Quantification of groundwater recharge in the city of Nottingham, UK

Y. Yang · D.N. Lerner · M.H. Barrett · J.H. Tellam

Abstract Groundwater is an important and valuable resource for water supply to cities. In order to make full and wise use of the asset value, a clear understanding of the quantities and sources of urban groundwater recharge is needed. The water supply and disposal network is often an important source of recharge to urban groundwater through leakage from water mains and sewers. An approach to establishing the spatial and temporal amounts of the three urban recharge sources (precipitation, mains and sewers) is developed and illustrated using the Nottingham (UK) urban aquifer. A calibrated groundwater flow model is supplemented by calibrated solute balances for three conservative species (Cl, SO_4 and total N), thus providing four lines of evidence to use in the recharge estimation. Nottingham is located on a Triassic sandstone aquifer with average precipitation of 700 mm/year. Using the models, current urban recharge is estimated to be 211 mm/year, of which 138 mm/year ($\pm$40%) is from mains leakage and 10 mm/year ($\pm$100%) is from sewer leakage. The wide confidence intervals result from the scarcity of historical field data and the long turnover time in this high volume aquifer, and should be significantly lower for many other aquifer systems.

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Introduction

Urban groundwater is an important and valuable resource for potable water supply and industrial use, but is at risk from polluting landuses. Over-abstraction has traditionally been a concern, causing falling water tables and problems such as continual refurbishment of water supply infrastructure and land subsidence. Problems of too much groundwater are more common in the later stages of city development such as in the UK and elsewhere in Europe, and can be caused by both reducing abstractions and increasing recharge. Rising water tables can be nuisances and hazards, by flooding basements and tunnels, mobilising pollutants in unsaturated zone, and affecting foundations. A recent trend in the UK has been a decrease of urban abstraction due to pollution problems at public water supply wells and a downward trend of industrial pumping. In contrast, there is growing pressure on rural groundwater because of its good characteristics for exploitation.

Reassessment of the value of urban groundwater resources is timely, in order to balance the complex and conflicting issues of environmental impacts, sustainable use, and pollution risks. As part of this reassessment, tools are needed to quantify groundwater balances in urban areas. In particular, urban recharge is a complex and poorly understood process. The objective of this paper is to investigate the use of multiple solute balances to estimate recharge components in an urban environment. The city of Nottingham, UK, is used as a case study.

Sources and pathways for recharge

The classical view that cities reduce recharge because of the high proportion of impermeable surfaces has being recognised as incorrect (Lerner 1986; Price and Reed 1989; Foster and others 1994). Not only are there opportunities for precipitation to bypass impermeable surfaces, but substantial quantities of water are imported for public and industrial supply and a proportion of these imports finds its way to groundwater. Thus the main sources of recharge water in urban areas are precipitation and the water supply and disposal systems.

As land surfaces are altered by cities, so the hydrological pathways for precipitation are altered (Fig. 1). The main change is the interception of rainfall by relatively low permeability surfaces such as roofs, roads and other paving. The resulting storm runoff must be disposed of, and UK towns have a variety of types of sewer networks. Some have separate foul and storm sewers, while others have combined sewers that mix both types of water. The roof runoff from at least the rear part of many houses is directed to soakaways to reduce the sewer loading and will increase recharge.

Road and other paving might be expected to increase runoff, and they certainly do for higher intensity events. However, some careful research (Hollis and Ovenden 1988) showed that runoff from smaller events was low. They suggested that evaporation accounted for some of the missing water and the remainder was infiltrating through the road, particularly through gaps between kerbstones and cracks in the gutters. Even where storm sewers exist, they are unlikely to be watertight and leakage will occur. The leakage can be inwards or outwards, depending on the depth of the sewer relative to the water table, and the nature of any permeable surrounding, such as sand or gravel placed as support material in the trench. This permeable path allows water entering the sewer to flow laterally until it finds an opening. Conversely, water and pollutants leaking out of sewers can flow laterally to find a convenient route to penetrate deeper into the aquifer (Lerner and others 1993).

Different approaches are used for storm water disposal in other climates and cultures, and these often enhance recharge. In rapidly urbanising cities or arid climates, there may be no storm sewer system at all and runoff accumulates in ditches and depressions. Runoff retention basins (Ku and others 1992), infiltration basins (Appleyard 1993), recharge boreholes (Telfer and Emmett 1994), and permeable pavements (van de Ven 1990) are used in many cities. Although designed to control the quantity and quality of runoff, they also provide recharge routes that bypass the soil zone.

The water that is brought into cities for public water supply is a major part of the water balance. Lerner (1990) quoted some examples from around the world. Imports ranged from 14 to 7500 mm/year, when the average flow was expressed in the same units as precipitation; water supply to the case study city of Nottingham is 700 mm/year, which is roughly equivalent to the rainfall. This water supply is distributed through the city to consumers, and then is collected for waste. It can find a variety of routes to recharge groundwater, as illustrated in Fig. 2.

Consumptive use of water by humans and in cooking is small, and the main consumption is in plant watering. Even in the UK, water consumption for gardening can be high, with peak daily demands in a hot, dry period being three times the average, mainly due to garden watering. Over-irrigation of parks and gardens is common in urban areas. The water is not usually paid for by usage, so there is little incentive to save. A corollary of amenity irrigation is the high probability of excess irrigation giving rise to groundwater recharge, particularly with sandy and permeable soil.

Leakage from water mains is a major source of urban recharge. The network of pressured, unseen pipes can never be in perfect condition, and leaks are a recognised feature of water supply sizing and design. Loss rates of 20–25% are considered normal in the UK (Price and Reed 1989), and are similar to the proportion of rainfall that becomes recharge in the UK. A portion of this loss may be legitimate usage (Lerner 1988), and part of that which is leakage may be intercepted by sewers, which normally lie deeper than water mains. Thus estimating recharge from water mains is difficult, but it can exceed that from precipitation, even in temperate and humid climates.

Some issues of sewer leakage are discussed above in the

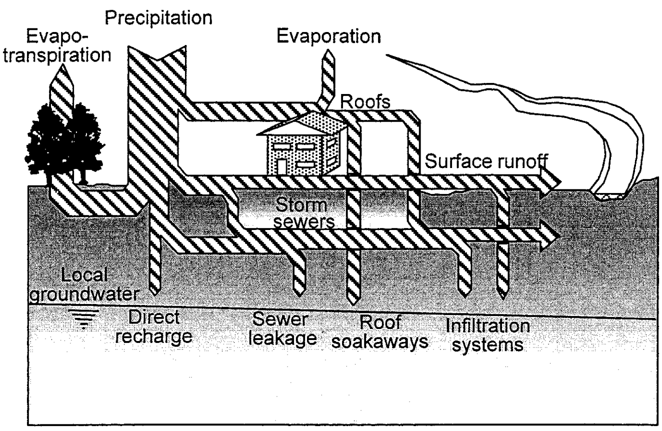


Fig. 1
Conceptual model of urban pathways from recharge sources – precipitation to recharge

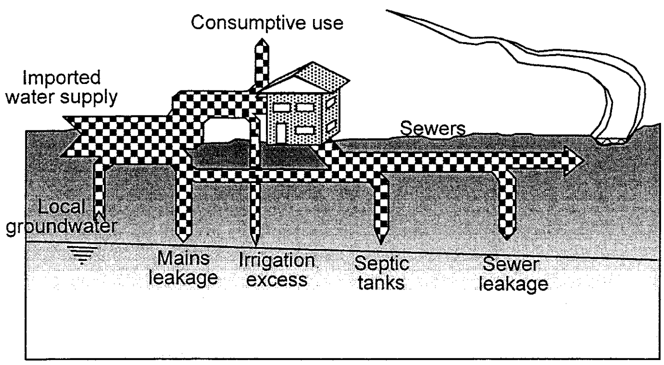


Fig. 2
Conceptual model of urban pathways of water supply to waste and recharge

context of storm systems, and apply equally to foul sewers. They can be expected to leak unless specially designed not to, leakage can be inward or outward, and the quantifying the losses is difficult. Recent reviews of literature on sewer leakage and data from UK cities (Lerner and others 1994; Misstear and others 1995) showed that, although leakage occurs, there are almost no estimates of quantities, and no proven methods of identifying and quantifying them.

Not all cities have complete sewer systems to remove wastewater for treatment, although they are almost universal in Northern Europe. In the UK for example, 97% of all households are sewered, not just in urban areas. The alternative disposal route for waterborne waste is through septic tanks and other soakaway systems. If used, these clearly return most wastewater to the subsurface, as well as the chemicals it carries. Well-designed soakaway systems can reduce the microbiological and organic loads of sewage, particularly in intergranular aquifers with a substantial unsaturated zone and sufficient travel time. However, they will not significantly reduce the nitrogen loading.

