ground (above the standing water level) and this was topped by a 2 m bentonite seal to prevent short-circuiting contamination from the well head. Finally, sand was added to finish the hole (Fig. 2c). Hydraulic integrity of each level was tested by pumping while monitoring upper and lower level intervals to ensure no connection. Once completed, each piezometer was disinfected with sodium hypochlorite solution (50 mg/L). Complete field sampling details may be found in Morris *et al.* (2006).

Theory of estimating sewer leakage using urban water mass balance

In an unconfined aquifer situation, as in Doncaster, leakage from sewer or water mains pipes, respectively, pass through the unsaturated soil zone eventually reaching the water table. Other forms of recharge such as precipitation, stream or river leakage or on-site drainage similarly reach the saturated zone via the unsaturated zone. Therefore, urban groundwater recharge represents

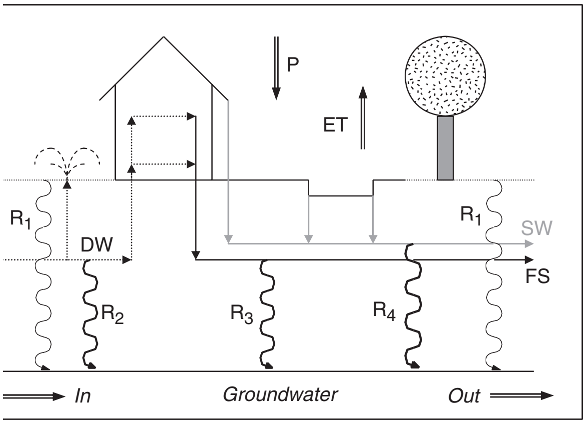

Fig. 3. Schematic of the urban water cycle. Dotted lines indicate mains water supply, full lines indicate foul water (black) and stormwater (grey) system.

a mixture of varied inputs. On a small spatial scale, these different inputs may vary significantly depending on the position of the sampling point relative to pipeline leaks and on the time of sampling (Wolf & Hoetzl 2007). However, at the suburb or city scale, the small-scale variations in volume and concentration are integrated over large areas (Foster 2001). On this scale it is, therefore, possible to use urban water balance calculations to estimate the different contributions to urban recharge.

The water cycle in a small urban area, such as a housing district with mains water supply and wastewater connections, is shown schematically in Fig. 3. The urban water balance can then be calculated assuming a balance of water inputs and outputs to the area [Eq. (1)]. The equation does not consider any possible river leakage because rainwater recharge in the study area occurs mainly through the stormwater system and not through direct surface drainage. Equation (1) basically shows all inputs to the urban area on the left hand side and the outputs on the right hand side.

$$P + DW = ET + SW + FS + GW \tag{1}$$

where P is the precipitation, DW the mains water supply, ET the evapotranspiration, SW the pluvial drainage, FS the foul sewer volume and GW the recharge to groundwater with all values given as mm/year.

Considering the entire thickness of the unconfined aquifer and that there is no leakage occurring to or from lower aquifer units, the groundwater flux (GW) can be split into five major recharge fluxes while also considering lateral flow through groundwater movement [Eq. (2)].

$$GW = R_P + R_{DW} + R_{FS} + R_{SW} + in - out. \tag{2}$$

The simplifying approach presented here is applicable for slow-moving groundwater bodies with relatively short time scales (<10 years), as applicable in the Doncaster case.

The concentration of each groundwater quality component is governed by the same mass balance [as in Eqs (1) and (2)] but is affected by various processes such as die-off and sorption (e.g. for micro-organisms), adsorption, ion exchange or dissolution/precipitation reactions, all of which can occur in the Permo-Triassic sandstone (e.g. Taylor *et al.* 2006; Rueedi *et al.* 2007). The general mass balance for groundwater concentrations C_{GW} is therefore given in Eq. (3) where it is assumed evapotranspiration losses are conservative and groundwater inflow and outflow are negligible.

$$PC_P + DWC_{DW} = ET + SWC_{SW} + FSC_{FS} + GWC_{GW} + S, \tag{3}$$

and

$$C_{GW} = \frac{R_1C_1 + R_2C_2 + R_3C_3 + R_4C_4}{GW}, \tag{4}$$

where C is the source concentration (mg/L), R_1 the natural recharge through precipitation (mm/year), R_2 the mains leakage (mm/year), R_3 the foul sewer leakage (mm/year), R_4 the pluvial drain leakage (mm/year), S the sink or source (mg/m^2).

