THE IMPACT OF URBAN GROUNDWATER UPON SURFACE WATER QUALITY: BIRMINGHAM – RIVER TAME STUDY, UK.

by

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ABSTRACT

A field-based research study has been undertaken on the River Tame within the industrial city of Birmingham, UK, to understand better the influence of urban groundwater discharge on surface-water quality. The 8 km study reach receives ~6% of its total baseflow (60% of which is groundwater) from the underlying Triassic Sandstone aquifer and flood-plain sediments. An integrated set of surface water and groundwater flow, head and physical/chemical data was collected from installed riverbed piezometers and existing monitoring across the aquifer. Field data and supporting computer modelling indicated the convergence of groundwater flows from the sandstone/drift deposits and variable discharge to the river (0.06 to 10.7 $m^3d^{-1}m^{-1}$, mean 3.6 $m^3d^{-1}m^{-1}$), much of which occurred through the riverbanks. Significant heterogeneity was also observed in groundwater quality along and across the river channel. Key contaminants detected were copper, nickel, sulphate, nitrate, chlorinated solvents, e.g. trichloroethene, and their biodegradation products. Groundwater contaminant concentrations were generally lower than expected and ascribed to dilution and natural attenuation within the aquifer and riverbed. High concentration plumes were detected, but their effect was localised due to substantial dilution within the overlying water column of the river. Estimated contaminant fluxes were not found to reduce significantly the present surface water quality, which is poor (>30% is pipe-end discharge). Comparative studies elsewhere and further elucidation of heterogeneity and natural attenuation controls are recommended.

To my wife Yumi and sons Max Leo and Ben

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CHAPTER 1. INTRODUCTION

1.1 Project outline

The development of conurbations in the vicinity of river systems is common. Such urbanisation may cause contamination of land and underlying groundwater by a wide range of substances (Lerner et al, 1996), with the attendant risk of contamination of urban surface waters. Also, any improvements achieved in the quality of surface water, as a result of better control of industrial effluent discharges and industry closures, may be limited by the long-term release of pollutants from contaminated land to the underlying groundwater that subsequently discharges to urban river systems. Poor quality surface water will have a negative impact on the local ecology, on the potential for potable supply and on the amenity value of the river, both locally and perhaps for a considerable distance downstream.

Impetus for the study of the impact of contaminated land on baseflow and urban surface water quality is driven by several factors. These include UK legislation covering the regulatory assessment of liability with respect to local surface water receptors (Environmental Protection Act, 1990); and the new European Commission Water Framework Directive (Council of Europe, 2000). The latter provides for integrated catchment management of both groundwater and surface water. In addition, the processes occurring in the groundwater/surface-water interface (the hyporheic zone) merit study as they may reveal ‘natural attenuation’ of contaminants which could limit the impact of the contaminants on the surface-water quality (Environmental Protection Agency, 2000). The above considerations have provided the underlying rationale for the current research.

The quantification of contaminant fluxes from contaminated land to surface waters via the groundwater pathway is poorly understood and documented. Previous work has generally been on a local scale and has either addressed groundwater geochemical flux for limited determinands to simple rural catchments (Gburek et al.,1999), or has focussed upon the discharge of specific contaminant plumes to the surface water (Lorah et al.,1998). Contamination of urban groundwater and surface waters has also been examined on a regional scale (Ator et al., 1998, Lindsey et al.1998). However, little has been done to link local and regional investigations, or to quantify the groundwater contaminant flux to a river from an entire conurbation for a large suite of determinands. This provides a further rationale for the current research.

1.2 Aim and objectives

The overarching aim of this research is to investigate the impact of contaminated land and groundwater on urban surface-water quality.

Four main objectives have been identified to meet this aim:

1) to characterise and quantify the contribution of groundwater-derived contaminants to the surface water quality of an urban river at the subcatchment scale;

2) to investigate the physical and chemical processes controlling contaminant flux across the groundwater/surface water interface;

3) to investigate the processes controlling the temporal and spatial variations in groundwater flow and contaminant flux to the river; and

4) to develop suitable monitoring methods to quantify contaminant flux to the river.

1.3 Approach and thesis layout

The objectives were met via a case study approach involving field investigations supported by computer modelling. The study area comprises the River Tame as it flows through the industrial city of Birmingham, UK. The population of the city amounts to over one million and urbanisation covers an area of some 250 km^2. Research was focused on a 7.3 km river reach traversing an alluvial flood plain overlying a bedrock unconfined sandstone aquifer. This was supplemented by additional information on surface-water quality and flow collected 8 km upstream and downstream of the main reach. The study area was selected on the basis of previous work that had identified a wide range of organic and inorganic groundwater contamination within the sandstone aquifer that was thought to contribute to an increase in river baseflow across the study reach.

Foundation for the study was provided by a literature review of the previous research on groundwater/surface water quality and flow interactions. The research accessed was a combination of process/theory and case studies (Chapter 2). Extensive archive data on the regional hydrology and hydrogeology were examined to characterise the study setting and facilitate with the design of the fieldwork programme (Chapter 3). In order to estimate the geochemical mass flux from the aquifer to the river, information on the groundwater and surface water quality and flows in the Tame Valley were obtained from archive data and field investigations. Tools and methods were developed to measure groundwater flow and obtain water quality samples across the groundwater/surface-water interface. Fieldwork included the installation and monitoring of a river-bed-piezometer network combined with the monitoring of existing piezometers in the aquifer. An overview of the research undertaken is presented followed by a more detailed description of the methods employed (Chapter 4).

Surface water flow data were analysed to determine the groundwater component. These results were compared with groundwater flows estimated from data obtained during field investigations of the riverbed and riverbanks. Transient river-aquifer interactions during river flood events were also examined using a purpose built pressure logging system (Chapter 5). The results of the field investigations were used to develop a conceptual model of the groundwater flow system in the Tame Valley. This conceptual model was tested by using several numerical modelling tools to simulate groundwater/surface-water interactions at different scales (Chapter 6). The numerical models were used to investigate the distribution of groundwater flow to the river.

The levels of urban contamination in the groundwater and surface water were determined for a large suite of organic and inorganic determinands and compared with 'natural background' quality. This was measured at Sutton Park, the source area of one of the Tame's tributaries, located on the aquifer 5 km northwest of the study area. The urban surface water and groundwater quality distributions were examined to determine the spatial and temporal trends in the data. Changes in water quality that occurred across the groundwater/surface-water interface were examined for evidence of the natural attenuation of contaminants (Chapter 7). The quality and flow data were combined to produce estimates of the geochemical mass flux from the groundwater to the river (Chapter 8). The generic relevance of the conclusions drawn from the case study were considered with reference to the possible implications for the new

European Commission Water Framework Directive (Chapter 9). Recommendations for further research are made building on the insights gained from the work completed to date.

CHAPTER 2. A REVIEW OF GROUNDWATER/SURFACE WATER INTERACTIONS

2.1 Introduction

Groundwater and surface water are often in hydraulic continuity and form a single hydrological system. Previous practice, however, has often been to manage these resources in isolation. Clearly, an integrated approach is required to best manage what is effectively a single resource within many river-basin settings. The aim of this chapter is to describe the key processes that occur during groundwater/surface water interactions and to review previous case studies of this interaction. More detailed theory on specific processes will be introduced as necessary in the subsequent chapters.

2.1.1 Groundwater flow to rivers

The interaction of groundwater and surface water bodies is complex and dependent on many factors including the topography, geology, climate, and the position of the surface water body relative to the groundwater flow system (Sophocleous, 2002, Winter, 1999 and 2002). Local, intermediate and/or regional groundwater flows may discharge into surface waters, (Figure

2.1) and surface water may on occasion recharge the groundwater (Toth, 1963, Nield et al., 1994). Hence a reduction in flow or contamination in one will often affect the other.

Woessner (2000) states that groundwater flow to a river channel is dependent on:

1. the distribution and magnitude of the hydraulic conductivities within the river channel, the associated fluvial plain sediments and the underlying bedrock;
2. the relation of river stage to the adjacent groundwater gradients; and
3. the geometry and position of the river channel within the fluvial plain.

Groundwater is influent to the river channel when the groundwater head at the channel interface is greater than the river stage. Conversely, surface water is effluent from the channel when the river stage is higher than the groundwater head. Groundwater through-flow may occur when the groundwater head is higher than the river stage on one bank but lower on the other (Townley et al., 1992). Groundwater flow may also occur parallel to the river in which case only limited groundwater/surface water exchange may occur. During periods of high recharge, groundwater 'mounding', (ie, rapid increases in head) may occur in the thin unsaturated zone adjacent to surface-water bodies which may temporarily influence groundwater/surface water interactions (Winter, 1983).

A key reference by Winter et al. (1998) summarised the current understanding of groundwater/surface water interactions and their relationship to water supply, water quality and the aquatic environment, in a variety of settings. A unifying framework based on the concept of hydrologic landscapes is used to present conceptual models containing common

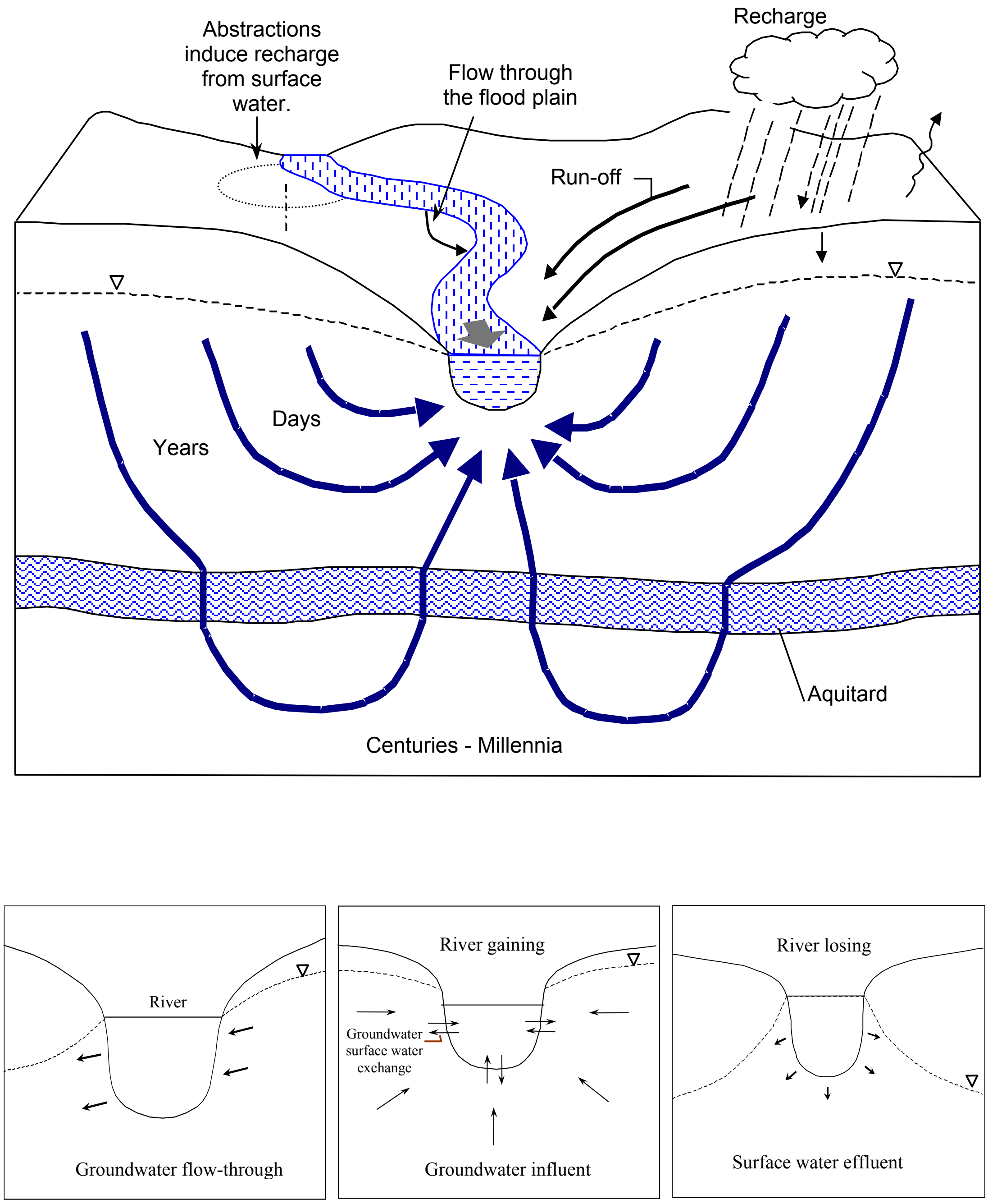


Figure 2.1 Conceptual model of groundwater flows to a river

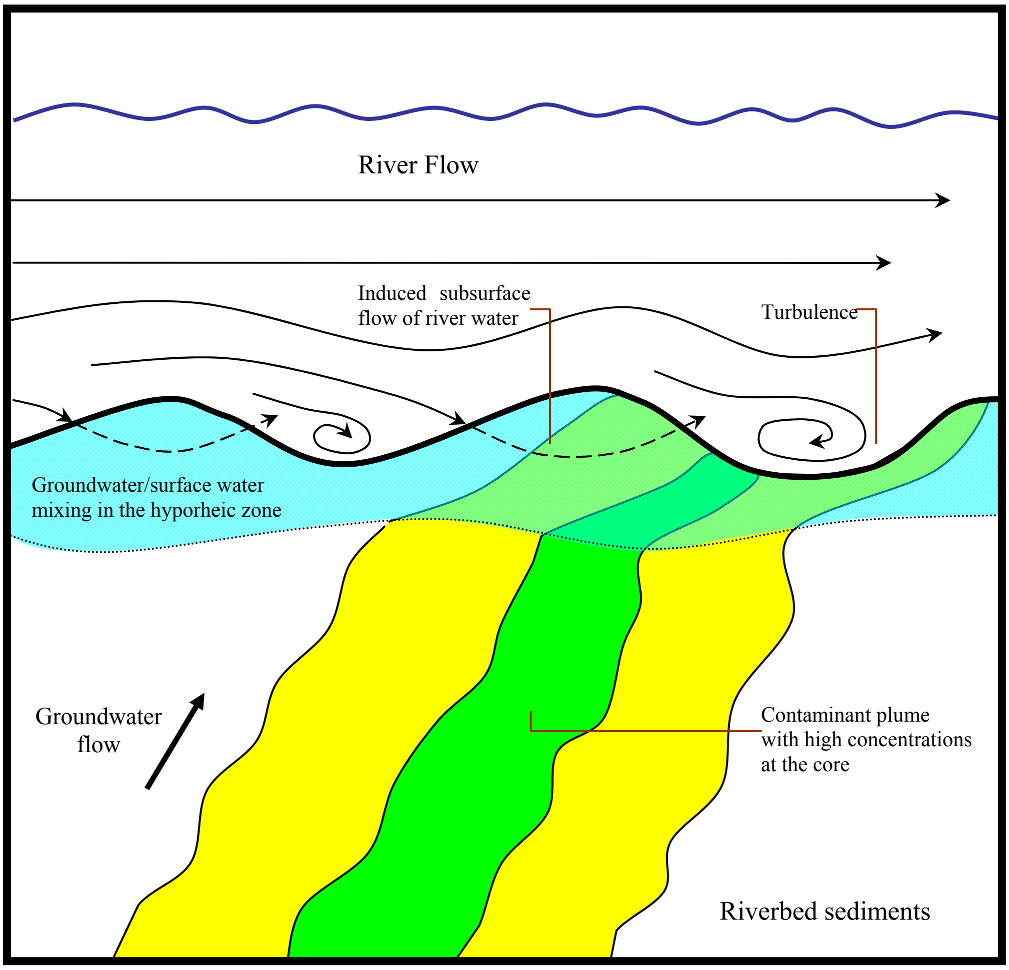
features of groundwater/surface water interactions in five general types of terrain: mountainous, riverine, coastal, glacial and dune, and karst.

2.1.2 The groundwater/surface water interface

Interactions between groundwater and surface water occur across a transition zone within the beds of lakes, rivers, or seas (Henry, 2002). In the case of a river, a 'hyporheic zone' develops where mixing between groundwater and surface water occurs (Biksey et al., 2001). This is an ecological term that refers to an ecotone where both groundwater and surface water are present within a stream bed along with a specific set of biota (Conant, 2000). The flow of river water over variations in the surface of the riverbed may cause localised variations in pressure that induce flow through the riverbed, causing groundwater/surface water mixing (Figure 2.2). The extent of this mixing zone may range from centimetres to hundreds of meters if surface water flows through the flood plain sediments are considered (Woessner, 2000, Wroblicky et al., 1998). The zone is heterogeneous, dynamic and dependent on the surface water and groundwater head distribution, river flow, riverbed hydrogeology and bedform (Fraser et al., 1998).

Large gradients in concentration and environmental conditions often exist across the transition zone (Boulton et al., 1998). These affect the spatial and temporal distribution of aerobic and anaerobic microbial processes as well as the chemical form and concentration of nutrients, trace metals and contaminants. Microbial and biological activity may lead to biodegradation of organic contaminants, reducing levels by several orders of magnitude within this zone (Conant, 2000). The hyporheic zone is important ecologically because it may store nutrients (and potentially contaminants), transform compounds biologically and chemically, and

(a)



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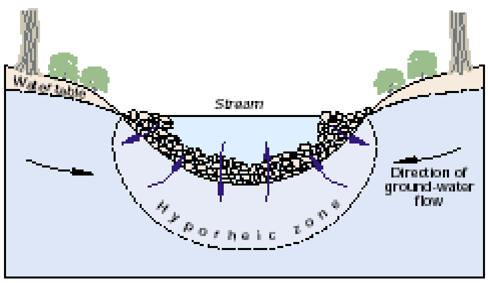


Figure 2.2 Schematic section through a riverbed, (a) longitudinal (b) lateral (after Winter et al., 1998).

provide refuge to benthic invertebrates that are the base of the aquatic food web (Battin, 1999, Barnard et al.,1994). Hyporheic organisms are likely to show the effects of pollution from discharging groundwater before organisms within the water column and so provide an indicator of the impact of contaminated groundwater (Environmental Protection Agency (US), 1998).

Surface water exchange and storage within the hyporheic zone influences downstream nutrient and contaminant transport, and may be associated with enhanced biogeochemical transformation of these compounds in the surface water. Hyporheic flow paths are typically small but if rates of chemical reactions are rapid enough and the volume of exchange great enough then substantial modifications of surface water quality may occur (Choi et al., 2000, Packman et al., 2000, Harvey et al., 1993).

2.1.3 Groundwater contamination

The natural background chemistry of groundwater resulting from recharge composition and mineral dissolution can be substantially modified by a wide range of contaminants that may be present as different phases within the subsurface environment. These include; synthetic organic compounds, hydrocarbons, metals and other inorganics, and pathogens such as viruses and bacteria. Conant (2000) summarised the factors that may help to determine the impact of a contaminant present in the subsurface on a surface water body. They include:

1. physical and chemical characteristics of the contaminants;
2. geometry and temporal variations in the contaminant source;
3. transport mechanisms (advection and dispersion);
4. reactions (reversible and non-reversible).

The physical and chemical properties of the contaminant determine its mobility and toxic effect. A contaminant may move through the subsurface as a pure liquid or gas phase, as a dissolved phase, in particulate form or attached to colloids. Soluble compounds may be transported readily within the groundwater, and attain high levels of concentration. Less soluble compounds will occur in low concentrations but may provide a long-term source of contamination. Advective transport of dissolved phase contaminants within the groundwater is seen as the primary mechanism by which subsurface contaminants may impact upon surface water systems.

The sources of contaminated groundwater may be spatially restricted point sources such as an industrial spill or waste dump, or more diffuse sources such as arise from the widespread application of agricultural fertilisers and pesticides. Point sources tend to give rise to narrow plumes (Rivett et al., 2001) which migrate with the groundwater flow and may eventually discharge to the surface water (Figure 2.3). In the USA more than 75% of the contaminated land categorised under the government's 'superfund' sites lie within 0.5 miles of a surface water body and more than half had an impact on surface water in some way (Environmental Protection Agency (U.S.), 2000).

The initial contaminant concentration in the groundwater will depend on the mass and distribution of the contaminant in the source area, the rate of groundwater flow and the physical-chemical-biological processes controlling contaminant dissolution (Fetter, 1999). Contaminants derived from the land surface may take a considerable time to enter the groundwater if a large unsaturated zone is present. Groundwater contaminant concentrations in the source area may vary with time or may give rise to discrete pulse-type inputs.

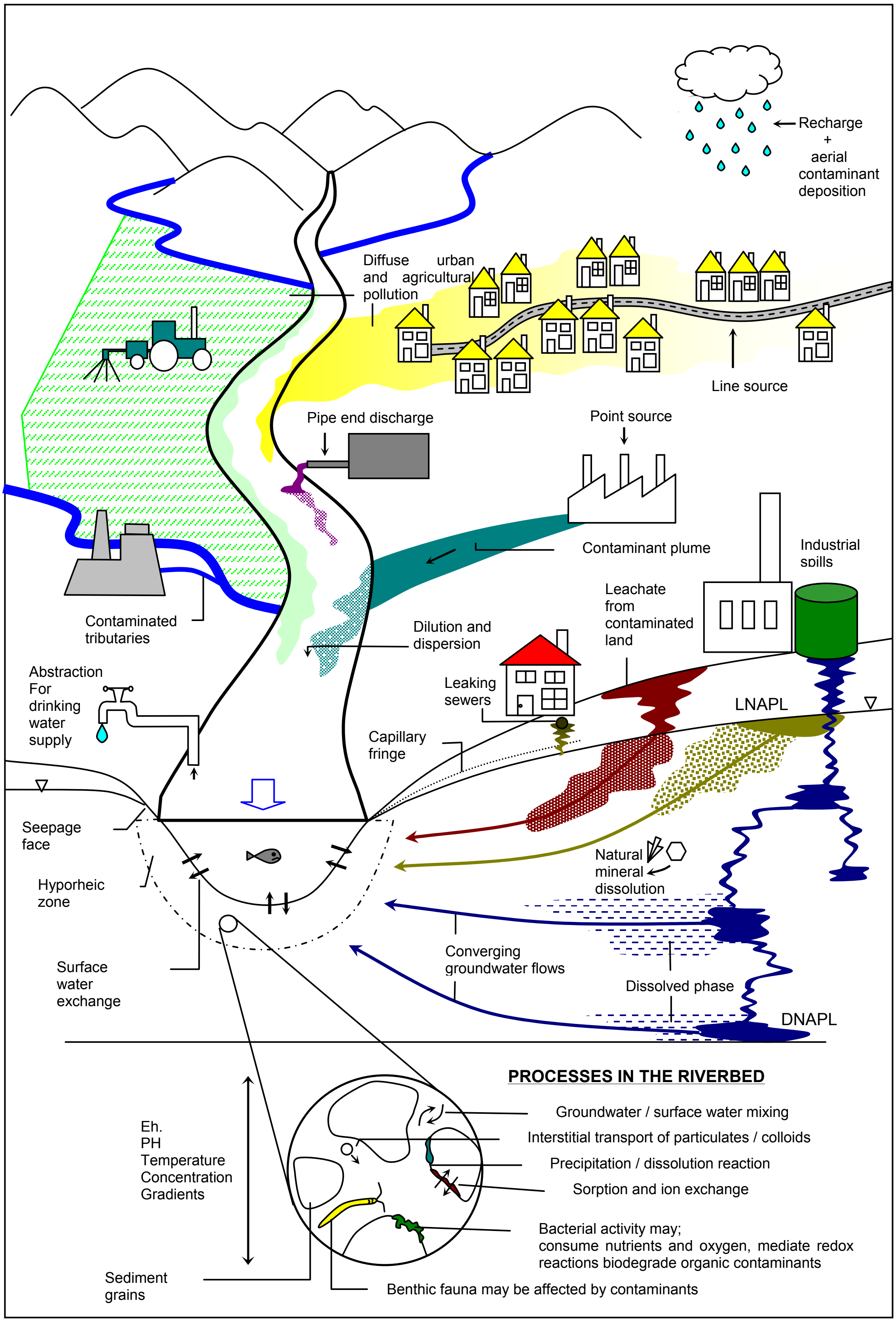


Figure 2.3 Conceptual model of contaminant inputs to a river.

Contaminant concentrations within the groundwater will be modified en route to, and across, the groundwater/surface water interface. Dispersion will result in spreading and mixing of the contaminant plume with cleaner groundwater. However, lateral dispersion within an aquifer is generally low and the plume remains narrow relative to its length, the highest contaminant concentrations being within the central core (Rivett et al., 2001). When the plume reaches the hyporheic zone, more turbulent conditions are likely to exist as the groundwater mixes with the surface water and is ultimately diluted in the surface-water column.

During transport, the contaminants may undergo reversible reactions such as adsorption, precipitation/dissolution and ion exchange, and non-reversible reactions such as biodegradation. Reactions may be reversible only under certain conditions. For example desorption of heavy metals occurs under conditions of low pH, and therefore contaminants may be effectively removed from the system until conditions change. The types of reactions that occur are dependent on the local conditions, and these may vary considerably along the contaminant flow path from the source area, through the aquifer to the groundwater/surface water interface. There are key processes controlling the movement and transport of contaminants across the groundwater/surface water interface.

1. Bacterial action which may play an important role in catalysing reactions (e.g. sulphate and nitrate reduction) and directly degrading some organic compounds. The groundwater/surface water interface often has a high nutrient content and anoxic conditions which are conducive to bacterial action.
2. Adsorption of contaminants to sites on the surrounding aquifer material. This reaction is generally reversible in which case it will not alter the total mass flux of the contaminants but it may significantly retard their transport allowing extra time for other processes to

occur such as biodegradation. For organic contaminants, the degree of sorption is often proportional to the content of organic carbon which is generally much higher in the riverbed sediments relative to the surrounding geology (Schwarzenbach et al., 1981). For inorganic contaminants, clay minerals, organic matter and oxides/hydroxides all have a sorption and exchange capacity which may retard contaminant transport (Appelo and Postma, 1999).

3. Rapid changes in pH, E_h, and mixing of waters of significantly differing concentrations occur across the groundwater/surface water interface. Groundwater chemistry which may have been in equilibrium will adjust rapidly to the new conditions, perhaps leading to sudden mineral precipitation. Iron oxides are a common example of precipitation occuring when acidic, oxygen-poor groundwater mixes with higher pH, more highly oxygenated surface waters.

Contaminant movement and transformation across the groundwater/surface water interface are poorly understood. In light of this a workshop was held to summarise the existing knowledge base on the groundwater/surface water interface and to develop strategies for an improved understanding of the effect of contaminated groundwater discharge through it. The proceedings of this workshop provide a key reference on this subject area (Environmental Protection Agency (U.S.), 2000).

It is predicted that by 2010, half the world's population of 6,500 million will live in towns or cities, and that much of the urban growth will be in developing countries (Morris, 2002). Increasing demand will be placed on both urban groundwater and surface water resources for domestic and industrial use. Unfortunately, urban growth is often associated with degradation of water quality which limits the usefulness of both groundwater and surface water resources (Petts et al., 2002). The occurrence of contaminated groundwater related to urbanisation is well known, both in the U.K. (Barrett et al., 1997, Lerner et al., 1996, Ford et al., 1994, Rivett et al., 1990) and abroad (Eiswirth et al., 1997, Kacaroglu et al., 1997, Appleyard, 1995). Likewise, the occurrence of poor quality urban surface waters is extremely common (Kayabalý et al., 1999, Ziegler et al., 2001). However, urban groundwater and surface water interactions are less well understood.

2.2 International Case Studies

To characterise the impact of urban groundwater on surface-water quality requires a broad multidisciplinary approach involving hydrological, hydrogeological, hydrochemical and ecological elements. To cover all these elements is generally beyond the scope of a single study and previous work has often focussed on a particular groundwater and/or surface water flow and/or quality issue. The scale of the studies can be divided into those that provide detailed but localised information on a specific contaminant plume or process across the river bed, and those that provide more general regional information on surface-water quality and flow. Only limited work has been done on scaling up the effects of local processes to the catchment scale. The urban groundwater system generally lies between these two extremes of scale, having many potential contaminant sources but covering a limited extent of the total catchment area.

2.2.1 Water quality studies

Relationships between groundwater and surface-water quality have been investigated for nutrients, pesticides , volatile organic compounds and metals (Lindsey et al., 1998, Bevans et al., 1998). A major study of the surface water and groundwater quality in the Potomac River Basin, USA, revealed a substantial impact from anthropogenic sources of contamination related to urbanisation (Ator et al., 1998). Chlorodane, DDT, PCBs, mercury and lead were all detected in streambed sediment and aquatic tissues derived from persistent sources, such as groundwater, within the river basin. Grischek et al. (1996) investigated urban groundwater quality beneath the city of Dresden in Germany and found the influent surface water from the River Elbe to have only a limited effect. Hartwell (1997) investigated the transport of the herbicide atrazine between the Cedar River and groundwater in Iowa, USA. The study found that groundwater from the adjacent alluvial aquifer was the principal source of atrazine in the river during base-flow conditions.

Lorah et al. (1999) and Conant (2000) investigated the discharge of chlorinated solvent plumes to surface water systems and found evidence for enhanced biodegradation across the interface. Schwarzenbach et al. (1981) conducted some of the earliest work on groundwater/surface water contaminant interactions based on laboratory simulations of field conditions beneath a river valley in Switzerland. They conducted a ‘classic’ set of sorption studies and showed sorption to organic carbon to be a key process in retarding the transport of non-polar organic compounds across the groundwater/surface water interface.

Metal contamination resulting from acidic groundwater associated with mining activities has been widely researched (Benner et al., 1995, Kimball et al., 2002). Paulson (1997) investigated the transport and fate of Fe, Mn, Cu, Zn, Cd, Pb and SO_4 in a groundwater plume and in the downstream surface waters in the Coeur d'Alene Mining District in the US. He found that upon mixing with higher pH surface water the metals were lost from solution in the following order: Fe>Al>Pb>Cu>Mn>Zn> Cd. Less than 10% of the dissolved Zn and Cd were lost, despite a 5 km journey through both the groundwater and surface water regimes. Paulson and Balistrieri (1999) used model simulations and laboratory experiments to examine the removal of Cd, Cu, Pb and Zn in acidic groundwater during neutralisation by ambient surface waters. They concluded that hydrous Fe oxides and particulate organic carbon are more important than hydrous Al oxides in removing metals from the groundwater. Tessier et al. (1996) found that trace metals present in water influent to two lakes in Canada sorbed directly to the OH functional groups of iron and manganese oxyhroxides and organic matter found in the lake sediments. Kuwabara et al. (1999) found that water flowing through the bed of lake Coeur d'Alene caused a significant flux of metal from the lake sediments to the water column.

2.2.2 Flow Studies

The groundwater support of river baseflow and its relationship to the underlying geology has has long been recognised and studied (Meyboom, 1961, Mau et al., 1993, Pinder et al., 1969, O'Conner 1976,). Several studies have investigated the relationship between reduced surface water flows and increased levels of groundwater abstraction (Sophocleous 2000). A comprehensive study on the interaction of the Equus Beds alluvial aquifer and the Arkansas River, USA, found that abstraction for irrigation significantly reduced river baseflow levels and induced surface water seepage into the aquifer (Ziegler et al., 2001). The response of groundwater to transient surface water levels has been used to derive estimates of the aquifer property S/T (Reynolds, 1987, Workman et al., 1997, Erskine 1991). Devito et al. (1996)

investigated groundwater/surface water interactions and the groundwater contribution to wetlands on the Canadian Shield and demonstrated the control of morphology and shallow subsurface geology on the hydrology of valley bottom swamps. Wroblicky et al. (1998) investigated the groundwater flow and seasonal variations in the hyporheic zone for two first order mountain streams in the US. The streams were in alluvial material with different hydraulic properties derived from bedrock types comprising welded tuff and sandstone. Numerical modelling was used to simulate unconfined transient flow. Sensitivity analyses indicated that changes in the hydraulic conductivity of the alluvial and streambed sediments and variation in recharge rates have greatest impact on the magnitude, direction, and spatial distribution of stream/groundwater exchange.

2.2.3 Combined Quality and Flow Studies

Case studies of groundwater/surface water quality and flow interactions have generally been conducted in a rural setting (Cey et al., 1998, Pionke et al., 1998) in order to avoid the complexities associated with urban rivers which have many inputs (Ellis et al., 2002, Appendix 1). Gburek et al. (1999) investigated the flow and major ion contributions to baseflow in an upland watershed in Canada. The study showed increased ionic concentrations, including nitrate, within groundwater baseflow derived from an area of agricultural land compared with low ionic concentrations from forested areas. A chemical mass balance (i.e concentration x flow) was used to investigate the contribution of groundwater from different parts of the study area. A simple model was developed based on land use to explain nitrate concentrations within baseflow.

Diffuse nitrate pollution of surface waters from groundwater has been investigated in several studies in the USA (MacNish et al., 1998). (Bachman et al., 1997) used river baseflow analyses and surface water quality sampling to calculate nitrate loading to Chesapeake Bay.

The median groundwater contribution was estimated as 56% of the total nitrate load under baseflow. Harvey et al. (1998) examined the effect of enhanced maganese oxidation (and co-precipitation of trace metals) in the hyporheic zone on basin-scale geochemical mass balance in a drainage basin contaminated by copper mining.

2.3 UK Case Studies

Published data on groundwater quality and flow interactions in the UK are generally limited. A few of the key studies are briefly presented.

A study of trace-element concentrations in the major rivers entering the Humber estuary showed elevated levels in the rivers draining industrial urban catchments (Neal et al., 1996). The relationship between concentration and flow was used to evaluate the sources of trace elements; a negative relationship to flow implied a groundwater source. Within the urban rivers, most elements in the dissolved phase exhibited negative relationships, indicating that point-source discharges provided the major source of trace elements. Roberts et al. (1998) investigated nitrate contamination of groundwater in the Lincolnshire Limestone resulting from surface water recharge related to local groundwater abstractions.

Of particular concern in the UK has been the widespread detection of pesticides at low levels within groundwaters and surface waters (Croll, 1991). Clark et al. (1991) investigated pesticide occurrence in the River Granta and the underlying groundwater within the chalk aquifer of Cambridgeshire. The maximum levels of pesticide were found in the river during periods of high flow. However, even under low-flow conditions, levels of pesticide were persistent and are probably derived from the groundwater. Of particular interest in the study, owing to their widespread detection and/or application, were the pesticides atrazine, simazine, isoproturon, chlortoluron and tri-allate.

Hooda et al. (1997) found the impact of diffuse phosphorous loading on six small catchments in Scotland to be greatest during summer groundwater-fed baseflow conditions. Birtles, (1978) modelled water quality in the River Severn by separating the river hydrographs into discrete components each of which had separate, conservative, surface water quality parameters. These comprised, direct run-off, baseflow from the Triassic Sandstone deposits, baseflow from superficial deposits and effluent returns. Younger et al. (1993) investigated groundwater/surface water interactions in the Thames basin.

Research currently in progress in the UK is summarised by Young et al. (2002). New research initiatives are under way to provide support for integrated groundwater/surface water monitoring and assessment for sustainable catchment management. These include the Lowland Catchment Research Programme, LOCAR, (Peach et al., 2000, LOCAR, 2002) and the Catchment Hydrology and Sustainable Management (CHASM) group which will investigate highland catchments (CHASM, 2002). The recently concluded Land-Ocean Interaction Study (LOIS, 1998) was undertaken to quantify the chemical fluxes entering the North Sea from the surrounding countries, including the UK. Extensive data sets were collected for the Humber catchment which incorporate the groundwater contribution to the total geochemical flux to the sea. Most of the current research is focussed on rural catchments and there is a lack of ongoing research into urban groundwater/surface water interactions.

2.3.1 UK Legislation

The new Water Framework Directive (Council of Europe, 2000, Environment Agency, 2002) has been introduced to ensure an integrated approach to the management of catchment surface and groundwater quality and flow. Consultations are under way in the UK on the implementation of this directive (DEFRA, 2002). It will be implemented in conjunction with other legislation such as Part IIa of the Environmental Protection Act 1990 designed to maintain and improve the quality of surface and groundwater. Under the legislation, the impact of land causing pollution of controlled waters must be assessed by comparison to the background water quality and suitable standards such as drinking water and environmental quality standards (EQS) for surface waters (Smith, 2002). Contaminated land and contaminated groundwaters arising, where found to cause significant impacts on the surface water are likely to require remedial action (Rivett et al., 2002).

2.4 Monitoring methods

Comprehensive reviews of the existing analytical, numerical, field and chemical investigative techniques for groundwater/surface water interactions have been carried out by HRU (2001) and Winter (1995). On the catchment scale, the most widely-used approach is river hydrograph separation and baseflow recession analyses to derive the groundwater component of river flow (Meyboom, 1961, Mau et al., 1997, Gustard et al., 1992). Several workers (Pinder et al., 1969, O’Conner 1976, Birtles 1978) have used differences in groundwater and surface water chemistry and total dissolved solids to resolve further the groundwater component of the river hydrographs. At the smaller scale of the river reach, stream tracers have been used to determine river discharge accretion and characterise river subsurface water exchange (Harvey et al., 1996, Kimball, 2002).

Piezometers and piezometer nests have been used in several studies to obtain water samples and head measurements within the groundwater/surface water interface (Dean et al., 1999, Cey et al., 1998, Lee, 1980, Henry, 2000). Other in situ sampling devices have also been developed such as the colonisation corer (Fraser et al., 1996) and the bead tube sampler (Moore, 2000). Passive sampling techniques based on diffusive gradients across semi-permeable membrane devices have been used to detect both hydrophobic organics and heavy metals in aquatic environments (Vroblesky et al., 1997; Church et al., 2000). A number of devices have been developed to measure the seepage across a lake or riverbed (Lee, 1977; Carr et al., 1980; Isiorho, 1999) although the methods often encounter errors when used in fast-flowing river environments (Libelo et al., 1994). The temperature gradient across the groundwater/surface water interface has been used to determine flow rates (Silliman et al., 1995, Evans et al. 1997). Several of the investigative methods identified in the literature review were adapted for use in the present study.

2.5 Modelling methods

A review of analytical and numerical methods of modelling river-aquifer interactions are presented by HRU (2001), Winter (1995) and Younger (1989). Analytical solutions have been developed from early simulations of 1D groundwater interaction with fully penetrating streams (Rorabaugh, 1964) to meet a range of applications. These include the simulation of bank storage and groundwater fluctuations during river flood events (Serrano and Workman, 1998, Reynolds, 1987, Hunt, 1990), travel times for surface water into contiguous aquifers (Heij, 1989) and the effects of pumping groundwater on stream flow (Spalding et al., 1991).