Generalisation about the overall effect of urbanisation is not possible across all cities due to the variable geologies, climates and infrastructures (Thomas and Foster 1986). Pathways for precipitation recharge are more numerous and more complex than in the rural environment, and many of them can enhance recharge to compensate for any loss of direct recharge due to impermeabilization. There will be substantial recharge from the water supply system, which is expected to range from 90% of the supply in cool, unsewered cities to 10% in cities with well maintained mains and sewers (Foster and others 1994). The complexity of urban infrastructure will always make it difficult to measure or estimate recharge rates. A major need is for methods that can estimate areal average recharge rates for use in modelling and water resources studies.

Outline methodology

The traditional approach to estimating urban recharge has been a combination of water balance with groundwater modelling. Most examples do not consider all of the sources and routes of recharge. For Lima, Peru, Lerner and others (1982) obtained water supply and leakage rates for all zones of the city and used these to estimate recharge from mains. They were fortunate that rainfall and storm runoff was not important; however, they took no account of sewage leakage. Recharge rates were confirmed by calibration of a groundwater model. A similar approach to mains and sewage was used for Birmingham (UK) by Greswell and others (1994) and for Liverpool (UK) by Rushton and others (1988). In the latter case, calibration of the model suggested that recharge was higher and the additional water was ascribed to storm water systems. For Coventry (UK), Lerner and others (1993) made an attempt to estimate leakage from the foul sewer network by comparing water supply inputs and sewage outflows. The differencing of large numbers, combined with the likelihood that both inward and outward leakage occurred, meant that little confidence could be placed on the results.

Our approach has been to use solute balances to supplement and refine a standard water balance and groundwater modelling study. This has yielded additional information on both recharge sources and their net effect on groundwater. The initial groundwater model provided an estimate of total recharge, and solute data were intended to quantify the contribution to the total of each of the three main sources – precipitation, leaking mains and sewage.

Barrett and others (in press) discuss the whole range of solutes in urban water to identify those which may be useful as markers of individual recharge sources. They conclude that no simple markers are yet available, especially for recharge quantification. Solute balances require the use of conservative species for which historical data are available over the whole area of interest. In the case of Nottingham, this restricts the choice to chloride (Cl), sulphate (SO_4) and total nitrogen (N) as other major species in the aquifer can be non-conservative or have poor data availability. There are no geochemical reactions involving Cl that are likely to affect groundwater concentrations. Sulphur can be found in several species (HS^-, H_2S, SO_4^{2-}, HSO_4^-). The Nottingham aquifer is aerobic everywhere (except possibly close to point sources of pollution where strong degradation might occur) and the predominant form of sulphur in all the inputs is SO_4. Hence it is a reasonable assumption for a regional study that SO_4 is the dominant form and that there are no important sources and sinks of sulphur. Nitrogen inputs will be as oxidised (NO_3^-, NO_2^-), reduced (NH_4^+, NH_3^0) and organic forms, with latter two coming from sewage. Because the aquifer is predominantly aerobic, all nitrogen is expected to convert rapidly to NO_3, and there will be little loss by reduction to N_2. This agrees with field evidence, where reduced forms were only found occasionally and at low concentrations (Barrett and others, in press). Hence it is a reasonable assumption that total N can be used as a conservative solute for this aquifer. These three solutes are present in all the potential recharge sources but in different ratios; hence the information content of the solute balance is maximised by using all three solutes.

Multi-solute balances have been used by Adar and others (1988) and Adar and Neuman (1988) to quantify recharge to the Aravaipa Valley aquifer (USA). Their conceptual model assumes steady state hydraulic and chemical conditions, which permits the use of an optimisation technique to identify contributions to the zones of their mixing cell model. Neither condition applies to Nottingham, or to most urban aquifers where recharge and solute loads change with city development and landuse changes. We have used trial and error to fit the solute inputs, and then analysed the sensitivity of the results to changes in the initial assumptions.

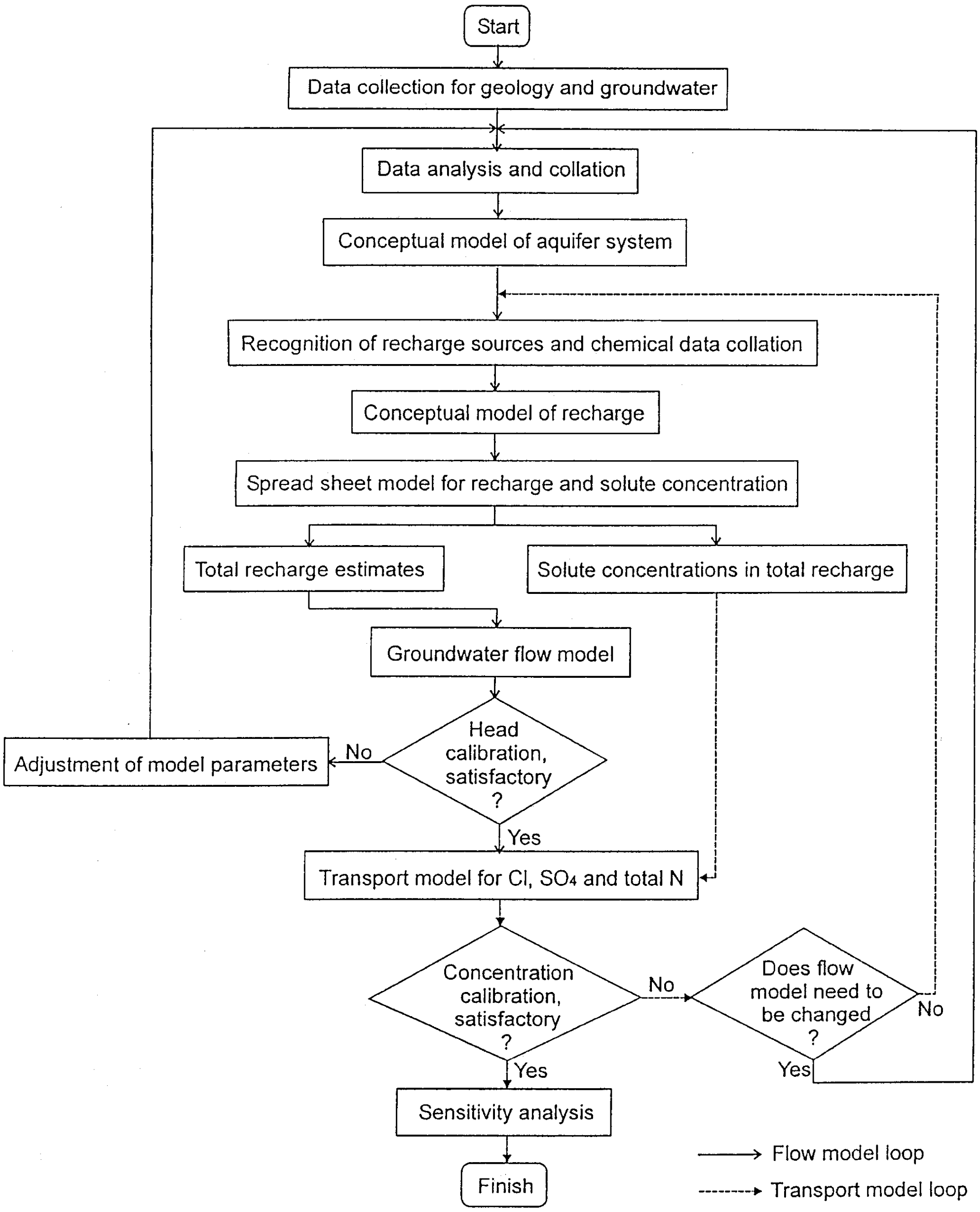


Fig. 3
Flow chart of quantification methodology

An outline of the procedure is shown in Fig. 3. There are two iterative loops, one each for the flow and solute transport models. Linking the loops is a spreadsheet model that calculates total recharge and average solute concentrations for each spatial division (zone) and temporal division (stress period) for both the flow and solute models. The spreadsheet model contains all the assumptions and data about urban growth, impermeabilization, water supply, leakage and solute concentrations in source waters.
The groundwater flow model is calibrated first. The important outcome is estimates of total recharge for each zone and period. Attention then transfers to the three solute transport models for Cl, SO_4 and N. Each uses the average concentration in total recharge, which is adjusted during calibration by adjusting the relative proportions of the three recharge sources. Once all models are calibrated, a sensitivity analysis is performed.

Hydrogeology and landuse in the study area

The geology of the Nottingham area is summarised in Fig. 4 (BGS 1981). The major bedrock of the study area includes sandstones, siltstones, mudstones and limestones (Charsley and others 1990). The strata in the district dip 1.5–4° towards the southeast. The most extensive spreads of unlithified Quaternary sediments occupy the Trent valley and tributary valleys such as that of the River Leen.