All parameters included in the water and contaminant balance are associated with an uncertainty that can be estimated according to the general law of error propagation taking into account first order errors only [Eq. (6)] and assuming a normal distribution for all parameters.

$$\mu_Y = f(\mu(X_i)), \tag{5}$$

$$\sigma_Y^2 = \sum_j \left(\frac{\partial f}{\partial X_j}\right)^2 \sigma_j^2, \tag{6}$$

where

$$Y = f(X_i) \tag{7}$$

where X_i is the water balance component (e.g. FS, GW, R_1, etc.), Y the mixing ratio, μ the arithmetic mean, σ the standard deviation, σ_j the uncertainty of water balance component and σ_Y the uncertainty of mixing ratio.

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 uncertainties. However, provided these parameters are conservative, all should show the same mixing ratio. Rueedi *et al.* (2005) applied a solution to calculate total mixing ratios $\overline{m}$ and uncertainties $\overline{\sigma}$ using conservative hydrochemical parameters P [Eqs (8)–(10)].

$$\overline{m} = \frac{\sum_{g=1}^{n} m_g w_g}{\sum_{g=1}^{n} w_g}, \tag{8}$$

$$\overline{\sigma} = \sqrt{\frac{1}{\sum_{g=1}^{n} \sigma_g^{-2}}} = \sqrt{\frac{1}{\sum_{g=1}^{n} w_g}}, \tag{9}$$

with

$$w_g = \frac{1}{\sigma_g^2}, \tag{10}$$

where g is the groundwater parameter and n is the number of parameters considered.

Doncaster urban mass balance components

Several datasets were analysed to estimate the urban water balance of the study area. Weekly precipitation and potential evaporation data from the UK Meteorological Service MORECS system (1970–2004) are used to obtain an average annual precipitation of 692 ± 114 mm/year. The UVQ model was calibrated using these datasets and information gained from landuse maps, population statistics and satellite images (provided by Doncaster Metropolitan Borough Council) and supplemented by direct field observation (Morris *et al.* 2006). UVQ produces values for stormwater runoff from impermeable areas using two field parameters: the runoff-effective paved areas and the maximum initial loss. The effective paved areas were estimated from housing maps as well as field observations of the three different pavement types (roads, roofs and pavements). Initial loss values relied on direct measurements from other studies; Table 2 summarises the key UVQ parameter values. These estimates take into account that, even though sealed areas are fully connected to the stormwater system, contributions to recharge can still result from leaky gutters or pavements that drain towards green areas. Paved areas around houses were observed to be often poorly connected to the stormwater system and therefore assigned an effectiveness of only 40%. The UVQ model, driven by the climatic data, estimates an average stormwater volume of 105 mm/year for the period 1970–2004; uncertainty is estimated to be approximately 10% (Table 2).

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 and were used to calculate an average water supply volume of 1.22 million m^3/year for the study area. Mains leakage was estimated based on hourly night-time flow records from Yorkshire Water and corrected for domestic and commercial night-time uses such as toilet flushing, etc. Leakage rates in the control zones range from 1.1 to 5.3 L/property/h, with an average for the year 2003 of 1.9 ± 0.4 L/property/h (corresponding to an average leakage rate of 9.9% of all imported water).

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

Water balance	(mm/year)	Average domestic property statistics	
Precipitation *P*	692 ± 114[a]	Area (ha)	603[b]
Evapotranspiration *ET*	–	No. of domestic properties	8323[c]
Total imported water *DW*	203 ± 12[c]	Average occupancy	2.52[d]
Mains leakage R_2	22 ± 5[c]	Road area (ha)	92[b]
Garden irrigation I	10 ± 3[c]	Paved area (ha)	39[b]
Total foul sewage volume *FS* (*DW-R*2-I)	171 ± 7	Roof area (ha)	74[b]
Stormwater Volume *SW*	118 ± 12[e]	Effective road area (%)	95
Ratio *SW* : *FS*	0.69 ± 0.08	Effective paved area (%)	50
Total urban recharge	150–200[f]	Effective roof area (%)	95
		Maximum initial loss (mm)	2

[a] From MORECS database.
[b] From water supply statistics.
[c] From UVQ.
[d] From GIS analysis.
[e] From domestic statistics.
[f] From literature.

Gardens were watered using mains water, which can account for significant volumes of water during summer time. Therefore, the water supply records were corrected for this direct 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 7300 L/property/year (corrected for leakage), corresponding to 4.9% of total supplied water. This amount ranged from 2600 to 14 000 L/property/year from areas with denser housing and smaller gardens to values in the older part of the suburb with large gardens.