The onset of readily available computing power has led to an increase in the use of numerical models to simulate groundwater/surface water interactions. These have been successfully applied to numerous field studies for different hydrogeologic settings and scales (Modica, 1993, Jorgensen et al., 1989, Younger, 1989, Rushton et al., 1989, Sophocleous, 1991). Most models use idealised stream geometry and the Darcy equation to simulate transfer of water across the streambed sediments. In this case groundwater discharge = hydraulic conductivity x hydraulic gradient x cross sectional area. This may often be an oversimplification of the natural system. Previous workers (Rushton et al., 1979) indicate that the relationship between discharge and hydraulic gradient is not always linear, with variations occurring in the hydraulic conductivity, particularly if flow reversal occurs. Near surface water bodies, convergent flow with a considerable vertical component often occurs which must be accounted for if hydraulic conductivity is anisotropic.

The realistic simulation of transient surface water flows and groundwater flows across a region is complex. Surface water and groundwater flow equations may be coupled and solved simultaneously by iteration. However, more commonly the surface water heads are set (from field data or separate modelling) or river baseflow is determined as output from a groundwater model (Younger, 1989). The widely used groundwater flow model MODFLOW (Macdonald and Harbaugh, 1988) has been adapted to incorporate a 1D simulation of unsteady flow in open-channel networks, but still utilises the simple Darcy equation for water transfer. Other models have been developed to investigate groundwater/surface water interaction for variably saturated groundwater systems for losing streams (Riesenauer, 1963) and under conditions of non-uniform recharge adjacent to a surface water body (Winter, 1983, Cooley 1983). Models have been used to evaluate the effect of groundwater pumping on stream flow for water resources and environmental impact purposes (Sophocleous, 2000, Eberts and Blair, 1990). Flow and transport modelling has been used to investigate solute (contaminant) transfer between groundwater and surface water on the regional scale (Duffy and Lee, 1992, Jakeman et al., 1989). Geochemical models such as the USGS model PHREEQC have been used to simulate solute fate and transport across the groundwater/surface water interface (Van Breukelen et al., 1998).

2.6 Conclusions

Groundwater/surface water quality and flow interactions are complex and variable between and within catchments. Further work is required to better understand the link between quality and flow processes, particularly in an urban setting where research to date has been limited. The groundwater/surface water interface is ecologically important and is a major pathway for mass transfer between the subsurface and surface. Contaminant and nutrient concentrations within both the groundwater and surface water may be altered significantly during their passage through the groundwater/surface water interface. The interface often shows high environmental and concentration gradients. Significant processes that occur across this zone that may affect contaminant fate and transport include; sorption and precipitation reactions and bacterial activity which may mediate chemical reactions and degrade organic contaminants. The key references identified in the review of groundwater-surface interactions were, Sophocleous (2002), Winter et al. (1998), EPA (2000), and Woessner (2000). Useful case studies of groundwater quality and flow interactions were Cey et al. (1998) and Gburek et al. (1999).

CHAPTER 3. STUDY SETTING

The River Tame drains a 408 km^2 highly urbanised catchment covering part of the West Midlands conurbation, which currently supports a population in excess of 1.8 million, and includes Birmingham the UK's second largest city. The Tame forms part of the larger drainage system of the River Trent, which eventually drains to the North Sea. It rises to the east of a ridge that forms part of the major watershed of central England with the River Stour in the Severn catchment to the west draining to the Bristol Channel. The three upper arms of the Tame (Figure 3.1) rise in the 'Black Country' towns of Wolverhampton, Walsall and Oldbury before joining to flow eastwards through the northern part of Birmingham. The river baseflow is supported by contributions from sewage effluent and other industrial discharges plus a significant contribution from groundwater which may increase the flow by >20% across the research area. The 24 km study reach lies between the gauging stations of Bescot and Water Orton and work has been focused on the central 7.4 km section that overlies the unconfined Birmingham Aquifer.

3.1 Historical Background

The West Midlands conurbation has been a population centre for the past 200 years and a major centre for heavy industry since the industrial revolution. Rapid development of the area occurred during the industrial revolution based on the exploitation and processing of raw materials from the South Staffordshire Coal Field and local ironstone deposits. A series of canals (from 1800) and later railways (from 1840) were constructed connecting various

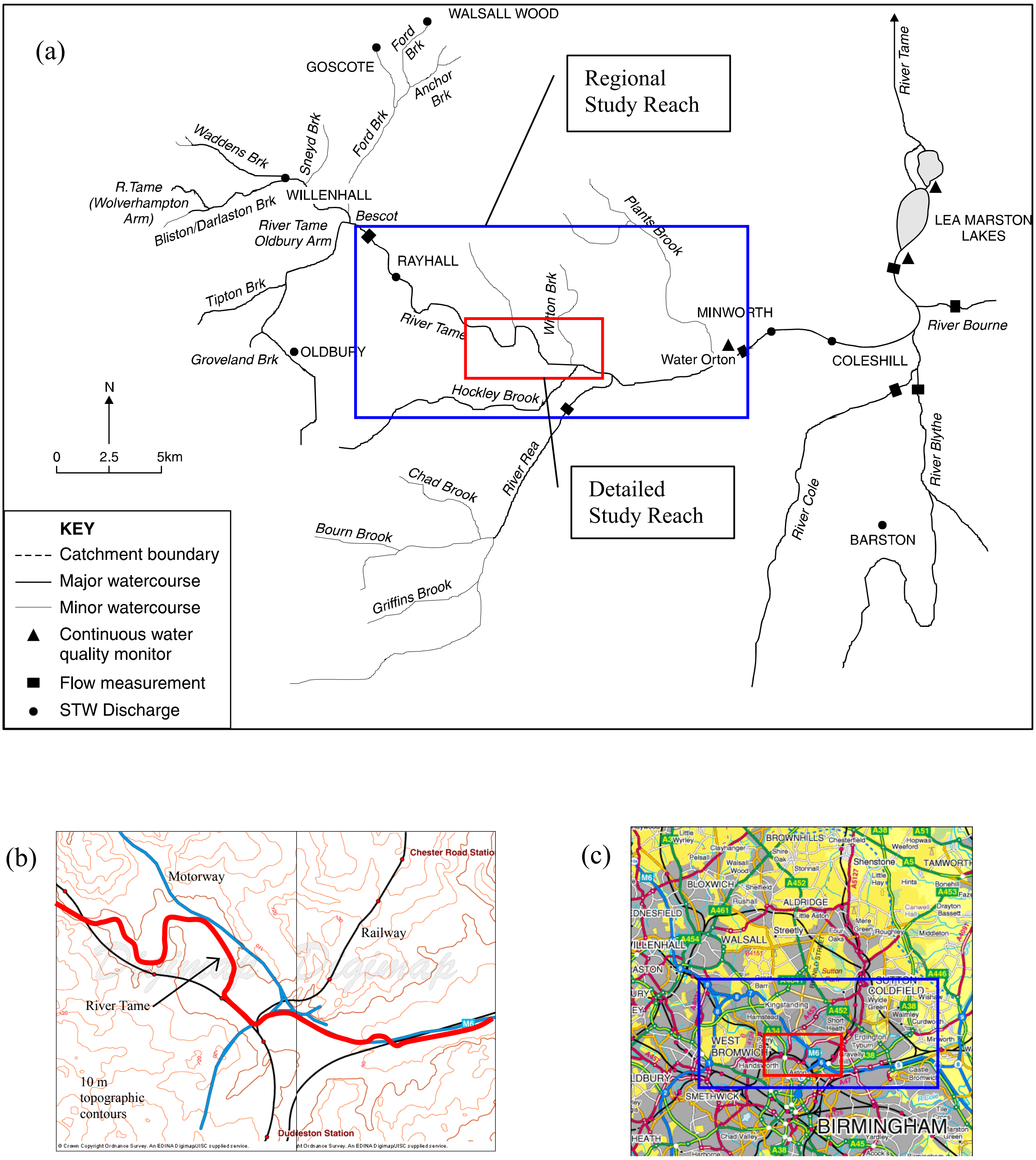


Figure 3.1 (a) Catchment map of the Upper Tame (b) Topography of the detailed study reach (c) Regional Setting

sections of the city with the rest of the country and facilitating the rapid growth and spread of urbanisation. The sparsely populated semi-rural area was transformed into a series of gradually expanding towns that have eventually coalesced to form the West Midlands conurbation. The region became a major centre for metal working and manufacturing industries with productivity reaching maximum levels during the two world wars. Because of this development, a large demand was placed on both surface water and groundwater resources for industrial and public supply.

Contamination of soils, groundwater and surface water occurred as a result of the urbanisation with little or no legislation in place to control it. This pollution combined with sewage effluent and industrial discharges began to seriously effect river quality during the 19th Century. By 1870 Birmingham City had major problems with the disposal of sewage effluent and the continued deterioration of water quality in the Tame prompted the construction of the Elan Valley reservoirs in Mid Wales to achieve an adequate public supply. This had the effect of increasing flow in the Tame which received the increased levels of sewage effluent. Because of overloading of the sewerage system untreated effluents often entered the river and by 1918 the Tame supported no fish life and by 1945 virtually no aquatic life (Clay, 2000). Today, the Tame, which once supported salmon and trout has no sustainable fish population in the study area (Crabtree et al., 1998). Groundwater quality in the Birmingham Aquifer also showed a substantial deterioration with many public supply wells polluted and abandoned by 1920 (Greswell et al., 2000).

Since the 1950s' modification of the sewerage system, a reduction in heavy industry and the implementation of better pollution control has helped to improve the surface water quality.

But under the General Quality Assessment (Environment Agency, 1998) the Tame is still categorised within the worst classes E/F (poor/bad) with severe impacts reported on the river ecology. Rapid urban run-off from precipitation events can generate high surface water flows and large pollutant loads that add to the hostile conditions for aquatic life.

The long history of industrialisation and urbanisation has left a legacy of contaminated land, which has caused significant inorganic and more recently organic contamination of the underlying groundwater. The contribution of sewage effluent and industrial pipe end discharges to the surface water quality and flow are relatively well understood but little is known about the contribution of groundwater across the study reach.

3.2 Land use

Heavy industry and housing grew up side by side with little or no legislation to control contamination of soils, groundwater and surface water.

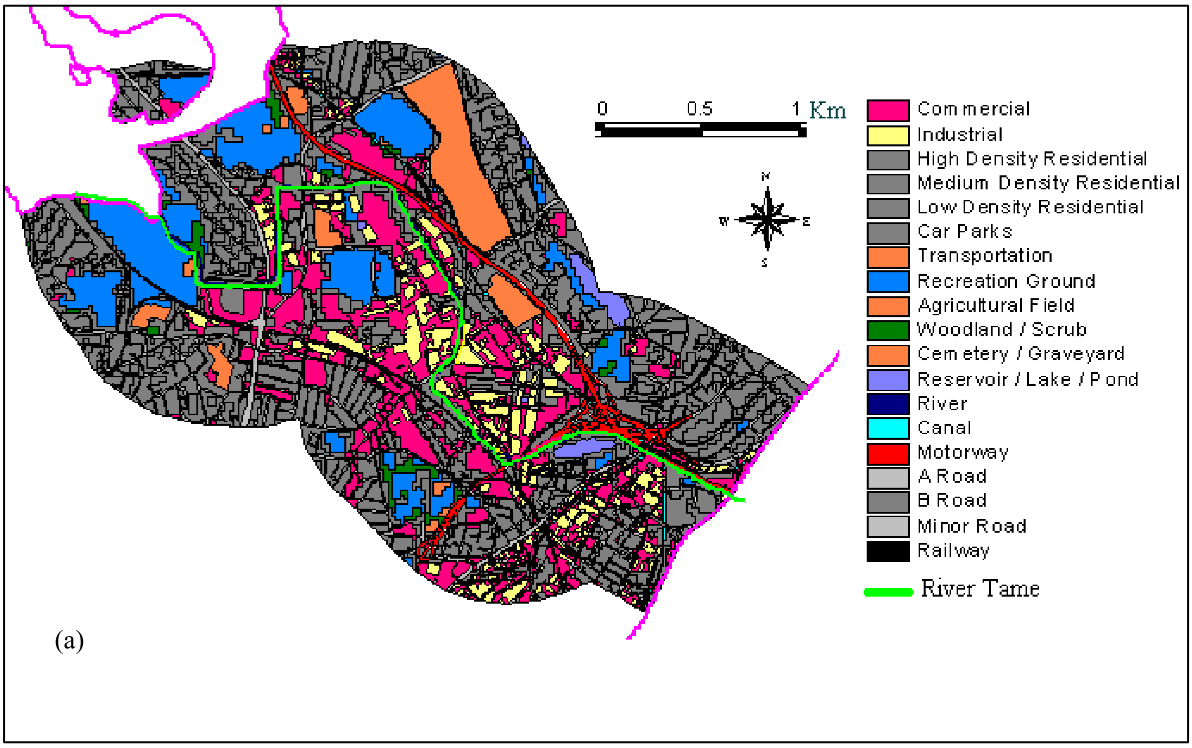
Studies of urban groundwater pollution throughout the UK (Lerner et al.,1996) and in nearby Wolverhampton (Bridge et al., 1997) identified associations between particular contaminants and land use. Sampling of the Birmingham Aquifer has identified groundwater contamination including heavy metals and chlorinated solvents associated with industrial land use (Rivett et al., 1990, Ford et al.,1994). Metal working and related industries which constitute 59% of the industry overlying the Birmingham Aquifer (Rivett et al., 1990.) are thought to be among the most significant polluters.

A high concentration of industry is located in the Tame Valley which is particularly vulnerable to groundwater pollution owing to the limited thickness of the drift and the shallow water table. Recent studies (Thomas, 2001) of land use on the unconfined Birmingham

Aquifer show industry to comprise 19% of land use within the Tame Valley adjacent (<1 km) to the river (Figure 3.2). Residential areas cover 40% of the area and may give rise to groundwater pollution due to leakage from the attendant sewers and water mains. Household gardens may show heavy metal accumulation due to the disposal of fossil fuel residues, household refuse, fragments of paint and atmospheric fallout. The application of garden fertilizers and pesticides may also be detrimental to groundwater quality. Roads are the other major land category (25%) and have many associated contaminants including Pb from vehicle emissions, spills of petroleum BTEX compounds and Na and Cl from road de-icing salt. A more detailed study of the land use and the associated contamination across Birmingham may be found in Shepherd (2002).

3.3 Hydrology of the River Tame

The River Tame is typically 8-12 m wide, 0.2-2 m deep with average dry weather flow velocities of 0.1-0.8 ms^{-1}. The river more than doubles its mean discharge across the study reach between Bescot, 182 mega litres per day (Mld^{-1}) and Water Orton 397 Mld^{-1}. Given the occurrence of rapid urban run-off, flood events are relatively common throughout the year. These are generally short lived (1-2 days) but may attain discharges of up to 70 m^3s^{-1} (6048 Mld^{-1}) at Water Orton. On average there may be 10 major flood events per year lasting 24 hours each and attaining discharge levels of >60 m^3s^{-1} (5184 Mld^{-1}) at Water Orton (J.West, 2000). Gently undulating terrain surrounds the river as it flows down a relatively shallow gradient which remains fairly constant along the length of the reach at 0.0013. The highest elevation of 283 m above



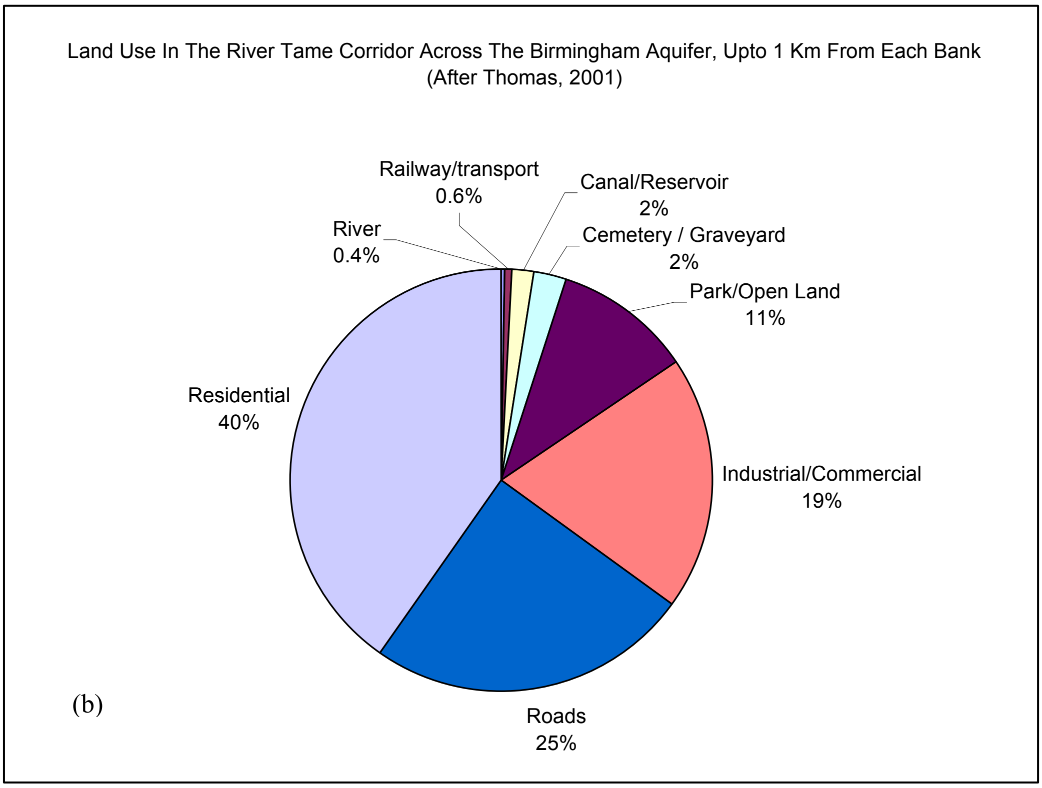


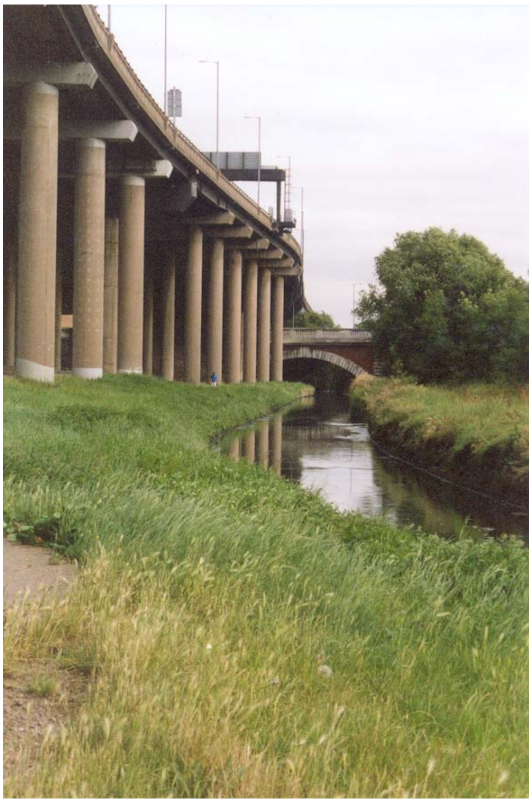
Figure 3.2 (a) Land use map within 1 km of the River Tame, (b) Categorisation of land use

Ordnance Datum (maod) occurs to the south west at Romsley Hill and the lowest point of 70 maod is at Kingsbury, north of the Lea Marston Lakes (Figure 3.1).

The river has been extensively modified from its original form as a meandering braided river system on a broad flood plain. More than 50% of the study reach has undergone engineering works for flood defence purposes, including strengthening bank sides and straightening some sections of the river. The channel bottom remains natural and unlined over most of its length, apart from some concrete-lined sections beneath the M6 motorway (Figure 3.3). Some of the tributaries have been brick lined (River Rea) or culverted (Hockley Brook). Bed materials range from sub angular cobbles through gravel, sand and silt and include many artefacts. Weed growth within the channel is limited during the winter months but is abundant during the summer and is thought to have an effect on the hydrological regime by reducing flow velocities (Clay, 1999).

Aside from groundwater discharge, sewage effluent is a major component of dry weather flow in the river, with 55% of flow at the Lea Marston Lakes (Figure 3.1) attributed to sewage discharge. Efforts to improve the surface water quality have led to the closure of some of the smaller sewage treatment works (STWs) and the centralisation of treatment at larger plants. Several STWs discharge upstream of the study area and one within it, at Rayhall (Figure 3.1a), which is intended to operate at a constant discharge of 30 Mld^{-1}. Sewage in excess of this capacity is transported from Rayhall to Minworth STW, by the recently constructed Black Country Trunk sewer, and discharged downstream of Water Orton. The STWs are a major control on the surface water flow balance and the water quality within the river and introduce a significant temporal variability to both. The daily fluctuation in surface water flows related

(a)



(b)

Figure 3.3 (a) Photograph of the River Tame by the M6 Motorway, (b) Installation of riverbed piezometers.

to the STW takes ~6 hours to propagate from Bescot to Water Orton. A rhodamine dye tracer experiment (Clay, 2000) undertaken along the study reach showed an overall travel time of 20 hours (0.33 $msec^{-1}$) for the dye between Bescot and Water Orton.

3.4 The Geology of the Tame Catchment

The River Tame rises in the west on the Carboniferous Middle Coal Measures and flows eastwards across the Silurian Wenlock Shale and the Upper Coal Measures before reaching an unconformable contact with the Triassic Sandstone of the Birmingham Aquifer. The Tame continues to flow eastwards for 7.3 km over the sandstone (Figures 3.4 and 3.5) crossing the Birmingham fault onto the down-thrown Mercia Mudstone which form the eastern division of the study area. Outcrops are rare (Figure 3.6) and the bedrock is generally covered by superficial drift deposits of glacial and alluvial origin (Figure 3.7) ranging in thickness from 1 to 40 m. The principal focus of the study is related to the interaction of the river with the underlying Triassic Sandstone of the Birmingham Aquifer.

Information on the solid and drift geology was derived from recent geological maps and memoirs (Powell et al.,2000) of the British Geological Survey and a study of rising groundwater levels within the Birmingham Aquifer (Knipe et al.,1993). These data summarise a considerable body of earlier work combined with more recent mapping. Owing to the extensive urban land cover and the lack of outcrop in the study area, the geology (Table 3.1) is based in large part on the interpretation of water-well borehole logs and site investigation reports.

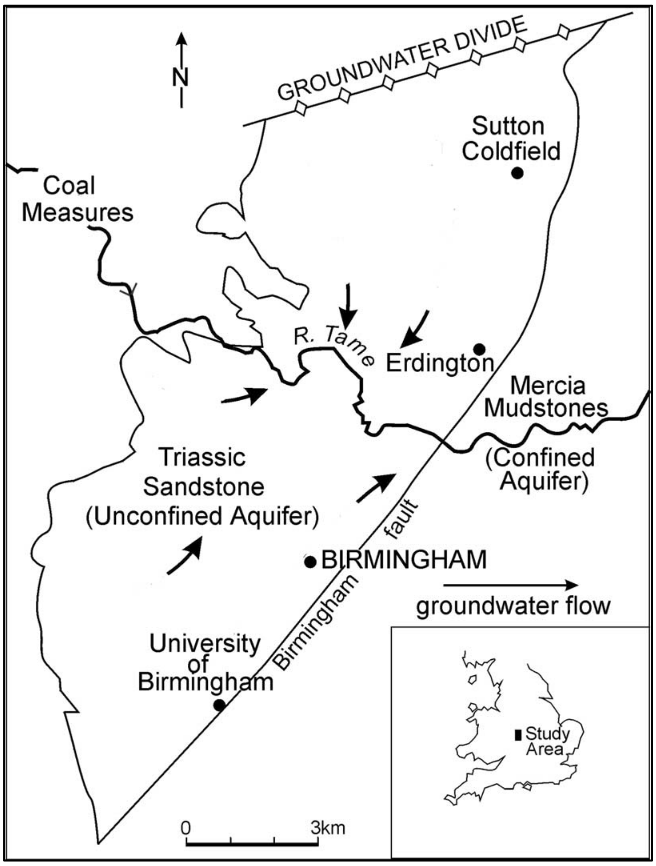


Figure 3.4 Schematic geology of the study area

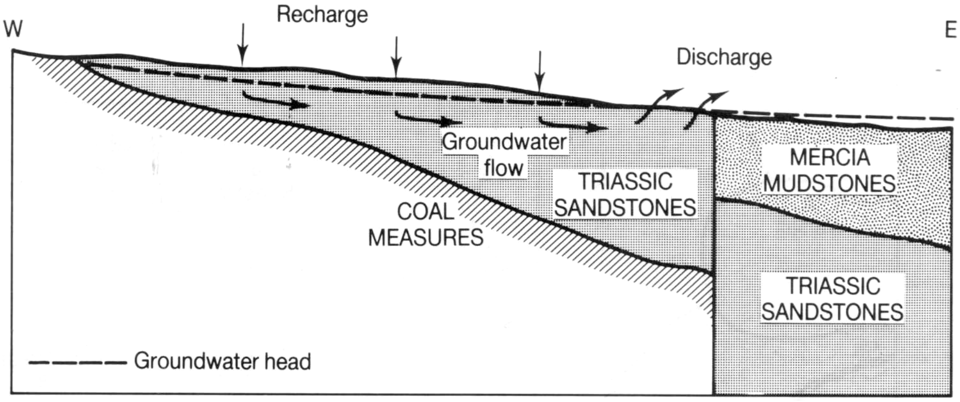


Figure 3.5 Schematic geological cross of the study area (Jackson et al., 1983)

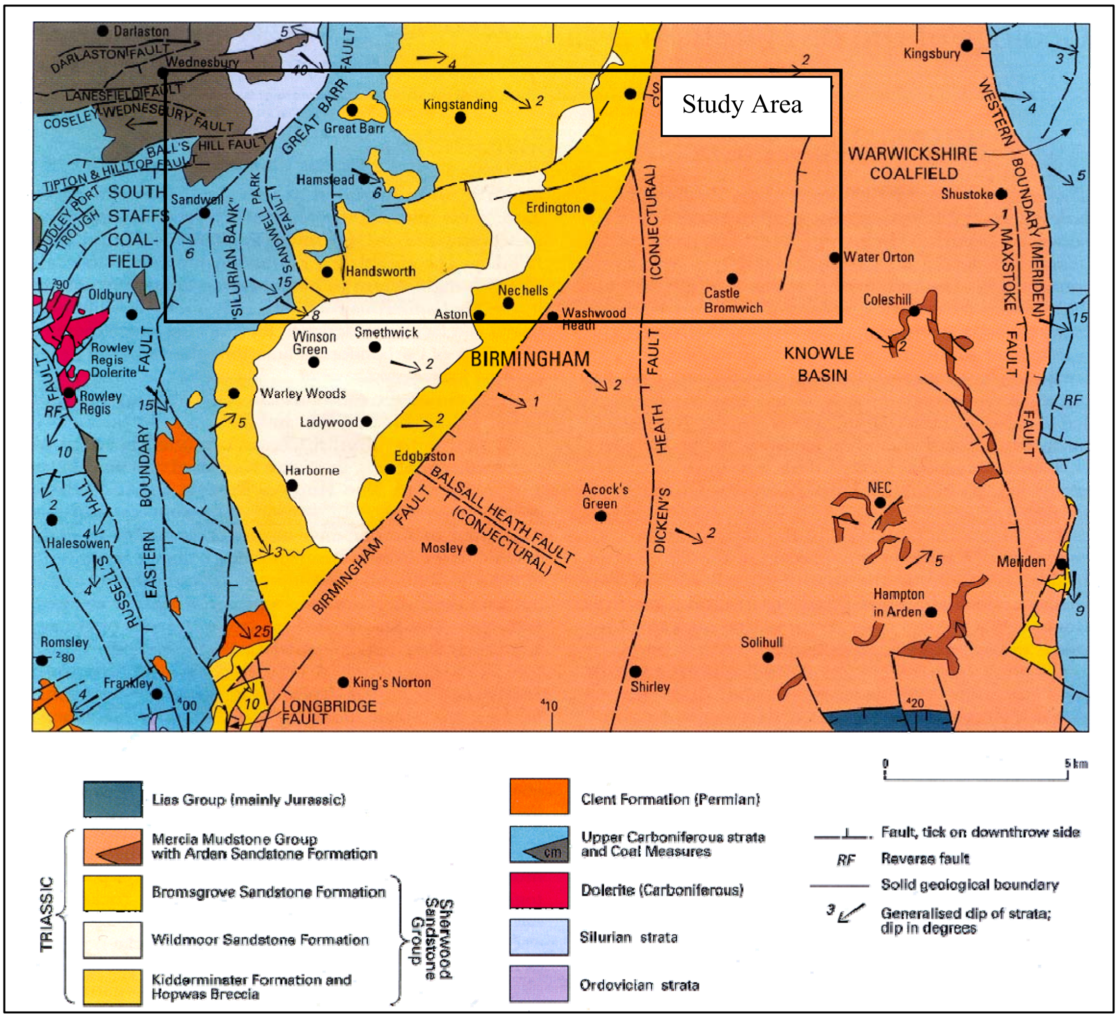


Figure 3.6 Solid geology of the Tame catchment (Powell et al., 2000)

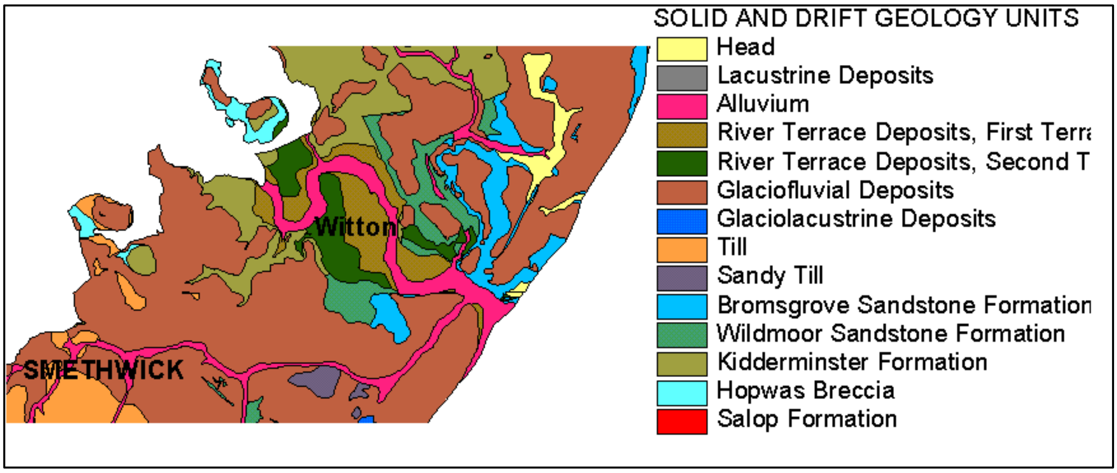


Figure 3.7 Drift and solid geology across the unconfined Birmingham Aquifer (Thomas, 2001)

Table 3.1 Description of geological units

Age and Unit	Description	Thickness (m)	Distance Downstream from Bescot (km)
RECENT			
Made Ground	Comprising mine spoil, sand and gravel mixtures often with substantial quantities of ash and slag and rubble.	1-6	
Alluvium	Channel and terrace deposits – sand, gravel and clay	0.5	
Glacial Drift	Till – boulder clay, Glaciolacustrine-clay, silt, fine sand, peat Glaciofluvial – sand and gravel	0-40	
TRIASSIC			
Mercia Mudstone	Red-brown mudstone, minor, dolomitic siltstone and sandstone beds and nodules of gypsum	<365	15.9 – 23.8
Bromsgrove sandstone	Red-brown medium to coarse grained, subangular arkosic sandstone, pebbly in places interbedded with mudstones layers	<120	13.9 – 15.9
Unconformity			
Wildmoor sandstone	Orange-red fine grained micaceous, soft sandstone with sparse mudstone layers	<120	13.5 – 13.9
Kidderminster sandstone	Red-brown pebbly sandstone, pebble conglomerate, medium to coarse grained sandstone with sparse mudstone layers	<120	8.6 – 13.5
Unconformity			
Hopwas Breccia	Coarse quartzite breccia and pebbly sandstone	<30	
Unconformity			
PERMIAN			
Clent Formation	Breccia with mudstone matrix, thin beds of sandstone and mudstone	<100	
Unconformity			
CARBONIFEROUS			
Upper Coal Measures	Sequences of mudstones and sandstone with thin beds of limestone	<650	2.9 – 8.6
Unconformity			
Middle Coal Measures	Mudstones, sandstones and seat earth and productive seams of coal and ironstone	<200	0 – 0.4 0.9 – 2.9
SILURIAN			
Wenlock shale	Mudstone and nodular limestone	<150	0.4 – 0.9

(Information compiled after Powell et al., 2000)

3.4.1 Solid Geology

Carboniferous Coal Measures

The units present are the Upper and Middle Coal Measures comprising a cyclical sequence of sandstone, mudstones, seat earths and occasional thin nodular limestone horizons. The Middle Coal Measures contain the productive coal seams of the South Staffordshire Coalfield.

The Triassic Sandstone

The Triassic Sandstone forming the Birmingham Aquifer comprises a thick sequence of sediments accumulated during subaerial deposition in a desert environment (Allen et al, 1997). The sediments are fluviatile in origin and derived from a major braided river system with evidence of aeolian dunes merging laterally into the water-laid deposits. The deposits lie unconformably on the Carboniferous Coal Measures and are succeeded by the conformable Mercia Mudstones above. The sediments range from friable to cemented depending on the degree of calcite cementation. The aquifer forms part of the Sherwood Sandstone Group, and comprises three units.

Kidderminster Formation (lower)

Comprising interbedded sandstone, breccia and conglomerate, the formation is believed to represent part of a major river system flowing northwards with wadi mouth fan type deposits. The unit comprises red-brown, medium- to coarse-grained, cross-bedded pebbly quartz sandstone and pebble conglomerate with sparse mudstone layers

Wildmoor Sandstone Formation (middle)

The Formation comprises finer-grained deposits of sandstone with fewer pebbles indicating a gradual reduction in sediment load and transport capacity and including some aeolian deposits. The unit consists of orange-red, fine-grained, micaceous soft sandstone with sparse mudstone layers

Bromsgrove Sandstone Formation (upper)

This lies unconformably on the lower divisions and consists of a series of upward fining units from sandstone to mudstone. The overlying junction with the Mercia Mudstone group is typically gradational with an upward increase in mudstone and siltstone, and a decrease in sandstone beds. The unit comprises red-brown medium to coarse grained, sub-angular arkosic sandstone, pebbly in places and interbedded with mudstones layers.

Mercia Mudstone Group

The mudstones of Upper Triassic age were laid down on arid alluvial plains as mudflats and ephemeral lakes, at times connected to the sea at other times drying out giving rise to halite and gypsum deposits. The unit comprises red-brown mudstone, minor, dolomitic siltstone and sandstone beds. Nodules of gypsum occur throughout the formation.

3.4.2 Superficial Drift Deposits

The unconsolidated superficial deposits of the region have a complex distribution reflecting the glacial and post-glacial history of the Pleistocene and Recent time (Figure 3.7). The glacial deposition from ice sheets and melt-water reflects at least three periods of ice advance and retreat. However, correlating units across the area associated with these different phases is problematic and Horton (1974) proposed a subdivision based on four lithological types summarised as follows (Knipe et al, 1993).

1. Glacial till deposited beneath the ice sheets and retreating glaciers. The unit consists of boulder clay and unbedded drift comprising unsorted rock debris in a sandy clay matrix.
2. Glacio-lacustrine deposits formed in ice-dammed lakes. The unit comprises clay, silt, fine-grained sand and peat.
3. Glacio-fluvial deposits derived from meltwaters flowing from or beneath the glaciers.
4. Interglacial deposits formed in shallow lakes and marshes between periods of ice advance. The unit comprises humic silt and clay with some peat beds.

The glacial deposits cover a buried landscape significantly different from that of the present and including several buried valley systems with sand and gravel fill running on different courses to the current drainage pattern. The distribution of the drift types is highly variable across the region but in general the boulder clay is generally found on the higher ground and lining the sides and base of the main depressions (Knipe et al., 1993).

Postglacial deposits in the district are primarily products of the erosion of earlier glacigenic deposits and subsequent deposition under fluvial and lacustrine conditions. Alluvial deposits comprising sands, gravels and clays are found in the valley bottoms of all the current water courses. Flood plain deposits up to 1 km in width occur in the Tame Valley and two additional stages of river terrace deposition have been identified that predate the recent alluvium.

Man-made excavations and mining activities are common in the region, with numerous shafts sunk into the South Staffordshire Coal Field and quarrying of sand and gravel deposits. Made ground deposits are extensive, particularly in the Tame Valley, where large amounts of material have been deposited on otherwise marshy ground. The deposits comprise mine spoil, sand and gravel mixtures, often with substantial quantities of ash, slag and rubble.

3.4.3 Structure

The Coal Measures underlie the west of the region and to the east are unconformably overlain by the Triassic sandstone that makes up the central area. The sandstones dip gently to the east at less than five degrees. Within the unit subdivisions, the Wildmoor Formation lies conformably on the Kidderminster Formation, with an unconformable contact with the overlying Bromsgrove unit which, in turn, has a gradational contact with the overlying Mercia Mudstone. The Birmingham Fault is a major structural lineament which juxtaposes the sandstones of the central area against the Mercia Mudstones to the east. The Birmingham Fault is a northeast-southwest trending normal fault with a downthrow of between 50 m and 200 m on the eastern side which has resulted in the erosion of the original overlying Mercia Mudstone .

3.5 Regional Hydrogeology

The primary water-bearing unit is the Triassic Sandstone of the Birmingham Aquifer which is unconfined over an area of 106 km^2 (Thomas, 2001) in the central portion of the study area. To the east, the Mercia Mudstones are considered impermeable and the Birmingham Aquifer becomes confined beneath them. The northern boundary of the Birmingham Aquifer is defined by an anticline and to the south by a series of faults. To the west the Carboniferous

Coal Measures support some abstraction from minor aquifer units comprising highly fractured multi-layered sandstones inter-bedded with shale. Little natural connectivity is thought to exist with the overlying Triassic and the erosional surface of the Carboniferous forms the effective base of the Birmingham aquifer .

The Birmingham Fault is a low permeability feature, and a significant drop in piezometric head (<55 m in 1976) has developed across it during periods of high groundwater abstraction (Jackson et al., 1983). Some continuity does exist between sandstone units juxtaposed at depth and limited flow does occur across the fault through permeable units in the overlying drift.