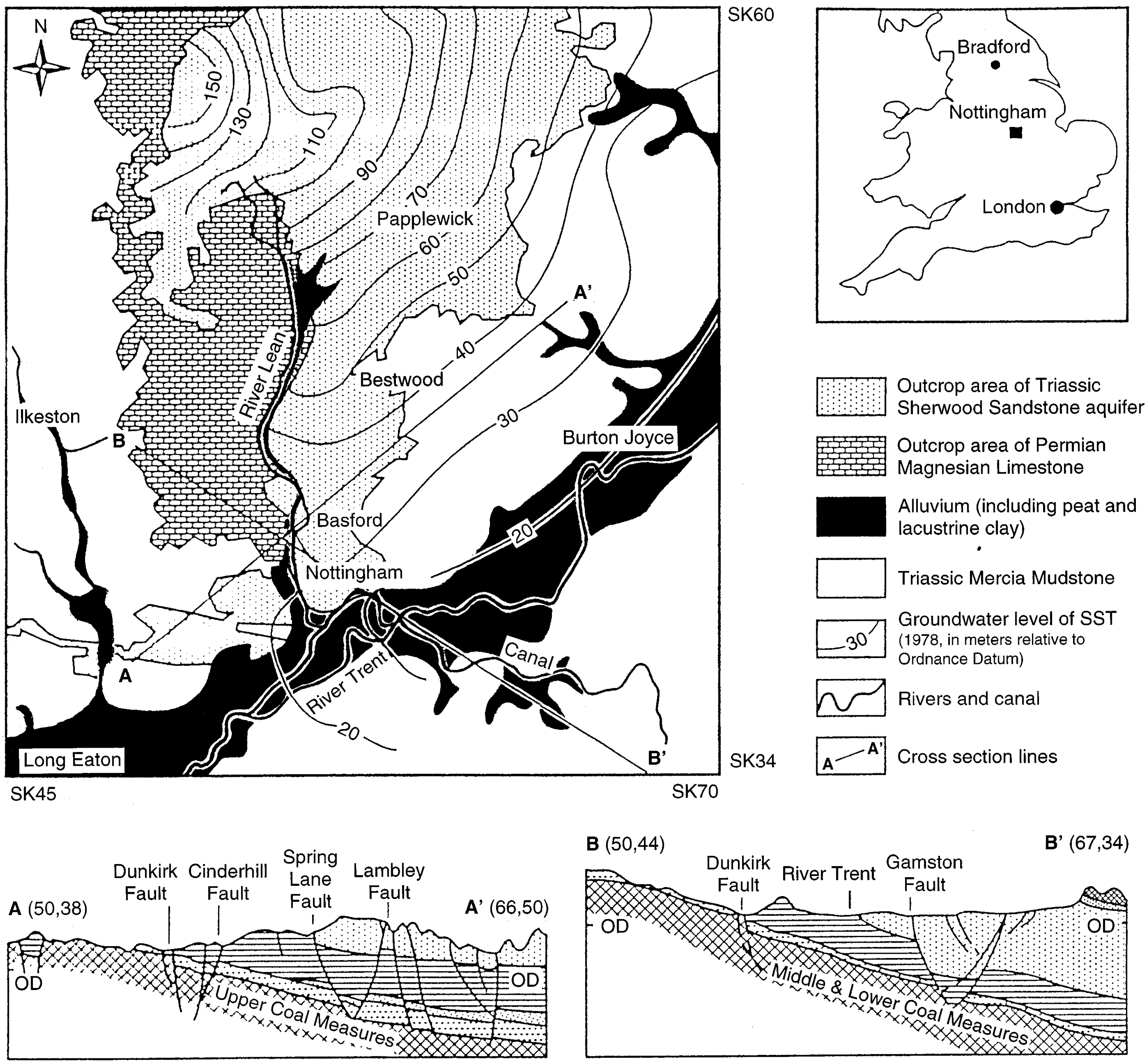


Fig. 4
Geology and groundwater of the Nottingham area (after BGS 1981; Charsley and others 1990)

These include till, sand and gravel, silt and clay. Hydrogeologically, the city of Nottingham lies on the Sherwood Sandstone Group, one of the most important UK aquifers. Permian Marls separate this from the underlying Lower Magnesian Limestone, a minor aquifer. The Sherwood Sandstone aquifer is extensively used for public and private water supply. The sandstones, where not impacted by human activities, produce a high yield of good quality groundwater. The aquifer is unconfined over much of the study area with little superficial cover away from the valley bottoms. To the east and south of the study area the Mercia Mudstone Group confines the aquifer.

The region is drained by the River Trent and its tributaries. The relatively broad and flat Trent river valley varies from 15–30 m above Ordnance Datum (AOD, approximately sea level), contrasting with the land north of the Trent which rises to over 160 m AOD. The modern city of Nottingham is thought to have its origin in Roman or pre-Roman times and became a major trading centre in the Middle Ages (Charsley and others 1990). Expansion occurred in the nineteenth century with the development of local coal mining and associated major industries in the city. The rapid urban growth of Nottingham commenced in the 1870s. The city is currently known for the manufacture of pharmaceuticals, bicycles, telecommunications equipment, cigarettes, and for its knitting and textile industries. Surrounding the city are coal mines (open cast), sand and gravel extractions and gypsum mining (from the Mercia Mudstone Group). The major rural landuse is for arable farming. Figure 5 shows the development of the city within the study area (Edwards 1966).

Regionally the groundwater flows from the northwest to the southeast with physical boundaries to the west and south. To the west the Sherwood Sandstone is in contact with the Permian, with a slight inflow from it. To the south there is the Cliffton Fault which acts as a groundwater barrier. To the east the aquifer dips under the Colwick Formation and Mercia Mudstone Group. The aquifer extends a long way to the east and south, but borehole evidence from this area suggests transmissivities are reduced due to compaction and cementation, probably making the formation ineffective as a water resource. The system continues much further north (Rushton and Bishop 1993); a flowline has been adopted for the northern boundary of this study.

Precipitation, leakage from water mains, and leakage from sewers are the major recharge sources. Various figures have been given for recharge to the aquifer. Lamplugh and others (1914) give figures of 4.5–10 inches per year (114–254 mm/year), with Land (1966) giving similar figures of 114–267 mm/year, dependant on rainfall. Rushton and Bishop (1993) have an average figure for this area of 239 mm/year, which was produced using a node-by-node soil moisture balance model.

The aquifer has a large volume of water in storage and a long turnover time (specific yield 15%, aquifer thickness 65–150 m from the north to the south). There is a long history of urban development and of groundwater abstraction with the first steam-powered public supply well installed in 1858. These factors combine to create complex and changing patterns of recharge, groundwater flow and solute movement, and require a transient distributed model to represent them.

Groundwater flow simulation

A groundwater flow model was developed from the hydrogeological conceptual model of the area, using MODFLOW (Harbaugh and McDonald 1996). Rushton and Bishop (1993) initially simulated the area as part of a regional resource model. Gebbett (unpub. data 1996) used this information to develop a model of the Nottingham area between National Grid References SK 520365 and SK700570. The current model is a refined version of the latter, with a basic grid spacing of 500 m, telescoped down to 250 m and 125 m for the city centre.

The Sherwood Sandstone aquifer was represented as a 2-D depth-integrated aquifer, unconfined to the west and confined for the eastern area. The western boundary is a specified inflow from the Permian, the eastern one is a no-flow boundary. The southern boundary is represented as no-flow except for a segment of specified outflow to the River Trent. The northern boundary is also modelled as no-flow, and represents a flowline running in a southeasterly direction. The River Leen is simulated as a river boundary (a series of leaky nodes). The calibrated aquifer properties were 10 m/d for hydraulic conductivity, 0.15 for specific yield and 0.0005 for confined storage.

A steady state simulation was used for conditions prior to 1850, which was when the first significant stresses were applied to the system. This provided the initial condition for the transient model, which was run from 1850 to 1995. The 145 years were divided into 13 stress periods, ranging from 2-years to 26-years long, and simulation time steps of one year. Abstraction and recharge were constant in each stress period. For the first 48 years, the dates of the stress periods were fixed by the starting of major pumping stations. From 1898 until 1960, landuse changes fixed the stress periods as they controlled recharge changes. From 1970 the periods were decades. The stress periods, with their recharge and abstraction, are shown in Table 1.