Cunningham *et al.* (2004), present a detailed analysis of the local pipe network using geographical information system (GIS) techniques to analyse the pipe asset provided by the water utility. The pipe network is categorised according to age, material type, diameter and joint type and here a selection of key statistics are highlighted (Fig. 4). It can be seen that the information on pipe ages of the sewage system is limited; the pipe asset database of

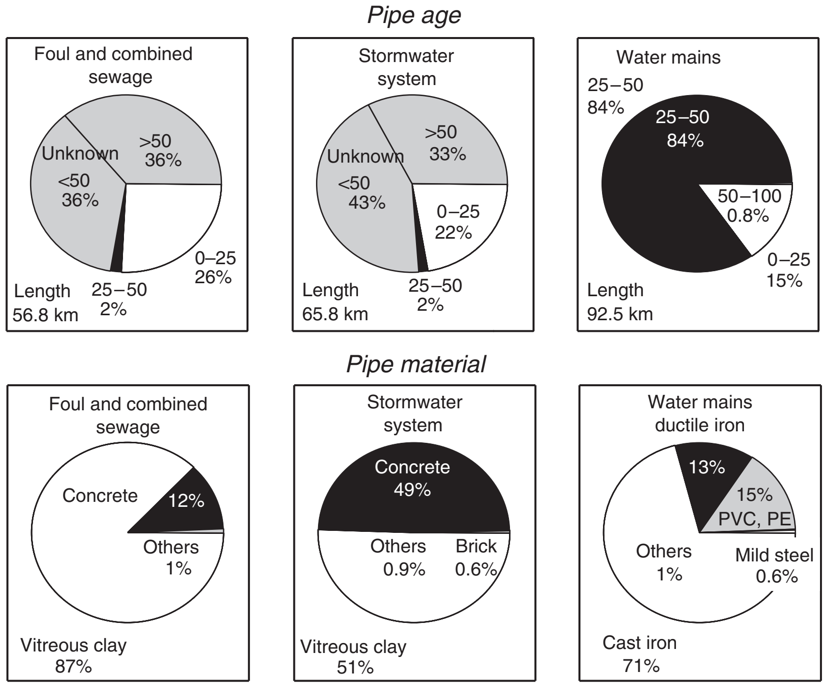

Fig. 4. The top row shows age distributions (in years) of the pipe network in Bessacarr with the total length displayed in the bottom left corner. The lower row displays the percentage distribution of pipe materials in use.

both sewerage and stormwater systems provide age information for only 28% of their entire length. The remaining 72% are of the same age as the households they drain and so it can be concluded that 36% of sewers and 33% of storm water pipes are over 50 years old. 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. Figure 4 shows that 99% of foul and pluvial drainage systems are built with vitreous clay and concrete pipes while the water mains are almost exclusively of ductile and cast iron materials.

An important unknown parameter, apart from the leakage rates from stormwater and foul sewage system, is the natural recharge from green areas and gardens. This output of the UVQ was found to be very sensitive to the choice of soil parameters. A natural recharge (under unpaved areas) of approximately 200 mm/year was estimated by Binley *et al*. (2002) near Hatfield, 12 km NE of Bessacarr–Cantley. Ragab *et al*. (1997) modelled similar values and 200 mm/year also gives an acceptable calibration of water levels against observed values for the steady-state subregional model (Morris *et al*. 2006). Yang *et al*. (1999) used a mass balance approach to estimate a value of 239 mm/year for urban recharge in Nottingham while Brown & Rushton (1993) used a value of 110 mm/year to calibrate the first regional Nottingham–Doncaster groundwater model.

Results

Doncaster field sampling

The different sampling campaigns (Table 3, also showing arithmetic means and standard deviations) show only small (seasonal) variations, as also found by Taylor *et al*. (2006) and Cronin *et al*. (2003). Natural urban recharge was taken as the average chemical composition of water unaffected by urban recharge, as given by CFC-12 (dichlorodifluoromethane or Freon 12) results; see Morris *et al*. (2006). These groundwaters feature low bicarbonate alkalinity and are strongly undersaturated with respect to calcite indicating a lack of calcite in the aquifer matrix (Rueedi *et al*. 2007). Rainwater and supplied mains water quality in the research area are given in Stuart *et al*. (2004). Raw sewage was analysed quarterly and, in addition, a further 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 undertaken during dry periods but the composition was similar to that consistently observed in shallow piezometers at the Bolton Hill site.