Extensive mining of the Coal Measures has taken place historically and it is unknown what effects the now abandoned shafts may have on groundwater flow to the river. In many cases the shafts were back-filled and the original locations lost. The shafts are, however, limited to the outcrop of the Coal Measures and the thin western extremity of the Birmingham Aquifer.

The River Tame and its tributaries are the primary focus of natural groundwater discharge in the region with flows trending in a north or south direction toward the river. The unconfined Birmingham Aquifer is thought to provide the most significant groundwater contribution from the bedrock and for this reason the section of river overlying the aquifer was selected for detailed study. The influence of the drift deposits on flow to the river may be significant. In some areas low vertical and high horizontal conductivity may lead to considerable horizontal interflow through the drift possibly discharging to the river without recharging the underlying aquifer. During historical periods of peak abstraction and high drawdown, sections of the

Tame were effluent to the underlying aquifer (Land, 1966, Jackson et al., 1983). At the present time, however, the recovery in water levels implies that the Tame is now receiving groundwater discharge over most of its length with, perhaps, the exception of localised areas adjacent to abstractions.

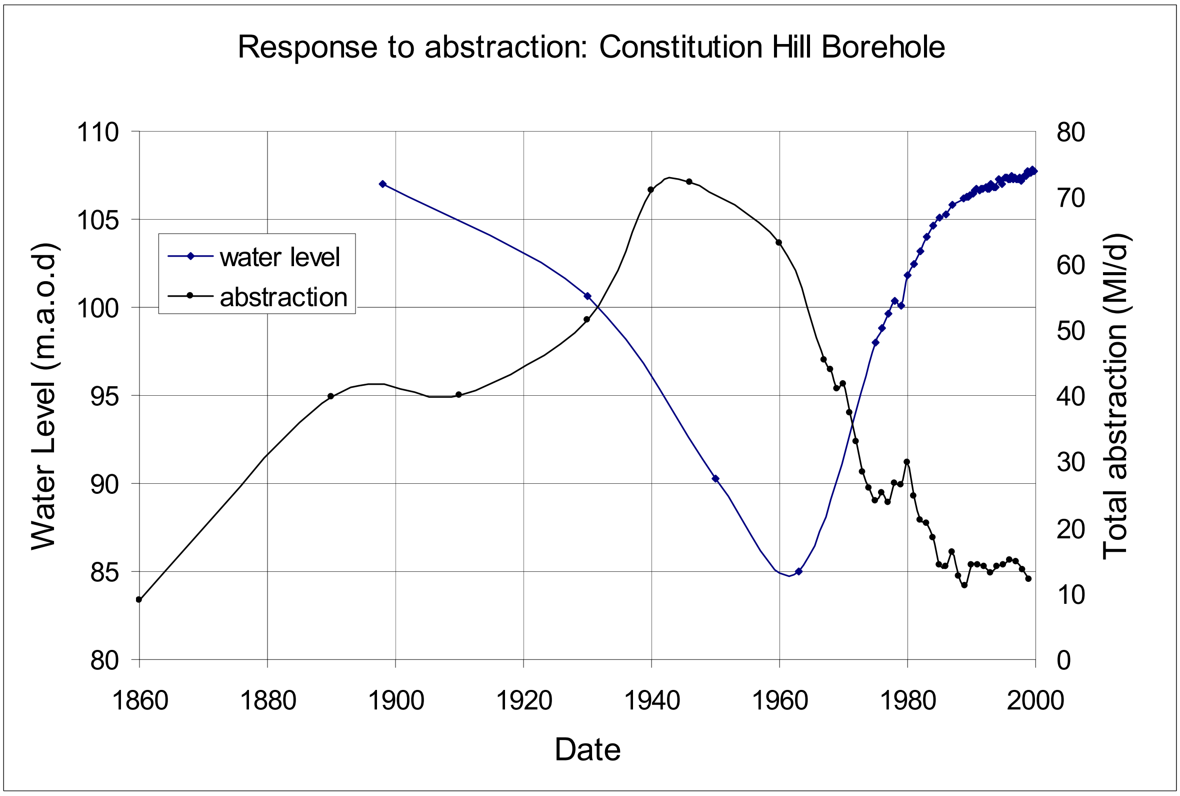
The Birmingham Aquifer has supported substantial abstraction in the past for industrial purposes and limited public supply, reaching a peak of 60 Mld^{-1} (Figure 3.8) during the 1950s. This produced drawdown in some areas of up to 30m below the original postulated levels (Figure 3.9a and b). Industrial closures and changing practices has led to a decline in abstraction to approximately 13 Mld^{-1} (Environment Agency, 1999) and a consequent rise in the water table (Figure 3.9 c and d). An investigation was undertaken (Knipe et al., 1993) into the engineering implications of the groundwater rise because of concern about foundation stability, basement flooding and the mobilisation of contaminants from previously unsaturated waste.

The limited data available from pumping tests indicate a range in transmissivities of between 150 – 300 m^2d^{-1} for the Birmingham Aquifer (Knipe et al., 1993). Values of conductivity and porosity assigned to the aquifer subdivisions by Lovelock (1977) are as follows:

Formation	$K_{horizontal}$ md^{-1}	$K_{vertical}$ md^{-1}	Porosity
Kidderminster	3.5	2.7	0.29
Wildmoor	1	0.83	0.27
Bromsgrove	0.93	0.53	0.28

Table 3.2 Hydraulic properties of the subdivisions of the Birmingham Aquifer

Figure 3.8 Abstraction from the Birmingham Aquifer and its effect on water level at the Constitution Hill Borehole, modified from Greswell (1992).



* Note pre 1960 data are taken from estimates used in groundwater modelling by Greswell (1992) undertaken for the CIRIA report (Knipe et al., 1993). Data post 1960 comes from the National Rivers Authority and the Environment Agency.

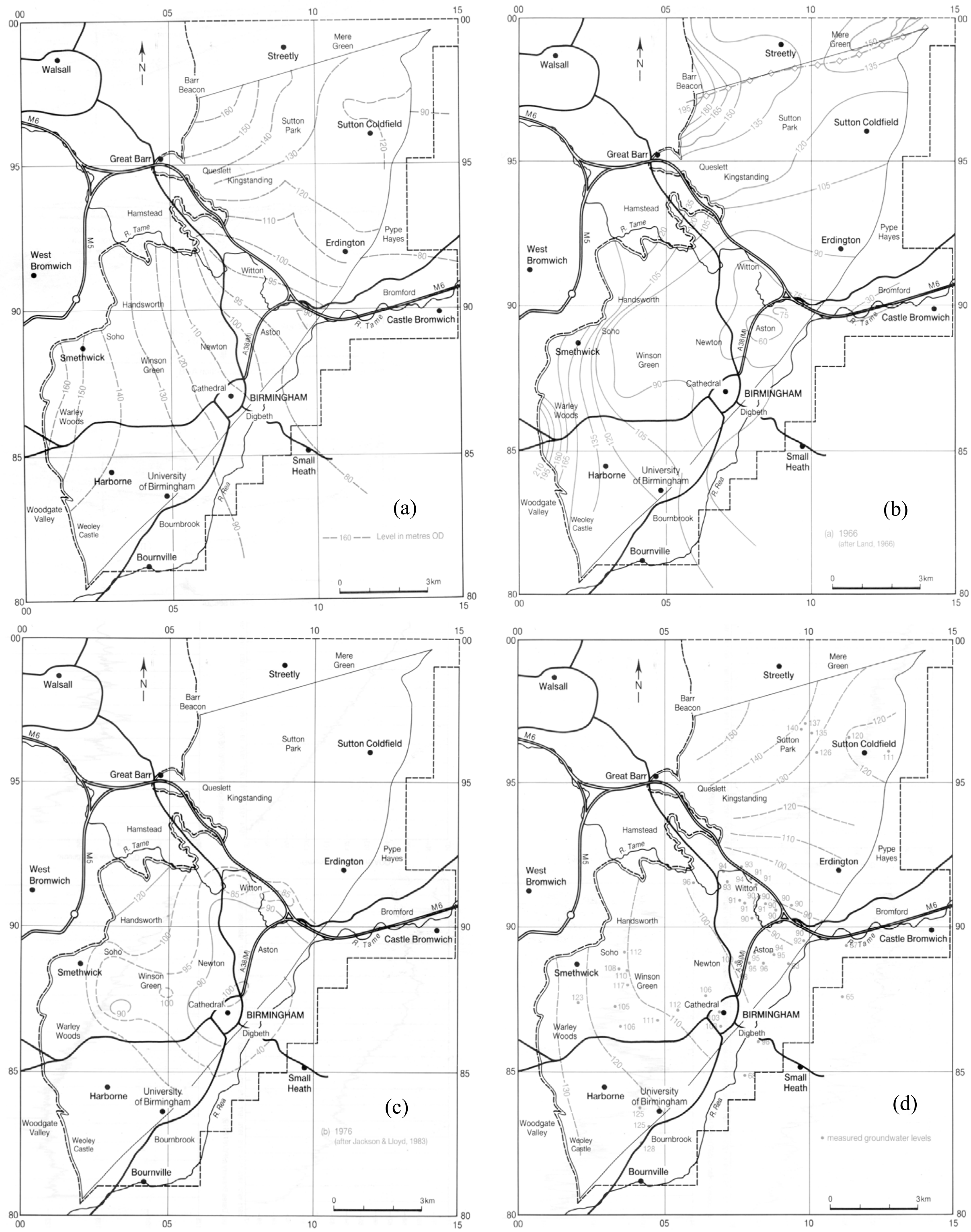


Figure 3.9 (a) Postulated water table in the Birmingham Aquifer from pre-abstraction times (b) Water table in 1966 (c) Water table in 1976 (d) Water table in 1988/89. (Figures adapted from Knipe et al., 1993).

The importance of fracture flow has yet to be fully investigated but is believed to be significant in some cases. The calcite cementation of the formations varies from strong to weak. A value of 0.15 for specific yield and 0.0005 for the confined storage coefficient are thought to be generally representative of the aquifer (Knipe et al.,1993). The increasing occurrence of mudstone beds adjacent to the Mercia Mudstone contact is thought to produce multi-aquifer conditions (Ramingwong, 1974). The impact of these mudstone beds throughout the aquifer is unknown, but may significantly reduce vertical conductivity, with fractures acting as the dominant control on vertical flow. The extent of the active groundwater system is highly dependent on the degree of vertical connectivity, with the likelihood that the older groundwater at depth will be relatively stagnant with long residence times in the aquifer.

Recharge is complex and has a high spatial variability in the urban catchment. The distribution of precipitation across the region is not uniform with average annual rainfall of 800 mm in the south-west and only 650 mm in the north-east (Powell et al., 2000). Run-off is high due to the extensive urban coverage and recharge may vary considerably over relatively short distances of tens to hundreds of metres dependent on land use and coverage. The reduced recharge from precipitation may be supplemented by seepage from the extensive Birmingham canal network and leakage from sewers and water mains. Recharge from mains leakage into the aquifer has been estimated as 600 $mmyr^{-1}$ equivalent to 25% of Birmingham's public supply (Lerner et al., 1996) the majority of which is derived from Wales via the Elan Aqueduct. A major control on recharge to the underlying bedrock is the conductivity, thickness and distribution of the superficial drift deposits. Large thicknesses of boulder clay are present in some areas (e.g. more than 40 m in the vicinity of Smethwick) and this restricts the level of recharge to the bedrock. A geographical information system has been used by Thomas (2001) to investigate the complex nature of recharge in Birmingham based on land use and drift geology.

3.6 Hydrochemistry and Contamination of the Birmingham Aquifer

The extended period of urbanisation and industrial activity in the study area has had a substantial impact on the groundwater quality, with widespread contamination of the shallow portions of the unconfined aquifer, and local to some boreholes within the confined section. Despite this, many parts of the aquifer contain water of good quality and sustainable use of the aquifer for public supply has been investigated (Greswell et al., 2000). Research by Jackson (1981), Ford (1990) and Rivett (1989) highlighted the distinct variations that occur in the hydrochemistry of the aquifer between the confined and unconfined sections, and between shallow and deep waters.

The natural hydrochemistry is a bicarbonate system dominated by dissolution of calcite cement with high sulphate concentrations in some areas associated with the dissolution of gypsum in the upper part of the Bromsgrove Sandstone Formation and the Mercia Mudstone Group. Water from the Kidderminster and Bromsgrove Sandstone Formations is oversaturated with respect to calcite, but undersaturated in the Wildmoor Formation owing to the lower content of calcite cement. A 20 m thick zone of decementation reported (Ford et al., 1992) at the top of the unconfined formations has developed as a result of acidic recharge water primarily related to industrial pollution. Contributing factors to the acidification include microbial activity increasing the PCO_2 in the shallow groundwater e.g. microbially mediated oxidation of hydrocarbons, industrial acid spills which may be locally important, and oxidation of sulphide minerals present within the Quaternary deposits. The majority of the unconfined aquifer is oxygenated (2 to 10 mg l^{-1} O_2) with Ca and HCO_3 the dominant ions and Eh 300-350 mV. The Wildmoor Formation contains waters that have relatively lower pH than the other formations and SO_4 and/or NO_3 and/or Cl are generally the dominant ions. In the confined aquifer, reducing conditions prevail, Ca and SO_4 are the predominant ions, dissolved oxygen (DO) is generally zero and nitrate is absent.

Three age groups for the waters have been defined (Figure 3.10) on the basis of carbon isotope analyses (Jackson et al., 1983) and corroborated by the distribution of anthropogenic contamination. These comprise:

1. modern and contaminated waters within the shallow portions of the unconfined aquifer;
2. intermediate waters (2000 to 4000 years) beneath group one waters in the unconfined aquifer and in sections of the confined aquifer close to the Birmingham fault; and
3. old waters (>6500 years) within the confined section and at depth in topographic lows in the base of the unconfined aquifer.

As is the case in all the hydrochemical studies involving abstraction wells within the aquifer, samples are drawn from over a considerable depth interval resulting in mixing of waters of different origin. Therefore, waters in the intermediate group may reflect induced mixing of modern and old waters within the well or natural mixing within the aquifer. Water quality data from the abstraction wells must be considered in the light of several factors that may introduce bias into the sampling.

1. The abstraction wells used in the surveys were almost all from sites within the industrial land-use category which comprises <5% of the total surface area of the unconfined aquifer (Ford et al., 1994).

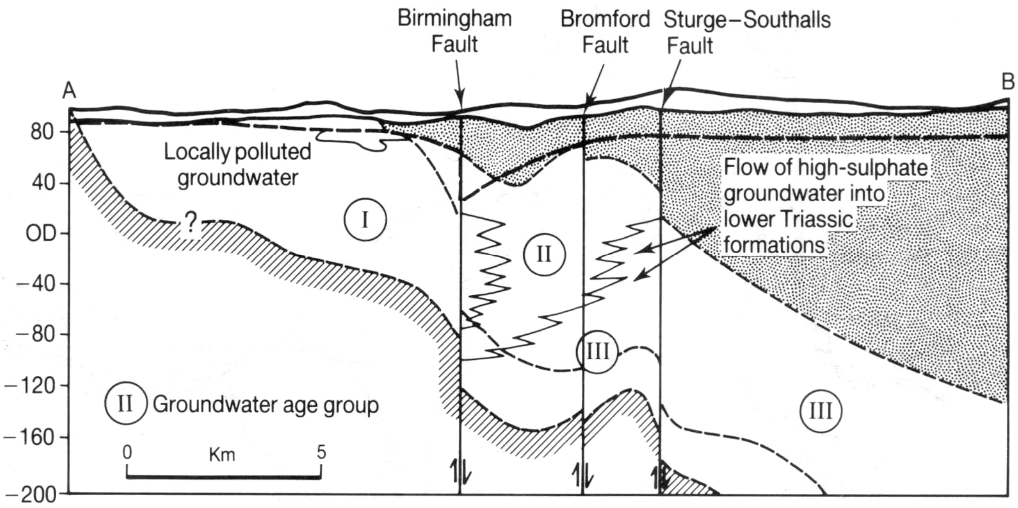


Figure 3.10 Schematic cross section of the distribution of groundwater age groups within the Birmingham aquifer (Jackson et al., 1983).

2. Mixing of different waters occurs over the large open-screen intervals which is likely to result in a dilution of contaminants from a narrow depth interval, and a change in chemical equilibrium conditions.
3. Most of the abstraction wells are associated with industrial supply and are therefore at most risk of contamination by the local industry.

The carbon isotope study (Jackson et al., 1983) indicated a limited flow of modern waters across the Birmingham Fault and the travel of high sulphate intermediate group waters downwards in areas adjacent to the fault zone. Contaminant studies (Ford et al., 1994, Rivett et al., 1989) have detected a limited occurrence of modern pollutants within sections of the confined aquifer indicating connectivity, perhaps via an unsealed borehole.

The hydrochemistry of the aquifer is modified by the amount and quality of the recharge it receives, primarily from rainfall and water mains leakage. The natural quality of rainfall in Birmingham is subject to considerable chemical loading from urban sources prior to entering the groundwater system. Historically induced seepage from the River Tame and tributaries may have occurred over several decades during periods of high drawdown perhaps introducing considerable amounts of poor quality surface water into the aquifer underlying the valley.

Numerous diffuse and point sources of different groundwater contaminants occur across Birmingham including industrial wastes and processing chemicals, sewage, road de-icing salts, urea, domestic wastes, fertilizers, construction wastes, historical animal waste, human burial and wet cleaning processes. Ford (1990) observed anthropogenic impacts on inorganic water quality throughout the unconfined aquifer with the exception of two boreholes cased to a depth of >135 m. In all surveys the abstraction regime of the sampled well was found to be a major influence on temporal quality variations. Intermittent pumping was found to take high proportions of shallow water. Evidence was also found of migration of contaminants down the side of borehole casing. Rises in groundwater have led to a reduction in the unsaturated zone increasing the vulnerability of the groundwater. Closure of industry and reduction of abstraction mean migration of contaminant plumes off site is now more likely to occur.

Of the 28 inorganic determinands measured from 70 abstraction wells (Ford et al.,1994) only nitrate and barium regularly approached the European Community drinking water standards. However, localised areas showing high levels of groundwater contamination, including heavy metals, were observed. The principal organic contaminants detected in the groundwater are chlorinated solvents (Rivett, 1989) which could be directly related to metal work degreasing or dry cleaning industries. Little correlation was found between the distributions of organic and inorganic contaminants, probably reflecting differences in physical transport processes,chemical interactions and histories of chemical usage. The worst case of inorganic pollution was found in west Birmingham and the worst case of organic contamination was within the Tame Valley. The extent of microbiological contamination within the Birmingham Aquifer is unknown but a study of shallow groundwater in the Triassic Sandstone beneath Nottingham has revealed the widespread occurrence of bacteria and bacteriophage marker species related to leakage from sewers.

Temporal variations in groundwater quality have been investigated by Ford et al., (1994), Rivett, (1989) and Taylor and Rivett, (1999). Contaminant travel times to the water table have been estimated at 20 years beneath Birmingham city centre which has a deep unsaturated zone. Travel times are thought to be more rapid in areas where the water table is close to surface such as the Tame Valley. It was noted that some areas with a shallow water table in the Tame Valley had fast recovery in water quality owing to the rapid addition of high quality recharge and discharge of contaminated groundwater to the river system. Ford (1990) found a decrease in groundwater quality beneath the city centre since the survey by Jackson (1981), but an increase in quality in the Tame Valley. This is may be related to the lag between contaminant travel times to the water table in the different locations. A general decline in pH values in the abstraction wells was also observed between the 1978 (Jackson, 1981) and 1989 (Ford, 1990) surveys. Chlorinated solvent concentrations were found to be stable over a ten-year period (Taylor and Rivett, 1999) and the solvents are thought to derive from dissolution of dense non-aqueous phase liquid (DNAPL) pools within the aquifer which represent a long term source.

Better inorganic water quality showed no relationship to thicker overlying drift deposits. Rivett (1989) found a correlation between thicker deposits of drift and lower levels of organic contamination. However, this may also be related to the fact that organic contaminants are relatively recent and are therefore undergoing increased lag times through the drift before eventually reaching the water table.

3.6.1 Inorganic Contamination

Land use is a significant factor in contaminant occurrence with the highest levels of salinity, sulphate, chloride, sodium, boron and total heavy metal concentrations reported by Ford (1990) to be associated with metal working sites. Conductivity logging and depth sampling

(Ford, 1990) from nine boreholes revealed that all determinands with the exception of nitrate, which is more widespread, exhibit a well-defined trend of decreasing concentration with depth.

Elevated levels of pH have been observed in the shallow groundwater as a result of the depleted levels of calcite in the shallow zone. This has resulted in increased mobility of pH-sensitive contaminants, including trace metals. Inorganic contaminant attenuation characteristics of the sandstone (Buss et al.,1997, Mitchener, 2002) are governed by the presence of calcite cement and the oxyhydroxide grain coatings, and the few percent clay component, mostly kaolinite. Laboratory experiments return cation exchange values of 1 to 20 meq/100g. Stagg (1997) looked at metal transport on colloids but found it to be insignificant in relation to the total metal content of the groundwater.

Elevated levels of the trace metals copper, zinc, chromium, nickel and cadmium have been detected related to land use, primarily associated with the metal working industry. This is despite the expected limitations on transport by sorption processes. This may be a result of low pH, ion pairing, and metal loading greater than the local aquifer's sorption capacity; also, some anionic forms of chromate are highly mobile.

Nitrate distribution is generally more diffuse and extends to greater depths than the other contaminants reflecting the long term and wide distribution of its numerous sources. Nitrate is not present in the confined section owing to the reducing environment. Nitrate shows no relationships to land use, with average concentrations of 50-65 mgl^{-1} (as NO_3) across the unconfined aquifer. Unusually low concentrations of nitrate have been recorded in shallow boreholes perhaps suggesting reduction in association with organic pollution. Sulphate levels may be raised by atmospheric fallout estimated at 275,000 kgy^{-1} (Ford, 1990), but the primary loading is directly related to industry. Metal working industries return the highest average values of 192 mgl^{-1} compared with 25 mgl^{-1} for the service industries.

High natural background levels of barium were observed in association with low sulphate concentrations. Boron shows a clear correlation with land use and is indicative of contamination by boric acid which is much used in the metals industry. Median values for metal industry boreholes were 450 μgl^{-1} with service industry boreholes at < 20 μgl^{-1}. Substantial concentrations of boron (30-150 μgl^{-1}) were detected in the confined aquifer related to release during gypsum dissolution. High levels of strontium (0.5-10 mgl^{-1}) are associated with natural occurrence within the confined aquifer. High levels of iron and manganese are found to occur naturally as coatings within the sandstone and within the drift deposits, providing a plentiful source for dissolution within the groundwater.

3.6.2 Organic Contamination

A survey of organic water quality (Rivett et al., 1990) indicated the chlorinated solvents Trichloroethene (TCE), Trichloroethane (TCA), and Tetrachloroethene (PCE) to be the main organic contaminants of the Birmingham Aquifer. The high levels of contaminants that were detected were generally related to solvent user sites. The most widely occurring contaminant was TCE which shows the greatest consumption of the chlorinated solvents by industry within Birmingham. The solvent has been widely used since 1928 as a metal degreaser. TCA was introduced as a replacement for TCE in 1965 but its use has declined because of its detrimental effect on the ozone layer. PCE is used almost exclusively in Birmingham by the dry cleaning industry. Sorption experiments indicate the low organic carbon contents, from core analyses, are not a dominant factor in the transport of organic contaminants within the aquifer (Shepherd., 2002).

3.7 Summary

The Birmingham and West Midlands region has a long industrial history dating back to the beginnings of the industrial revolution in 1800, and this, together with urban development, has led to widespread contamination of the groundwater. Previous work has identified inorganic contaminants within the Birmingham Aquifer, including heavy metals and organic contaminants, primarily chlorinated solvents. This industrial footprint on the urban groundwater in the Birmingham area, and its implications for future river basin management and protection policy, warrants further investigation, hence the rationale for my research.

The River Tame was selected for study because it runs from east to west through the heart of the urban/industrial area, and because some of the contaminated groundwater within the Tame Valley discharges to the river. A 23.8 km stretch of the Tame was chosen for the study as it is known to receive an increased (20%) inflow of groundwater, in particular, along a 7.4 km section that flows over the Birmingham Aquifer. This section was targeted for a detailed study of the contaminant flux via groundwater to the river. In addition to the bedrock aquifer, there is groundwater flow to the river through the alluvial gravels deposited on the flood plain.

Site selection was also based on the fact that the 23.8 km reach of the Tame is conveniently delimited by two gauging stations, Bescot at the upstream limit and Water Orton downstream.

The river along this stretch is typically 8-12 m wide, 0.2-2.0 m deep, with average dry weather flow velocities of 0.1-0.8 ms^{-1} and a mean discharge at Water Orton of 397 Mld^{-1}. The relatively small size of the river enables easy access for the installation of research instrumentation.

The main potential drawback in selecting the study area was identified as the pipe-end discharges to the surface water. There is no control on these discharges which can vary considerably from day to day. However, this is one of the inevitable challenges when undertaking research in an urban catchment environment.

CHAPTER 4. MONITORING NETWORKS AND METHODS

The fundamental aim of this research is to determine the significance of contaminated groundwater flow in contributing to the water quality of the groundwater/surface water interface and the overlying water column of an urban-river system. To this end data were collected from surface water surveys, a river-bed piezometer network specifically installed for the study and a previously installed shallow groundwater monitoring network in the vicinity of the Tame. The objective of this chapter is to detail the techniques employed in collecting field data on groundwater and surface water quality, head distribution and flow and the subsequent methods of data and sample analyses. Groundwater modelling methods are described separately in Chapter 6.

Data were collected over a period of three years (1999-2001) with the majority of fieldwork undertaken in the summers of 2000 and 2001. The range of fieldwork undertaken was greatly enhanced by the creation of several MSc and BSc projects designed to contribute to the overall PhD remit. The following projects were field managed by the PhD: MSc projects - Hogan,1999, Dowell, 2000, Henstock, 2000, Singleton 2001; BSc projects - Moylett, 2000, Fuller 2001, Littler, 2001. This greatly facilitated data collection. The sampling of shallow groundwater across the northern section of the Tame valley was undertaken in conjunction with the PhD research of Shepherd (2002).

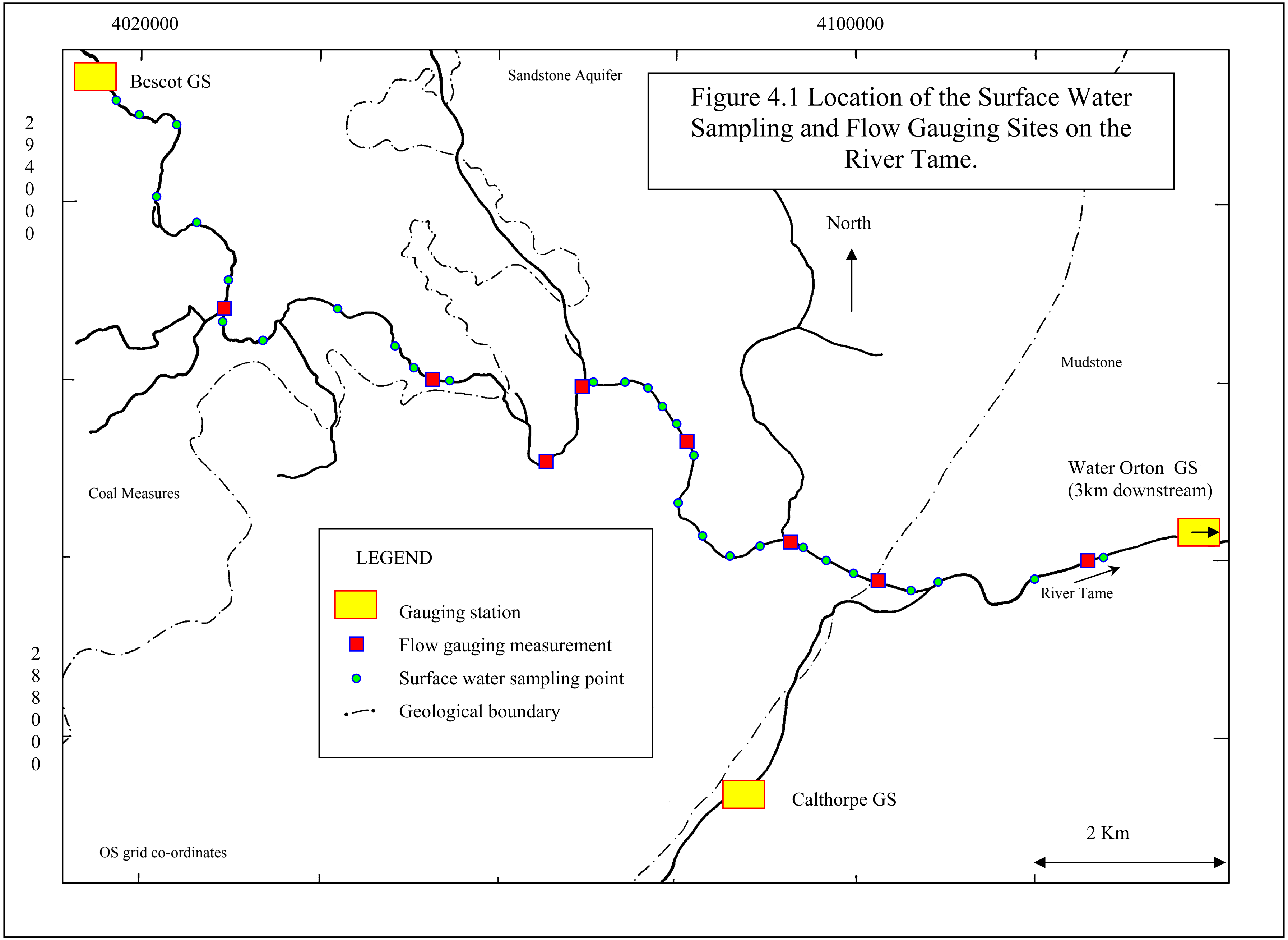
Because of the broad scope of the investigation and the wide range of methods employed, an initial overview is presented of the techniques and monitoring networks used, together with the locations of sampling and measurement points. The reader may then refer to the subsequent sections if a more detailed description of each methodology is required. The sections are divided as follows:

- Archive data
- Surface water flow gauging
- Surface water quality sampling
- Groundwater quality sampling
- Groundwater head measurements
- Characterisation of the riverbed sediments
- Riverbed temperature survey
- The groundwater contribution to baseflow
- Sample analyses.

4.1 Overview

The 23.8 km study reach of the Tame is delimited by the Environment Agency's flow gauging stations at Bescot (upstream) and Water Orton (downstream). The main focus of the study was on the central 7.4 km stretch of the river that flows over the unconfined Birmingham Aquifer (Figure 4.1).

Surface water sampling was undertaken at ~ 200-500 m intervals along the river to determine the variation in surface water quality (Figure 4.2). Results were compared with the water quality data from 96 piezometers specifically installed in the riverbed at depths of 15-200 cm.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Bescot GS
Sandstone Aquifer
Figure 4.1 Location of the Surface Water Sampling and Flow Gauging Sites on the River Tame.
North
Mudstone
Water Orton GS
(3km downstream)
Coal Measures
LEGEND
Gauging station
Flow gauging measurement
Surface water sampling point
Geological boundary
River Tame
Calthorpe GS
2 Km
OS grid co-ordinates

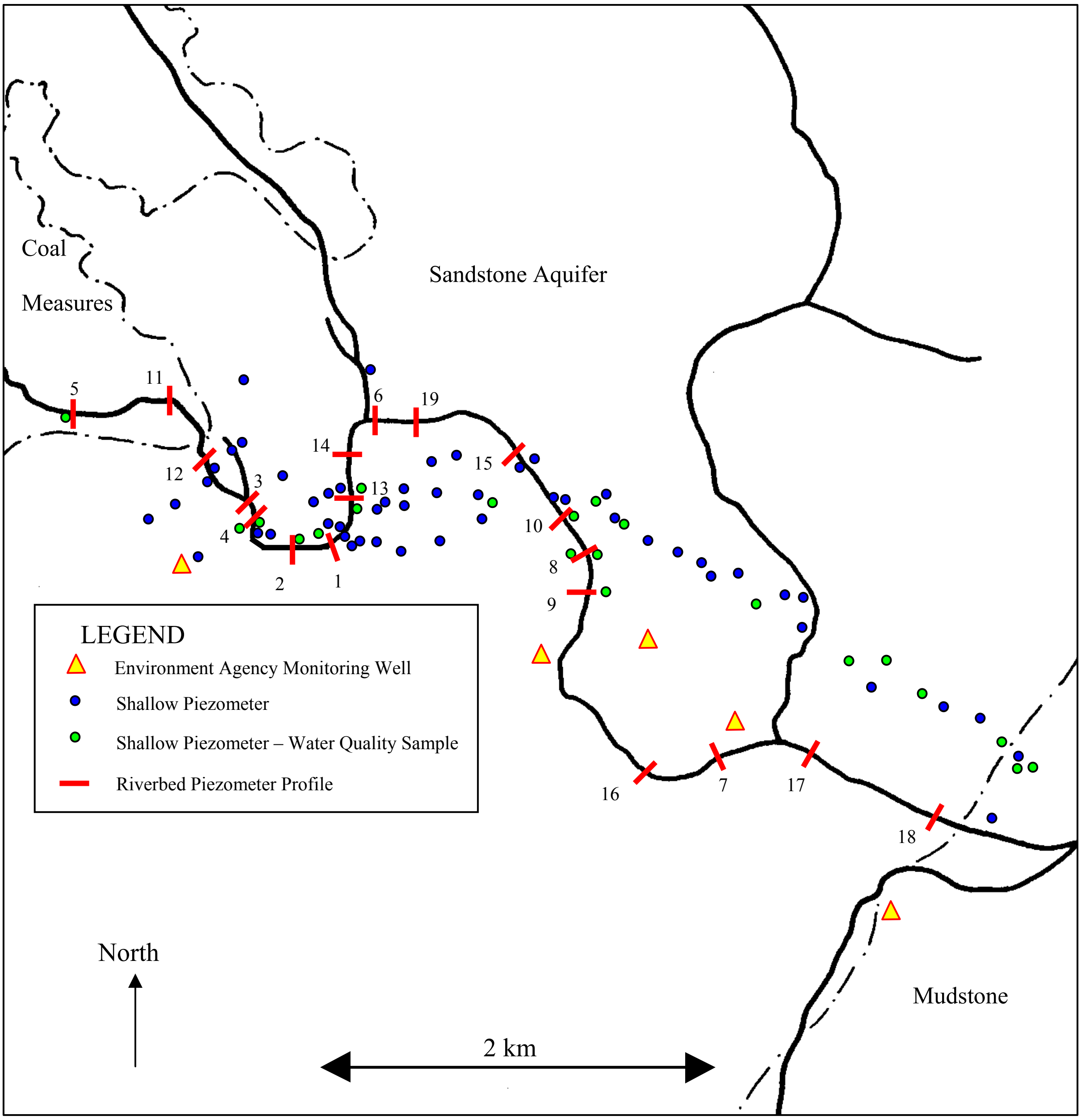


Figure 4.2 Location of the Riverbed Piezometer Profiles, Shallow Piezometers and Agency Monitoring Wells in the Tame Valley

The riverbed piezometers formed a series of 18 lateral profiles, comprising 2-5 piezometers, so that variations in the water quality and head across the channel could be studied. Five multilevel piezometers were installed to examine changes vertically through the riverbed across the groundwater-surface water interface. Groundwater quality data were also obtained by sampling a series of shallow piezometers (5-20m) adjacent to the riverbanks and across the northern section of the Tame Valley. This was supplemented by data on the deeper groundwater from the sampling of industrial abstraction wells (Shepherd, 2002). As a comparison with the urban water quality study, a background survey was undertaken in the natural undeveloped area (6 km^2) of Sutton Park to the north of Birmingham. Water samples were collected from springs, surface waters, 12 specifically installed riverbed piezometers and 4 shallow piezometers (Figure 4.3).

Inorganic sample analyses were for a full suite of cations and anions, plus the measurement of field parameters, including dissolved oxygen, pH, Eh, conductivity and alkalinity. Organic analyses were for a full suite of volatile organic compounds including the chlorinated solvents and, for a few samples, vinyl chloride. During the course of the research a total of 324 water samples were collected and analysed (Table 4.1). Temporal variability in water quality was assessed by repeated sampling over the three-year period.

Sample Type	Number of samples analysed
Surface Water	128
Riverbed Piezometers	139
Shallow Groundwater	43
Abstraction Wells	14

Table 4.1 The number and type of samples collected during the research.