Five historical recharge estimates were obtained for each of the dates when landuse was mapped (1877, 1914, 1939, 1945, 1965; see Fig. 5), as described below in the section on recharge quantification. These recharge estimates were used for the stress periods as shown in Table 2. The total recharge over the 145 years has not changed much, in

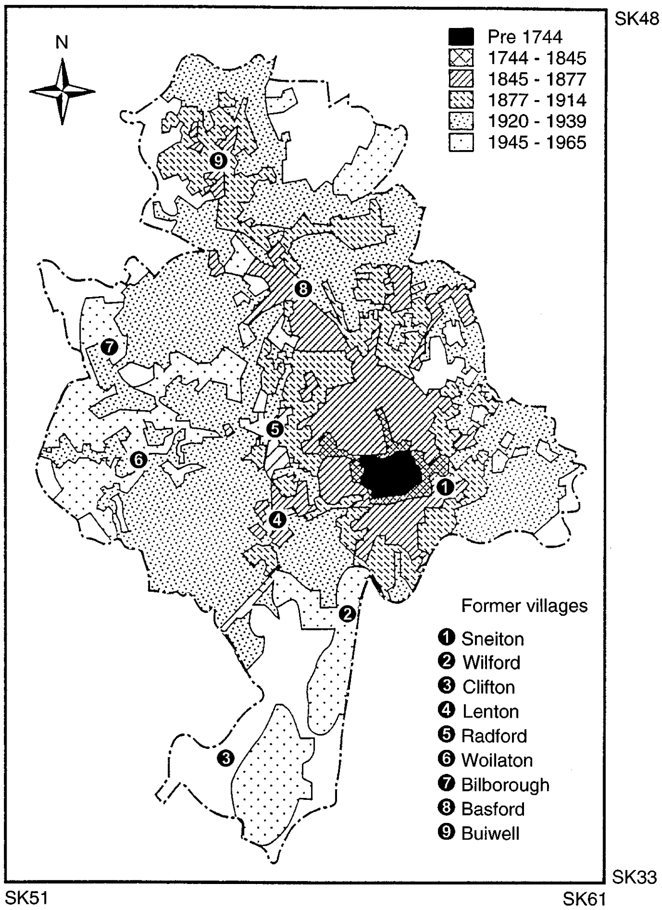


Fig. 5
Urban development within the study area

Table 1
Stress periods used to simulate the Nottingham aquifer

Stress period	Years	Length (years)	Date of urban recharge estimate	Total recharge (m^3/d)	Abstraction (m^3/d)
1	1850–1857	7	1877	66185	1818.4
2	1858–1871	14	1877	66185	10910
3	1872–1881	10	1877	66185	17729
4	1882–1884	3	1914	65465	18939
5	1885–1886	2	1914	65465	30304
6	1887–1898	12	1914	65465	28304
7	1899–1914	16	1914	65465	37393
8	1915–1940	26	1939	64779	34225
9	1941–1958	18	1945	64720	35825
10	1959–1970	12	1965	64705	44061
11	1971–1980	10	1965	64705	47551
12	1981–1990	10	1965	64705	47300
13	1991–1995	5	1965	64705	47300

Table 2
Loading zones and the delay of recharge reaching the water table

Zone	Description	Depths to groundwater (m)	Average delay (years)
1	Rural unconfined area	0–40	10
2	Basford industrial area, urban unconfined with shallow boreholes	0–20	5
3	Daybrook area, urban unconfined area by deep water table	5–50	15
4	City centre industrial area, urban unconfined with shallow groundwater table and subdued topography	5–10	4
5	Urban confined area	30–50	80
6	Rural confined area	50–70	200

common with other UK studies of Birmingham (Greswell and others 1994) and Liverpool (Rushton and others 1988) because water supply losses compensate the effects of impermeabilization.

These first estimates of total recharge from each period were found to be satisfactory for calibrating the flow model, probably because they were based on two previous modelling studies of this aquifer, and on other urban groundwater models in the UK. If the aquifer had not been so well studied, it is likely that several iterations around the “flow model loop” of Fig. 3 would have been necessary.

There were no calibration data for the steady-state model. For the transient model, 14 hydrographs were obtained from the Environment Agency, dating from 1970 or later. Some limited pre-1970 data were also available in Lamplugh and others (1914). The calibration was by trial-and-error. Figure 6 shows some typical example comparisons of model predictions and historical data under different land uses. The water levels in observation boreholes do not always represent the regional heads (Lerner 1989), so the aim of the calibration was not precise simulation of field heads, but to identify the trends in recharge in the urban area. Generally the model produces good matches between observed and computed values. The groundwater flow patterns are similar over the whole period, but heads fall about 10 m from 1850 to 1995.

Solute transport simulations

MT3D^{96}, a widely used solute transport package (Zheng 1993), was used to simulate the movement of solutes through the Nottingham aquifer. MT3D^{96} is a sophisticated package, solving the advection-dispersion equation in three dimensions, and incorporating sorption and first order decay if required. It is possibly more sophisticated than justified by the amount and quality of data available for the Nottingham aquifer. We considered developing a simpler model, similar to the mixing cell model used by Adar and Neuman (1988). However, such models suffer from serious numerical dispersion in transient transport simulations, and would require an interface with MODFLOW to obtain flow information for this study. Our approach has been to recognise the limited spatial and temporal data on solute inputs and to use a correspondingly simple discretisation with an accurate solution of the transport equations.

The temporal and spatial divisions followed those of the flow model. There were 13 stress periods (Table 1) and six spatial zones, as shown in Fig. 7 and described in Table 2. The latter were based on zoning of geology, depth of groundwater, landuse, topography and urban activity. The grid used to solve the transport equations was the same as that for the flow model, that is with 125–500 m spacings.

Depth-averaged simulations were used, similar to those of Adar and others (1988). Clearly most solute inputs are at the water table, and a 3-D pattern of concentrations will have developed. Little information exists on vertical variations. All the samples used for calibration came from fully penetrating, pumped boreholes, and so can be expected to be depth-averaged. Hence the calibration samples match the model formulation, and a 2-D simulation seems a reasonable approximation, certainly for this first study of the aquifer.

The initial solute concentrations, which represented pre-urbanisation conditions, were estimated on the basis of geology, landuse, historic chemical data (Lamplugh and others 1914) and the model calibration. Table 3 gives the values finally adopted. The five urban recharge periods were used to estimate the loadings of solutes, as discussed below. The additional data required were related to solute transport (dispersivity of 50 m, effective porosi-

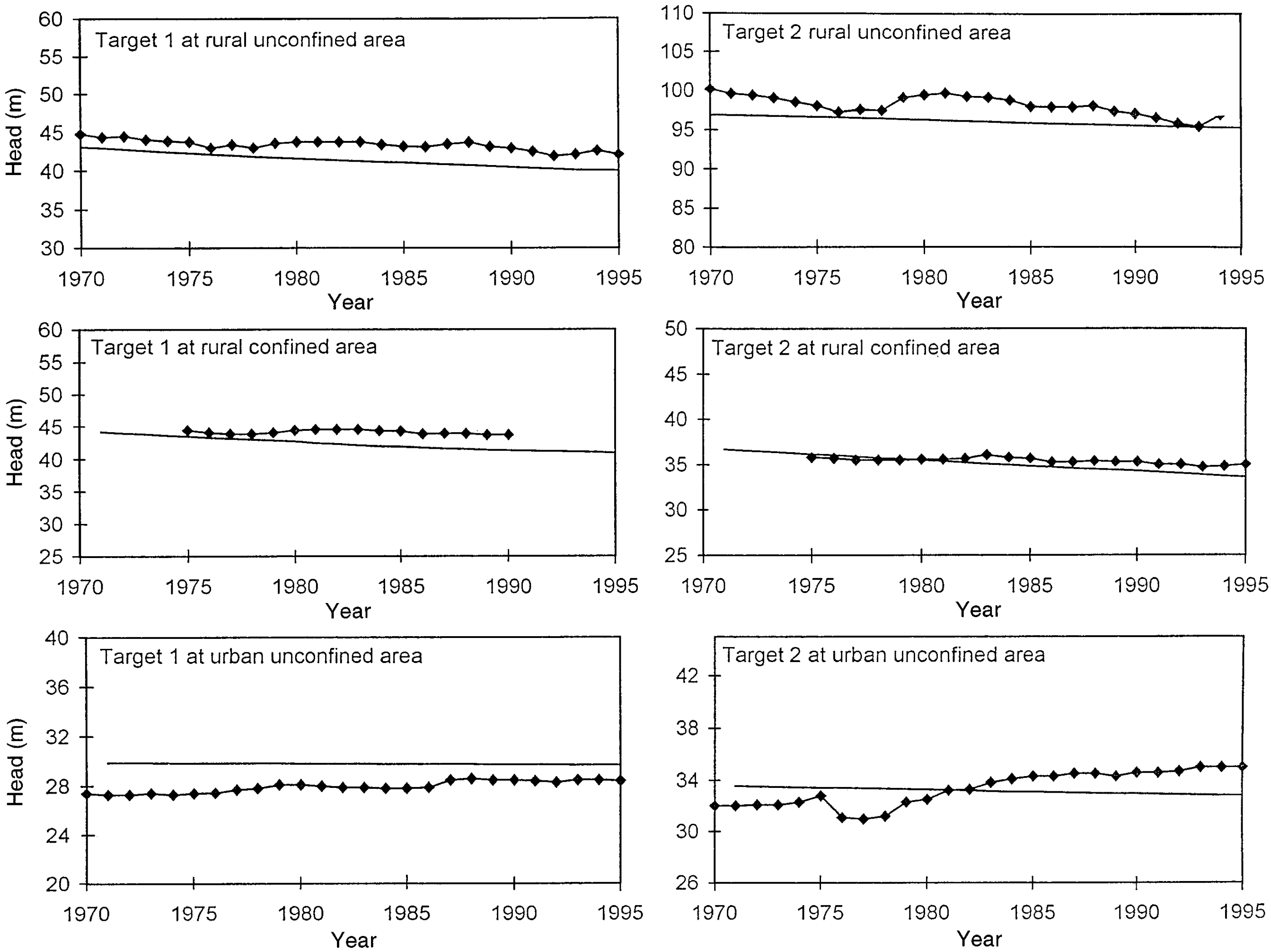


Fig. 6
Selected calibration results for groundwater levels from the flow model (*solid lines* are calculated, *lines with symbols* are field data)