Hydrochemical signatures of foul sewage and stormwater differ considerably for some components (Table 4) and indicate that the most tracers with most potential to detect sewage influence are K, Na, HCO_3 and B. Cl is of limited use while Ca, Mg and SO_4 cannot be used because

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

Hydrochemical parameters (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 ± 2.16	45.9 ± 6.3	19.5 ± 1.6	39.4 ± 4.1	18.1 ± 1.3	32.5 ± 3.9	204.8 ± 26.4	7.36 ± 1.32	0.063 ± 0.012	0.233 ± 0.058
McA14	5.68 ± 0.12	19.2 ± 2.7	23.5 ± 0.9	66.1 ± 4.7	18.7 ± 1.7	46.5 ± 3.3	249.9 ± 4.4	5.17 ± 1.27	0.059 ± 0.011	0.174 ± 0.116
McA21	5.44 ± 0.04	24.3 ± 3.8	23.7 ± 0.1	63.5 ± 1.2	13.7 ± 0.9	59.9 ± 3.7	226.4 ± 21.0	9.11 ± 0.94	0.055 ± 0.013	0.213 ± 0.078
McA28	4.64 ± 0.65	12.6 ± 1.4	23.7 ± 0.6	80.0 ± 1.9	15.6 ± 0.6	147.0 ± 5.2	147.1 ± 3.7	12.13 ± 1.21	0.065 ± 0.002	0.184 ± 0.021
McA36	3.70 ± 0.14	20.6 ± 2.1	22.1 ± 0.7	69.9 ± 3.4	20.8 ± 2.3	121.7 ± 10.4	148.3 ± 28.1	7.09 ± 0.42	0.015 ± 0.008	0.218 ± 0.016
McA45	3.05 ± 0.15	20.0 ± 0.8	18.0 ± 0.3	60.7 ± 2.5	26.4 ± 1.7	103.1 ± 4.4	82.5 ± 3.5	18.93 ± 2.14	0.015 ± 0.008	0.199 ± 0.002
McA60	2.14 ± 0.33	12.2 ± 2.6	10.6 ± 0.1	38.2 ± 2.3	32.7 ± 0.7	36.5 ± 0.2	63.2 ± 11.1	11.73 ± 1.12	0.031 ± 0.010	0.156 ± 0.054

The depth interval referencing system refers to the multilevel location and then to the depth of metres below ground level, e.g. BH16 is Bolton Hill multilevel, depth interval at 16 mbgl.

Table 4 Hydrochemical composition of different recharge sources

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[a] ± 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[a] ± 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

[a]Represents total amount of nitrogen (nitrate, nitrite and ammonium) as mg N/L.

of their low ratios; NO_3 and P are usually nonconservative in soils and groundwater. Hence, Na, Cl, Mg, HCO_3 and B were found to be useful to detect stormwater recharge though the contrasts between the signatures of stormwater and the background recharge are much smaller than for foul sewage. Concentrations of HCO_3, B and K are consistently higher in the most shallow 30 m in all four multilevel piezometers (Fig. 5) reflecting the depth of penetration of groundwater contamination by foul sewage (Table 5). Higher concentrations of SO_4, Ca and Mg are observed at intermediate depths at both Bolton Hill and McAuley School indicating gypsum dissolution but the origin of high concentrations of Cl with only slightly elevated levels of Na in depths of 25–40 mbgl at Bolton Hill is unclear though the possibility road salting followed by some Na exchange is possible.

These results represent the average urban recharge quality of the four locations and were used to estimate the different recharge volumes, particularly leakage from the sewage network. Horizontal flow velocities in the study area are in the order to 1–10 m/year and vertical velocity in the order of 1 m/year and so groundwater