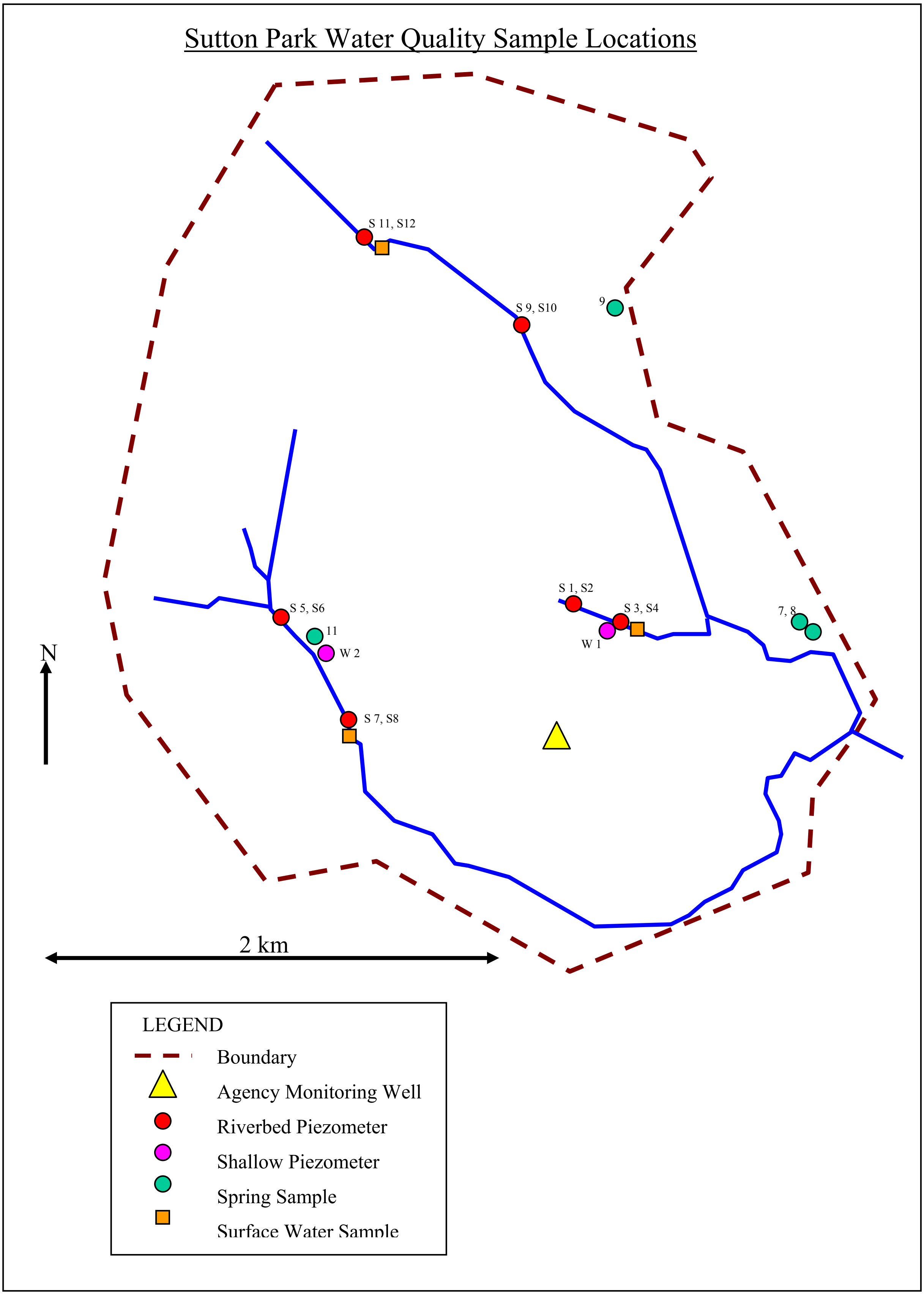


Figure 4.3 Water quality sample locations for the survey of Sutton Park

Water quality data were collected in order to:

(i) estimate the contaminant mass flux to the river in conjunction with groundwater and surface water flow measurements;

(ii) examine the variation in water quality across the groundwater-surface water interface associated with such processes as biodegradation and redox reactions;

(iii) assess the water quality in relation to the U.K. toxicity standards for aquatic life (environmental quality standards) and drinking water;

(iv) examine the differences in groundwater quality between the riverbed and the shallow and deep monitoring wells and the 'background' Sutton Park samples;

(v) and to observe the changes in the surface water quality that can occur across the study reach.

Archive surface water flow measurements from the gauging stations at Bescot, Water Orton and Calthorpe (Figure 4.1) were supplemented by additional fieldwork involving surface water flow gauging across the aquifer. The data were examined to determine the groundwater contribution to baseflow and the degree of temporal variability in this.

A characterisation of the riverbed sediments was undertaken by performing falling head tests in the riverbed piezometers and taking riverbed sediment cores upon which sieve analyses were carried out. Samples of the riverbed sediment were analysed to determine the fraction of organic carbon (F_{oc}) content to assist with studies on the sorption and retardation of organic contaminants on their passage across the groundwater/surface water interface.

Local groundwater flows across the riverbed were estimated using the Darcy flow and radial flow equations in combination with head and conductivity data from the riverbed and riverbank piezometers. Measurements of the temperature gradient across the groundwater-surface water interface were taken across and along the riverbed to give a qualitative indication of groundwater discharge to the river. Vertical temperature profiles from multilevel piezometers were used to calculate rates of groundwater flow.

Groundwater head and river stage data were collected during the project and used in conjunction with archive data to model groundwater flow. This was undertaken on a large scale based on groundwater flow across the river flood plain (1.8 km^2) and on a local scale (centimetres – metres) based on flow to the river channel. Groundwater/surface water interactions under transient (river flood) conditions were examined using continuous head data collected on river stage and groundwater levels in the adjacent riverbank.

The riverbed piezometer profile 8 (Figure 4.2) was selected for detailed study of both flow and groundwater quality. Piezometers were located on both banks, 5 riverbed piezometers were installed across the channel of which 3 were multilevel installations. This profile was sampled on 4 occasions during the study period and detailed modelling of groundwater surface water interaction was carried out.

Field equipment specifically designed and built for the project included drivepoint mini piezometers, seepage meters, a temperature probe, a peristaltic pump, a riverbed coring device and pressure transducer and logging systems to fit inside the narrow diameter shallow piezometers. The seepage meter was found to be unsuitable for the River Tame environment and was not used further after initial trials (Section 4.9.2).

More information on the equipment construction, installation and use can be found in the following sections together with details of the sampling, chemical analyses, archive data and methods of data analyses.

4.2 Archive data.

A considerable body of data for use on the project was obtained from a number of sources as follows:

Environment Agency Data

- Continuous discharge measurements for the Tame from the gauging stations at Bescot, Calthorpe, Water Orton 1990-96, 1998, 1999.The data were obtained through liaison with the Department of Civil Engineering, Birmingham University.

- Continuous water quality measurements for the River Tame at Water Orton for limited determinands, including temperature and conductivity, 1990 – 1996, 1998, 1999.

- Monthly spot water quality measurements from four locations on the Tame 1990-1999, for Ammonia, Dissolved Oxygen, Copper, Nickel and Total Hardness (Appendix 2).

- Flood defence survey data of several hundred river cross-sections (Appendix 3).

- Hydrograph data from 13 deep Agency monitoring wells (Appendix 4).

- Licensed discharge and abstraction information from the Agency for the Tame and the Birmingham Aquifer (Appendix 5).

- Geological logs, hydraulic conductivity estimates and sieve analyses from 20 shallow piezometers drilled on the banks of the Tame for flood defence investigations (Appendix 6).

Birmingham University School of Earth Sciences

- Historical data on inorganic groundwater quality, (Jackson, 1981, Ford, 1990, Hues, 1998).

- Historical data on organic groundwater quality, (Rivett, 1989, Brennan 1999, Taylor, 1998).

- Surface water quality sampling profiles along the Tame, (Hogan,1999)

British Atmospheric Data Centre (BADC)

- Historical rainfall data from > 10 weather stations in the Tame catchment.

Severn Trent Water Company

- Geological logs from ~ 70 boreholes drilled as part of the Black Country Trunk Sewer Extension (BCTSE) site investigation along the Tame valley (Appendix 7).
- Dipping records (1993 –2000) from piezometers installed for the BCTSE (Appendix 8).

The fieldwork programme was designed to provide more detailed information on the groundwater/surface water quality and flow interactions at a finer scale along the study reach and across the groundwater/surface water interface.

4.3 Surface water flow gauging.

In addition to the archive data from the gauging stations at Bescot, Water Orton and Calthorpe, flow measurements were also undertaken on the study reach between the stations (Figure 4.1). This was to define better the discharge accretion along the study reach and to examine the relationship with the underlying geology (Appendix 9).

4.3.1 River discharge measurements

Discharge measurements collected during periods of dry weather flow from a series of cross sectional profiles at intervals of several kilometres along the river were designed to assess discharge accretion along the study reach. Sites were selected to enable easy access to the river in areas with a relatively uniform channel profile and minimal turbulence. The measurements were taken within the space of a single day to minimise variations due to baseflow recession. The method was time consuming and labour intensive with a significant travel time between sites. This limited data collection to an average of four sites in one day per team (two people).

Flow meters of both propeller and electromagnetic type were borrowed from the School of Civil Engineering . After initial trials the propeller type ‘Ott’ meter was found of limited use owing to the presence of weed within the channel. An electromagnetic meter was used to collect the majority of readings. The meter was calibrated by technicians using the School of Civil Engineering’s flume tank.

The method of data collection followed was as outlined by (Dingman, 1994). A measuring tape was stretched across the channel and fixed to each bank. Measurement points were spaced at 0.5 m intervals across the channel and the depth was recorded at each location. Flow velocity measurements were taken for each point at a depth of 0.6 m*(total depth) as previous workers (Clay, 1999, Cey et al., 1998) have found that this is representative of the average flow rate over the total depth. A fixed point was chosen to act as a stage reference mark. The river stage was taken at the beginning and end of the profile measurement to give an indication of any variation in discharge from upstream that may have occurred during the course of the profile measurement.

Because of time constraints it was not possible on every occasion to repeat the velocity measurements back across the profile so as to obtain an estimation of the measurement repeatability. However, for the readings taken on May 21 and repeated on May 22, 2001, there is good correlation, indicating that the method shows good repeatability. The digital flow meter provided an average velocity over a one minute period from readings taken at one second intervals which reduced the error from this source. Error estimates of +/- 15 % are indicated by Cey et al, (1998) under similar conditions.

The data were analysed using the mid-sectional method to estimate the total discharge across the profile. The total discharge for each profile was then plotted against distance down river to give an indication of the discharge accretion. Discharge measurements taken on six different days were combined with other available data (Clay, 1999, Knowles, 2000) to provide an indication of the variability of baseflow discharge accretion along the study reach.

There was a significant problem in determining the inputs from tributaries and industrial discharges between the measurement points, and the diurnal variation in discharge from sewage treatment works upstream. This added additional uncertainty in the calculation of the actual accretion related to baseflow input over the study reach.

4.3.2 River cross sectional discharge calculation

Stream discharge measurements were carried out as detailed in Section 4.2.1. Values of flow velocity were obtained at 0.5m intervals across the river at a level of 0.6 m of the total river depth at each measurement location. The velocity at 0.6m of the total depth is generally representative of the average velocity occurring over the entire depth interval. The total river discharge may be calculated using the mid-section method (Cey et al.,1998):

$$Q = \sum_{i=1}^{n} (X_{i+1} - X_i)(U_i Y_i + U_{i+1} Y_{i+1}) / 2$$

Q= total discharge

X_I = distances to successive velocity measurements from the river bank

U_i= velocity measurement at each interval

Y_i= depths at each interval

4.4 Surface water quality sampling.

Surface water samples were obtained along the 23.8 km section of the River Tame between Bescot and Water Orton gauging stations (Figure 4.1, Appendix 10). A series of ~40 samples were taken to form a longitudinal profile along the river, and this was repeated four times during the project. Daily variations in water quality were examined at one site by repeat sampling through the course of a day. On four occasions surface water samples were taken in conjunction with river discharge measurements to enable estimates of total mass flux within the river to be made. Surface water sampling was conducted only during periods of dry weather flow with no rainfall having occurred within the preceding three days. A sampling device was constructed to allow representative samples to be collected from the approximate mid-depth of the river. A weighted ceramic jar with a cork seal was lowered on a string into the river and the seal removed only when the required depth was attained to prevent sample bias (e.g. volatilisation) which may occur close to the river surface. Samples for volatile organic compound (VOC) analyses were placed directly into the sample vial without filtration. A series of samples were taken from different positions in the river channel at the same time to determine the degree of variation in concentration across the channel. As part of a comparative study a total of three surface water samples and two samples of spring water were taken from Sutton Park.

4.5 Groundwater quality sampling.

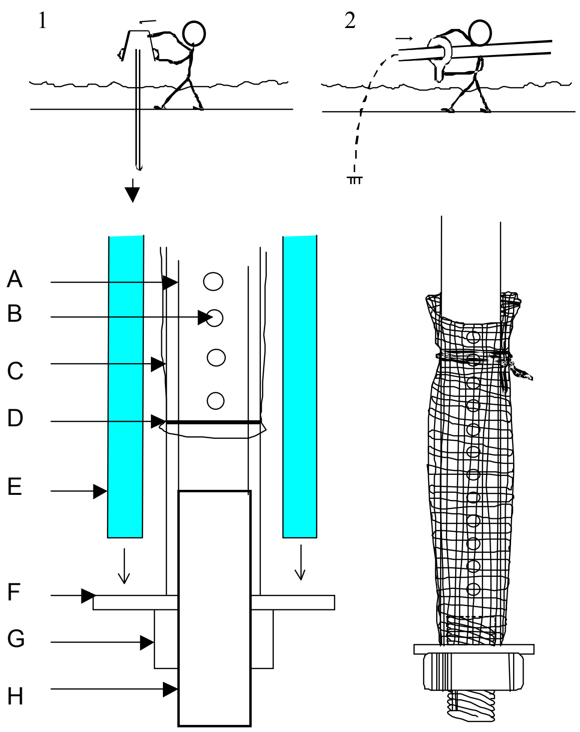
4.5.1 Riverbed piezometers

Several methods were considered for assessing groundwater quality and flow through the bed of the Tame. Both seepage meters and mini drive point piezometers have been used by previous workers (Cey et al, 1998, Lee et al,1980 and Carr et al,1980) to quantify localised groundwater discharge through lake and river beds. The two methods were given field trials to assess their suitability to the local conditions and the data requirements. The riverbed mini drive point piezometers were selected as the primary tool for the investigation.

4.5.1.1 Construction of Riverbed Mini Drive Point Piezometers (MDPs)

Piezometers were constructed of flexible 13mm OD (10mm ID) high density polyetheylene (HDPE) tubing with a drivepoint screwed into the open tube (Figure 4.4) using 3.5 cm of stainless steel studding thread (10mm dia) capped with a washer and nut. Located behind the head was a 10cm open section of drilled holes screened with nylon mesh (100 microns) secured by stainless steel wire. The piezometers were inserted inside a 2 m length (20mm ID) steel tube and driven by hand into the riverbed using a fence post driver to depths of up to 2 m. The driving tube was removed leaving the MDP firmly in place. The MDP was sealed with a rubber cork to prevent the growth of algae within the tube and to prevent artesian flow which would disturb the natural system. The MDPs are firmly anchored within the riverbed making them impossible to remove by hand (reducing the likelihood of theft) and this combined with the MDP's flexible nature prevented damage occurring during the three year study period despite several large flood events.

A total of 96 MDPs were emplaced as a series of lateral and vertical profiles across the channel at 18 different locations along the study reach (Figure 4.2, Appendix 11). High chloride concentrations exist in the surface water owing to sewage treatment work discharges and generally low concentrations were recorded in the MDPs. This implies that it was primarily groundwater being sampled at depths of greater than 20 cm. Installation depths for



A - HDPE tubing (ID 10mm)
B - Drilled Hole (Dia 10mm)
C - Fine Nylon Mesh
D - Stainless steel wire tie
E - Driving Tube (ID 20 mm)
F - Stainless Steel Washer (ID 10 mm, OD 30 mm)
G - Stainless Steel Nut (ID 10 mm)
H - Stainless Steel Studding Thread (Dia 10 mm)

Figure 4.4 Construction of Mini Drive-point Piezometer the MDPs ranged from 5 cm to 200 cm with the typical installation depth at 50 cm. Strong correlations were found between the chemistry of samples from the river bed piezometers and the larger permanent piezometers located 10 m from the adjacent banks. This reinforced confidence in the use of the MDPs.

Multilevel sampling arrays have been used by previous workers (Dean et al, 1999, Lorah et al 1999) to enable the investigation of the processes (including the degradation of organic compounds) that may be occurring across the groundwater/surface water interface. For this study a total of five multilevel arrays were installed at profiles 5, 7 and 8. The method of installation simply involved driving in single MDPs to different depths within an area of $1.5m^2$. A total of 12 MDPs were installed in Sutton Park (Figure 4.3).

4.5.1.2 Sampling methods

Water samples from the riverbed piezometers were obtained using either a portable 12 VDC peristaltic pump or a hand vacuum pump (Mighty Vac). Samples taken using the hand pump were extracted directly from the piezometer in the river channel into a 300 ml conical flask which was then brought to the bank and emptied into a beaker for further processing. This method was used only for sampling carried out in 2000. For 2001 sampling work, a portable peristaltic pump was constructed. The pump was connected from the bank via a ~12 m section of LDPE tubing with an OD of 10 mm which fitted directly into the riverbed piezometer forming a tight seal. Samples for inorganic analyses were initially collected in a 250 ml beaker prior to filtration. Only the minimum necessary amount of water was withdrawn to meet the sampling requirements in order to prevent the occurrence of downward vertical flow and possible contamination by surface water.

Samples for VOC analyses were collected from a piece of LDPE tube of length sufficient to contain the sample volume. This was connected directly to the river bed piezometer. A fresh section of LDPE tube was used at each sample location to prevent cross contamination. The VOC sample was collected after all other sampling had been carried out. This was to ensure a large volume of fluid had passed through the sampling tube to reduce the impact of sorption when the final VOC sample was taken. To reduce the risk of VOC sample degassing /volatilisation the sample tube contained a full column of water back to the peristaltic pump and the sample was taken from the base of this column. Three standing water volumes were purged prior to the commencement of sampling. It was noted that the piezometers appeared to be 'developed' during pumping with subsequent sampling runs yielding faster discharge rates owing to the removal of fine sediments adjacent to the piezometer screen.

4.5.2 Shallow monitoring wells

Groundwater quality (and head) data were obtained from two sets of shallow piezometers (<20m) within the sandstone aquifer and overlying drift deposits located adjacent to the river banks and across the northern section of the Tame Valley (Figure 4.2).

The older piezometer set (~70 holes) was drilled for Severn Trent Water Company as part of the 'Black Country Trunk Sewer Extension Project' during the latter part of 1993. The holes were drilled by rotary and percussion techniques with an average hole diameter of 121 mm to depths of between 10 to 35 metres. The holes were then back filled with arisings to the bottom of the response zone within the sandstone. The response zone comprised a one metre section with a bentonite seal at top and bottom, filled with a sand filter and containing a PVC piezometer (ID20 mm) with a 30 cm screen installed 30 cm from the base of the zone

(Gabriel, 1993). The hole was covered with a standard 13x13 cm stop-cock cover. Many of the holes are no longer accessible but at least 30 remained open during the course of the project.

The second set of piezometers was drilled during summer 2000 as part of the Environment Agency River Tame Asset Survey. The holes (ID 150 mm) were drilled to depths of up to 10.5m by light percussion rigs within 15 metres of the riverbank. Completion was similar to that employed in the Severn Trent holes with the response zones ranging from 1 to 3 metres in width (Caudell, 2000).

A total of 20 shallow piezometers were sampled representing ten point samples adjacent to the river and ten across the northern side of the Tame Valley up to 2 km from the river.

In Sutton Park, four shallow piezometers were sampled (Figure 4.3). These formed part of a network installed by South Staffordshire Water Company to monitor the effect of a nearby public supply well.

4.5.2.1 Sampling methods

A low-flow Waterra pump was used with HDPE tubing (ID 10mm, OD 13mm) and stainless steel ball valves. Dedicated tubing was used for each hole and left in place between sampling. A total of three ball valves were used and rotated between each site, those not in use were soaked in distilled water. If possible, three well volumes (inclusive of the gravel pack) were purged prior to sampling but this was not always possible owing to the slow recovery time of some piezometers. In each case a note was made of the purged volumes. Samples for inorganic analyses were initially collected in a 1 litre measuring cylinder prior to filtration. Samples for VOC analyses were poured directly without filtration from the standing water column within the pump tubing into the sample vial which was sealed immediately.

4.5.3 Deep abstraction wells

The sampled wells abstract water for industrial use from depths up to 100 metres within the Triassic Sandstone Aquifer. A single sample was collected at each site, in some cases directly from the pumped flow and in others from the storage tank. The abstraction wells provide a composite sample of considerable lateral and vertical extent that depends on a well's abstraction rate and associated capture zone. Pumping regimes, screen length and depth varied considerably between sites but the samples are thought to be generally representative of deep groundwater within the aquifer. Sampling of the holes has been undertaken by previous workers (Jackson 1981, Rivett, 1989, Ford, 1990, Taylor 1998) over a period of two decades. The most recent survey (Shepherd, 2002) was undertaken in conjunction with the shallow groundwater sampling programme to allow comparisons of the relative levels of contamination.

4.6 Groundwater and surface water head measurements.

A considerable amount of historical head data (Section 4.2) are available for the Birmingham Aquifer. Weekly to monthly dipping records were obtained for 70 Severn Trent shallow piezometers (1993-2000) drilled for the BCTSE and for five Environment Agency monitoring wells (1970-2000) within the Birmingham Aquifer < 1km from the river. Riverbed level data were available as part of an Environment Agency flood defence survey.

Access was possible to 40 of the shallow (<20 m) piezometers within the sandstone aquifer and overlying drift deposits located adjacent to the river banks and across the northern section of the Tame Valley. Data were obtained from these piezometers on occasion throughout the project during sampling runs or during investigation of groundwater/surface water interactions.

4.6.1 Measurement procedure

Measurements of the depth of the water table below the piezometer collar were obtained using a standard electronic dip meter. Groundwater head values were then calculated by subtraction from the collar elevations. Water level measurements are considered to be accurate to within +/- 0.5 cm.

In order to allow detailed (continuous) temporal investigation of transient groundwater and surface water interactions, four pressure transducer and logging systems were designed and constructed (Section 4.5.3). Measurements of river stage and head from piezometers on the adjacent river bank were collected at five minute intervals over several two-week periods from different locations (Appendix 12).

The water level within each piezometer was recorded prior to the insertion of the pressure transducer and subsequent to the removal of the transducer. Upon removal it was necessary to allow sufficient time (> 5 minutes) before dipping to enable the recovery of the water level as the transducer comprised a significant portion of the volume of the standing water column in the narrow 20 mm ID piezometers.

Water level was calculated for the set-up by subtracting the depth of the transducer from the collar elevation and then adding the recorded head value from the transducer. The dipped level recorded at the start was taken as the actual water level and if necessary used to apply a correction to the transducer reading. The correction was sometimes necessary because the transducer does not hang freely within the narrow piezometer.

Transient water levels within the river were monitored using the pressure transducer clamped securely inside a 30 mm ID section of polyethene pipe with nylon cable ties. This pipe was fixed securely to the wire gabions that reinforce the channel sides in many places. Owing to the possibility of theft or vandalism it was necessary to make the equipment as unobtrusive as possible. The data logger and power pack for the system were concealed within dense undergrowth on the channel side.

Head measurements within the riverbed piezometers were possible by direct observation of the water level through the semitransparent HDPE tubing. Relative differences between piezometer head and river level were measured by placing a section of 20 cm ID clear plastic tube around the piezometer to isolate the river surface and reduce the effect of river flow and surface ripples. River levels and hence piezometer levels were calculated by comparing them with a known benchmark on the riverbank.

4.6.2 Borehole locations and survey data

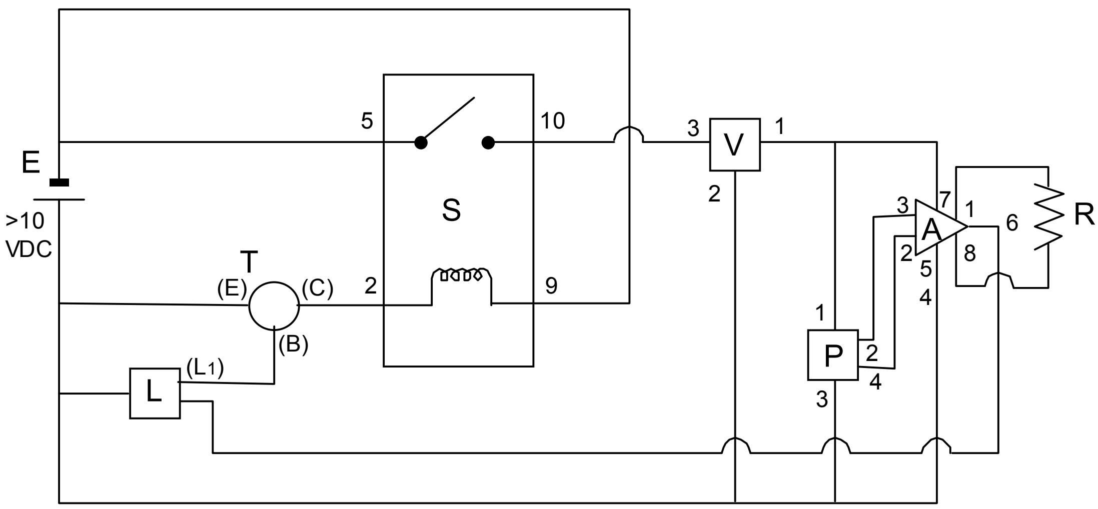
The details of the Severn Trent piezometers included collar locations and elevations according to ordnance datum. These boreholes were used as fixed reference points from which to survey in the flood defence boreholes, riverbed piezometers and river stage markers using a tape and prismatic level with stadia. Maps at 1:1250 scale were available from Severn Trent showing borehole locations. Environment Agency maps were available at a scale of 1:1000 showing the River Tame and environs and marking the location of surveyed river sections and flood defence boreholes.

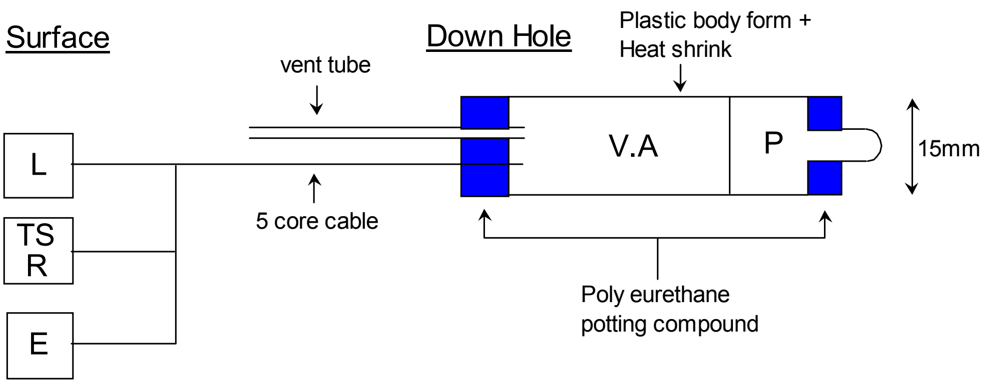
4.6.3 Pressure Transducer Logging System

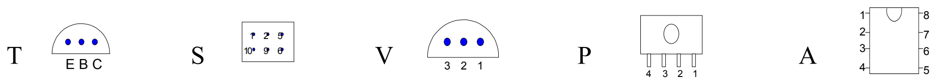
To obtain information on groundwater and river interaction it was necessary to collect continuous head data from both the river and adjacent bank-side piezometers. The cost and the size restrictions of the piezometers (ID 20mm) precluded most of the commercially available pressure logging systems. The urban setting meant a high risk of theft /vandalism necessitating the concealment of all equipment beneath a standard size 13x13 cm metal stop-cock cover. It was decided to construct a pressure transducer and logging system that would meet the space requirements and have low component cost in case of loss. A benefit of in-house construction was the ability to adjust the transducer range and resolution to each individual site.

The voltage recording data loggers and pressure transducers were readily available from parts catalogues and construction was simple (Figure 4.5) and achieved by soldering on to single-sided circuit board. Surface components consisted of the power source (two 9V batteries) and the voltage data logger (either an 8-bit or 16-bit device) housed in accessible waterproof containers. Downhole components comprising the transducer unit were waterproofed by encasing in heat-shrunk plastic and sealing with polyurethane potting compound. After construction, the transducers were calibrated by plotting voltage versus head which showed a linear relationship. Sensitivities of 1cm to 1mm over a 3m range in water level depending on whether the 8-bit or the 16-bit data logger was used. The 8-bit logger had a trigger output

Figure 4.5 Pressure Logging System







Component Connections

E - DC power supply > 10 VDC [in this case 2 x 9V DC batteries in series]
L - Data logger [Gemini Tiny Talk (8 bit) voltage input,0-2.5 VDC (RS#TK-0702)*[1], resolution 10 mv] L1 – trigger output
Or [ERTCO Volt101 (16 bit), 0-15 VDC ,no trigger (CP#U-38010-25)*[2], resolution 0.5 mv]
T - Darlington small signal transistor [(RS#157-7157)] Gemini logger only
S - Relay [Omron G5V-1, 5 VDC (RS#369-343)]
V - Voltage Regulator 10 V [(RS#298-7151)]
P - Pressure Transducer 0-5 psi [gauge, compensated, 0-50mv, (RS#286-6580]
A - Amplifier [Burr-Brown, single supply micro power instrumentation amplifier (RS#285-8126)]
R – Resistor to set the gain [Gain = 5+ 200k/R, used 4.7k but this may be altered for greater sensitivity of measurements over a smaller range.]

*[1] Radio Spares UK catalogue number *[2] Cole Palmer UK catalogue number

Components V,P,A are located downhole. For this system a range of 3.3m provides a resolution of 1 cm with the Gemini and 1 mm with the ERTCO (however the ERTCO has no trigger system requiring power to be on all the time). With a total component price of £100 the whole system can be constructed to fit a 20 mm ID piezometer and be concealed within a 13x13 cm stop cock cover that enabled the transducer to be powered up for only the measurement period, thus preserving battery life. Data loggers are interrogated using PC software and can be used in real time recording mode with a laptop in the field.

The system successfully collected continuous data illustrating the interplay between river flood events and rises in adjacent riverbank groundwater levels. Another use of the pressure logging system was to take head measurements from the riverbed MDP profiles by anchoring the transducer at a fixed point on the bed of the river and connecting by tube in turn to each of the MDPs. The data logger is then monitored in real-time mode via a laptop on the riverbank. The system was used with some success but requires further improvement. It should also be possible to incorporate the system for use in the falling head tests performed on the MDPs.

4.7 Characterisation of the riverbed sediments.

4.7.1 Riverbed sediment coring

A sediment coring tool was developed in order to characterise the river bed materials of the Tame. The tool comprised a 2m length of PVC pipe (ID 10cm) with a shorter (0.5m) section of narrower (ID 9cm) PVC pipe inside acting as a plunger. The mode of operation was to drive the tool into the riverbed and extract a sediment core using suction to prevent loss of the saturated material upon withdrawal of the tool. To generate the suction, a close fit was required between the inner and outer tubes. This was achieved by wrapping plastic sheeting about the inner tube to increase the diameter (ideally O rings would have been used). The ends of the inner tube were sealed and the plastic sheeting secured by the use of two large rubber bungs.

A series of cores was obtained across the river channel from areas adjacent to the riverbed piezometer profiles 5, 1, 8 and 17. A maximum penetration depth of 0.5 m was achieved and the success rate for each coring attempt was ~ 25%. However, the method was simple and easily repeatable, and this allowed sufficient samples to be obtained.

The core tool was driven into the riverbed by hand using a fence-post driver until no further progress could be achieved. The depth of penetration was recorded prior to removal of the tool. The sample was removed from the core tool by pushing on the upper surface of the plunger, and depositing the sample on a plastic sheet. The sample was allowed to drain, clearly labelled and then placed in a wooden core box for transportation to the laboratory. The core was subsequently logged and sampled for F_{OC} analyses (Appendix 13) and grain size distribution analyses (Moylett, 2000, Appendix 14).

The coring process was found to be problematic owing to a number of factors:

1. an armoured surface to the riverbed comprising large clasts made initial penetration of the coring tool difficult;
2. the unconsolidated and saturated nature of the deposits made them difficult to retain in the core tube;
3. high flow velocities in the river led to the wash out of sediments as the core tube was removed; and
4. large cobbles particularly at depth tended to jam inside the coring tool and damage it.

In order to overcome these problems the following methodology was used. The initial 10 cm of material comprising the armoured surface layer was removed by hand. The core tool was designed with a plunger in the barrel above the sediment to develop a suction in order to retain the sample. On reaching maximum depth the tool was slowly withdrawn and prior to complete withdrawal tilted upstream within the hole to as close to horizontal as possible. The tool was then swiftly withdrawn from the river with the open end facing downstream.

4.7.2 Falling head tests in the riverbed piezometers.

Falling head tests were performed to obtain estimates for the conductivity of the river bed materials (Appendix 15). The conductivity values obtained could be used in conjunction with river stage and piezometer head differences to derive a general estimate of flux to the river using the Darcy equation (Section 4.8.4). Given the low cost and ease of installation it was possible to obtain a large number of measurements at different locations to assess the heterogeneities within the riverbed. The MDPs were generally emplaced in the base of the channel not immediately adjacent to the bank sides. However, later modelling work has indicated the likelihood of high flow zones through the bank and seepage faces. Installation of MDPs in these zones may be problematic in some cases because of the presence of rock-filled baskets used to support the channel sides.

4.7.2.1 Slug Test Analyses

The Hvorslev method (Hvorslev 1951) was used to analyse the data collected from falling head tests performed on the riverbed piezometers. The value for the conductivity of the riverbed sediments was derived using the following variant of the Hvorslev equation.

$$K = \frac{r^2 Ln(Le/R)}{2LeT_0}$$

K = hydraulic conductivity cm s^{-1}

r = radius of well casing (0.5 cm)

R= radius of well screen (0.65 cm)

Le =length of well screen (10 cm)

T_0 =Time for water level to fall to 37% of the initial level

The data for each test were plotted as h/h_0 on the y axis (log scale) against time on the x axis.

h = water level at time t, h_0 = water level when t = 0.

A straight line was fitted to the points using the trend line function in Excel to derive two constants describing the line in the form:

$y = ae^{-bx}$

T_0 was then derived using the equation:

$$T_0 = \frac{(Log(0.37/a))}{-b}$$

A spread sheet was used to calculate the value of K for the bed sediments which would contain both vertical and horizontal components. A typical slug test would involve a 50 cm column of water which would penetrate approximately 200 cm^3 of sediment (assuming porosity of 0.2). This is a small volume compared to the river bed as a whole and as a consequence of heterogeneities within the bed sediments a large spread of results is to be expected for piezometers within the same profile.

4.7.3 Grain size analyses by the Hazen and Shepherd methods

The hydraulic conductivity of unconsolidated sediments is directly related to the packing of the particles and the void spaces between them which are a function of the grain size distribution. As median grain size increases so will permeability. An increase in the standard deviation indicates a more poorly sorted and less permeable sample. Coarse sediments are more sensitive to the degree of sorting than fine sediments, and unimodal samples have greater permeability than bimodal samples (Fetter, 1994).

The riverbed sediments of the Tame comprise unconsolidated sand and gravel which are easily subjected to grain size analyses. Samples were selected from the material obtained during riverbed coring. The samples were dried before being passed through a standard series of sieves and the weight of each fraction recorded. A grain size distribution was then constructed by plotting grain size versus percent finer by weight and subjected to the analyses outlined below.

The uniformity coefficient is the ratio of the grain size that is 60% finer by weight to the grain size that is 10% finer by weight. This was calculated for each sample to give a measure of the degree of sorting.

$Cu = d60/d10$ $Cu < 4$ = well sorted $Cu > 6$ = poorly sorted

The Hazen method (Hazen, 1911) may be used to estimate the hydraulic conductivity for sandy sediments from the grain size distribution curve. The method is appropriate when the grain size that is 10% finer by weight is between 0.1 and 3.0 mm. The Hazen approximation is:

$K=C(d10)^2$

K = hydraulic conductivity (cms^{-1})

d10 = the effective grain size, (10% finer by weight) (cm).

C = coefficient based on sorting see Table 4.2, ($cm^{-1}s^{-1}$)

Table 4.2 Representative values of the Hazen coefficient for different grain sizes and degrees of sorting (Fetter, 1994).

Sediment character	Range of 'C' coefficient ($cm^{-1}s^{-1}$)
Very fine sand, poorly sorted	40 – 80
Fine sand with appreciable fines	40 – 80
Medium sand, well sorted	80 – 120
Coarse sand, poorly sorted	80 – 120
Coarse sand, well sorted, clean	120 – 150

Additional work has been carried out by Shepherd (1989) in which he developed a relationship between grain size and conductivity based on field and laboratory data from 18 published studies. The general formula for the relationship is

$K = Cd_{50}^{j}$

K = hydraulic conductivity (feet day^{-1})

C = a shape factor

d_{50} = the mean grain diameter (mm)

j = an exponent

The shape factor and exponent (Table 4.3) are greatest for texturally mature sediments which are characteristically well sorted with uniformly sized particles with a high degree of roundness and sphericity.

Table 4.3 Representative values of the Shepherd shape factors and exponents for different levels of sediment maturity (hydraulic conductivity values derived will be in feet day^{-1}).

Sediment Character	Shape factor ‘C’	Exponent ‘j’
<u>Texturally mature</u>		
Glass spheres	40000	2
Dune deposits	5000	1.85
Beach deposits	1600	1.75
Channel deposits	450	1.65
Consolidated sediments	100	1.5
<u>Texturally immature</u>		

4.8 Riverbed Temperature Survey

Temperature gradients exist in groundwater. Groundwater temperatures at depth are relatively constant when compared to the fluctuations in surface water temperatures that vary on several time scales from daily to annual. Previous workers (Silliman et al.,1995), have used the high contrast between summer and winter groundwater and surface water temperatures from direct measurements to identify areas of groundwater discharge within a river reach and over larger areas by remote sensing techniques (Souto-Maior, 1973). At a smaller scale, the temperature contrast and the response to surface water temperature fluctuations have been used to identify the extent of the hyporheic zone within the riverbed for a pool and riffle system (Evans et al., 1997).

4.8.1 Lateral temperature profiles

Several temperature profiles were measured at different locations laterally across the Tame and at one short longitudinal section within 2m of the river bank (Appendix 16). Lateral profiles were measured at 0.5m intervals and the longitudinal profile at 10m intervals. Two temperature measurements were taken at each location, one at a depth of 10cm below the stream bed and one in the river water immediately above the riverbed. The two measurements were then compared graphically to give a qualitative estimate of where groundwater inflow was occurring based on the temperature difference. Measurements were taken during the summer at which time the surface water temperature was expected to be considerably warmer than the average groundwater temperature of $12^{o}C$ as measured during sampling of abstraction wells in the Birmingham Aquifer (Jackson, 1981, Ford, 1990).

A specially constructed temperature probe was used for the survey. Difficulties were encountered in driving the probe into some sections of the river and temperature readings were slow to stabilise making the procedure time consuming. Unrealistic increases in temperature for both the surface and groundwater were noted in one profile which may have resulted from heating of the apparatus by direct solar radiation. Therefore, the most meaningful data were thought to be the temperature gradient between the surface water and the sediment water rather than the absolute temperature values. The method provided a qualitative method that suggested the inflow of colder water through the bed sediments.

4.8.2 Construction of a temperature probe

The temperature probe was constructed from a 2m section of PVC pipe (ID 20mm) with a steel drive point mounted on one end. A series of holes were drilled to form a screened section between 5 and 10cm above the drive point. A K-type thermocouple with a 2m extension was inserted into a protective plastic tube (2mm ID) leaving the two ends exposed. The active end of the thermocouple was then located within the screened section and sealed from the empty tube above using a waterproof sealant. The upper end of the thermocouple was then connected via a plug to a hand-held thermocouple reader.