ty of 0.2), choice of a solution method (hybrid method of characteristics), and solute concentration boundary conditions (solute input from various sources of recharge). MT3D^{96} only models one chemical species at a time so three models were constructed, one each for Cl, SO_4 and N.

The calibration targets have been selected for the zones considering the length and quality of record of solute concentration (Downing and others 1970; Edwards 1966; Lamplugh and others 1914; Page 1970; Thomas 1969). The locations of the targets are shown in Fig. 7. The data available for calibration of the solute models were sparse and showed some fluctuations in time (see Figs. 8–10), which may have been due to short-term pollution loadings or variations in pumping rates. The calibration was achieved by adjustment of the solute loadings, and comparing concentrations at the targets with model predictions for the nearest nodes. The loadings represented weighted averages of the inputs of Cl, SO_4 and total N in the various recharge sources for the different zones and recharge periods.

The models (Figs. 8–10) showed good matches between calculated and observed concentrations except for SO_4 and total N at some targets (BH5 and BH6 in zone 2, and BH7 in zone 4). Pollution is known to have occurred at some of the sites, which may explain the difficulty in obtaining better matches. In the absence of better data, the calibration of the three solute models was considered to be acceptable, and they were taken to present the overall trend of the solute loadings and groundwater chemistry in the study area.

The average concentrations of the three solutes are shown in Fig. 11 for the rural unconfined (zone 1) and

Table 3
Initial concentrations of solutes (mg/l)

Solute	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6 and south of Trent
Cl	10	20	20	20	10	10
SO_4	10	20	10	20	5	5
Total N	3	3	3	3	3	3

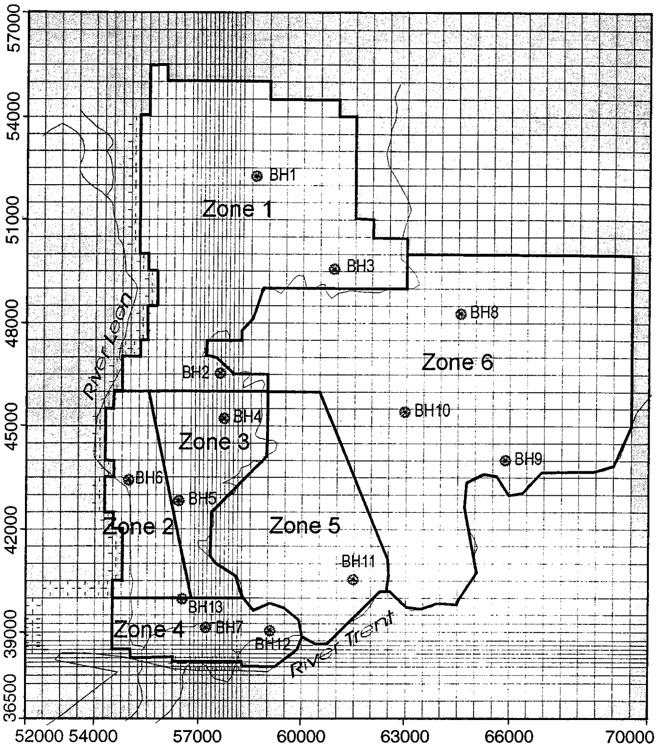
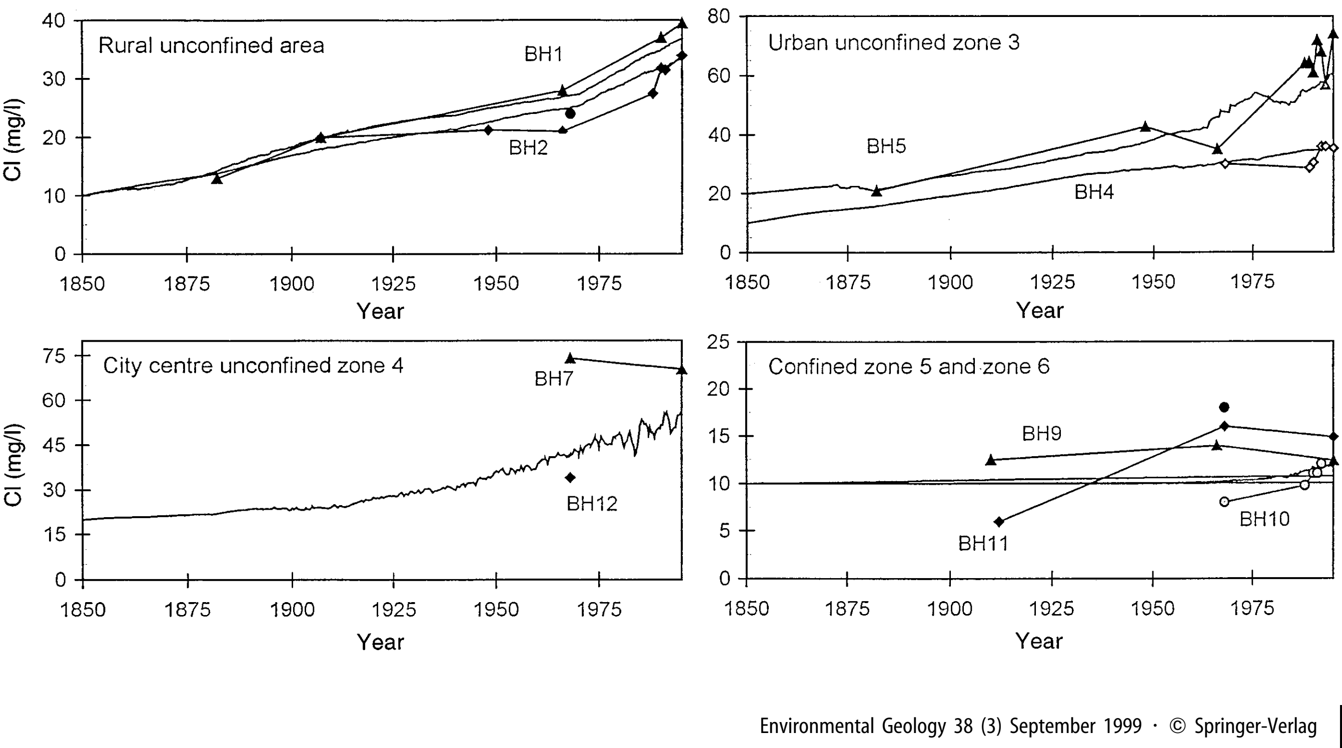


Fig. 7
Grid for groundwater and solute transport models, showing solute input zones 1–6

Fig. 8
Example calibration results for the Cl solute transport model (*lines with data points* are field data, others are calculated)



urban unconfined (zone 4) areas. All show rises over the period 1850–1995. Similar trends were seen in the other unconfined zones, but with smaller increases. In the confined areas the inputs remain the constant because of the long delays for recharge to reach groundwater.

Quantification of urban recharges

The flow and transport models described above used total recharge and its average concentrations as input data for simulations (Fig. 3). This section describes how these input data were built up using a spreadsheet model of recharge components and adjusted to provide satisfactory calibrations of the models. Total recharge is made up of three components, effective precipitation, mains leakage and sewer leakage. Six zones have been used to describe the spatial variability of recharge (see Fig. 7 and Table 2). Five recharge periods represent the temporal changes in recharge (Table 1) caused by the growth of the city (Fig. 5). Estimates for each period and zone of the amounts of, and solute concentrations in, each recharge component were combined to provide the overall values used to simulate groundwater flow and solute transport.