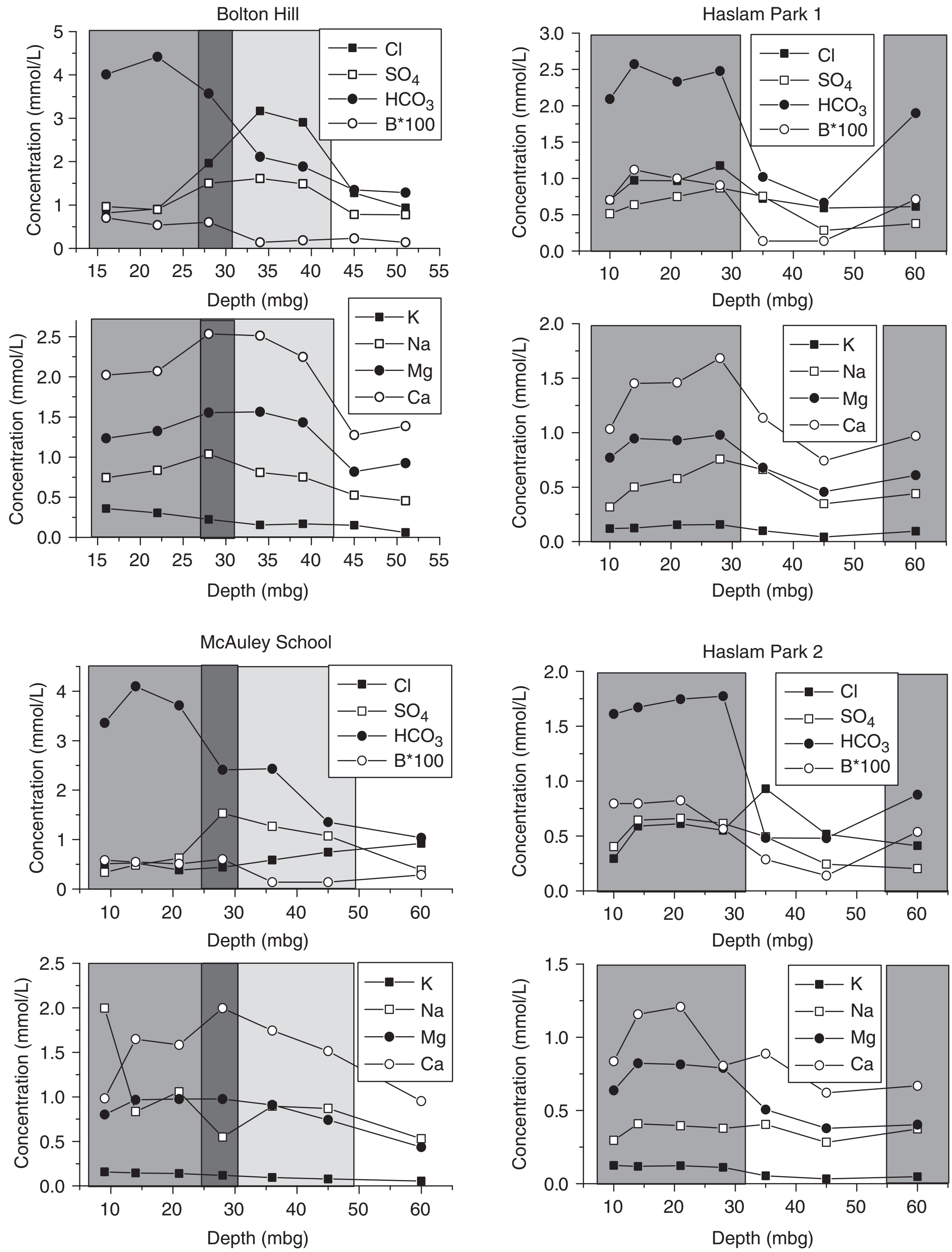

Fig. 5. Mean concentrations plotted against depth below ground. Different chemical signatures are highlighted with shaded areas – dark grey areas showing an admixture of sewage and light grey areas show an alternative source admixture (e.g. stormwater).

residence times to 30 mbgl are in the order of 10 years (Morris *et al*. 2006), in line with similar findings from Birmingham and Nottingham (Taylor *et al*. 2006). Horizontal to vertical dispersivity ratios are therefore in the order to 1 : 10 and so samples can be seen as representing mixing over horizontal distances of tens of meters. The volume of each recharge source and the estimated range of total recharge are listed in Table 2 and the respective

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/year total urban recharge

	Bolton Hill						
Total urban recharge (mm/year)	BH16	BH22	BH28	BH34	BH39	BH45	BH51
150	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
200	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						
	HP1 10	HP1 14	HP1 21	HP1 28	HP1 35	HP1 45	HP1 60
150	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
200	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						
	HP2 10	HP2 14	HP2 21	HP2 28	HP2 35	HP2 45	HP2 60
150	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
200	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						
	McA9	McA14	McA21	McA28	McA36	McA45	McA60
150	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
200	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

The depth interval referencing system refers to the multilevel location and then to the depth of metres below ground level, e.g. BH16 is Bolton Hill multilevel, depth interval at 16 mbgl. Standard deviations are given in the ± figures.

water qualities in Table 3. The natural background concentrations should be approximately two to three times the rainwater concentrations due to evapotranspiration (Walton 1981). Uncertainties of rainwater quality are assumed to be ±10%. Assumed uncertainties are 10% for all parameters except sodium and chloride where 20% uncertainty is assumed in recognition of sporadic road salting during winter.