4.8.3 Estimating groundwater flow from vertical temperature profiles

Several workers have investigated the use of vertical temperature profiles within groundwater to calculate vertical flow velocities. Bredehoft et al. (1965) developed a steady state analytical solution describing the vertical flow of heat and groundwater through an isotropic homogenous, fully saturated semiconfining layer by use of a type curve method. Silliman et al. (1995) consider the characterisation of downward flow velocities by the temperature response of shallowly buried (5-15 cm) thermistors to variable forcing temperatures applied at the surface of the riverbed. Silliman et al (1993) discuss the three extremes of flow that may occur within the river bed sediments.

1. The river is strongly gaining groundwater and sediment temperature will remain constant dominated by groundwater advection.
2. There is zero flux through the sediments and temperature will be dominated by conduction of the daily fluctuations of the surface temperature of the sediments.
3. There is downflow from the river to the subsurface, and sediment temperatures will reflect surface variations with a phase lag and a reduced amplitude which may be characterised by the advection and dispersion (conductance) equation.

These solutions consider only one dimensional flow and ignore other sources of thermal energy such as biological activity or chemical reactions within the sediments. The solution of Bredehoeft et al. (1965) was used to calculate a vertical flux from temperature measurements taken from a series of multilevel piezometers within the riverbed (Appendix 16).

The one dimensional equation for vertical heat and fluid flow may be written as follows:

$$\frac{\partial^2 T_Z}{\partial z^2} = -(c_0 \rho_0 v_z / k)(\frac{\partial T_z}{\partial z}) = 0$$

C_0 is the specific heat of water = 1.0 cal.g^{-1}, (4.18 J.g^{-1})

P_0 is the density of water = 1g.cm^{-3}

Vz is the vertical flow rate cms^{-1}

K is the thermal conductivity of the saturated sediments

The value used was for a Hudson River sand (Clark, 1966) with a moisture content of 30% and a value of 3.94 x 10^{-3} cal.$cm^{-1}s^{-1}.^{o}C^{-1}$ (1.65 x 10^{-2} J.$cm^{-1}.s^{-1}.K^{-1}$). This may be compared with the value of 2.3 x 10^{-3} cal.$cm^{-1}s^{-1}\,^{o}C^{-1}$ (9.62 x 10^{-3} J.$cm^{-1}.s^{-1}.K^{-1}$) used by Silliman (1995).

Equation 1 may be solved for the following boundary conditions (Figure 4.6)

T_z=T_0 at Z=0 and T_z=T_L at Z=L

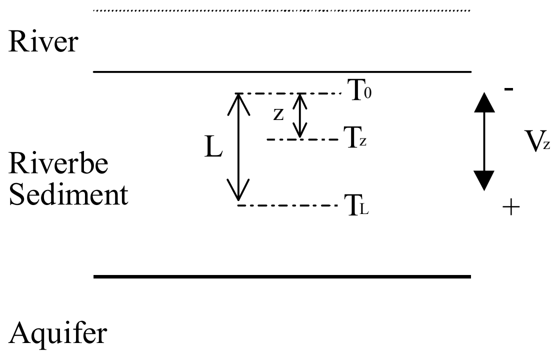


Figure 4.6 Schematic diagram representing variables in the steady state temperature calculation

T_z = the temperature at depth z

T_0 = the uppermost temperature measurement

T_L = the lowermost temperature measurement

L = the length of the vertical section over which temperature measurements extend

The solution is:

$$(T_z - T_0)/(T_L - T_0) = f(\beta, z/L)$$

where

$$f(\beta, z/L) = [\exp(\beta z/L) - 1]/[\exp(\beta) - 1]$$
$$\beta = c_0 p_0 v_z L/k$$

This may be calculated using a type curve fitting method as described by Bredehoeft (1965) or as in this case by using the solver function in an Excel spreadsheet to obtain the smallest error by varying Vz.

Investigation of the vertical temperature profile within the river bed was undertaken using measurements within the multilevel piezometers of profiles 5,8 and 7. Measurements were taken at the base of each piezometer using a 2 metre long K type thermocouple. Temperatures of the surrounding river water and any adjacent river bank piezometers were also taken.

4.9 Analyses of the groundwater contribution to baseflow.

The contribution of groundwater to baseflow was estimated on the catchment scale by the analyses of river gauging station data and on the local scale based on field data from investigations within the riverbed.

4.9.1 Analyses of river hydrographs

River flow comprises several components and the proportion of total river flow contributed by each will fluctuate with time. The following components contribute to the flow in the River Tame:

Run-off relatively short term component associated with precipitation events;

Interflow subsurface flow derived from a precipitation event which reaches the river without recharging the underlying groundwater system;

Bank storage the release after a flood event of river water that has entered the adjacent sediments with head gradient reversal during the flood event. This is at maximum during the falling limb of the flood hydrograph. An effect similar to bank storage may occur with the release from storage of groundwater unable to discharge during the flood event;

Baseflow is the long term discharge that sustains the river during dry weather flow (DWF) periods. This will contain a major component of groundwater discharge from the underlying bedrock aquifer and drift deposits. Anthropogenic inputs including treated sewage effluent and industrial discharges also comprise a significant component in this urban catchment.

The identification of the groundwater component of river baseflow is best accomplished under dry weather flow conditions when run-off and interflow are minimal. The groundwater discharge will fluctuate, depending on the regional head conditions. These will vary over time, and are dependent on recharge and abstraction rates. Groundwater discharge follows an exponential decline or recession during DWF until a recharge event occurs marking the start of a new recession period. This recession may be described (Fetter, 1994) by the expression:

$$Q=Q_oe^{-at}$$

Q= discharge at time t (m^3s^{-1})

Q_o= flow at the start of the recession (m^3s^{-1})

a= basin constant ($days^{-1}$)

t = time since the beginning of the recession (days)

Butler (1957) used a semilog plot of discharge against time to derive a near straight line for baseflow recession. He used the recession equation in the form:

$$Q = \frac{K_1}{10^{t/K2}}$$

K_1= groundwater discharge at the start of the recession (Q_o)

K_2= time increment corresponding to one log cycle change in discharge

Q = discharge

t = time since start of the recession

By integration of this expression it is possible to derive the total discharge over a time period (t_1 to t_2)

$$Total\ Discharge = \int_{t1}^{t2} Q dt = \left[\frac{-K_1 K_2 / 2.3}{10^{t/K2}} \right]_{t1}^{t2}$$

Meyboom (1961) used the integral between time equal to zero and infinity to derive an expression for the total potential groundwater discharge (Qtp) for the recession period.

$Qtp = K_1 K_2 / 2.3$

If the actual discharge from a recession is subtracted from the total potential discharge this leaves the remaining potential discharge (Qrp). If Qrp is subtracted from the new Qtp of the next recession following a recharge event then the recharge of this event may be calculated. An assessment of this method and several others was carried out by Mau et al. (1993) and it was found to give reasonable results. He noted that the calculated recharge for the year was generally 25% higher than the groundwater baseflow estimates and concluded that baseflow did not equate to recharge as is sometimes assumed. These losses may have been related to evapotranspiration from riparian vegetation or recharge to deeper groundwater flow systems that do not discharge to the river. An attempt was made to calculate recharge using this method for two rainfall events of different intensity but the results contained a high degree of uncertainty and were not used. Data from more than one rainfall gauge within the catchment are required and estimates for multiple events are necessary to give meaningful information on the aquifer recharge. This is beyond the scope of this project.

The determination of the time t_0 for the start of the recession was taken from the first minimum to occur in the recession after run-off and interflow cease (Meyboom, 1961). This was compared against the empirical formula (Table 4.4) for the length of time over which overland flow occurs following a precipitation event (Fetter, 1994). This is based on an arid catchment, which may have similar high levels of run-off to an urban catchment.

$D = 0.827A^{0.2}$

D = days from storm peak A = drainage basin catchment area,

Gauging station	Catchment area (km^2)	Time to end of overland flow (days)
Bescot	196	2.38
Calthorpe (Rea)	74.3	1.96
Water Orton	408	2.75

Table 4.4 Time for overland flow to cease after a rainfall event

Environment Agency data from gauging stations at Bescot, Calthorpe and Water Orton were inspected and recession periods selected for analyses during periods of dry weather flow to avoid the impact of run-off. Exponential decay curves and semilog straight lines for the two different methods were fitted through the minimum values for each day to reduce the impact of intermittent sources such as the sewage treatment works (Appendix 17). The basin constants derived for each recession were compared and average values taken. These constants allow calculation of baseflow after any time period and are useful in calculating the maximum and minimum values to be expected for drought or high baseflow scenarios. The basin constants may be compared between catchment areas and between different time periods (years) to assess variability in the hydrogeological regime.

The baseflow component for the 24 km section of river between the gauging stations may be derived by subtracting the upstream baseflow (Bescot G.S.) and the major tributary baseflow (Rea at Calthorpe G.S.) from the baseflow at the downstream end (Water Orton). Further subtractions must be made for known industrial discharges and smaller tributaries. Data for these additional discharges are difficult to find as industry seldom discharges continuously at its consent levels and tributary flow is variable. Other unknown inputs will most certainly be incorporated within the baseflow estimates in addition to the groundwater contribution.

The 23.8 km of river channel between the gauging stations incorporates groundwater inputs from the following underlying geological units:

Mercia mudstone (7.8 km);

Triassic sandstone (7.4 km):

Carboniferous coal measures (8.5 km).

These lengths incorporate channel meanders, and straight line distances across the units are Mercia mudstone (7.0 km), Triassic sandstone (4.5 km), Carboniferous coal measures (6.25 km). Superficial deposits comprising alluvial gravel and glacial till overlie the bedrock over the entire length of the study area and may form significant aquifers in their own right, particularly the alluvial gravel in the river valley. The lack of a gauging station at either end of the Triassic sandstone makes it impossible to assign baseflow contributions from the separate geological units. Therefore, a uniform baseflow increment was assigned per metre length of channel basis across the entire reach which possibly underestimates discharge from the sandstone and overestimates it from the Carboniferous and mudstone units, based on the conductivity contrast.

Baseflow statistics were derived for the year 1999 by processing the gauging station data (15 minute intervals) using an Excel spreadsheet to remove periods of river flow impacted by run-off (Appendix 17). The data set was examined and the period of highest baseflow was selected. From the data an upper limit for baseflow discharge was set and a maximum slope of recession determined. Data exceeding these limits were filtered out and general statistics generated on the remaining data. The total time under baseflow conditions was compared to the number of days with rainfall multiplied by an estimate of the time for overland flow to cease in the catchment. An estimation of the total discharge and baseflow discharge for the year was calculated using a simple method of integration between each adjacent point that was considered as baseflow. The method employed was as follows:

Total discharge =

$$\sum_0^n 1/2[(Q_{n+1} - Q_n) * (t_{n+1} - t_n)] + Qn * (t_{n+1} - t_n)$$

Q_n = discharge at time t_n

Mau et al.(1993) list a number of more sophisticated methods for determining groundwater baseflow contributions.

The recession methods of analysis previously discussed consider baseflow discharges after a storm peak has passed and stable conditions of recession have resumed. However previous workers (Kunkle, 1965, Cey et al, 1998) have determined that considerable groundwater discharge occurs during the flood event itself. Hydrograph separation techniques have been employed using natural tracers such as electrical conductivity and oxygen isotope ratios to determine the groundwater discharge component of the flood hydrograph. These previous studies have dealt with rural catchments with a limited number of inputs. Urban catchments are more difficult to assess as they have multiple sources and for this reason hydrograph separation was not attempted on this project.

4.9.2 Seepage measurements

Seepage meters have been used with success to directly measure seepage to and from lakes and water courses (Lee 1977, Lee et al. 1980, Carr et al. 1980). The principle of the type of seepage meter used in this case was the measurement of volume change that occurred within a flexible plastic bag connected to an area of riverbed and confined by the apparatus over a known time period. The seepage can then be calculated by dividing the change in volume with time by the area contained by the meter.

Difficulties were encountered in seating the meter to a suitable depth (10 cm) in coarse gravel bed sediments, and high river flow induced errors in the seepage measurements as found by other workers (Libelo et al 1994, Isiorho et al 1999). The seepage meter did not permit the direct sampling of groundwater quality from depths below the groundwater and surface water interface (10cm). Emphasis was therefore placed on the riverbed mini drivepoint piezometers to be developed as the primary tool for the investigation.

4.9.2.1 Construction of a seepage meter

Seepage meters were constructed according to the basic design of Lee (1977) from 200 litre oil drums cut in half around the circumference. A 12 mm diameter hole was drilled in the top/base of the drum and a water tight cable gland installed (Figure 4.7). A 40 cm section of LDPE tube was inserted into the cable gland and sealed in place. The seepage meter was then driven to a depth of 10 cm within the river bed sediments. An attempt was made to remove all air bubbles from the meter before attaching a plastic food bag filled with 100 ml of water and no air. The bag was attached to the end of the LDPE tube with the aid of a rubber band. At the end of the sampling period the bag was emptied into a measuring cylinder and the new volume recorded.

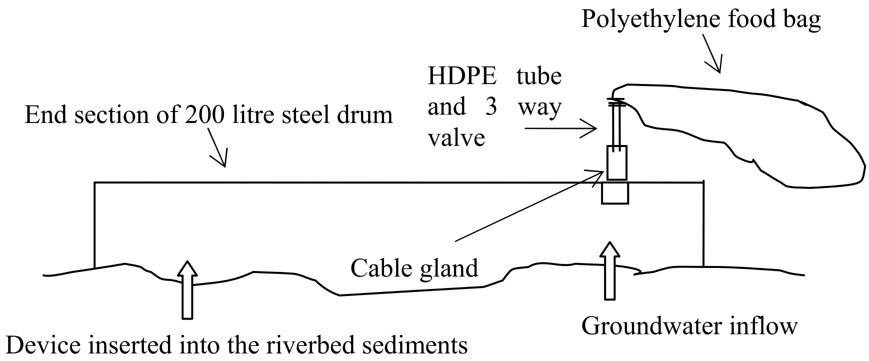


Figure 4.7 Construction of seepage meter (After Dowell, 2000)

4.9.3 Radial Flow Analytical Solution

Radial flow will dominate in the region immediately adjacent to the river. Estimates of the extent of the influence of radial flow from the river are estimated at 50 m (based on half of the saturated thickness for the aquifer (100m). An approximation of flow from the aquifer to the river may be based upon the Thiem equation ,divided by 2, if the radius of the stream (as a semi-circle) is less than the saturated thickness of the aquifer.

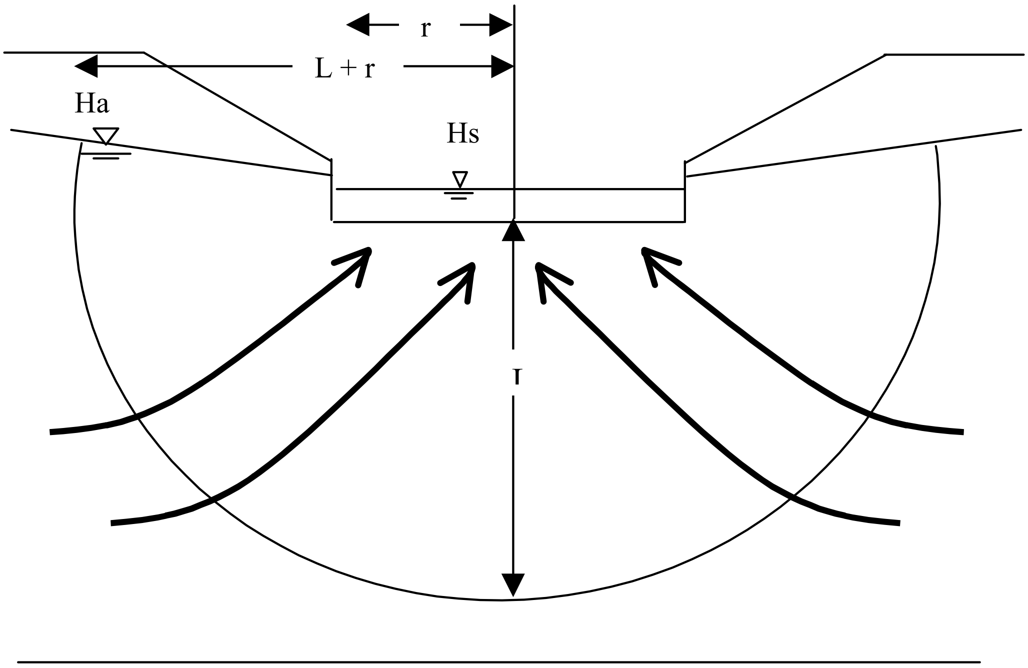


Figure 4.8 Schematic diagram for radial flow calculation

$$Q = \frac{k\pi(Hs - Ha)}{Ln\left[\frac{L+r}{r}\right]}$$

$$r = \frac{2A}{P}$$

Q = total discharge to the river from the aquifer per metre length of river ($m^3\ d^{-1}\ m^{-1}$)

k = conductivity of the aquifer ($m\ d^{-1}$)

Hs = head in the river (m.a.o.d)

Ha = head in the aquifer (m.a.o.d)

L = distance from the river bank to Ha (m)

r = the hydraulic radius of the river (m)

A = the cross sectional area of the river (m^2)

P = the wetted perimeter of the river (m)

The calculation assumes perfect radial flow perpendicular to the section and an isotropic value of k for the aquifer which at this scale is unlikely. A k value of 2 md^{-1} was used as a combined estimate for vertical and horizontal conductivity. There is also an assumption that the river bed sediments do not produce a significant drop in the values of Hs across the bed which would introduce error.

An Excel spreadsheet was used to calculate flow at all points where data were available from piezometers on the riverbank within the zone of radial flow (Appendix 18). The value of Ha was taken from the piezometer and Hs from the river level perpendicular to the piezometer at distance L. Values for P and A were based on survey data from agency flood defence profiles.

4.9.4 River Bed Sediment Controlled Darcy Flow Analytical Solution

The data derived from river bed piezometers may be used to calculate a flux through the river bed (Figure 4.9) using the one dimensional Darcy flow equation :

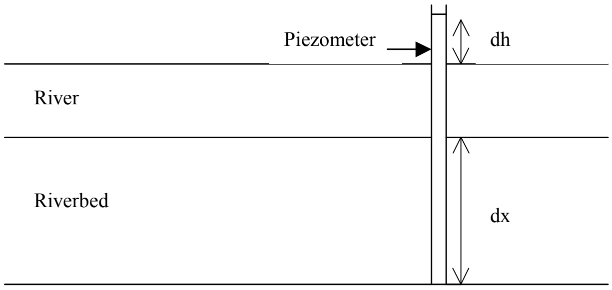


Figure 4.9 Schematic diagram for Darcy Flux Equation

$$q = -k \frac{dh}{dx}$$

q = the specific discharge (md^{-1})

k = conductivity of the riverbed material (md^{-1})

dh = the head difference between the river and the piezometer -steady state conditions assumed (m)

dx = distance from the piezometer screen to the river bed (m)

The values of k were determined from falling head test data analysed with the Hvorslev method. The value of k derived using this method has been taken as representative of the vertical conductivity but it will contain a significant horizontal component which, with a common anisotropy ratio of 10:1 (kx:kz), will have led to an over estimation of the vertical conductivity. The value of dh was derived at single time point under an assumed steady state condition. The data were processed with an excel spreadsheet to derive a flux estimate for each river bed piezometer (Appendix 18).

4.10 Sample analyses.

4.10.1 Chemical analyses

Sample analysis was undertaken at several different laboratories by different agents including the author, and laboratory technicians. A summary of the analyses undertaken is presented in Table 4.5. A description of each analyses is included in appendix 19 where the most detailed descriptions are provided for those analyses undertaken by the author. The results are presented in appendix 20.

4.10.2 Measurement of Field Parameters

Field measurements for pH, Eh, and conductivity were taken within a 300 ml plastic beaker filled at the start of the sampling run. The probes were agitated within the sample and then allowed to stabilise over a five-minute period. An alkalinity titration was undertaken on a separate 100 ml of filtered water. Dissolved oxygen analyses were undertaken using either a meter or via the Chemets ampoule colour change system. A detailed description of the methods and instruments used can be found in appendix 21.

4.10.3 Sample preparation and storage

Samples for inorganic analyses were filtered in the field with a syringe through 0.45 micron nitro-cellulose filters. An initial 30 ml of sample water was used to rinse the sample bottle prior to filling. Cation samples were acidified in the field with 0.5ml of nitric acid to keep the ions in solution. Cation and anion samples were collected in 125 ml plastic bottles with no head space. Samples to be analysed for mercury were collected in 250 ml glass bottles

Determinands	Method of analyses	Analysed by	Laboratory
Anions – F, Cl, NO_3, SO_4	Dionex Ion Chromatograph	Author	Environmental Health, Birmingham University
Ca, Cu, Cd, Cr, K, Mg, Na, Zn, Pb, Sr, Mn, Si, Fe, Ba, Al, Ni, B	ICP-AES	Technician	School of Earth Sciences, Birmingham University
Chlorinated solvents – PCE, TCE, TCA, TCM, CTC	ECD – Gas Chromatograph	Author	School of Earth Sciences, Birmingham University
Six toxic metals – Cu, Cd, Cr, Pb, Ni, Zn	ICP-MS	Technician	Environment Agency, Nottingham
Hg	Hydride generation and fluorescence spectroscopy	Technician	Environment Agency, Nottingham
As	Hydride generation and hydrogenolyses	Technician	Environment Agency, Nottingham
VOC – scan of extended volatile suite (52 compounds)	GC – MS	Technician	Environment Agency, Leeds
Vinyl Chloride	GC-MS	Technician	Groundwater Protection and Remediation Group, Sheffield University
Fraction of organic carbon (FOC)	LECO CS225 Carbon/Sulphur analyser	Technician	T.E.S. Bretby of Burton-on-Trent.
Dissolved Oxygen	Chemets titration, and D.O. meter	Author	Field site
Alkalinity	Hach Titration	Author	Field site

Table 4.5 Summary of the methods of chemical analyses undertaken.

containing the fixing agent potassium dichromate. VOC samples for analyses by the ECD method were collected in 137 ml amber glass bottles with teflon septa. VOC samples for broad scan analyses were collected in 40 ml clear glass vials with teflon septa. VOC samples were collected with no head space and stored upside down to minimise loss through the septa . The VOC samples were analysed within two weeks of collection. All samples were kept in an insulated cool box in the field prior to transfer to a fridge on return to the laboratory at the end of each day.

4.10.4 Precision and Accuracy of Inorganic Analyses

The accuracy of the inorganic analyses was checked by comparing the milliequivalents of the major cations and anions to derive the electroneutrality as the percentage ionic error of the solution (100*(Cations-Anions)/(Cations+Anions)). Upon examination of the data a series of negative ionic errors were identified associated with a malfunction in the ICP equipment during March 2001. The cation results from the surface water sampling undertaken in February have been discarded as a result of this. The error in the data was sporadic and only detected as result of sudden changes observed in the Ca content that should show only gradual variation along the surface water profile. The malfunction was not detected in subsequent analyses.

Average errors $(((\text{ionic error})^2)^{0.5})$ for each sample type were determined as abstraction wells +/-3.5%, shallow groundwater +/-6.9%, the riverbed piezometers +/-6.4%, and the surface water +/-5%. A link was found between sites of high contamination and increased error. Negative errors were found to be reduced when the effects of additional contaminants were included in the cation balance. A comparison of results for repeat sampling of a location at different times was undertaken and identified a sporadic under-estimation of alkalinity (>150 mgl^{-1} as $CaCO_3$). The alkalinity tests were usually conducted in the field to reduce the likelihood of carbonate precipitation and the error is likely to be associated with the titration process. In some cases positive errors were seen in association with high levels of SO_4 and NO_3 and it is suspected that some microbial reduction took place during sample storage, which was over a period of several weeks in some cases. A decrease in nitrate levels was observed when a set of analyses were repeated after a 5 day interval and smaller peaks on the ion chromatograph were often observed which may indicate the presence of sulphite and nitrite. Further investigation would be necessary to confirm this.

For the available 2000 and 2001 data set for which all cation and anion analyses were available, the mean error $(((\text{ionic error})^2)^{0.5})$ was +/- 5.5% (median error +/- 3.6). The error is considered acceptable for results derived from three different methods of analyses (ion chromatography, field alkalinity titration and ICP). The precision of each method of analyses is as quoted by the laboratory and verified by a number of repeat and standard analyses. The typical errors are given in Table 4.6.

Type of analyses	Typical error
Alkalinity	+/- 4%
Cations – ICP	+/- 3%
Anions – Dionex	+/- 10%
Chlorinated Solvents – ECD	+/- 10%
Environment Agency Metals	+/- 10%
Environment Agency VOCs	+/- 10%

Table 4.6 Typical errors for the different methods of chemical analyses

CHAPTER 5. GROUNDWATER FLOW TO THE RIVER TAME

The aim of this chapter is to present the analysis and interpretation of field and archive data to quantify the groundwater discharge to the Tame and to provide background information for the groundwater modelling presented in the next chapter. Estimates are derived for groundwater discharge that will be used in later chapters in conjunction with water-quality data to provide an indication of the total groundwater contaminant flux to the river from the Birmingham Aquifer.

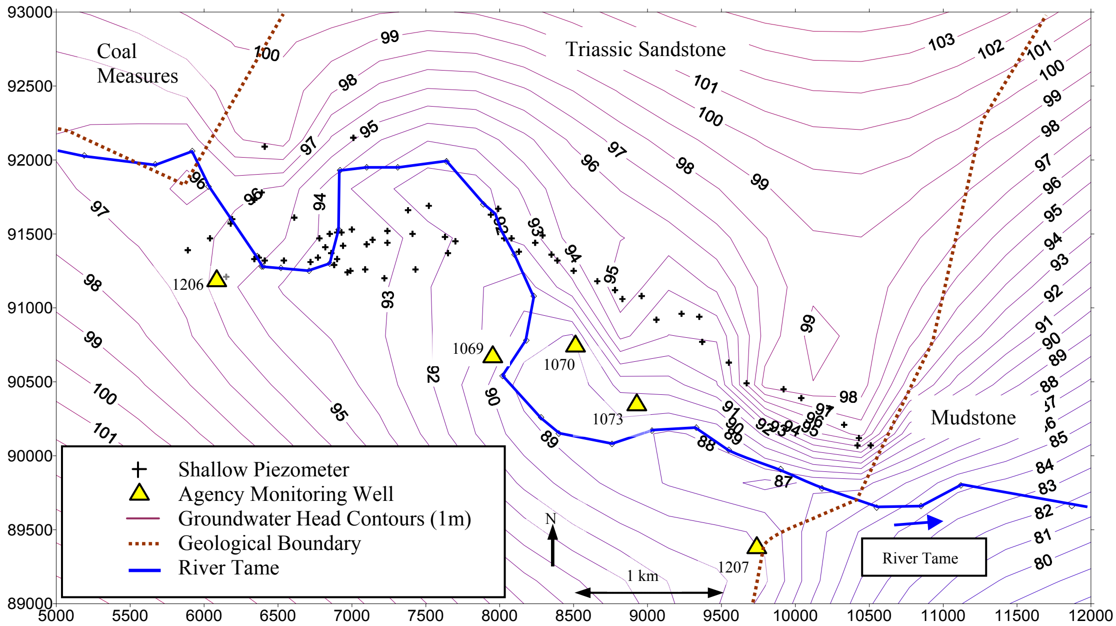
5.1 General Objectives

1. Investigate the groundwater system in the Tame valley
2. Investigate the surface water baseflow and determine the regional groundwater component.
3. Find evidence on a local scale for groundwater discharge through the riverbed.
4. Characterise the riverbed sediments and determine their control on groundwater discharge.
5. Derive estimates of groundwater flow through the riverbed.
6. Investigate groundwater surface water interactions during a river flood event.
7. Undertake a detailed site specific hydrogeological characterisation of a section of river channel and the adjacent river banks to provide information for the construction of a numerical model.

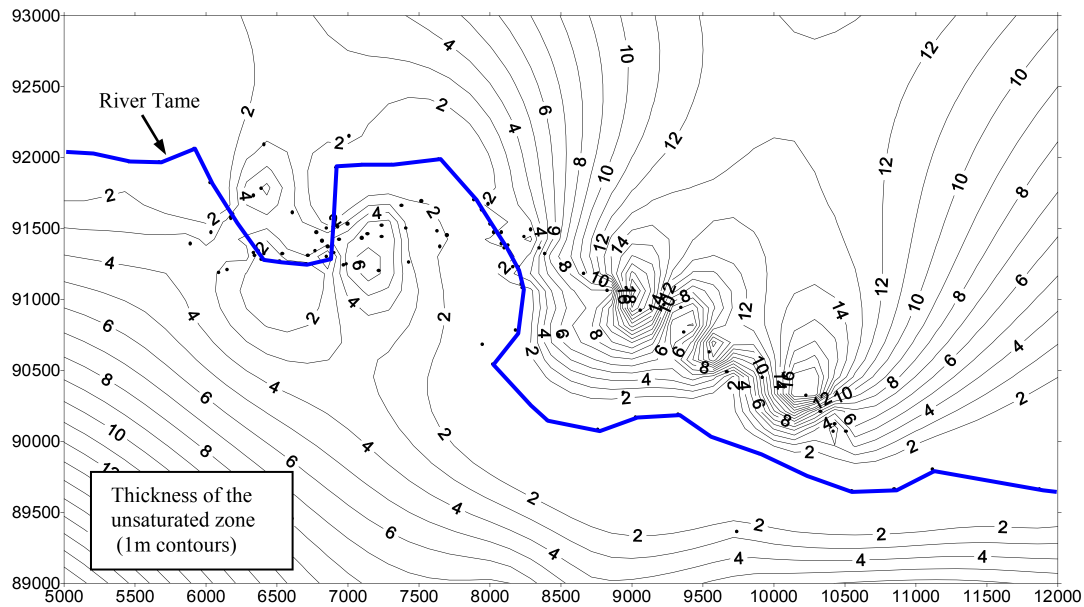
5.2 Groundwater in the Tame Valley

Previous investigations indicate that the River Tame is the natural sink for groundwater discharge from the Birmingham Aquifer. The kriged contour map of groundwater head in the Tame Valley (Figure 5.1a, Appendix 8) constructed using mean head data for shallow groundwater and river level indicates that groundwater flow is directed towards the river. The present course of the river lies close to the northern (left) side of the valley where flow through the aquifer is generally directly towards the river or the major tributary and high gradients are apparent associated with steep slopes in the topography. To the south of the river data are limited except for a detailed profile of shallow piezometers through the flood plain within the river meander centred (OS co-ordinates SP 0750091500). Here groundwater appears to flow across the flood plain away from the upper bend in the river indicating the possibility of a losing section in the river. The depth to the water table (Figure 5.1b) decreases from ~19 m on the sides to ~2 m at the base of the valley where flow will occur through the shallow alluvial gravel as well as the deeper sandstone aquifer. The reduced thickness of the unsaturated zone in the valley bottom may increase the risk of groundwater pollution from industry in this region.

The results of a MODFLOW model of the aquifer for the year 1989 indicate 22.3 Mld^{-1} of groundwater discharging to rivers and streams, 16.4 Mld^{-1} being abstracted and 13.6 Mld^{-1} entering storage (Knipe et al, 1993). The large amount shown to be entering storage in 1989 is a result of the groundwater rebound occurring after a reduction in the previously high levels of abstraction. The current reduced levels of pumping from the aquifer fluctuate about a generally stable trend of around 13.3 Mld^{-1} (Figure 5.2) estimated from Environment Agency borehole return data. Historically, during the period of peak abstraction from the aquifer



(a)



(b)

Figure 5.1 (a) Contours of groundwater head in the Tame Valley (b) Contours of unsaturated zone thickness.

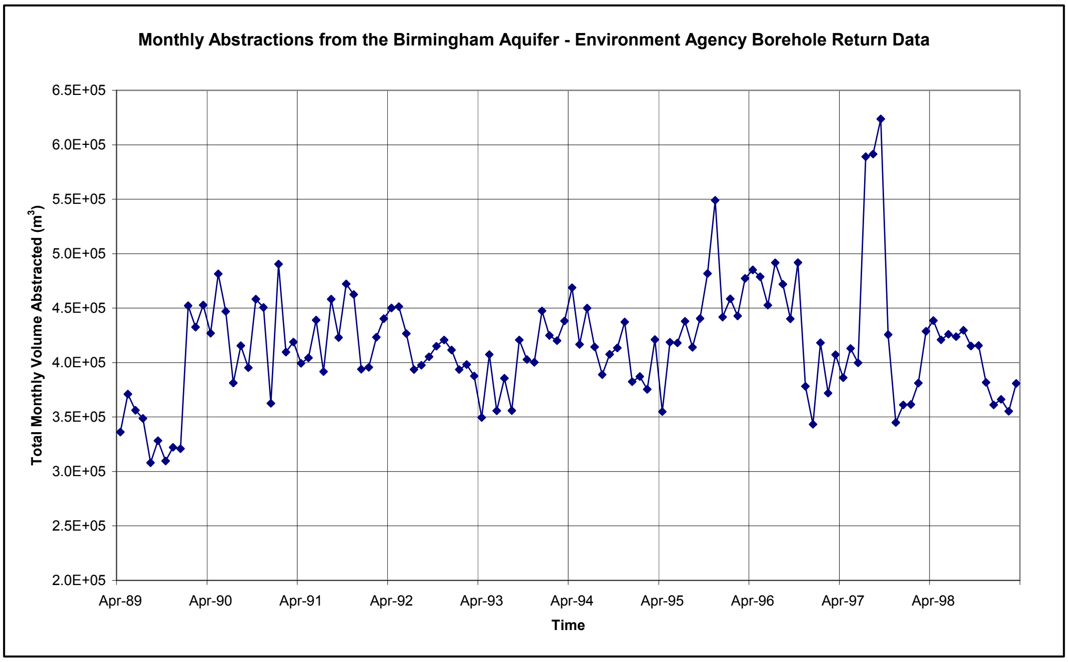


Figure 5.2 Monthly Abstractions from the Birmingham Aquifer

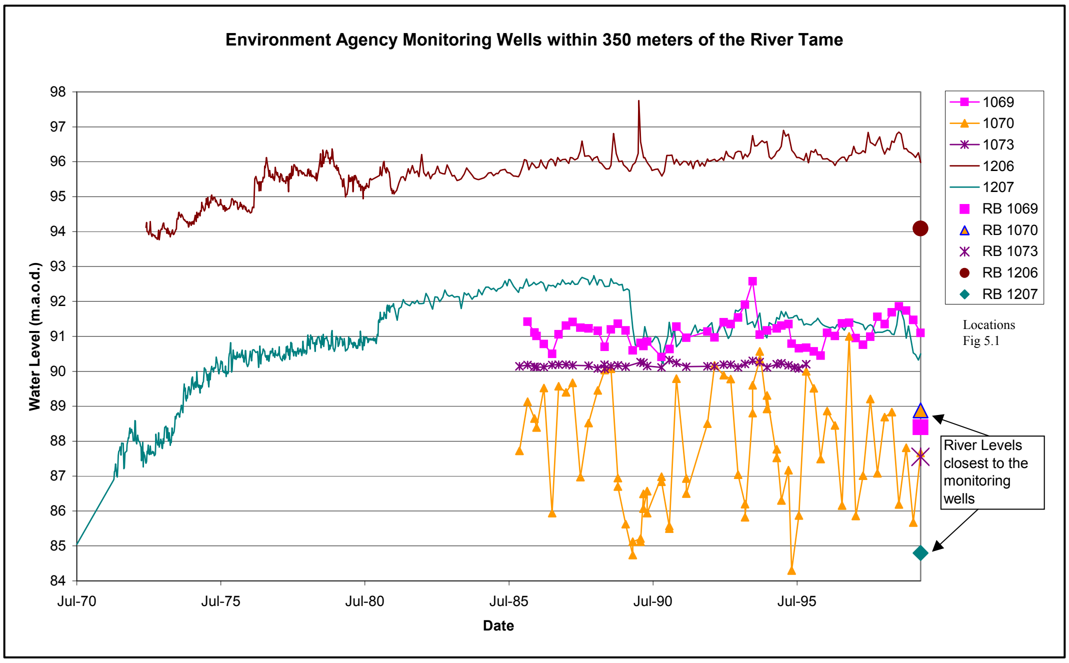


Figure 5.3 Historical water levels within 350 metres of the River Tame

(1900-1960) groundwater discharge to the river was much reduced and in some cases groundwater head was below river level, leading to the infiltration of surface water into the aquifer. Environment Agency monitoring well hydrographs (Figure 5.3, Appendix 4) indicate that by 2000 water levels in the Tame Valley had stabilised after rebound with only small variations due to local abstraction rates and seasonal recharge. Because of this rebound, groundwater discharge to the river has increased and stabilised at its highest mean level for the past 100 years, albeit with some variability resulting from seasonal fluctuations in the water table and river stage.

There is strong evidence that groundwater is discharging to the Tame on a regional scale across the Birmingham Aquifer and in order to quantify the discharge a study of the river baseflow was undertaken.

5.3 Investigation of the Surface Water Baseflow

5.3.1 The Surface Water Balance

Urban rivers are a complex system with multiple, time varying inputs from both natural and industrial sources, and it is difficult to know the exact contribution from each source at any one time. Environment Agency estimates (Davies, 2000, and Crabtree et al., 1998) of average dry weather flow and industrial discharges, from field data and catchment modelling (SIMCAT), were used to assess the contributions to flow in the Tame (Figure 5.4). The largest point discharges are from sewage treatment works that comprise an average of 30% of the flow at Water Orton and significantly influence the water quality of the Tame.