Total recharge

Total recharge was initially estimated by following the algorithms of Greswell and others (1994). They factored effective precipitation for the nature of the superficial geology and the density of urbanisation, and then added an allowance for leaking pipes. The formula used is:

$$\text{Recharge} = (\text{precipitation recharge} * F_d * F_u) + (\text{urban return flows} * F_d) \quad (1)$$

where F_d is a factor related to superficial cover, F_u is a factor for urban and industrial cover, and urban return

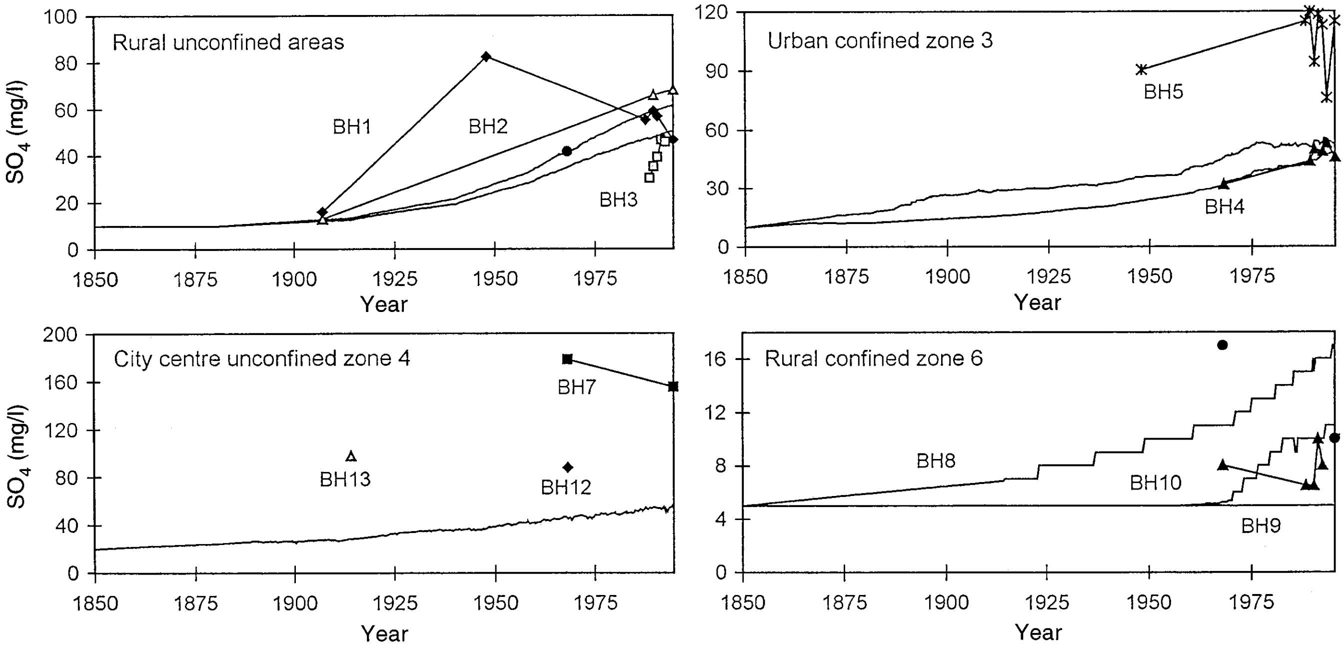
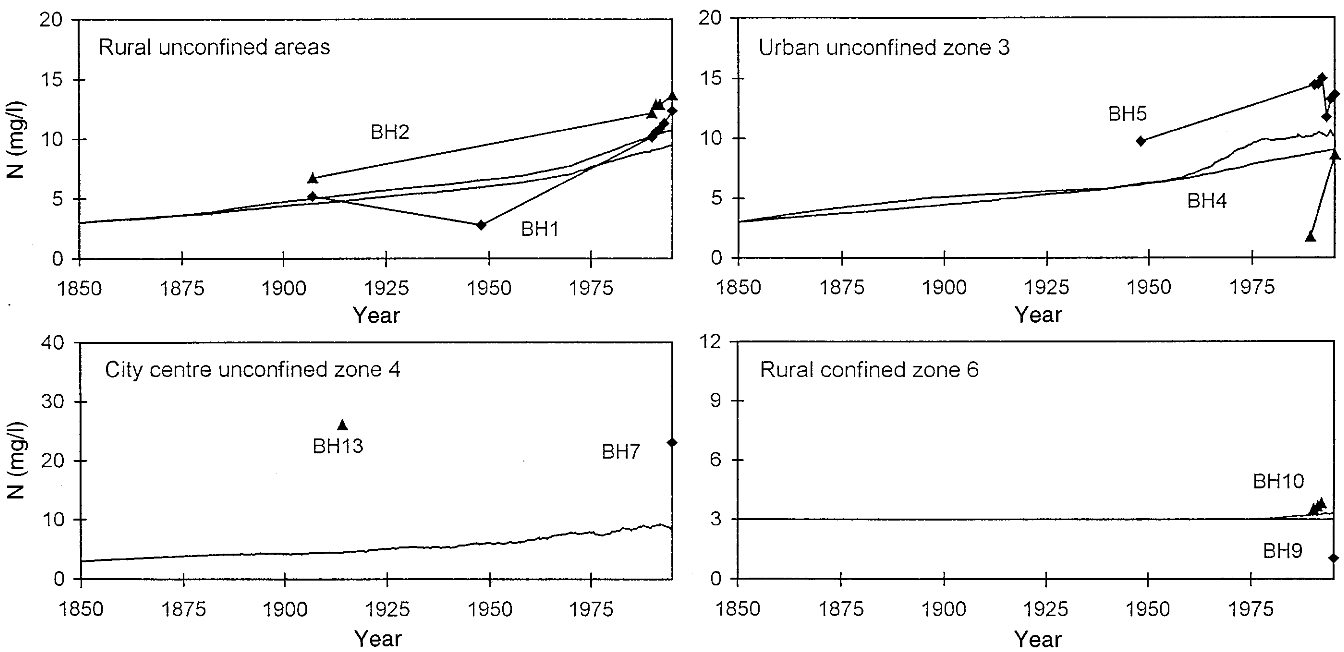


Fig. 9
Example calibration results for the SO_4 solute transport model (*lines with data points* are field data, others are calculated)

Fig. 10
Example calibration results for the total N solute transport model (*lines with data points* are field data, others are calculated)



flows are the leakage from public water supply mains and sewers. The values of Greswell and others (1994) were adopted, with the substitution of the Colwick Formation for superficial cover. The initial estimates of total recharge were found to give a satisfactory hydraulic calibration of the flow model (Fig. 6) and were not altered subsequently. However, the ratio of the recharge components was altered in order to calibrate the solute transport models.

Weighted solute concentration in total recharge

The concentration in recharge of each solute for each zone and period is given by a weighted average of concentration in each recharge component by

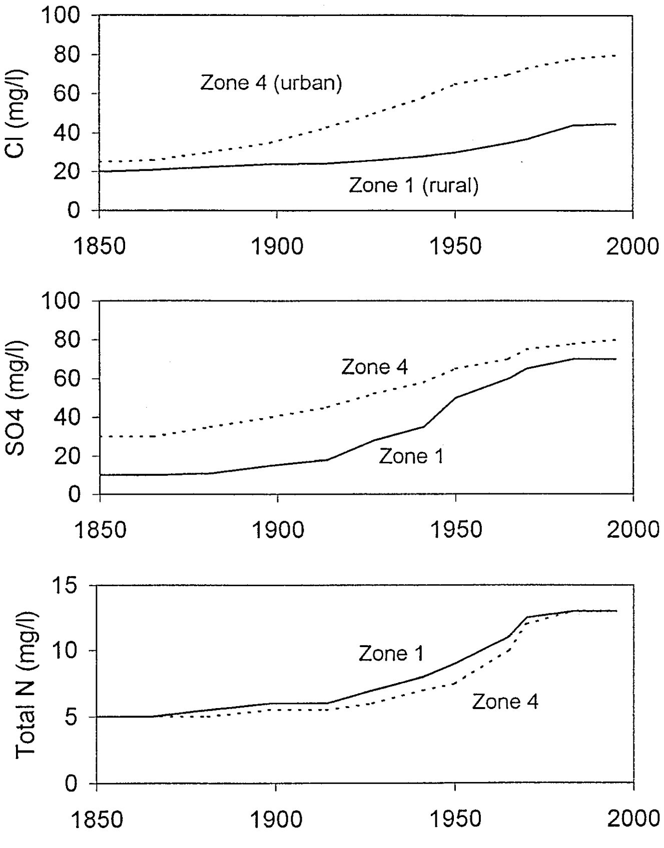


Fig. 11
Modelled solute loadings for Cl, SO_4 and total N (taking urban zone 4 and rural zone 1 as examples)

$$C(i,j,t)=\frac{1}{R(j,t')}[R_m(j,t')\,C_m(i,j,t') + R_s(j,t')\,C_s(i,j,t')+R_p(j,t')\,C_p(i,j,t')] \quad (2)$$

where C is the average concentration; C_m, C_s, C_p are the concentrations in mains, sewers and precipitation recharge; R is the total recharge; R_m, R_s, R_p are the recharges from mains, sewers and precipitation; i represents the solute (Cl, SO_4 and total N); j represents the spatial locations (zone 1–6); t indicates the stress period when solutes reach the water table (period 1–13); t' indicates the period when recharge occurs at the land surface, with t and t' related by $t'=t-d$, where d is the delay caused by the travel time of solute through the unsaturated zone. The recharge delay for each zone was estimated, assuming a moisture content of 10%, by $d=0.1WR$ where W is the average depth to groundwater. The depths and average delays used are given in Table 2 for each zone.
The recharge components are linked by

$$R=R_m+R_s+R_p \text{ for all } (i,j,t) \quad (3)$$

Concentration in precipitation recharge

A limited amount of historical data for Cl and total N concentrations in precipitation were available for the