If Eq. (5) is used to obtain sewer leakage rates, the remaining unknowns are the leakage from the stormwater and the foul sewage systems. The calculation of total sewer leakage was carried out for K, Na, Cl, HCO_3 and B as these are conservative and have a wide range of values between source and groundwater. SO_4, Ca and Mg are not considered in the final overall averaging process because they are potentially derived from congruent dissolution of gypsum (Rueedi *et al*. 2007).

As the chemical signature of stormwater is not easily distinguishable from natural recharge and, given that stormwater leakage appears limited, then recharge from stormwater leakage is considered part of the natural recharge. The contribution of sewer leakage, assuming 200 mm/year total recharge, is calculated for each parameter in all multilevel intervals and averaged [Eqs (8)–(10)] in Table 5. The resulting estimate uncertainty, when considering one parameter only in the uncertainty analysis, is of the order of 30–50%. Combining multiple parameters in the uncertainty analysis leads to standard deviations of 15–25% (Table 5).

The elevated concentrations found in the medium depth levels of Bolton Hill site (L28, L34 and L39) should be treated with care as the signal found there indicate an unknown additional source of chloride and sulphate, potentially due to road salting, which can also get into combined sewers quickly as a pulse on thawing or after rainfall. Table 5 shows that even the deepest levels containing the oldest groundwater seem to contain up to 20 mm/year of sewage. The shallower wells, however, contain typically between 20 and 35 mm/year of sewage, corresponding to 12–20% of total urban drainage or $7\text{–}12 \times 10^{-5}$ L/s/m of pipe. This corresponds well with findings from Liverpool, where sewer leakage was estimated to be of the same order as pressurised mains leakage (Howard 2001) and other studies from Europe and the United States (Table 6).

Assigning the weighting to the parameters can assess the usefulness of different parameters; sodium and potassium were found to be the most valuable 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. Sulphate was the least useful parameter

Table 6 Comparison of results of this study and other studies in Europe and the United States

Location	Method	Leakage (L/s/km pipe)	Reference
Doncaster, UK	Mass balance	0.07 to 0.12	This study
Prague, Czech Republic	Direct exfiltration measurement	3.5	Kohout *et al.* (2003)
Germany	estimated	0.012	Hoffmann & Lerner (1992)
USA	Exfiltration modelling	1.39–3.9	Gruenfeld (2000)
Aachen, Germany	Direct exfiltration measurement	<56	Decker & Risse (1993)
Dundee, Scotland	Extrapolating single leaks	2	Blackwood *et al.* (2005)
Rastatt, Germany	Monte Carlo Simulation	0.011–0.164	Wolf & Hoetzl (2007)

due to natural sources, potentially due to the large amount of sulphate found in construction materials (Howard 2001).

Nitrogen and phosphate levels in shallow groundwater should demonstrate higher values under sewage contamination influence though the highest concentrations of NO_3 are observed in the deeper levels least contaminated by sewage. 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 contrasts with other urban studies in the Sherwood sandstone that found significant increases in N in urban aquifers. More comprehensive studies (e.g. hydrochemical modelling, detailed field-study of single sewer leaks) on the storage of these two parameters in this aquifer are needed.

Conclusions

(1) A mass balance approach has been used to estimate sewer leakage proportions for a suburban area of Doncaster, UK. This was done using depth-specific hydrochemistry profiles from multilevel piezometers.
(2) Using estimated total urban recharge rates of 150 and 200 mm/year, the results show that 5–10% of total sewer volumes are lost to groundwater. The calculations suggest that the contributions to recharge from sewers are equivalent to 28 mm/year from foul sewer, 12 mm/year from stormwater and 22 mm/year from mains water.
(3) The results suggest that approximately 30–40% of total recharge in this urban study area may be coming from losses from the mains and sewage pipe networks, with the balance coming from other forms of infiltration.
(4) The statistical approach used in this study enabled 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.
(5) The results show the usefulness of multilevel piezometers to assess leakage rates from sewage system because these 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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