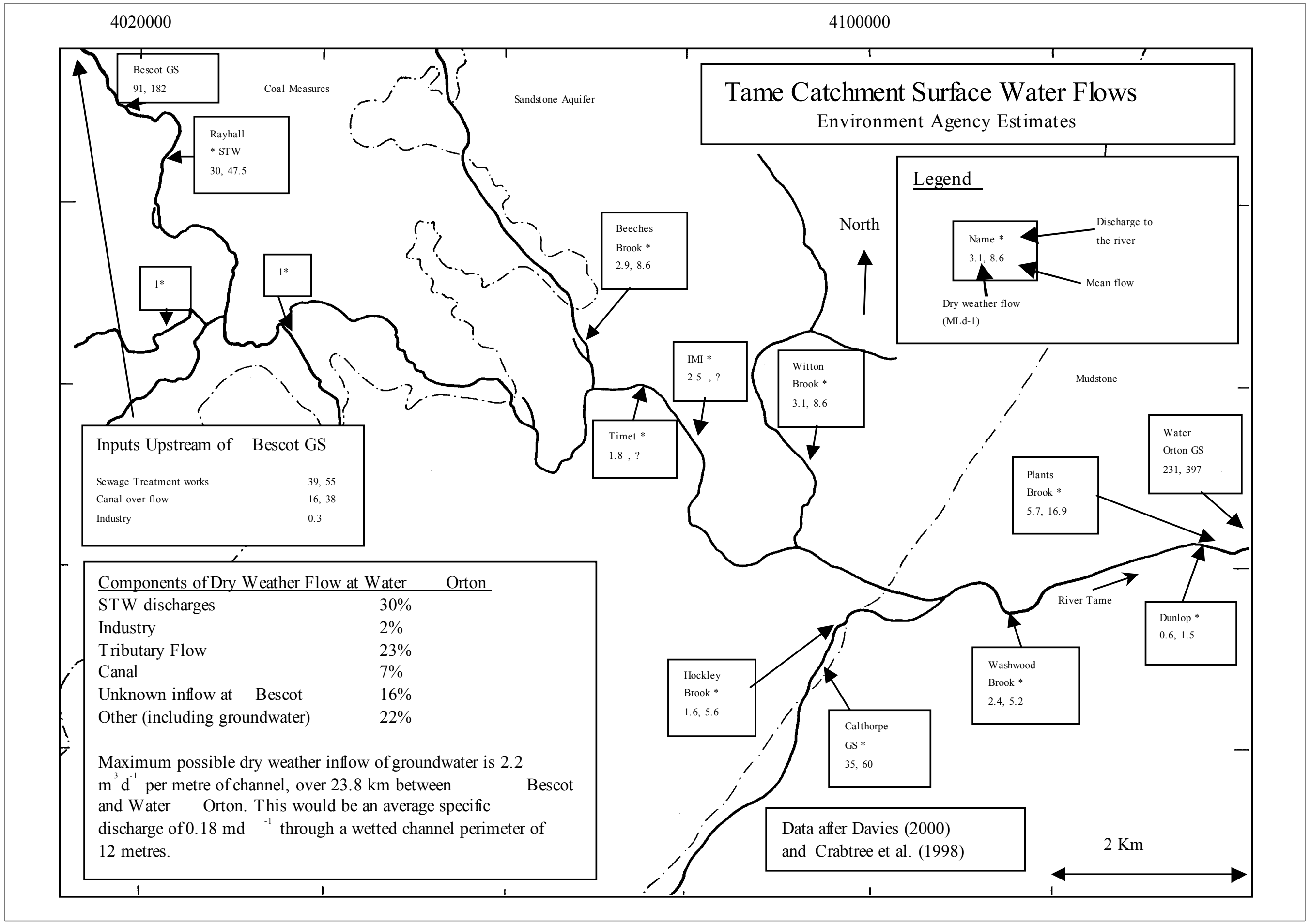


Figure 5.4 The estimated contributions to surface water flow

Diffuse and unknown sources are seen to increase dry weather flow by 22% between Bescot and Water Orton gauging stations. These unknown sources incorporate the groundwater contribution derived directly from the main channel between the two gauging stations. The total catchment groundwater component of the river flow may be even higher, up to 61%, if flow from tributaries is considered primarily groundwater derived. The unknown inputs are likely to include interflow and minor discharges from field drains and sewer overflow, although groundwater is thought to provide the main contribution. Over the entire 23.8km of the reach an average groundwater flow of up to 2.2 m^3d^{-1} per metre of channel may be discharging through the river bed. This equates to a specific discharge of 0.18 md^{-1} through an average wetted channel perimeter of 12 m. This is an average figure for the entire reach and takes no account of variations in the underlying geology. The Birmingham Aquifer would be expected to provide a larger groundwater contribution to dry weather flow than the adjacent and less permeable Coal Measures and mudstones. However the presence of old coal mining voids and shafts may considerably increase the relative groundwater contribution from the Coal Measures. River flow measurements were undertaken to define the baseflow accretion as a result of groundwater discharge specifically from the aquifer and are discussed later (Section 5.3.3).

The Environment Agency estimates are for an average set of dry weather flow conditions, but both surface water flows and the groundwater discharge are time variant. Therefore, an investigation was undertaken to establish the range of baseflow conditions and groundwater discharge to the river.

5.3.2 Baseflow Analyses

Data for 1999 taken from the Environment Agency gauging stations (Figure 5.5) on the Tame at Bescot and Water Orton and on the Rea at Calthorpe show the nature of flow in the heavily engineered catchment. Baseflow ranges between 2 and 6 m^3s^{-1} (173-518 Mld^{-1}) at Water Orton. The baseflow recessions are punctuated by large flood events with discharges ranging up to 106 m^3s^{-1} (9158 Mld^{-1}) associated with heavy precipitation and rapid runoff (Figure 5.6, Appendix 17). The flood events are generally short-lived (< 3 days). This is in line with the calculated duration of overland flow of 2 to 2.75 days. The changes in baseflow reflect the variation in groundwater discharge during the course of the year. The baseflow is seen to decline from high flow during the winter to the lowest flow occurring in July because of changing rainfall patterns (low recharge) and increased evapotranspiration during the summer.

The impact of the sewage treatment works (STW) discharges is seen (Figure 5.7) as a daily cycle on the Bescot and Water Orton hydrographs. Dry weather flow (DWF) in the Rea is primarily supported by groundwater discharge as STWs are absent. This groundwater is derived from the Triassic Sandstone and drift deposits to the west and the drift overlying the impermeable Mercia Mudstones to the east. The STW discharges cause variations in dry weather flows of 15 to 25% (0.4 to 0.6 m^3s^{-1}) over the course of the day at Water Orton. The discharge variations produce a corresponding change in river stage over a range of ~5 cm (assume channel width 12 m, and constant river velocity 0.5 m^3s^{-1}) and groundwater discharge to the river will fluctuate in response to this.

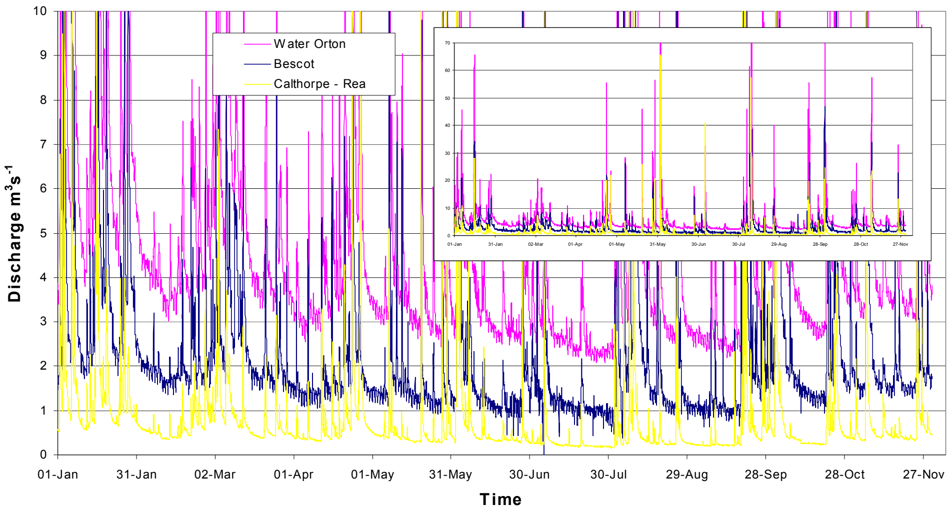


Figure 5.5 River Tame gauging station discharge measurements, 1999.

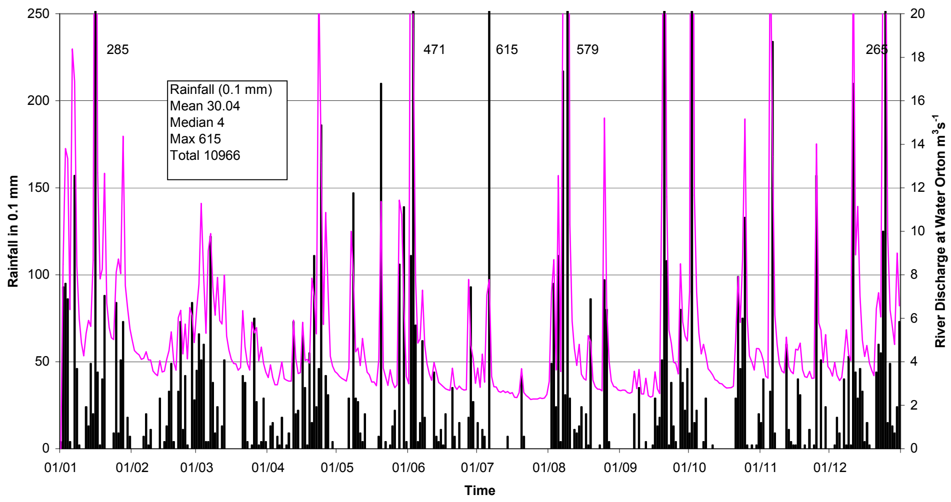


Figure 5.6 The association of rainfall and discharge in the River Tame, 1999.

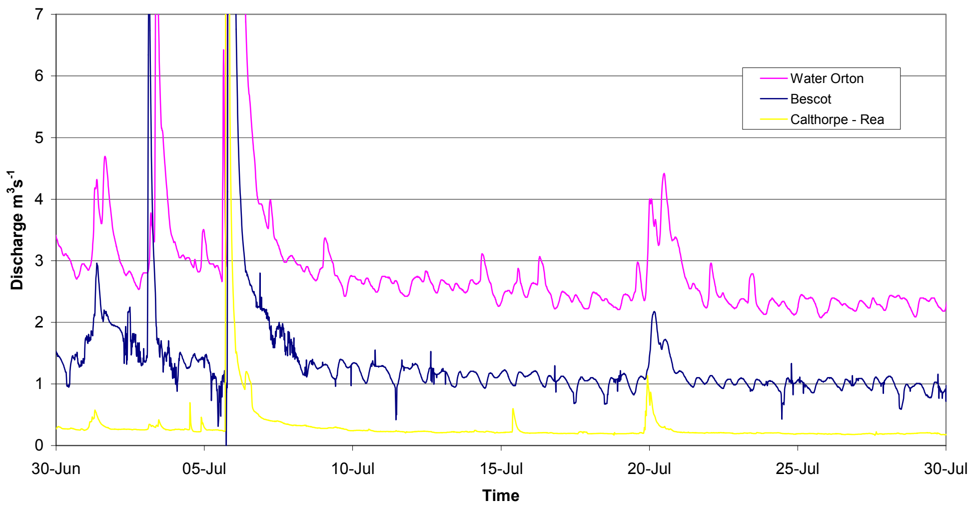


Figure 5.7 Summer baseflow in the River Tame, 1999.

The analyses of the river flow data using the baseflow filter (Section 4.9.1) removed the effect of runoff and wet weather conditions. The range in baseflow is estimated to be from 180 to 518 $Mld^{-1}$with a mean dry weather flow of 308 Mld^{-1} at Water Orton (Table 5.1). Estimates of discharges from known sources (including tributaries) across the study reach are 52.6 Mld^{-1} (dry weather conditions) and 77.2 Mld^{-1} (mean conditions). The increase in baseflow across the reach was obtained by subtracting the mean value of known discharges, plus the flows at Bescot and Calthorpe, from the flow at Water Orton.

Analyses of all discharge data					Baseflow filter applied			
Gauging Station	Mean	Median	Max	Min	Mean	Max	Min	Time under baseflow
	Mld^{-1}	Mld^{-1}	Mld^{-1}	Mld^{-1}	Mld^{-1}	Mld^{-1}	Mld^{-1}	%
Bescot	213	148	4040	27	131	225	48	68
Calthorpe	81	40	5671	13	38	86	13	79
WaterOrton	506	338	9150	180	308	518	180	68

Table 5.1 Baseflow analyses of 1999 gauging station data

This produced a range for the increase in flow of between 41 and 130 Mld^{-1} for the minimum and maximum periods of baseflow. This represents a direct inflow of between 1.7 to 5.5 m^3d^{-1} per metre length of channel (total 23.8 km) and comprises 23% to 25% of the flow at Water Orton. The inflow under mean baseflow conditions is 2.6 m^3d^{-1} inflow per metre length of channel (61 Mld^{-1}, 20% of flow), after the mean values of known discharges have been subtracted. On the catchment scale if tributary flow is considered as groundwater derived, then flow at Water Orton comprises between 40 and 67% groundwater. Clearly, the influence of groundwater contaminants on the dry weather surface water concentrations will vary considerably and will be dependent on the baseflow conditions and changing levels of industrial (STW) discharge. The mean values of baseflow derived using the filter technique were compared with values based on the analyses (Section 4.9.1, Appendix 17) of seven periods (54 days total) of baseflow recession during 1999. The results (Table 5.2) show a reasonable correlation with the recession estimates generally 14 to 18 % lower than the filter estimates.

Gauging Station	Average Baseflow – Recession Method (Mld^{-1})	Average Baseflow – Filter Method (Mld^{-1})
Bescot	113	131
Calthorpe	31	38
Water Orton	260	308

Table 5.2 Comparison of mean baseflow obtained using filter and recession analyses methods.

The effect of industrial discharges was subtracted from the mean baseflow to provide an estimate of diffuse baseflow discharge per unit of catchment area (Table 5.3). The values for Water Orton and Calthorpe are higher than Bescot and may reflect high levels of groundwater discharge from the sandstone and drift, though differences in recharge rates and land use will also be significant. A difference between catchments is also evident from the variation between the basin decay constants derived from the recession analyses (Section 4.9.1, Appendix 17). Water Orton and Bescot have average decay constants of 0.040 (d^{-1}) and 0.041 (d^{-1}) respectively whereas at Calthorpe the constant is 0.053 (d^{-1}).

Catchment Name	Catchment Area (km^2)	Known DWF industrial discharge. (Mld^{-1})	Mean Baseflow (Mld^{-1})	Mean diffuse baseflow discharge per unit area ($Mld^{-1}km^{-2}$)
Bescot	169	55	131	0.45
Calthorpe	74.3	0	38	0.51
Water Orton	408	90	308	0.53

Table 5.3 Estimates of mean diffuse baseflow discharge per unit of catchment area.

The more rapid decline in baseflow at Calthorpe is likely due to a lower storage potential of the smaller groundwater catchment area. As a comparison, the baseflow indices (Institute of Hydrology, 1998) are Water Orton 0.62, Bescot 0.66, and Calthorpe 0.47. These represent the baseflow, 1991-1995, as calculated using the method of Gustard et al. (1992), divided by the total flow. Similar values are derived using the filtered baseflow data for 1999 divided by the total flow, Water Orton 0.68, Bescot 0.68, and Calthorpe 0.56.

The Tame is estimated to be under baseflow conditions for 68% of the year using the filter method. Results from the university rainfall gauging station (Table 5.4) are assumed to be representative of the catchment and indicate that precipitation of >0.1mm per day occurs for 61% of the time. In order for the filter method of baseflow estimation to be correct precipitation below a certain level is assumed not to create sufficient direct recharge and surface run-off to effect baseflow. Inspection of the data (Figure 5.6) indicates that rainfall of <2 mmd^{-1} does not produce a visible effect on baseflow. This is in agreement with run-off calculations by Durr (2003) for the Tame catchment indicating threshold values of 0.5-2 mm of precipitation is required for run-off to occur. Precipitation events of >2 mmd^{-1} occur over 32% of the year. If a two-day duration of overland flow is assumed (Section 4.9.1) then the estimate of time under baseflow estimate of 68% may be an over-estimation. The Tame is assumed to be under baseflow conditions for between 36% to 68% of the year during which time the groundwater component of river flow is at its most significant.

Precipitation data 1999 – University of Birmingham Weather Station				
	Precipitation (mm)	Precipitation > than	Number of days	% of year
Mean	3	> 0 mm	224	61
Median	0.4	> 1 mm	156	43
Maximum	61.5	> 2 mm	117	32
Minimum	0	> 5 mm	57	16
Total	1096.6			

Table 5.4 Summary of precipitation data for 1999 from the University of Birmingham Weather Station.

Integration of the filtered river flow data was used to derive a value of 2.85 x10^7 m^3 for the years total baseflow discharge derived from unknown sources between Bescot and Water Orton. If this value is assumed to represent groundwater discharge and the groundwater is assumed to be in steady state then recharge may be approximated as 28% of the annual rainfall. In the same way, catchment recharge was calculated as 29% at Bescot and 20% at Calthorpe. These values will incorporate leakage from water mains in addition to recharge from precipitation. The 1999 yearly discharge to the main channel, derived from the Birmingham Aquifer directly, was approximated as 8.86 $x10^6$ m^3yr^{-1} based on the sandstone underlying a reach length of 7.4 km from the total of 23.8km. The total contribution from the aquifer as a whole may be significantly larger with a flow of 1.66 $x10^7$ m^3yr^{-1} from the Rea and 8.32 $x10^6$ m^3yr^{-1} from the lesser tributaries, which drain the aquifer and are groundwater fed.

5.3.3 River Flow Gauging

In order to determine the level of baseflow accretion, river flow gauging was undertaken. Measurements were carried out on ten separate days of dry weather flow during the period 7/7/99 to 23/5/01 at different points on the river between Bescot and Water Orton. An increase in flow across the reach is evident (Figure 5.8, Appendix 9) with base flow accretion ranging from $1x10^{-4}$ m^3s^{-1} (8.6 m^3d^{-1}) to $3x10^{-5}$ m^3s^{-1} (2.6 m^3d^{-1}) per metre length of channel. Taking into account known DWF discharges of 10.3 Mld^{-1}, groundwater discharge from the aquifer directly to the channel ranges between 1.2 m^3d^{-1} and 7.2 m^3d^{-1} per metre of channel. Several of the data series (e.g.02/06/00) fluctuate, sometimes displaying a fall in discharge levels downstream, indicating either that the river is not under steady dry weather flow conditions or an error in the flow measurements. There does not appear to be any obvious rise in accretion levels across the aquifer compared with the other types of bedrock geology (coal measures and the mudstones). This may be due to insufficient data coverage or perhaps it is an indication that the drift, in particular the alluvial gravel which is present along the entire length of the study reach, is controlling groundwater flow to the river. The most recent data were recorded on consecutive days (22^{nd} - 23^{rd} May 2001) and exhibit consistent discharges

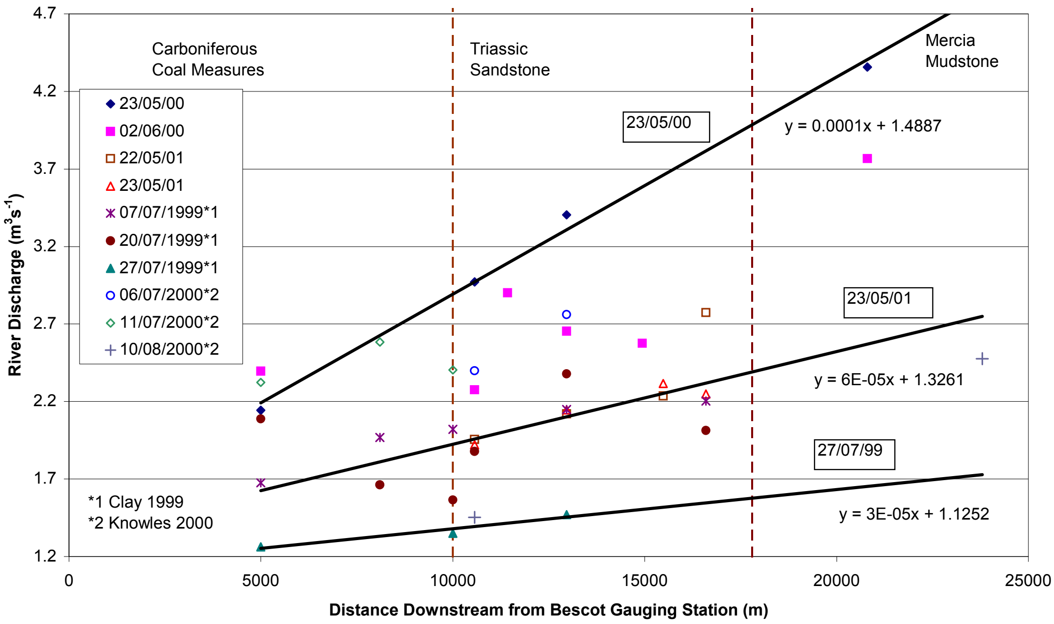


Figure 5.8 Dry weather discharge accretion along the River Tame.

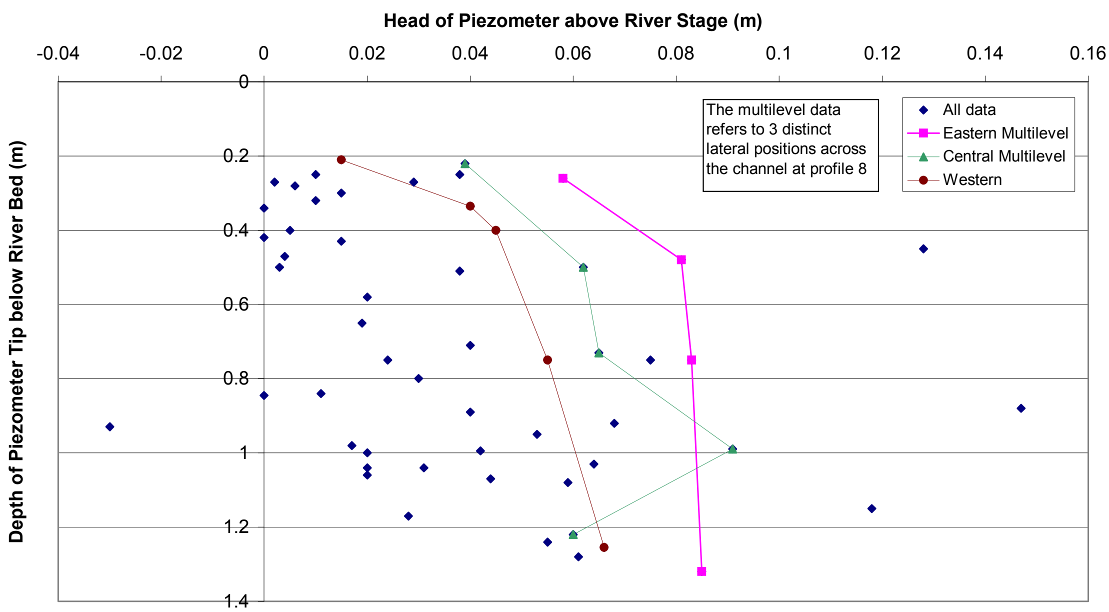


Figure 5.9 Variation in the difference in head between the river and the riverbed-piezometers with depth.

between each day. Values of known discharge may be subtracted from this data to provide a representative discharge of 4.4×10^{-5} m^3s^{-1} (3.8 m^3d^{-1}) per metre of channel across the aquifer under baseflow conditions. This estimate is similar to the average discharge of 3.6 m^3d^{-1} derived from baseflow analyses of 1999 data for the entire 23.8 km study reach with known dry weather discharges subtracted. Baseflow accretion is occurring across the entire study reach but a lack of data points makes it difficult to relate rates of discharge to the underlying geology. The regional increase in baseflow is thought to result primarily from groundwater discharge and investigations were undertaken to obtain evidence of this on the local scale.

5.4 Local Scale Evidence for Groundwater Discharge through the Riverbed.

The contrast between surface water and groundwater head and temperature measurements suggest that groundwater discharge to the river is occurring.

5.4.1 Piezometric Data from the Riverbed

Head in the riverbed piezometers was generally between 0 and 10 cm greater than the river stage across the aquifer over installation depths of 20 – 200 cm into the riverbed, with only a single negative head recorded (Figure 5.9, Appendix 11). The hydraulic gradient across the riverbed was seen to vary between –0.03 and 0.28 exhibiting large heterogeneity both in the longitudinal section and laterally across the channel at each profile (Figure 5.10).

The relationship between increasing head difference and depth is site specific, both within the longitudinal profile and the position laterally across the channel. There is a lack of any general correlation between depth and head difference but a relationship can be seen by considering data from within a vertical profile from a single location. The three vertical profiles from Profile 8 (Figure 5.9) all display increasing head with depth, with the most

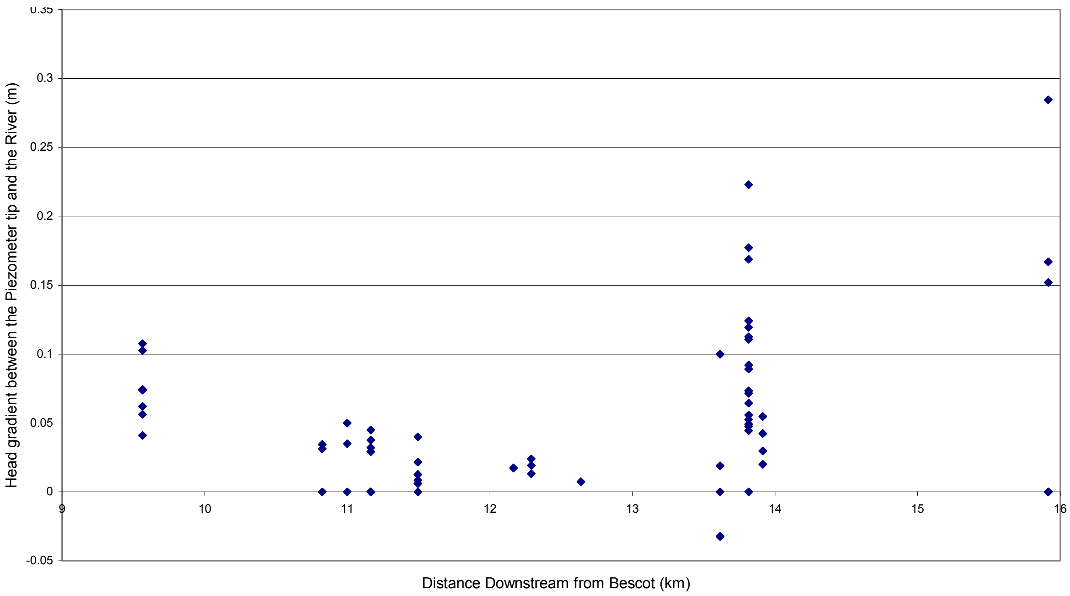


Figure 5.10 Variation in head gradient between the riverbed piezometers and the river with distance downstream.

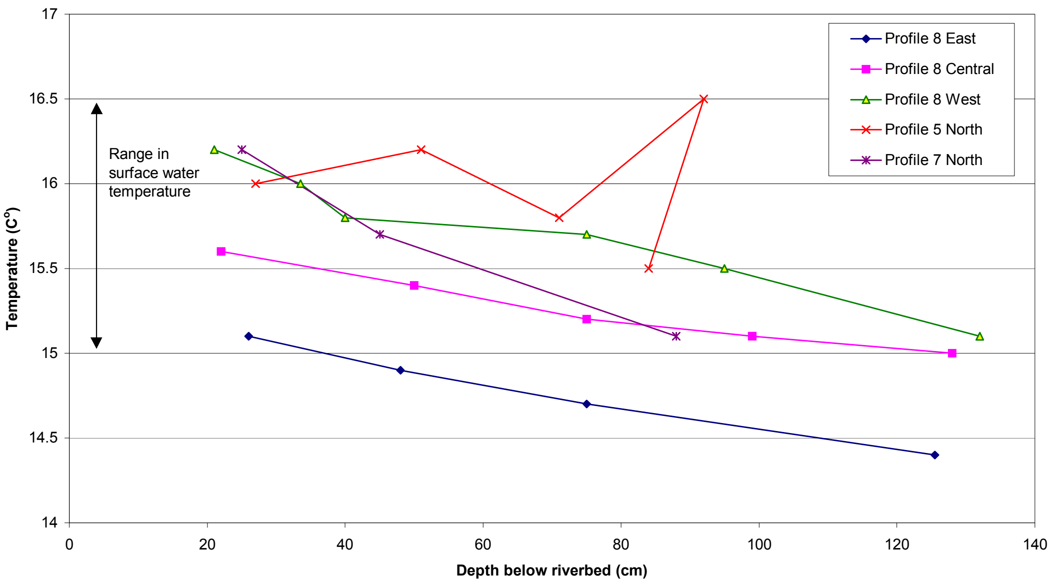


Figure 5.11 Vertical temperature profiles through the riverbed.

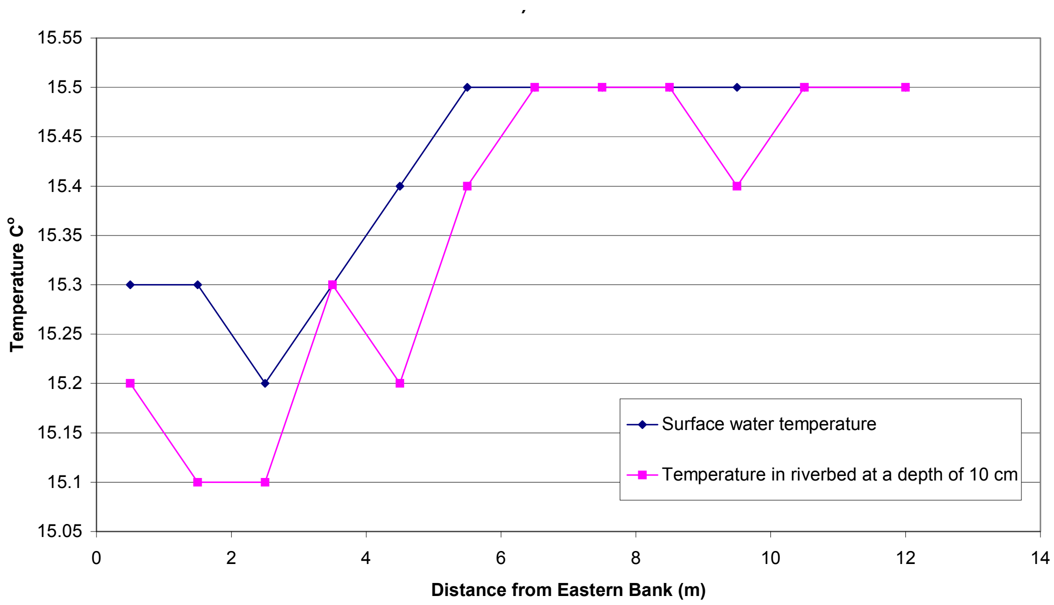
abrupt drop in head occurring over the uppermost section of riverbed. This drop may be due to lower sediment permeability but there is little slug test data to support this. The vertical profiles show the steepest hydraulic gradient across the eastern section of the channel closest to the valley side, the lowest gradient to the west associated with the flat-lying flood plain. If sediment permeability's are assumed uniform across the channel then the greatest groundwater contribution to surface water flow and quality will come from the eastern bank.

5.4.2 Temperature Data from the Riverbed

The temperature contrast between groundwater at a constant 12^{o}C and surface water, with a seasonal variation of 4.5 ^{o}C to 17 ^{o}C, was used to investigate groundwater discharge to the river (Appendix 16). Vertical temperature profiles (Figure 5.11) show the effect of the upward flow of cool groundwater on the downward conductive transmission of heat from the warm surface water during the summer. The three multilevel piezometers at Profile 8 display similar temperature gradients but show differences in the temperature/depth profile. This may be due to drift in the measurements, differences in flow rates or differences in temperatures of the groundwater sources. A similar type of variation between the three profiles is also noted on the head versus depth profile (Figure 5.9). The temperature profiles from the other sites show different gradients indicating significant heterogeneity in the system.

Temperature differences between the surface water and at 10 cm depth in the bed sediment indicated that cooler groundwater was discharging through the riverbed laterally across the channel (Figure 5.12a and b) and along the channel (Figure 5.13). The quality of some of the data is questionable as they display large apparent increases in surface water and groundwater temperatures across the profile. This is likely due to drift in the meter as a result of direct solar heating which may vary with changes in cloud cover. However, the change appears gradual

(a)



(b)

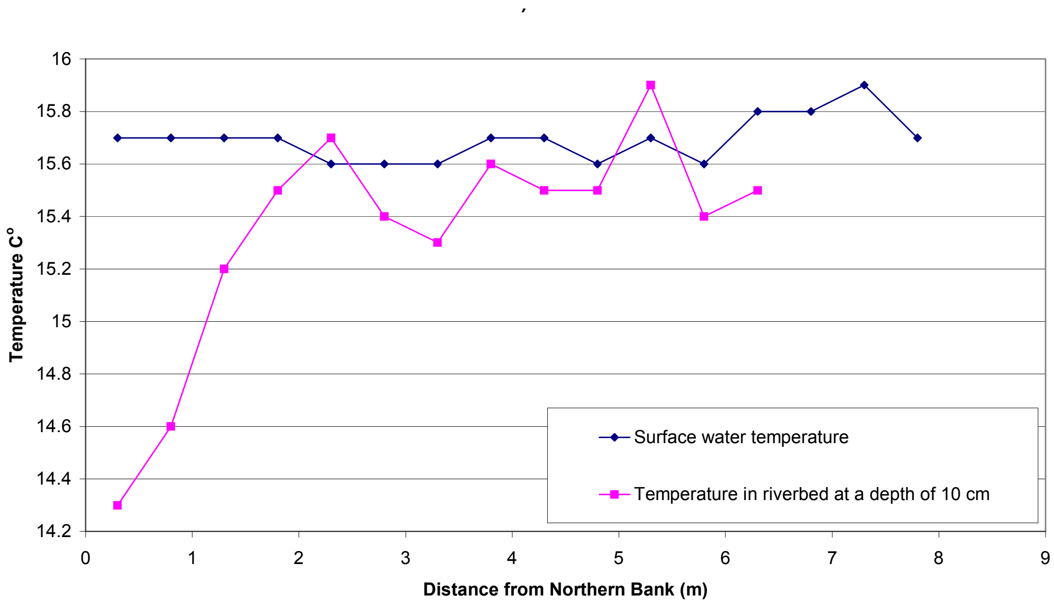


Figure 5.12 Lateral variations in surface water and riverbed temperature at 10 cm depth for (a) Profile 8 (b) Profile 1.

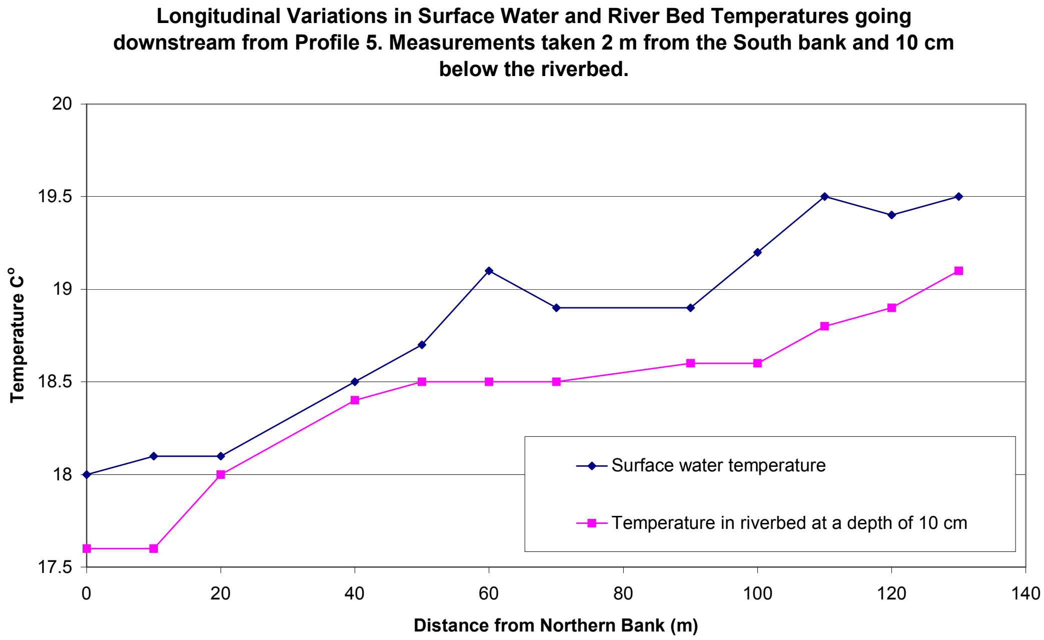


Figure 5.13 Longitudinal variations in surface water and riverbed temperatures.

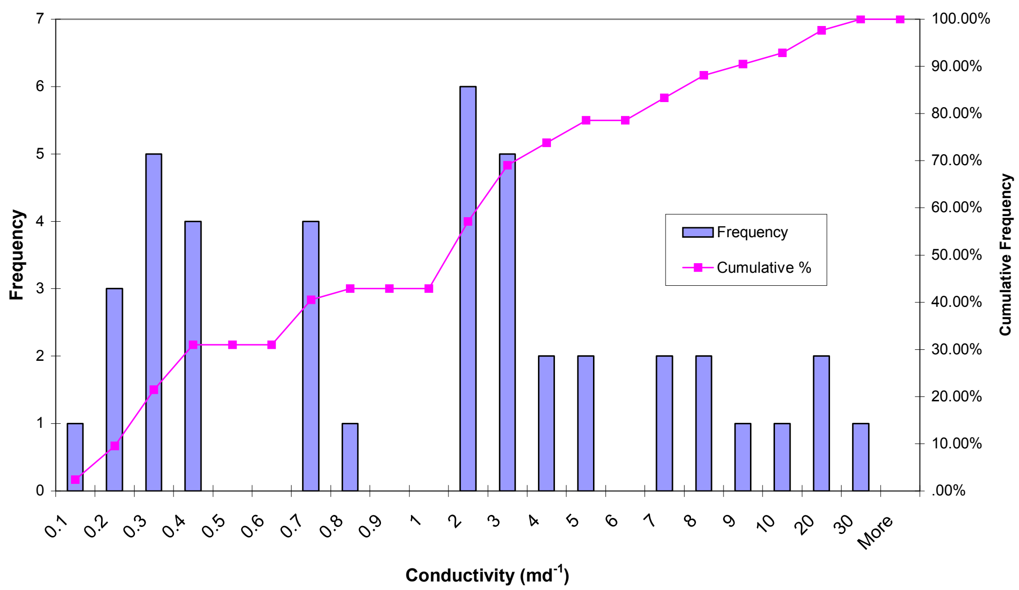


Figure 5.14 The frequency distribution of hydraulic conductivity in the riverbed

and both the surface water and groundwater temperatures at each point were taken in quick succession, therefore the data can be used to estimate the temperature contrast for each location. Each of the lateral profiles display the largest temperature difference close to the bank edge, indicating a zone of high groundwater flow. Points at which the temperature difference is zero suggest flow of surface water through the upper 10 cm of the riverbed. The data indicate the hyporheic zone to be deeper and more extensive in the central portions of the channel than adjacent to the banks. The variation in temperature contrast for the longitudinal profile indicates the river is gaining, with variable groundwater discharge over the section. The data also suggest that significant surface water flow does not occur through the riverbed sediments at depths greater than 10cm within 2 m of the riverbank in this area. Groundwater discharge is seen to occur through the riverbed and a major control on the rate of flow is the conductivity of the riverbed sediment. An investigation was therefore undertaken to characterise the riverbed sediments.