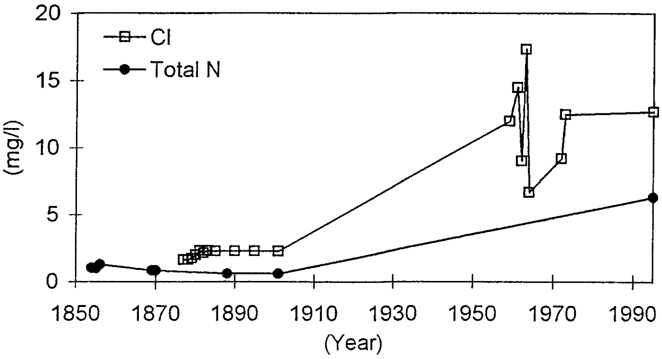


Fig. 12
Historical concentrations in precipitation in the study area

study area (Cawse 1976; Lamplugh and others 1914), see Fig. 12, and some modern data were collected for the project from 12 rain gauges around the study area. The only SO_4 data were from the current project.
Concentrations of all three species in recharge will differ from those in precipitation due to the concentrating effect of evapotranspiration, and the addition of any loadings that would be carried by recharge, such as fertilisers, landfills, industrial spillages or de-icing salts. A flux balance can express these processes

$$C_p(i,t)=\frac{1}{E}[PC_{pptn}(i,t)+L(i,t)] \quad (4)$$

where C_p, i and t were previously defined; E is effective precipitation, equal to recharge plus runoff; P is total precipitation; C_{pptn} is the solute concentration in precipitation; and L is the loading of solute due to agricultural or urban activities.
For rural areas, L was unknown. However, some boreholes had been drilled in the project to sample rural groundwater at the water table. These samples were assumed to reflect the overall process of Eq. (4), and shallow groundwater concentrations were used as estimates of C_p for the modern period. Concentrations in earlier periods were estimated by assuming a constant pattern of circulation between shallow groundwater and deep groundwater, and using historical concentration data for Cl and total N in deep groundwater

$$C_p^r(i,t)=C_{dgw}^r(i,t)\,\frac{C_{sgw}^r(i,13)}{C_{dgw}^r(i,13)} \text{ for } i=\text{Cl, total N} \quad (5)$$

where t is the stress period ($t=13$ for modern data), C_p^r is the concentration in rural precipitation recharge, and C_{sgw}^r and C_{dgw}^r are concentrations in shallow and deep rural groundwater respectively. In the absence of historical data, SO_4 concentrations were assumed to remain in a constant temporal pattern with Cl and total N, and were calculated

$$C_p^r(SO_4,t)=0.5\,C_{sgw}^r(SO_4,13)\left[\frac{C_p^r(\text{Cl},t)}{C_{dgw}^r(\text{Cl},13)}+\frac{C_p^r(\text{N},t)}{C_{dgw}^r(\text{N},13)}\right] \quad (6)$$

with $C_p^r(\text{Cl},t)$ and $C_p^r(\text{N},t)$ obtained from Eq. (5).

For urban areas, L, the loadings from industry, parks and de-icing were assumed to be insignificant, and Eq. (4) became

$$C_p^u(i,t) = \frac{P}{E} C_{pptn}(i,t) \quad (7)$$

where C_p^u is the concentration in urban precipitation recharge. The ratio of P/E estimated by Rushton and Bishop (1993) was used in Eq. (7) to estimate concentrations for the modern period. For the earlier periods, with consideration of changes in landuse and solute concentration in precipitation (Fig. 12), the same ratio of P/E was used to estimate historical $C_p^u(i,t)$ while historical C_{pptn} for SO_4 was estimated by a similar approach to the rural areas described above. Table 4 summarises the results of this procedure for concentrations in precipitation recharge (at the surface) for rural and urban areas. Table 5 gives the delayed solute concentrations arriving at the water table, and allowing for the mix of urban and rural landuses in some zones at some times.

Solute inputs from mains and sewage

Most boreholes used to supply the mains distribution system have not altered since their initial development.

Table 4
Estimated solute concentrations in precipitation recharge (mg/l) reaching the groundwater table after delay

Zone	Solute	1850–1881	1882–1914	1915–1940	1941–1958	1958–1995
	Cl	25	28.6	31.3	33.3	50
1	SO_4	12.1	17.6	34	50	71.4
	Total N	5	6.6	7.3	8.7	13.3
	Cl	5.5	6.5	20	30	35
2	SO_4	9	15	25	40	55
	Total N	3	4	6	10	18
	Cl	5.5	6.7	20	30	32
3	SO_4	8	15	25	38	50
	Total N	3	4	5	9	15
	Cl	5.6	6.8	20.3	30.4	35
4	SO_4	9	15	26.1	40.5	56.4
	Total N	3	4	6.6	10.7	18.9
	Cl	12	12	12	12	12
5/6	SO_4	6	6	6	6	6
	Total N	3	3	3	3	3

The assumption was made that mains water chemistry has been constant over time, in the absence of substantial historical data. The concentrations adopted are given in Table 6.

For leakage from sewers, concentration ranges were established using data collected during the project: Cl, 200–350 mg/l, SO_4, 80–150 mg/l and total N, 20–35 mg/l. The average concentrations were used for the spreadsheet modelling, and are shown in Table 6.

The mains and sewers were only present within the areas of urban development, which varied or expanded with time. The leakage from mains and sewers was effective in the urban zones as shown in Fig. 5 (Edwards 1966).

Recharge estimates

The spatial and temporal division of the total recharge into its various components was quantified by solving Eqs. 2 and 3. Average recharges (R) and their concentrations (C) are known for all zones and periods from the calibration of the models. Concentrations in each recharge source (C_m, C_s and C_p) have been estimated as described above. There are three unknowns (R_m, R_s and R_p) and three equations for each zone and period (one for each solute) and direct solutions were obtained. The result is summarised in Table 7, with the different recharge components expressed in mm/year. Zones 1 and 6 do not contain sewerage systems or mains distribution systems while zone 5 is confined and these types of recharge have not yet reached the water table.

As one would expect, the values of mains recharge generally increase with time whilst recharge from effective precipitation decreases as a result of urban expansion. Sewage recharge does not change much, except for fluctuations due to rounding errors. Total recharge is seen not to vary greatly with time, with only a slight decrease in the urban areas. Additionally total recharge is only very slightly less in the unconfined urbanised areas than in the unconfined rural area. Another clear result is that the recharge from water supply mains is important in the urban recharge budget, and therefore for groundwater resources. In recharge zones 2 and 4 mains water becomes the major recharge source from the 1915–1940 period. In zone 3, mains water never becomes the largest source, although it is almost as great as effective precipitation in the last period. Zone 3 is the most impacted by sewage leakage. Zones 2 and 4 are the oldest urban develop-

Table 5
Estimates of solute concentrations in precipitation recharge when leaving the soil zone (mg/l)

Stress period	Rural unconfined recharge (zone 1)			Urban precipitation recharge (zones 2–4)			Urban confined recharge (zone 5)			Rural confined recharge period (zone 6)		
	Cl	SO_4	N	Cl	SO_4	N	Cl	SO_4	N	Cl	SO_4	N
1958–1995	50	71	13.3	35	56.4	19	24	12	7.3	18.4	11	4.1
1941–1958	33	50	8.7	30.4	41	11	20	10	5.6	16	9	3.9
1915–1940	31	34	7.3	20.3	26	6.6	16.4	8.5	4.7	14.6	8	3.7
1882–1914	29	18	6.6	6.8	8.2	3	14	7	3.9	13	7	3.6
1850–1881	25	12	5	5.6	9	2	12.5	6	3.4	12.5	6	3.4

Table 6
Solute concentrations in recharge from mains and sewers (mg/l). These values were used for all stress periods

Recharge sources	Zones 2, 3 and 4 (all urban)		
	Cl	SO_4	Total N
Mains	45.3	50.7	5.6
Sewers	315	115	30

Table 7
Estimated recharge rates for each zone and period (mm/year)

Recharge source	Zone[a]	1850–1881	1882–1914	1915–1940	1941–1958	1958–1995
Mains	2	54	94	122	138	158
	3	27	33	51	64	93
	4	22	110	128	145	162
Sewage	2	5	4	3.4	3.3	6
	3	8	13	11	14	13
	4	9	7	6	7	8
Effective precipitation	1	239	239	239	239	239
	2	179	128	91.6	75.7	53
	3	182	168	150	134	105
	4	199	107	72	54	35
	5	50	49	49	49	49
	6	27	27	27	27	27
Total recharge	1	239	239	239	239	239
	2	238	226	217	217	217
	3	217	214	212	212	211
	4	230	224	206	206	205
	5	50	49	49	49	49
	6	27	27	27	27	27

[a] Zones 1, 5 and 6 have no mains or sewer recharge

ments, and it is possible that the mains distribution system is in a worse state of repair in these zones than in the more modern Daybrook area. However, this argument clearly does not apply for the sewerage system. A possibility would be that the older, nineteenth century, systems are not so prone to leakage as more modern systems, but this is only speculation.