5.5 Characterisation of the Riverbed Sediments

Falling head tests in the riverbed piezometers and sieve analyses of riverbed core material were carried out to determine the hydraulic conductivity of the riverbed sediments.

5.5.1 Falling Head Test Permeability Data

The results of 45 falling head tests in the riverbed piezometers were analysed using the Hvorslev method and gave conductivity ranging between 0.08 $md^{-1}$and 23 md^{-1} with an average of 3.13 md^{-1} and a geometric mean of 1.26 md^{-1} (Appendix 15). Three of the tests were repeated immediately and showed a variation from the original value of between 0.25 and 1.6 md^{-1}. The poor repeatability is a result of errors in the manual measurement process, which could be overcome if a pressure transducer and logging system were used. The frequency distribution (Figure 5.14) shows the large heterogeneity in the system and suggests a log normal distribution. This implies that the geometric mean is representative of the general conductivity of the sediments. This value contains both vertical and horizontal components and the vertical conductivity that controls the groundwater flow through the riverbed is likely to be lower. However the volume sampled is small (~200 cm^3) and other investigations (Bradbury et al., 1990) have suggested that hydraulic conductivity increases with the scale of measurement. In addition other studies (Cey et al., 1998) have recognised that clogging of the riverbed piezometer screen may reduce the estimated conductivity.

The conductivity estimates are very site specific and show large heterogeneity that requires a high sample density to determine average conductivity adequately. The spatial variation in conductivities along the study reach (Figure 5.15) may reflect changes in the sediment type but sample density is too limited to confirm this. High values at Profile 8 may be associated with coarse gravels.

5.5.2 Riverbed Sediment Core Data and Sieve Analyses

A total of 23 riverbed cores were taken to depths of up to 0.5m from Profiles 1,2,8,and 17 across the aquifer. The sediments typically comprised a 10 cm section of coarse gravel containing cobbles <30cm in diameter overlying mixed sand and gravel (Figure 5.16 a,b,c). Clasts were primarily quartzite derived from reworked alluvial deposits and the overlying glacial drift. Less than 5% of the clasts were of anthropogenic origin (fragments of brick, slag, coal and discarded metal artefacts). Sieve analyses were performed on 41 samples (Moylett, 2001, Appendix 14) and based on the d50 value the finest sample was classified as a fine sand and >60% of the samples were classified as gravel. In general, (Figure 5.17) particles < 10

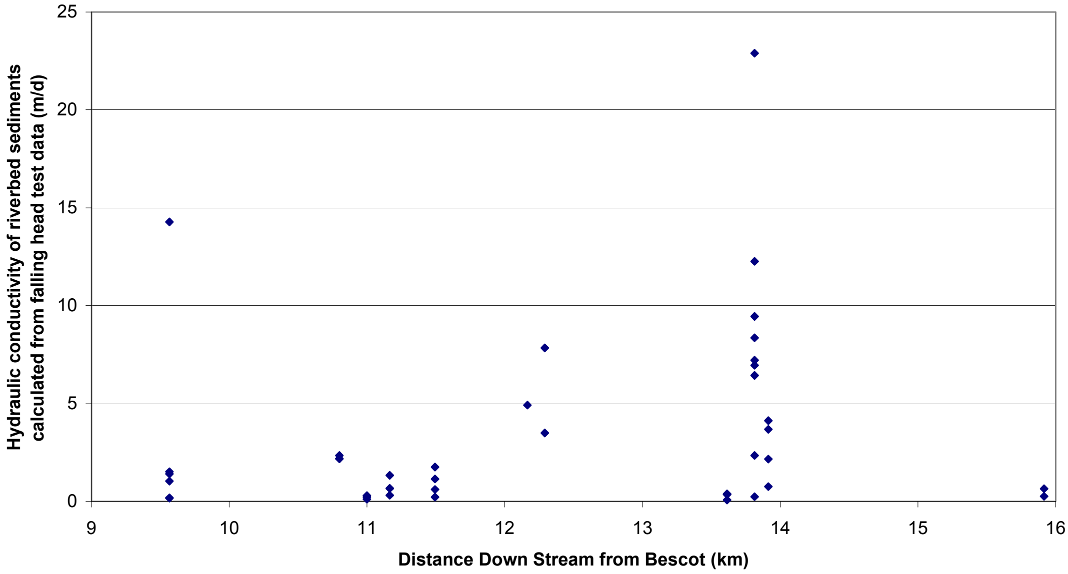


Figure 5.15 Distribution of riverbed conductivity values downstream

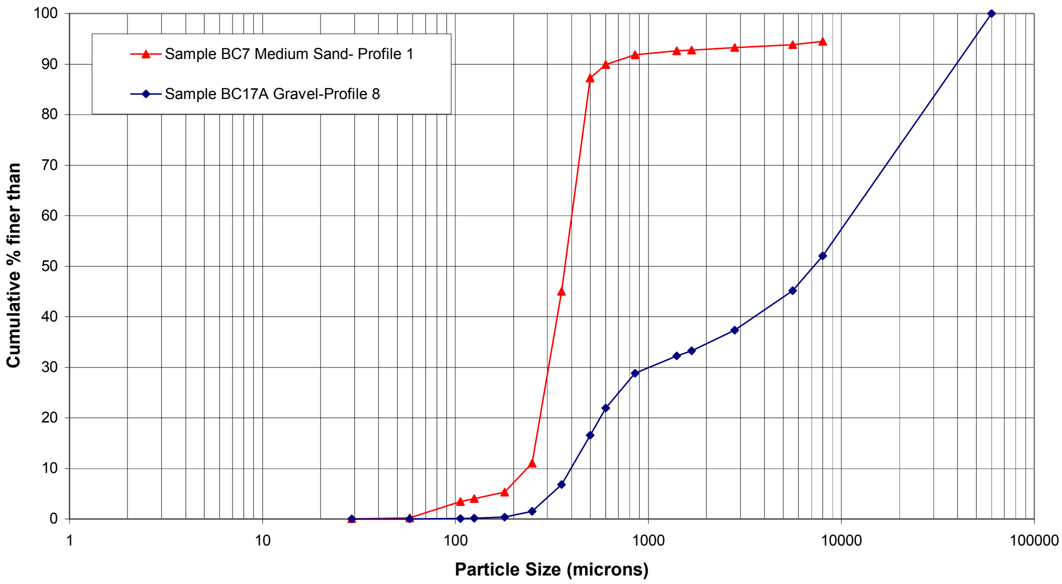


Figure 5.17 Typical grain size distribution curves for riverbed sediments in the Tame.

Figure 5.16 (a) Riverbed core BC7 from Profile 1

Figure 5.16 (b) Riverbed core BC7 from Profile 1 (Close-Up)

Figure 5.16 (c) Riverbed core BC17 from Profile 8

microns (medium silt) were rare as may be expected due to transport and erosion with river flow velocities commonly >0.5 ms^{-1} and > $1ms^{-1}$ during flood events. Conductivity values were calculated using the Hazen (1911) and Shepherd (1989) methods. Both methods were found unsuitable for the gravel, giving unreasonably high values of conductivity (Figure 5.18). For the 15 samples classified as sand, the Hazen conductivity ranged from 22 md^{-1} to 94 md^{-1} averaging 41 md^{-1}, and the Sheperd conductivity ranged from 11 md^{-1} to 91 md^{-1}averaging 31 md^{-1}. Laboratory falling head permeability tests (Moylett, 2001) were carried out on seven small-diameter cores kept in the core tube after extraction from the riverbed. The results ranged from 0.25 to 3.68 md^{-1}, with an average of 1.3 md^{-1}.

The conductivity values derived from the particle size analyses are on average an order of magnitude larger than those derived from the field and laboratory falling head tests. One reason for this is that the sieve analyses provide a bulk permeability for a mixed sample over a large vertical range (.>20 cm) and sample volume (>1500cm^3). This does not account for the impact of stratification with thin less permeable units that will reduce the vertical conductivity. A finer sampling interval is required on the core to obtain a better estimate of the vertical conductivity from the sieve analyses.

The sediments comprising the riverbanks are of a different type to the modern riverbed sediments as the river flow regime has been changed radically. Reworking of these older sediments under recent high flow conditions will, most likely, remove the fines and increase the conductivity. Flood defence investigations by the Environment Agency (Caudell, 2000) gave mean conductivity of 6 md^{-1} from Hazen analyses of 6 particle size distributions and a mean of 4md^{-1} from Hvorslev analyses of rising head tests for these riverbank deposits.

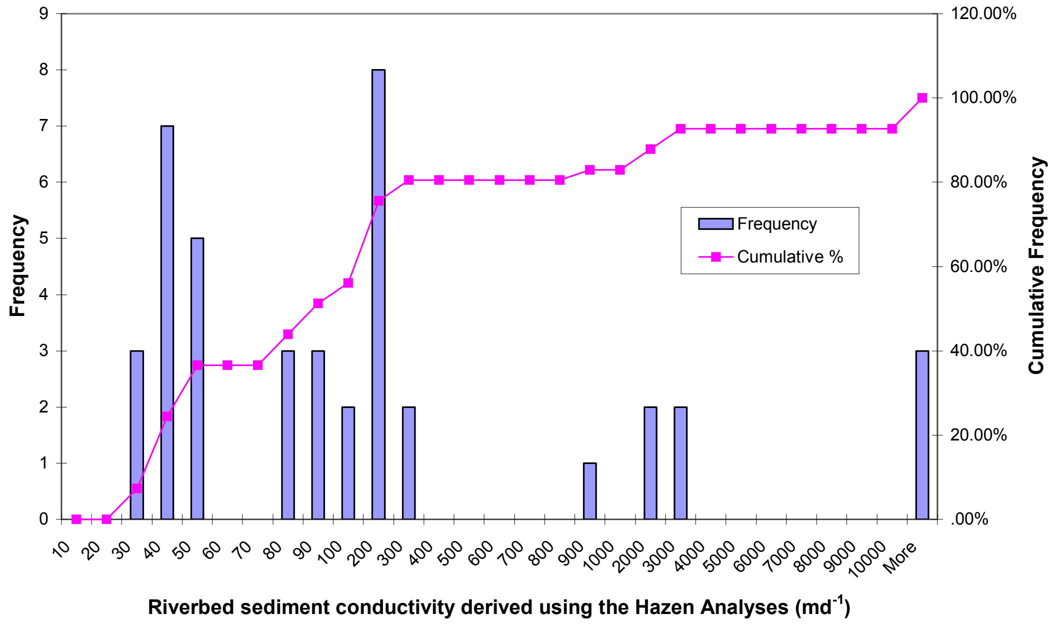


Figure 5.18 Frequency distribution of Hazen conductivity values for riverbed samples.

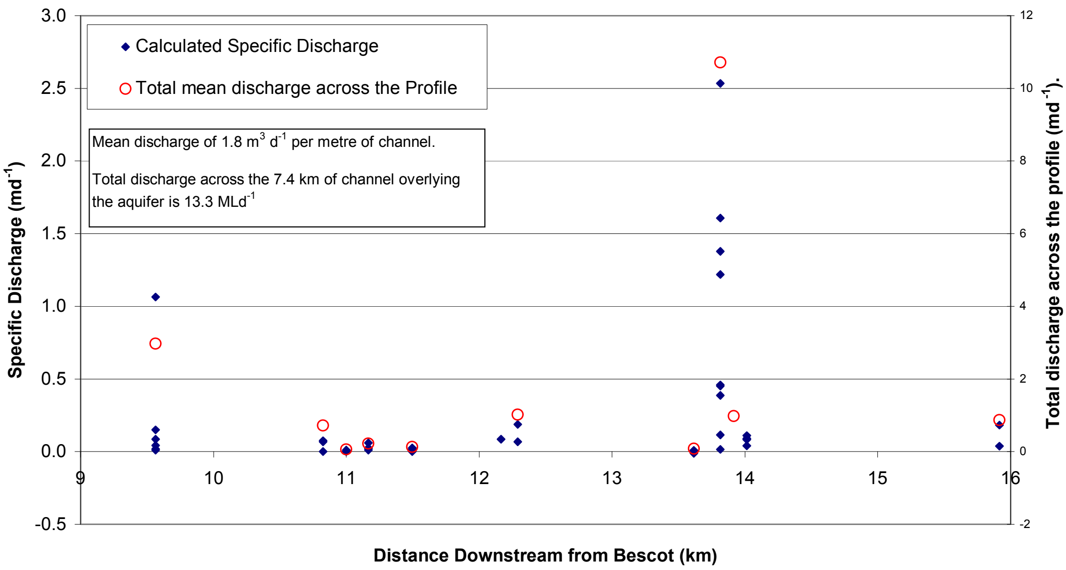


Figure 5.19 Specific discharge through the riverbed calculated using the Darcy flow equation.

The geometric mean conductivity of 1.26 md^{-1} from falling head analyses is considered to provide a general estimate for the combined vertical and horizontal riverbed hydraulic conductivity. The Hazen and Sheperd values of >11 md^{-1} may be more representative of the horizontal conductivity of the riverbed. Higher values of horizontal compared with vertical hydraulic conductivity in the riverbed would suggest a significant amount of lateral hyporheic flow might occur. The horizontal conductivity of the riverbanks is estimated as 5md^{-1} from the flood defence borehole data.

The riverbed sediment is an important control on groundwater flow rates and using the values derived for hydraulic conductivity in conjunction with piezometric data it was possible to calculate local rates of groundwater discharge.

5.6 Estimates of groundwater flow through the riverbed.

One-dimensional analytical solutions for Darcy flow, radial flow and temperature distribution were used to calculate groundwater discharge through the riverbed based on field observations of head, conductivity and riverbed temperature gradient.

5.6.1 Darcy Flow Estimates

Darcy flow estimates were calculated based on the head potential and conductivity measured in the riverbed piezometers (Section 4.9.4). The calculated specific discharges range from –0.01 to +2.53 md^{-1} (Table 5.5) and display considerable variability along the length of the study reach and across the channel (Figure 5.19, Appendix 18).

Summary of Darcy Specific Discharge Calculations (md^{-1})	
Mean	0.28
Median	0.07
max	2.53
95th %ile	1.40
min	-0.01
5th %ile	0.00

Table 5.5 Summary of Darcy calculation specific discharge through the riverbed

The total specific discharge for each profile was calculated based on profile width and the profile mean specific discharge. The mean total discharge for the combined profiles was 1.8 m^3d^{-1} per metre of channel as compared to 3.8 m^3d^{-1} calculated from the river-flow gauging. Several possibilities exist for the difference in results:

1. The flow-gauging estimate includes discharges from other sources in addition to the groundwater.
2. The falling head tests may underestimate the conductivity used in the Darcy flow calculations.
3. The piezometers used to calculate the Darcy flow were generally located towards the centre of the channel bed and no estimates of flow were derived from the riverbanks. Flow

through the channel sides may be considerable due to higher horizontal conductivity compared to the lower vertical conductivity that dominates flow through the riverbed.

4. The Darcy flow calculations are calculated using data from a period of minimum baseflow.
5. Perhaps there are narrow zones of high groundwater flow associated with geological features that focus groundwater discharge and which are not identified by the limited data coverage.

5.6.2 Radial Flow Estimates

Radial flow estimates were calculated (Section 4.9.3) using river stage and water levels from nine piezometers located within 15 m of the river bank. Values of discharge ranged from 0.18 to 1.53 m^3d^{-1} per metre of channel (Table 5.6, Appendix 18) with an estimated total discharge to the 7.4 km of channel overlying the aquifer of 5.6 Mld^{-1}.

Summary of Radial Flow Discharge Calculations (m^3d^{-1} per metre of channel)	
Mean	0.76
Median	0.9
max	1.53
95th %ile	1.26
min	0.18
5th %ile	0.26

Table 5.6 Summary of radial flow discharge calculations

The radial discharge is directly proportional to the conductivity, which in this case was set at 2 md^{-1}. The mean radial discharge of 0.76 m^3d^{-1} per metre of channel is considerably lower than the 3.8 m^3d^{-1} calculated from the river-flow gauging. This implies an underestimation of the conductivity that would need to be increased to 10 md^{-1} to raise the discharge levels to 3.8 m^3d^{-1}. Radial discharge is also sensitive to the length of the wetted perimeter used, which in this case was calculated from river width and average water depth. The inclusion of a 20 cm seepage face increased the mean discharge by 13% to 0.86 m^3d^{-1}. Variation of the head difference between the aquifer and the river by +/- 10cm produced changes in the mean discharge of +32% to 1 m^3d^{-1} and –30% to 0.53 m^3d^{-1}.

Discharge is spatially variable along the reach (Figure 5.20) and between opposing banks as seen in the cases where piezometers are located on both sides of the river on the same profile. There is no obvious trend in the distribution of discharge or any direct relationship to the distribution of the Darcy flow estimates. Both estimation methods yield similar ranges in discharge, radial 0.18 – 1.53, Darcy 0.06 – 5.46 m^3d^{-1} per metre of channel (the Darcy estimate was calculated using the geometric mean specific discharge for each profile multiplied by the wetted perimeter, Appendix 18).

5.6.3 Estimates of Flow from Measurements of Temperature Gradient

Vertical temperature gradients derived from five multilevel piezometer profiles were used to calculate (Section 4.8.3) rates of vertical groundwater flow ranging from –0.029 to 0.065 md^{-1} (Table 5.7, Appendix 16).

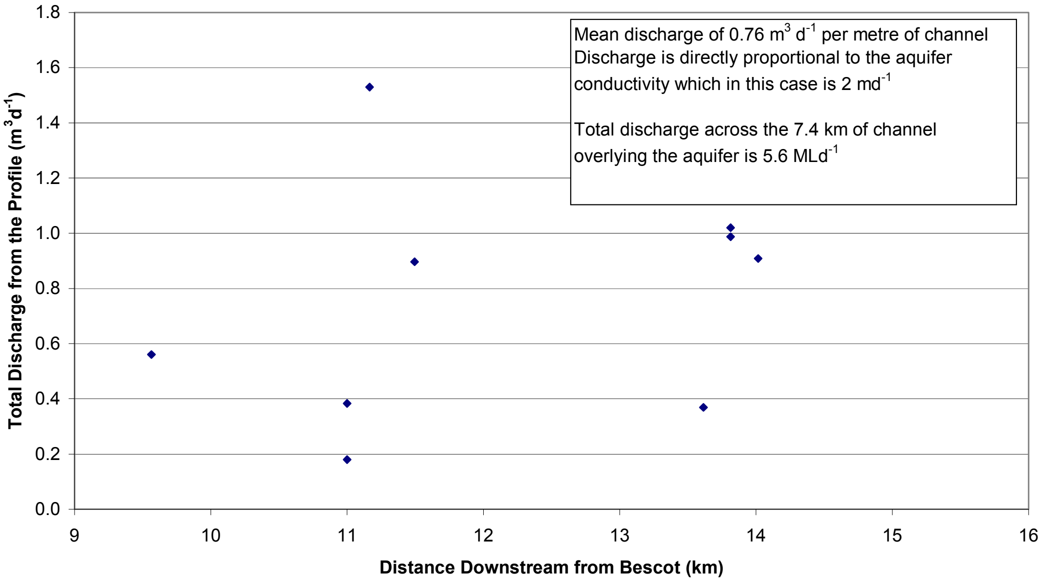


Figure 5.20 Variation between the calculated discharge for each profile using the radial flow equation.

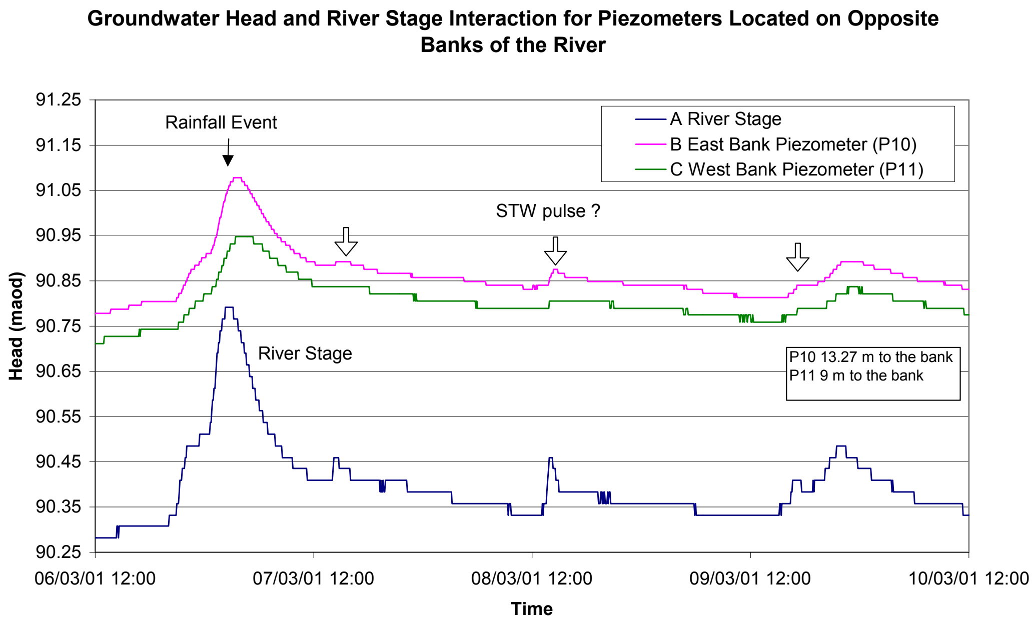


Figure 5.21 Groundwater head and river stage interactions

Location	Specific Discharge (md^{-1})
Profile 8 - West	0.022
Profile 8 - Central	0.035
Profile 8 - East	0.009
Profile 5 - North	-0.029
Profile 7 - North	0.065

Table 5.7 Summary of flow estimates derived from the vertical temperature gradient

The vertical flow rates obtained using the temperature method are at the lowest end of the estimates derived from the Darcy flow calculations. This may result from the data collection technique overestimating the groundwater temperature at depth. The probe may have been affected by the temperature gradient resulting from conduction within the standing water column of the piezometer as the temperature was not taken on a pumped sample. In addition, biochemical activity may also increase the groundwater temperatures and this is not taken into account in the analytical solution. There is a high degree of uncertainty in the temperature flow estimates though they may provide an indication of the minimum flow rates at each location.

Groundwater is seen to discharge to the surface water system across the aquifer under dry weather conditions with stable river levels. However, under wet weather conditions with rapid rises in river level, groundwater discharge is likely to be reduced or reversed in some cases.

An investigation of the groundwater/surface water interaction was undertaken for these conditions.

5.7 Groundwater and Surface Water Interactions

Continuous monitoring of river stage and adjacent groundwater levels indicated a good connection between the aquifer and the river. A rapid response of the water table to rises in river level was recorded in piezometers < 15 m from the riverbank at each of the six sites investigated (Appendix 12). River level rise due to rainfall events and STW discharges are seen to produce different responses in the piezometers (P10, P11) located on opposite banks at Profile 8 (Figure 5.21). The rise in water table occurs when groundwater no longer able to discharge under conditions of increased river stage enters temporary storage in the bank. The gradient between the river stage and the water table is seen to fall during the rising limb of the river flood hydrograph before returning to pre-flood levels, subject to any increase resulting from recharge to the aquifer. Gradient reversal was not observed in any of the flood events, for which a maximum stage rise of <60 cm was recorded. This indicates that although a reduction in groundwater discharge occurs during the flood there is no inflow of surface water to the aquifer.

Amplitude damping and time lag effects are apparent from the response of the observation piezometers at distance from the river forcing head. The degree of damping and lag is dependent on the relationship between storage and transmissivity (S/T), higher values of S/T being associated with increased reduction of amplitude and longer lag times. Changes in the groundwater response were noted between each observation site, highlighting differences in the geology and associated values of S/T. Variations were also noted in the groundwater response at a single site between different flood events and within the same event, indicating temporal changes in the S/T characteristics. The most likely explanation for this is that S/T varies during an event as a function of unsaturated flow, and differences in available storage across the capillary fringe as the water table fluctuates. The variation between events is associated with the different wetting and drying histories leading to differences in the moisture content profile.

During the period 11th to 22nd May 2001, a series of flood events occurred in rapid succession after a period of dry weather flow conditions (Figure 5.22). Both piezometers at Profile 8 displayed damping and lag effects that decreased between each cycle indicating reduced unsaturated storage due to moisture retention from the previous cycle. Piezometer P11 was seen to give a greater response to the third peak than the second despite the maximum head in the river being less on the third cycle. A different response was seen to the river forcing head between the rising and falling limbs of any one event. Initial groundwater rise was rapid and reduced towards the peak, and considerable lag between the piezometer and the river was evident on the falling limb. This was due to available storage increasing progressively away from the capillary fringe as the water table rose. The falling limb was subject to gravity drainage and displayed a rapid initial drop in the water table which slowed with the onset of delayed yield.

The groundwater discharge to the river varies in response to surface water fluctuations, the soil moisture conditions and the local groundwater level. A detailed study of the area around Profile 8 was undertaken to identify the local hydrogeological controls and provide data for the construction of a numerical model to investigate groundwater/surface water interactions.

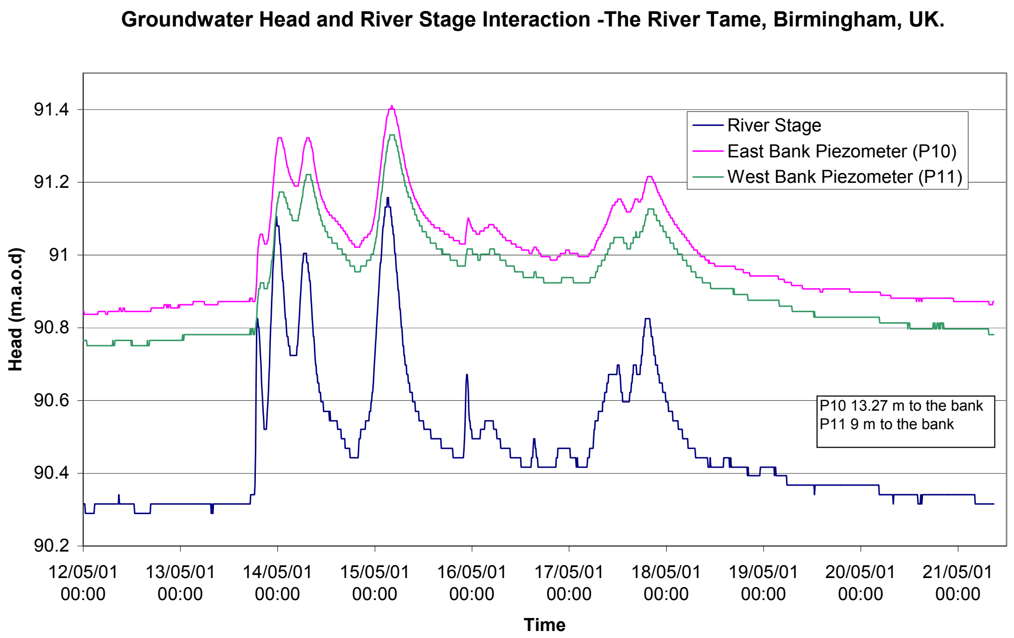


Figure 5.22 Groundwater head and river stage interaction (12/5/01 – 21/5/01).

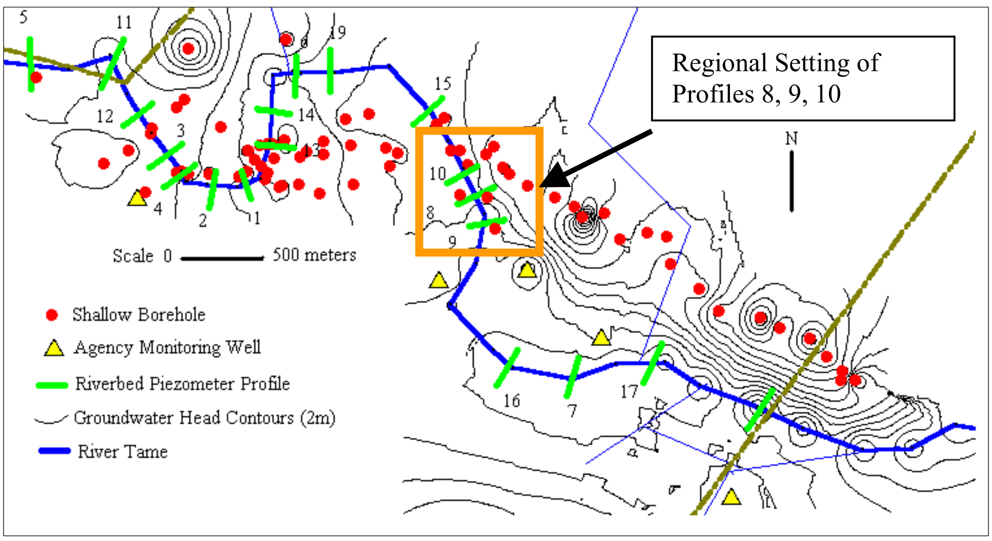


Figure 5.23 The regional setting for Profiles 8,9,10.

5.8 The Hydrogeological Setting of Profile 8

Profile 8 lies within the central part of the Tame Valley (Figure 5.23) where the river flows in a south-easterly direction on the downstream portion of a river meander bend. To the west lies the flood plain and to the east, the steeper topography of the valley side. The river at this location is 12 m wide and 30 cm deep, with a gravel bottom. The banks comprise a mixture of natural form and rock basket support. There is a large available data set for the site comprising geological logs, hydraulic conductivity estimates and water-level dipping records (Appendices 7,8, and 9). These were derived from piezometers adjacent to the river and multilevel piezometers within the river bed (Figure 5.24).

Geological sections were constructed for the area based on projected borehole logs (Figure 5.25a and b). The position of the water table in Section 1 indicates that groundwater flow to the east of the river occurs through the bedrock sandstone unit to within 175m of the river (BH 19). In this region, flow also starts to occur through the overlying gravel and weathered sandstone. To the west, flow occurs through the sandstone and the overlying gravel of the flood plain.

A study of historical head data (1993-95) indicates that flow to the river is parallel to the line of section with little seasonal variation (Figure 5.26) which supports the use of a 2D profile model. Piezometric data show an average temporal variation of < 60 cm in the regional groundwater head over three years. The influence of river flood events on water levels is short lived (< 3 days) and limited to the near river zone, <50m from the bank.

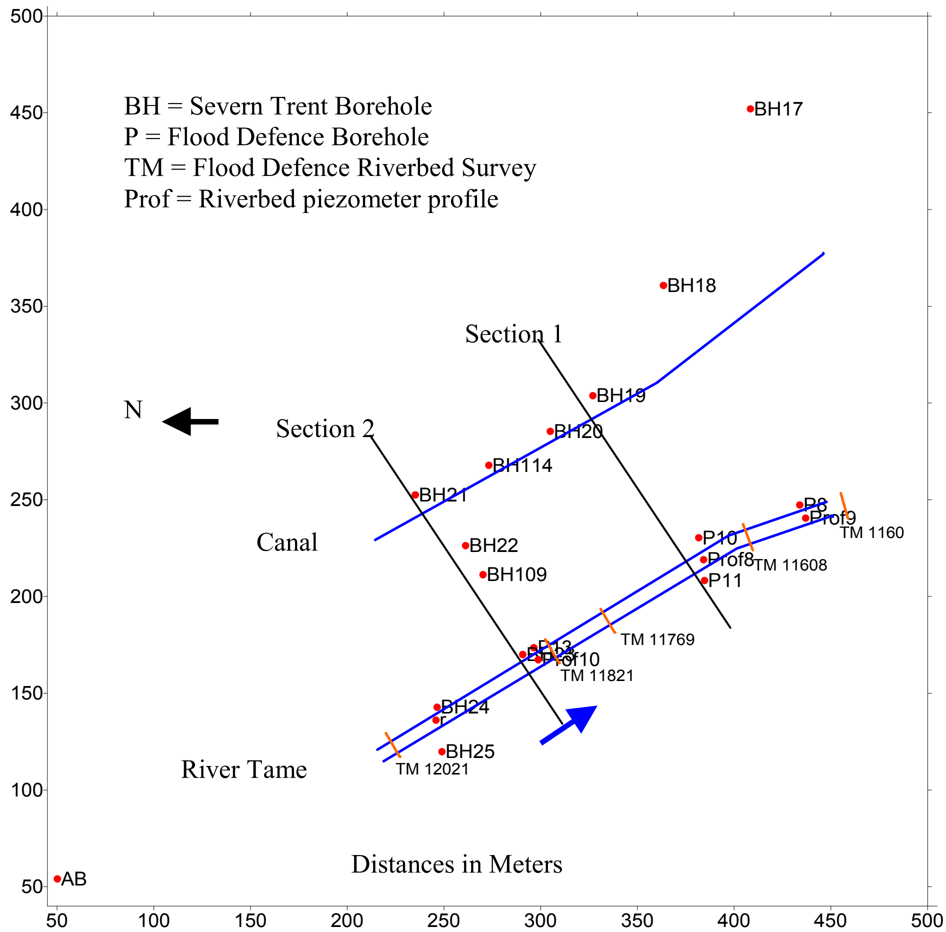
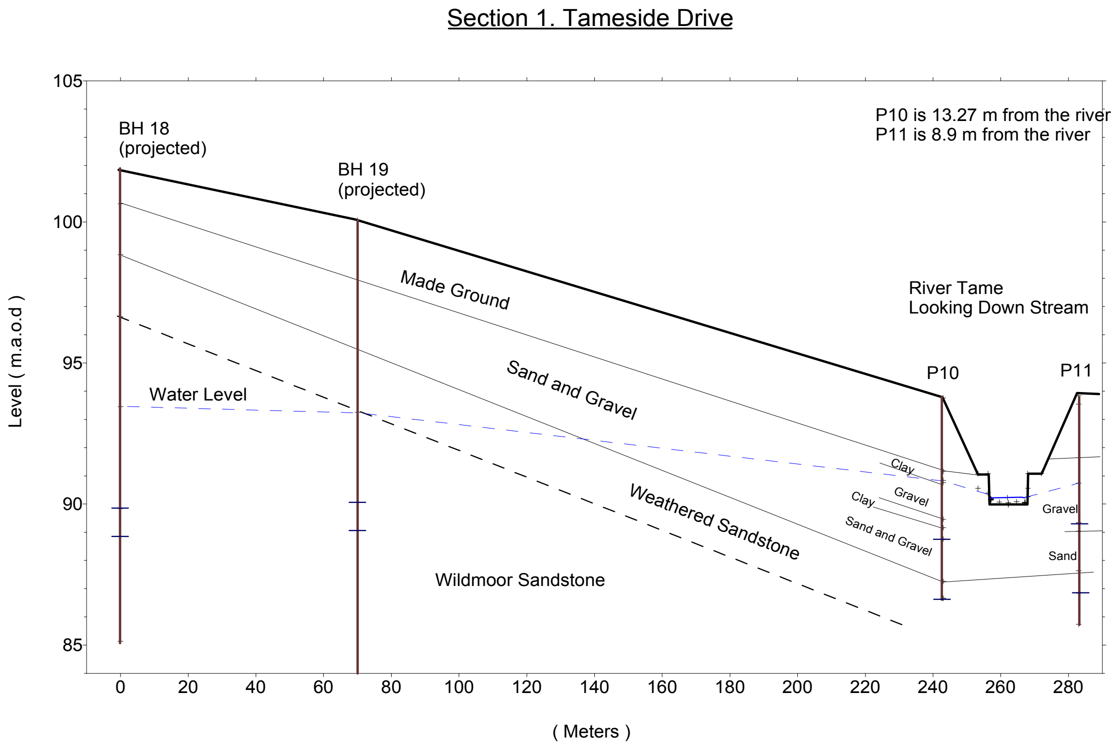


Figure 5.24 The location of boreholes adjacent to Profiles 8,9 and 10.



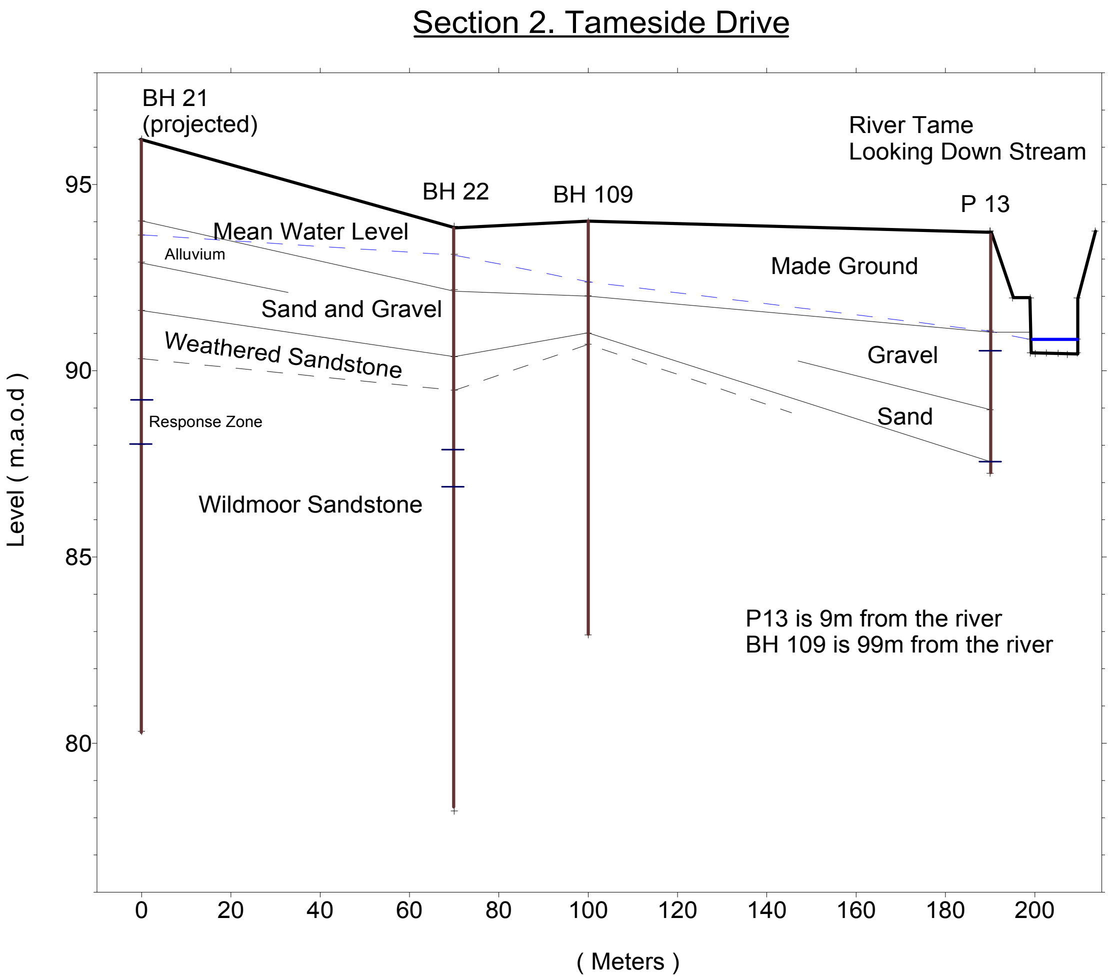


Figure 5.25 Geological cross sections through (a) Profile 8 (b) Profile 10.