Sensitivity analysis

The recharge quantification was carried out using chemical data for both groundwater and recharge sources. Unfortunately there is a great deal of the uncertainty in the process of hydrogeochemical development and groundwater formation. Whilst the current study collected relatively detailed information on groundwater and recharge source water quality, it still cannot account for temporal and spatial variations as the deep boreholes sampled are subject to uncertainties such as on-site contamination.

Whilst the mains water quality data are detailed and reliable, the sewage quality information is based on averages over the entire city, with no account of local variability. Additionally, each sample represents a snap shot. It is clear even from this study that sewage quality is variable with time. Further uncertainties exist for both recharge and groundwater quality over time. Whilst extensive literature searches have been carried out to obtain as much historical information as possible, the quantity of data available for the solute modelling calibration is still relatively small. The quality uncertainties are compounded by uncertainties relating to the groundwater flow model and consequently the zone recharge budgets. Again this relates to the limited historical data available.

In the light of these potential uncertainties a sensitivity analysis was conducted. The aim of this was to assess how accurately the quantification procedure could distinguish different ratios of the recharge components. For instance, the solute load in sewage is high, so how much change in sewage recharge will make a detectable difference to groundwater concentrations?

Two scenarios were applied to the model for the period 1850–1995. Scenario 1 was to change the proportions of the mains and effective precipitation recharge while keeping the sewage recharge constant. Scenario 2 was to keep the mains water contribution constant and vary the ratios of the sewage and effective precipitation recharges. In both cases the total quantity of recharge remains unaltered. Table 8 shows the sensitivity of predicted concentrations at each target borehole to changes in the ratios of recharge sources. Increasing proportions of mains water, relative to effective precipitation, increases the concentrations of Cl and lowers those of SO_4 and total N. Raising the proportion of sewage increases the concentrations of all solutes in recharge. The changes, which result from changing mains recharge by -40% to $+20\%$, are not large, and in most cases are within the noise in calibration data. This is best illustrated for Cl at BH5 in zone 3 (Fig. 8). Recent field observations have been in the range 56–74 mg/l. Table 8 shows that model predictions fall within this range for all the sensitivity analyses for mains.

The implication of the sensitivity analysis is that mains and sewage recharge can not be accurately estimated for this aquifer. Confidence intervals of $\pm 40\%$ and $\pm 100\%$ for mains and sewer recharge respectively may be appropriate. Two features of the case study lead to such wide intervals. Firstly the data for calibration are scarce in time and space. Secondly the high volume of the aquifer leads to a long solute turnover time, and the aquifer is not sensitive to changes in inputs.

Sensitivity coefficients show which input variables make the largest impact on model predictions, and which model outputs are most sensitive to changes in inputs, which helps to identify the data which are of most value for calibrating the models and hence for quantifying recharge. The sensitivity coefficient compares the change in calculated concentration to the change in recharge that produces that change:

Table 8
Sensitivity of the target solute concentrations in 1995 to the changes of mains and sewer recharge. Note: the analyses were conducting by changing one source of recharge (column 1), balancing this change by a change in precipitation recharge, and keeping the third source constant. The bold figures are the calibrated values

Source	Multi-plier	Cl concentrations				SO_4 concentrations				Total N concentrations			
		Zone 2 BH6	Zone 3 BH5	Zone 3 BH4	Zone 4 BH7	Zone 2 BH6	Zone 3 BH5	Zone 3 BH4	Zone 4 BH7	Zone 2 BH6	Zone 3 BH5	Zone 3 BH4	Zone 4 BH7
Mains	0.6	62	56	33	43	77	55	50	56	19	17	12	13
	0.8	64	59	34	49	75	55	49	55	15	14	10	11
	1	**70**	**60**	**36**	**55**	**73**	**54**	**49**	**54**	**13**	**10**	**9**	**9**
	1.2	76	67	38	57	72	53	47	51	8	7	7	5
Sewage	0	49	44	29	40	66	39	45	47	9	8	7	6
	1	**70**	**60**	**36**	**55**	**74**	**54**	**49**	**54**	**13**	**10**	**9**	**9**
	5	153	130	57	111	88	57	51	67	18	15	11	13

$$s = \frac{\frac{(C_n - C_0)}{C_0}}{\frac{(R_n - R_0)}{R_0}} \quad (8)$$

where C_n is the new groundwater concentration, C_0 is the original groundwater concentration, R_n is the new recharge rate and R_0 is the original recharge rate. Coefficients were calculated for each solute at each target borehole and then averaged for zones 2–4 (Table 9).

It is clear that the total nitrogen concentration of the groundwater is most sensitive to Scenario 1, the alteration of mains water to effective precipitation ratio. Increasing the mains proportion of recharge has the result of decreasing the total nitrogen concentrations. In Scenario 2, it is the chloride concentration in groundwater that is most sensitive to change, but the sensitivity coefficients are not as high as those for the total nitrogen under Scenario 1. Sulphate concentrations do not appear particularly sensitive to either scenario. The zone averages include both positive and negative coefficients, but show that none of the zones is particularly sensitive to inputs.

Overall the sensitivity coefficient values are not very high, reflecting the large changes in recharge sources within the model required to produce relatively small changes in groundwater quality. However, total N sensitivity has different signs for mains and sewage, and Cl and total N sensitivities have different signs for mains. Such opposing effects will make it easier to calibrate recharge models in cases where good data are available.

Table 9
Average sensitivity coefficients for solutes and zones

Item averaged	Scenario 1: Mains changed[a]	Scenario 2: Sewage changed[b]
Cl	0.34	0.25
SO_4	−0.24	0.08
Total N	−1.21	0.16
Zone 2	−0.32	0.19
Zone 3	−0.31	0.14
Zone 4	−0.37	0.19

[a] Average of runs with 0.8 and 1.2 multipliers
[b] Average of runs with 0 and 5 multipliers

Conclusion

There are two main sources of recharge to urban groundwater – precipitation, and water supply and disposal networks. Precipitation recharge is reduced by impermeabilization, but may be increased by soakaways, infiltration basins and leaking storm sewers. Leakage from water supply mains is commonly 25% of supply, much of which will become recharge. Leakage from sewers has rarely been quantified, but is likely to present pollution risks. Urban recharge is difficult to estimate due to the complexity of city infrastructure. Previous attempts to quantify have used water balances and calibrated groundwater flow models; most have neglected sewer leakage. This study used models of groundwater flow and three solutes (Cl, SO_4 and total N) to estimate all three recharge components (precipitation, mains and sewers) for the city of Nottingham, UK, over the period 1850–1995. Total recharge to the unconfined urban aquifer appears to have declined by about 8% over the period from an initial value of 230 mm/year; this likely to be within the margins of error. Leakage from water mains has grown significantly and is now the major contributor to recharge. Central estimates range from 93–162 mm/year over the city. A sensitivity analysis suggested that wide confidence intervals of ±40% should be associated with mains recharge due to the scarce groundwater quality data for calibration, and the slow response of the aquifer to changes of input.

The solute balance suggests that some recharge comes from sewer leakage, with estimates in the range 6–13

mm/year (±100%) over the city. There has been little change in these rates over the study period, perhaps reflecting the more widespread use of foul soakaways in the nineteenth century. Solute concentrations in Nottingham's groundwater were found to be not very sensitive to sewer leakage rates, leading to wide confidence intervals on the estimated leakage.
The method, developed in this paper, of multiple solute balances shows promise for estimating urban recharge components. The uncertainties on the results that arose in applying it to Nottingham were mainly due to lack of good historical data and the long turnover time of the aquifer. It will more successful when good historical data are available over a period that approaches the aquifer residence time.

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