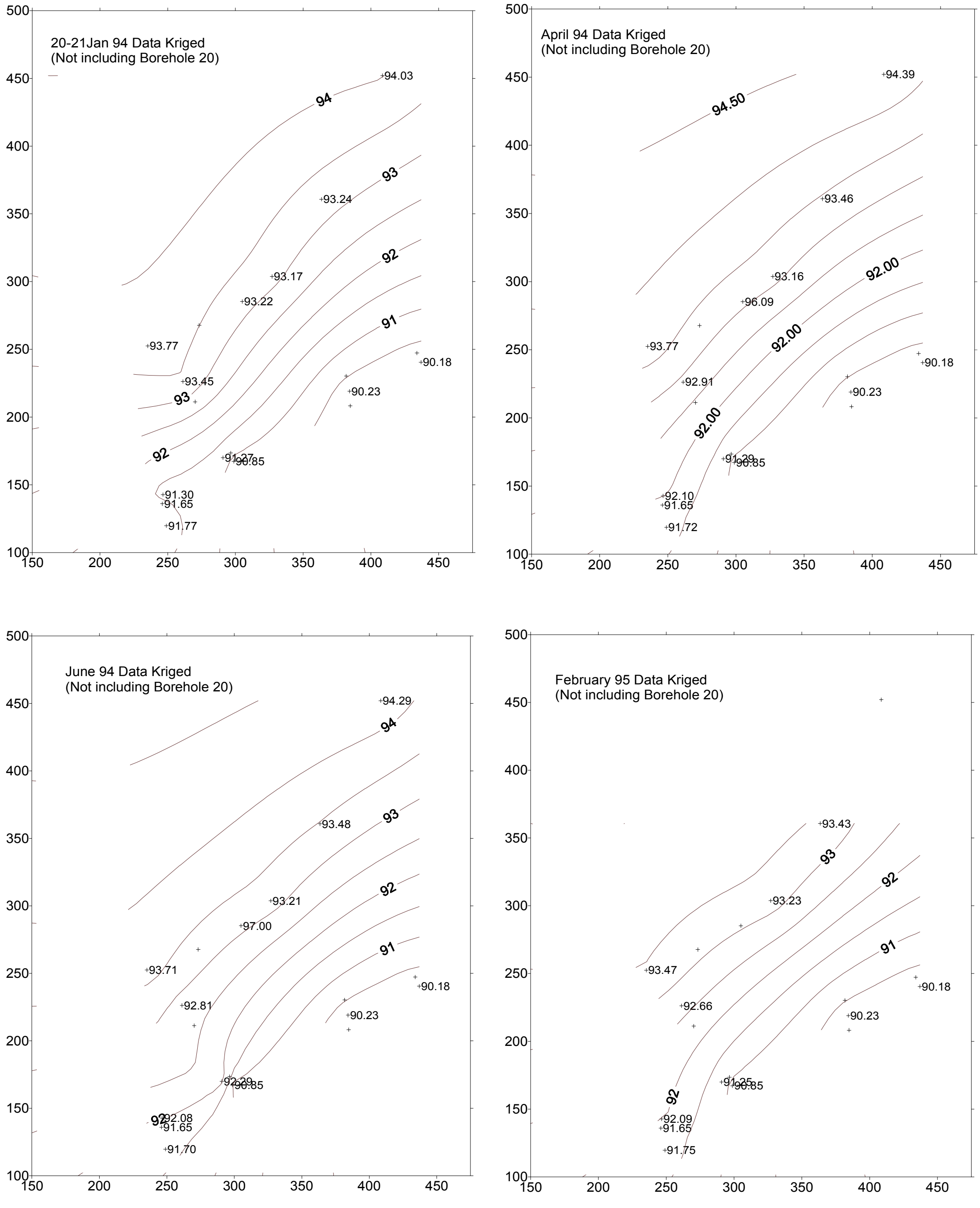


Figure 5.26 The seasonal variation in groundwater head contours by Profile 8.

5.9 Concluding Discussion

Baseflow analyses of river hydrographs indicate that 40-60% of dry weather flow at the lower end (Water Orton) of the study reach is derived from unknown sources including a major groundwater component. An estimated 20-25% of this is derived from groundwater discharging directly to the 7.4 km of channel that overlies the Birmingham Aquifer. Baseflow accretion also takes place as the river flows over the less permeable mudstone and coal measure bedrock formations, and it is suspected that a significant component of baseflow is derived from flow through surficial deposits of sand and gravel. Sewage Treatment Works provide a major contribution to river discharge and comprise 30% of the total dry weather flow at the end of the study reach. The Tame is estimated to be under dry weather flow conditions for between 39% and 68% of the year, during which time the groundwater component of river flow is most significant.

Groundwater levels in the Tame Valley have recovered from historical lows that occurred during the period of peak abstraction 1900 –1950 when considerable loss of surface water to the aquifer occurred. The Tame is now primarily, a gaining river with a very few localised areas displaying surface water loss.

Groundwater discharge to the river is both spatially and temporally variable. The river baseflow is at its lowest during the summer due to a reduction in groundwater discharge in response to lower levels of recharge. Baseflow analyses for the entire 23.8 km study reach estimated groundwater discharge to vary between 1.7 and 5.5 m^3d^{-1} per metre of channel during 1999 with a mean of 3.6 m^3d^{-1}. River flow gauging measurements taken across the aquifer indicate groundwater discharge in the range between 1.2 m^3d^{-1} and 7.2 m^3d^{-1} per metre of channel with a typical dry weather discharge of 3.8 m^3d^{-1}. There is no conclusive evidence that discharge to the river channel is higher as it flows across the Birmingham Aquifer than when it passes above the coal measures or the mudstone. The baseflow discharge directly to the river channel overlying the aquifer for 1999 was 8.86 $x10^6$ m^3yr^{-1} and the total groundwater discharge to surface water from the aquifer may be as high as $3.38x10^7$ m^3yr^{-1}.

The analytical methods of estimating groundwater discharge, based on spot measurements of hydraulic conductivity, head and temperature gradient, show spatial variability in the discharge from the sandstone aquifer in the range 0.06 to 10.7 m^3d^{-1} from individual lateral profiles. Average discharges to the river were 0.76 m^3d^{-1} per metre of channel (calculated from the radial flow method) and 1.8 m^3d^{-1} per metre of channel (calculated using the Darcy method). These estimates are considerably lower than the 3.8 m^3d^{-1} per metre of channel estimated for typical summer baseflow by direct measurement of discharge accretion in the river. A similar occurrence has been noted in other studies (Cey et al.,1998, Harvey et al., 1993) with estimates of Darcy flow amounting to 19-25% of the measured baseflow accretion. This may indicate an unknown geological control or a shortfall in the data coverage in relation to what is required to estimate flow in a highly heterogeneous system. The direct measurement of baseflow accretion is considered to provide the best estimate for the groundwater discharge. However, it must be regarded as an upper limit and it is reliant on knowledge of all the additional sources of discharge to the river which is problematic to obtain in an urban setting.

The Darcy flow estimates are very site specific to individual points within the riverbed and give a wide range in specific discharge values from –0.01 to +2.53 md^{-1}. A high sample density and coverage (including the riverbanks) is required in order to give a meaningful estimate of total discharge across any one lateral profile. Radial flow estimates provide an average discharge across a profile but are very dependent on an accurate knowledge of the local hydraulic conductivity. They also assume simple 2D flow conditions that seldom occur and require the installation of a piezometer on the riverbank. Estimates of flow using the temperature gradient are based on data of uncertain quality and range between –0.029 to 0.065 md^{-1}, and are an order of magnitude lower than the Darcy estimates.

The riverbed sediments comprise sand and gravel with conductivity ranging from 0.08 $md^{-1}$to >20 md^{-1} and generally do not provide a significant barrier to groundwater discharge from the underlying aquifer. The conductivity values derived from falling head tests form a log normal distribution with a geometric mean of 1.26 md^{-1} an order of magnitude lower than the minimum value of 11 md^{-1} derived from particle size fraction analyses. The difference may be related to the larger sample volume of the sieve analyses and the fact that it does not account for the occurrence of thin low conductivity layers which may control vertical conductivity.

The groundwater discharge to the river is temporally variable on a seasonal time scale and changes according to the levels of baseflow. Large variability in discharge is also seen over shorter periods of hours to days associated with rapid changes in river level due to rainfall run-off or STW discharges. Continuous monitoring data show a rise in the water table occurs as groundwater enters temporary storage in the bank under conditions of increased river stage. Levels of groundwater discharge decrease on the rising limb of the flood hydrograph and return to levels slightly greater than the initial conditions on the falling limb. Gradient reversal and loss of surface water to the aquifer was not observed in any of the riverbank piezometers for flood events with a maximum stage rise of <60 cm .

Amplitude damping and time lag effects are seen in the response of the observation piezometers to fluctuations in the river stage. Variations in the groundwater response at a single site between different events and within the same event indicate temporal changes in the S/T characteristics. This is may be due to unsaturated flow processes and differences in available storage across the capillary fringe as the water table fluctuates. The variation between events is associated with the different wetting and drying histories leading to differences in the moisture content profile which is a significant control on groundwater surface water interactions.

The groundwater and surface water systems interact as part of a single complex system that requires extensive data coverage to characterise adequately the high degree of heterogeneity. In order better to understand the processes occurring, a series of groundwater models were constructed at different scales based on a simplified interpretation of the field data.

CHAPTER 6. THE MODELLING OF GROUNDWATER FLOW TO THE TAME

6.1 General Modelling Objectives

The following are the modelling objectives developed as a result of the field investigations and designed to increase understanding of the groundwater flow and the transport of contaminants to the river.

1. To investigate the groundwater flows beneath the flood plain so as to identify the likely lateral paths of contaminant plumes from source area to discharge zone within the river.

2. To investigate the vertical components of groundwater flow paths near the channel.

3. To investigate the geological controls on groundwater flow to the river and, to determine the sensitivity of the system to each control including the seepage face.

4. To investigate the variations in groundwater discharge to the river along and across the river channel.

5. To investigate the effect of abstraction wells on the groundwater flow paths and groundwater flux to, or from, the river.

6. To investigate the effect of different steady state, dry weather flow river levels and the variations in regional head distributions according to increased or decreased recharge and abstraction rates.

7. To investigate groundwater/surface water interactions during a river flood event.

8. To investigate the impact of unsaturated flow processes and the capillary fringe on the fluctuation of the groundwater table in response to changes in river stage.

6.2 Modelling Tools

Five modelling tools were used to meet different aspects of the model objectives (Table 6.1).

Table 6.1 The selection of modelling tools to meet the different objectives

Modelling Tool	Objectives							
	1	2	3	4	5	6	7	8
Analytical Model, Steady State			✓					
Analytical Model, Transient							✓	
MODFLOW	✓		✓	✓	✓	✓		
FAT3D		✓	✓	✓		✓	✓	
UNSAT								✓

6.2.1 Analytical Model, Steady State Solution

The one-dimensional equation for flow in an unconfined aquifer may be expressed as:

$$Sy\frac{dh}{dt} = \frac{d}{dx}\left[k(h-b)\frac{dh}{dx}\right] + R$$

Sy = specific yield	[-]
k = hydraulic conductivity	[L/T]
h = hydraulic head	[L]
b = base elevation of the aquifer	[L]
t = time	[T]
x = distance	[L]
R = recharge	[L/T]

When Recharge is zero, the hydraulic conductivity and aquifer base elevation are homogeneous and the boundary conditions are:$h = h_1$ at $x = 0$ and $h = h_2$ at $x = L$ (Figure 6.1) the analytical solution is:

$$\frac{h^2 - h_1^2}{2} - b(h - h_1) = \left[\frac{h_2^2 - h_1^2}{2\mathrm{L}} - \frac{b(h_2 - h_1)}{\mathrm{L}}\right]x \qquad \text{Equation 6.1}$$

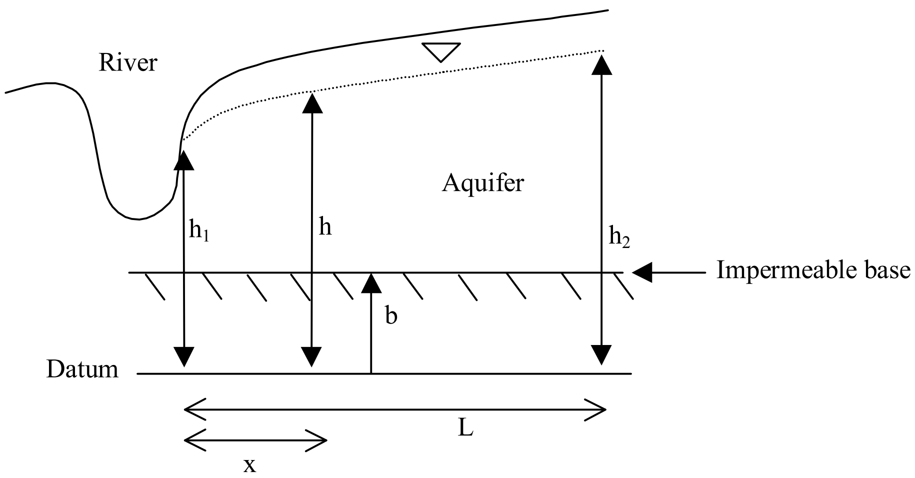


Figure 6.1 Schematic diagram for the analytical solution of unconfined flow to a river

Using an Excel spreadsheet, Equation 6.1 was used to calculate the base of the aquifer (b), given input values for h, h_1, h_2, l, and x obtained from field measurements (Appendix 22). The aim was to compare the effective thickness of the aquifer beneath the river with the expected thickness based on geological data. An investigation was also carried out to see whether h_1 was equal to the river stage, or contained an additional head component i.e. a seepage face.

6.2.2 Analytical Model, Transient Solution

Transient analytical solutions have been applied successfully in practical studies of ocean-aquifer (Erskine, 1991) and river-aquifer interactions (Reynolds, 1987). De Marsily (1986) presents the following analytical solution for aquifer heads given a sudden step change in river stage.

$$h(x,t) = h_0 . erfc\ (x\sqrt{\frac{S}{4Tt}})$$ Equation 6.2

h = change in head at distance x from the river at time t. [L]

t = time after the stage increment has occurred [T]

h_0 = sudden rise in stream stage at time t=0 [L]

S = Storage [-]

This solution fits the case of a semi-infinite confined aquifer initially in equilibrium with the stream at one boundary. It may be applied to the case of an unconfined aquifer if the stage change is small enough that the saturated thickness can be considered constant

The river hydrograph may be approximated as a series of stepped increases and decreases in head (h_{0i}) occurring at times t_i after time zero (the beginning of the event). The effect of each increment, h_{oi}, of the river stage on the head in a monitoring piezometer at distance x, is calculated using Equation 6.2, and combined by superposition to give a final distribution of head changes within the piezometer. The model is fitted to the field data from the piezometer by varying storage and transmissivity (S/T). An Excel spreadsheet was constructed to carry out the fitting automatically (Appendix 22). Error residuals of the form $head_{observed} - head_{model}$ were calculated for each time step. An optimum fit for S/T was obtained using the solver function in Excel. Optimisation was based on minimisation of the sum of the squared residuals.

6.2.3 MODFLOW

The three-dimensional, finite difference, numerical flow model program MODFLOW (original version 88/96) was used in conjunction with Groundwater Vistas (version 3). The MODFLOW code (McDonald et al., 1988) was selected as it has been extensively validated in many groundwater studies including a regional scale model of the Birmingham Aquifer (Greswell, 1992). Details of the model setup and calibration are given in Appendix 23.

6.2.4 FAT3D

The computer code FAT3D (Mackay, 2001) is a 3-dimensional block centred finite difference numerical model which solves the equation for confined flow.

$$\nabla(k\nabla h)-\sum_{i=1}^{n} Q_i\delta(x-x_i)+R=S\frac{dh}{dt}$$

k = hydraulic conductivity	$[LT^{-1}]$
h = hydraulic head	[L]
Q_i = point sink at location x_i	$[L^3T^{-1}]$
x = location	[L]
x_i = location of point sink	[L]
R = distributed source	$[T^{-1}]$
S = specific storativity	$[L^{-1}]$
t = time	[T]
δ = dirac delta function	$[L^{-3}]$

Details of the model setup and calibration are given in Appendix 24.

6.2.5 UNSAT

The computer code UNSAT (Mackay, 2001) is a one-dimensional, box-centred, finite difference numerical model which solves Richards equation for flow using Van Genuchten's (1980) equations for the soil moisture characteristics to simulate the unsaturated zone. Details of the model setup and calibration are given in Appendix 25.

6.3 Investigation of groundwater flow paths across the river flood plain

The modelling objective was to investigate groundwater flow paths, at the flood plain and channel scales, and in the vertical and horizontal planes to establish the likely movement of contaminant plumes from source area to discharge point within the river.

6.3.1 Regional conceptual model of groundwater flow through the flood plain

The Tame meanders across an urbanised flood plain comprising clays, sands, gravels and artificial made ground to a depth of 5 m. Underlying the floodplain and rising to form the valley sides, the sandstone aquifer has an effective base at >100 m, as defined by its contact with the top of the less permeable Carboniferous Coal Measures.

The sandstone aquifer is a high porosity, low to moderate permeability formation. Flow is generally considered to be single porosity though the effect of fractures is locally important. Hydraulic anisotropy is present in the sandstone, with lower vertical conductivity resulting from the presence of low permeability mudstone layers within the formation. The alluvial sands and gravels are thought to be high-storage/high-permeability units and may have higher conductivity along buried channels in a down valley direction. The made ground exhibits a large range of storage and permeability values. The presence of clay horizons may lead to localised semi-confined conditions. The riverbed material comprises sand and gravel and is thought to provide little resistance to flow.

The regional groundwater flows to the river from the north (left) and south (right) sides of the valley and the river flow increases across the area due to groundwater discharge. Groundwater flow occurs through the sandstone in areas with a deep water table (>15m) such as the valley sides. In areas with a shallow water table (<5m), such as the flood plain, flow occurs through both the bedrock sandstone aquifer and the overlying superficial sand and gravel deposits. The valley sides exhibit steeper head gradients with head gradients pointed directly towards the river. On the flood plain, the head gradients are shallower and groundwater flow occurs more obliquely to the river through the flood plain, depending on the orientation of the river meanders.

Recharge to the aquifer is complex and spatially variable. The primary recharge from precipitation is controlled by urban land cover and is supplemented by leakage from mains water, sewers and canals.

Industrial abstraction wells are present in the area and have caused significant recharge of surface water into the aquifer during the periods of major abstraction in the 1900s. The reduction in the abstraction rates has led to groundwater rebound. Groundwater levels are now essentially fully recovered in the Tame Valley. Small fluctuations (<20cm) in water level do occur in the deep monitoring wells within the valley associated with intermittent abstractions and variations in recharge.

6.3.2 Groundwater flow to a river meander - MODFLOW model

The MODFLOW model was selected to represent groundwater flow over an area of 1.8 x 1.8 km^2 covering a section of the Tame 3.8 km in length with a large meander feature (Figure 6.2). The MODFLOW model was chosen for the following reasons: it allows simulation of the unconfined aquifer as a single layer in two dimensions under steady state conditions; it allows aquifer properties, fixed heads and recharge to be assigned on an individual cell basis across a user-defined grid in the horizontal plain; it incorporates designated river cells that allow flow between the aquifer and the river to be modelled based upon the river stage and a river conductance term; abstraction wells and monitoring wells may readily be incorporated

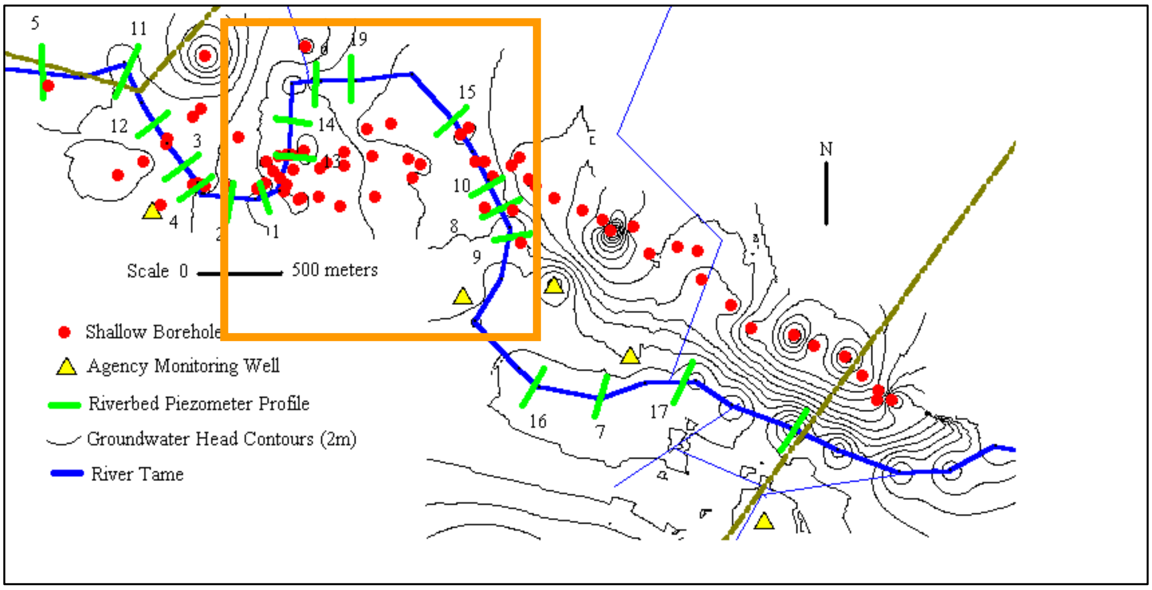


Figure 6.2 Regional Setting for MODFLOW Groundwater Flow Model

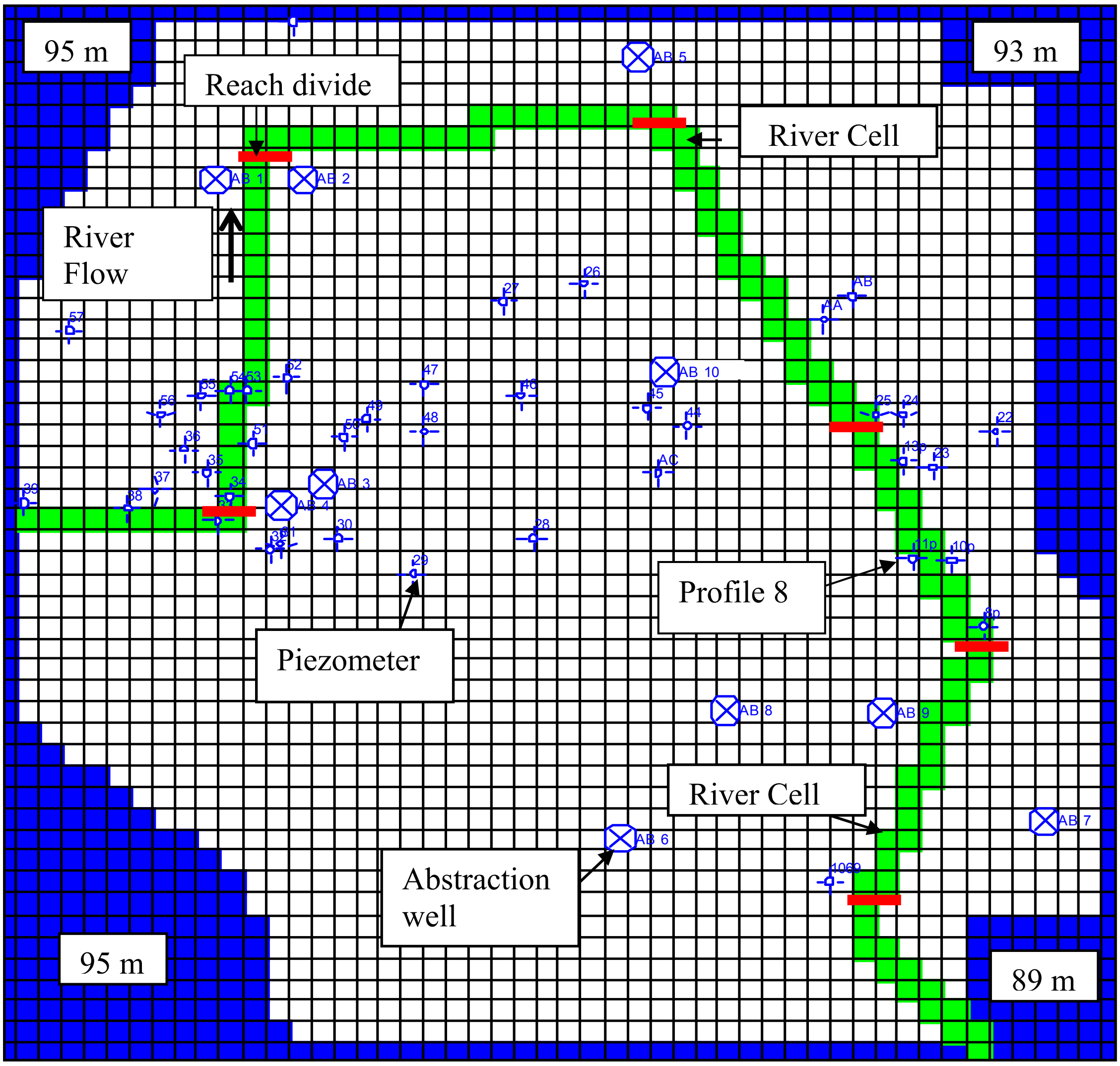


Figure 6.3 MODFLOW Model Grid and Boundary Conditions

into grid cells; and particle tracking function allows the visualisation of flow paths and the calculation of travel times.

6.3.2.1 Boundaries and grid layout

A uniform 50 x 50 grid, of cell size 36m, (Figure 6.3) was used with a single layer of 50m thickness of a uniform media type. The desired transmissivity was then obtained by varying the hydraulic conductivity. The grid was aligned approximately north-south with its centre at OS grid reference SP 0750091500 The ground elevation of the layer was set 1 m above the maximum fixed head values. No ground surface effects are therefore included in the model. The boundaries set for the model are as follows:

1. Fixed head boundaries around the perimeter of the model. Head levels were assigned on an individual cell basis using contour values obtained from mean head levels from piezometers and monitoring wells.
2. No flow at the base of the model.
3. River cells are assigned a fixed river stage and a conductance relating to the river surface area in the cell, the bed thickness and the bed hydraulic conductivity.

6.3.2.2 Model Parameters

<u>Hydraulic parameters</u>

Transmissivities in the range 10-320 m^2d^{-1} have been obtained from pumping tests within the Birmingham Aquifer (Jenkins, 1995). The sandstone underlying the model area comprises part of the Kidderminster formation with a geometric mean value for bulk permeability of 4.95 md^{-1} (Allen et al., 1997). The thickness of the unit above the underlying coal measures increases from 15m to 100m along the river reach. The actual effective thickness of the aquifer in relation to groundwater flow to the river is unknown owing to the presence of possible confining mudstone horizons within the sandstone. Aquifer units comprising alluvial flood-plain deposits and made ground, of unknown conductivity, overlie the sandstone. These exhibit a maximum thickness of 5 m in borehole geological logs. A uniform transmissivity for the layer was set in the range of 100 m^2d^{-1} to $600m^2d^{-1}$. No allowance was made in the model for any occurrence of lateral anisotropy as a result of depositional processes and buried palaeo-channels.

Recharge

An average recharge estimate of 0.00045 md^{-1} for the entire aquifer was obtained by Greswell (1992) amounting to 18% recharge from average rainfall of 736 mm $year^{-1}$ plus additional urban sources such as mains leakage. The effective rainfall (precipitation minus evapotranspiration) was multiplied by a modification factor incorporating drift type and thickness, and a modification factor for housing density and industry. Added to this were urban return flows multiplied by a drift modification factor. A later model that was available (Robinson, 2001) based on the earlier work, was used to determine the average recharge (0.00069 md^{-1}) over the 1.8 x 1.8 km^2 area of the current investigation. The level of recharge was probably higher than the average for the aquifer due to the limited thickness of permeable drift on the flood plain compared with the surrounding hills.

River-aquifer relationship.

Initial estimates of riverbed conductance were based on a cell length of 36 m, an average river width of 10 m and a conductivity of 2 md^{-1}. A riverbed thickness of 2 m was obtained from geological logs. The conductance was increased from 360 to 720 m^2d^{-1} during calibration of the model to take account of the presence of a seepage face and river lengths greater than the cell width. The river stage is well constrained by surveyed data for riverbed elevations and dry weather river levels.

6.3.2.3 Model Calibration

The model was constructed to reflect the style of the real system and to illustrate some of the important features of groundwater flow across the flood plain. As the model is simplistic and significantly constrained by the boundary conditions it was thought to be of little benefit to apply stringent calibration criteria.

The calibration of the model was carried out on a trial and error basis by altering conductivity, riverbed conductance, and recharge. The primary target was to achieve the groundwater discharge along the 3.8 km model reach that was consistent with field data. Baseflow analyses of 1999 gauging station data (Section 6.3.2) suggest a range in groundwater discharge to the river of 11 MLd^{-1} to 25 MLd^{-1} for the model. Dry weather flow gauging undertaken on 23/5/01 within the specific model reach, estimates a groundwater discharge of 8 MLd^{-1} to 16 MLd^{-1}, depending on which estimate of industrial and tributary discharges is used. The flow-gauging estimate lies within the range of values from the baseflow analyses, and is specific to the model reach rather than an average over 23.8 km as calculated from the gauging station data. Therefore, the model calibration criteria for groundwater discharge to the river were set as > 8 MLd^{-1} and <16 MLd^{-1}.

Average water-level data from 41 piezometers were used to provide an indication of the deviation in modelled head values from the real system. The model was kept simple with a uniform media type and the exact calibration to each piezometer level was not attempted. In general, the difference between modelled and observed heads was less than 0.5m and model head contours reflected the shape of the contoured field piezometric data.

The groundwater discharge to the river is strongly dependent on the close proximity of the outer boundary conditions (Section 6.7), and sensitive to changes in these (Section 6.9). In some places groundwater flows may be maintained at above natural levels by the fixed head boundaries. Significant groundwater abstraction is thought to occur within the model area but limited information is available on the quantity of abstraction and the pumping regime (intermittent). Therefore the calibration did not include the influence of the abstraction wells which are considered later (Section 6.8). For the preceding reasons the calibration was aimed at obtaining the minimum likely transmissivity. The parameters which yielded an acceptable discharge to the river of 9.7 MLd^{-1} were transmissivity 400 m^2d^{-1}, recharge 0.00069 md^{-1}, river conductance 720 m^2d^{-1}.

6.3.2.4 Sensitivity of parameters

The initial assessment of the sensitivity of the model was carried out indirectly during the calibration process based primarily upon discharge to the river and to a lesser degree on the deviation between the modelled and observed heads. The parameters of transmissivity and river bed conductance were selected for specific sensitivity analyses. Aquifer transmissivity values in the range 50 to 750 m^2d^{-1} were used and riverbed conductance values in the range 100 to 1500 m^2d^{-1}. The model discharge to the river was most sensitive to transmissivity. Riverbed conductance was of secondary importance. Riverbed conductance is of increased significance at low values (<100 m^2d^{-1}) but field data do not indicate a low conductance term. The model heads are most sensitive at low values of transmissivity <100 m^2d^{-1}; at values above this little change occurs between modelled and observed heads. Model heads are sensitive to low values of bed conductance (<100 m^2d^{-1}). The effect of different abstraction regimes and changing fixed heads by +/- 0.5 m in the river and perimeter boundary conditions was investigated (Sections 6.8 and 6.9).

6.3.3 Results and discussion

The model objective was to investigate groundwater flow paths across the flood plain to establish the likely movement of contaminant plumes in the horizontal plane from source area to discharge zone within the river.

The results of particle tracking (Figure 6.4) indicate groundwater flow is occurring down valley through the flood plain cutting across the river meander. This is significant when considering the source area of contaminants found discharging to the river.

Contaminant travel times to the river may vary significantly and can depend on which bank of the river the contaminant source is located. The most extreme case is seen in the northerly flowing reach of the river (between AB4 and AB1, Figure 6.3). In this area contaminants from a site located 100m from the river would have a travel time to the river of 2 years if located on the western bank but 16 to 18 years if located on the eastern bank, with flow occurring across the flood plain. Typical velocities of 50 m $year^{-1}$ were obtained for flow across the flood plain, with effective porosity set at 20%.

The results of the model were used to provide a regional context for more detailed modelling of a vertical cross section at the near channel scale to examine vertical flow paths within the aquifer.

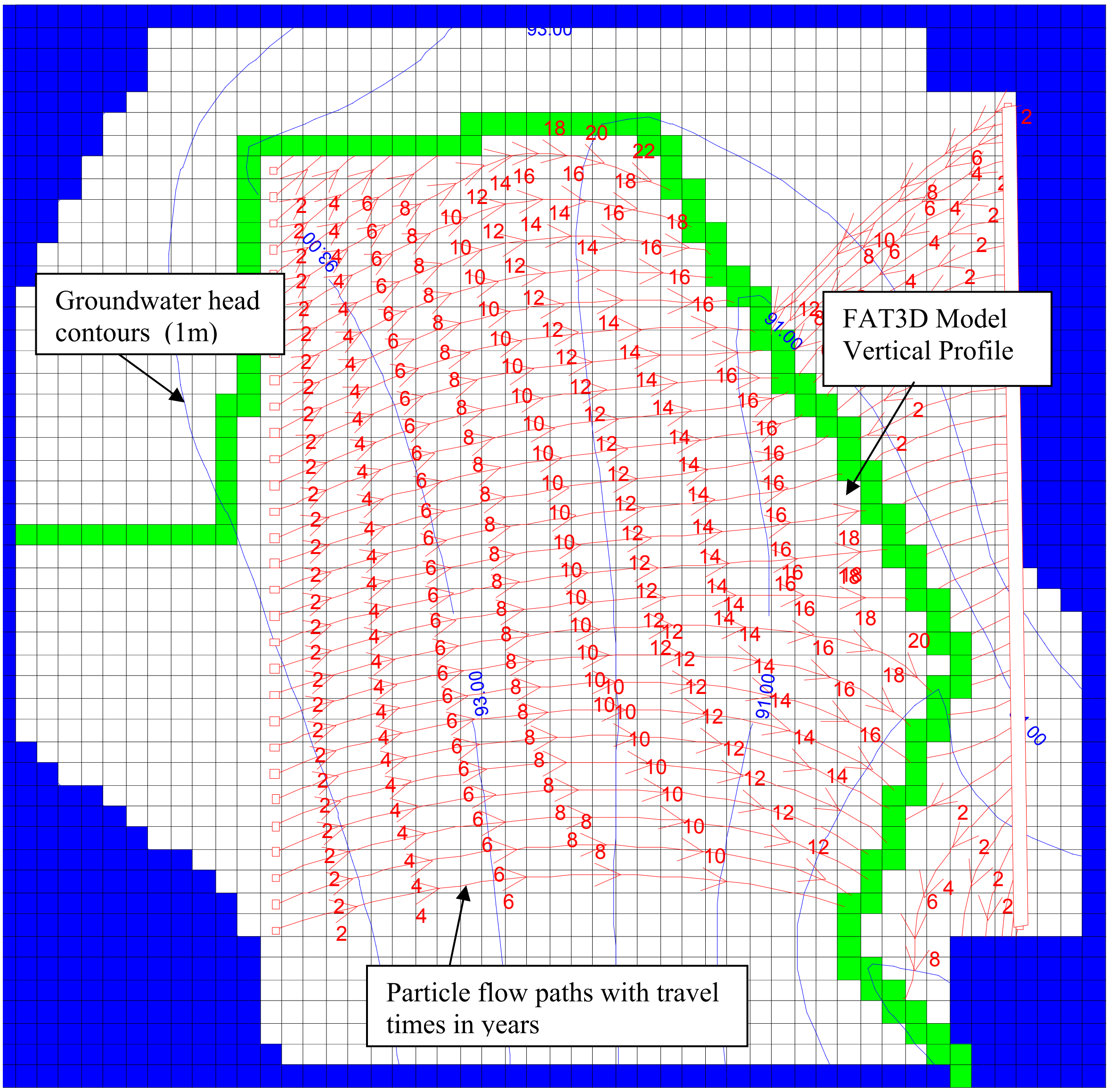


Figure 6.4 Results of Particle Tracking for the MODFLOW Model

6.4 Investigation of groundwater flow paths at the near channel scale in the vertical plane.

6.4.1 Conceptual model of groundwater flow to the river channel

The river meanders across an urbanised alluvial flood plain overlying the unconfined sandstone aquifer. The channel has been engineered in places to a uniform width of 12 m. The river banks are vertical to a height of 1.5 m above the riverbed. They are supported in places by permeable rock filled baskets (gabions). Dry weather river levels range from 0.3 to 0.5 m above a natural riverbed composed of sand and gravel. The regional modelling (Section 6.3) indicates that groundwater flow occurs towards the river from both sides of the channel in the selected model area. Flows discharging to the river may have many different origins and ages. Some will be recently recharged shallow groundwater flowing through the drift. Older, slower-moving, groundwater will be derived from deeper within the sandstone aquifer. Flow direction may change from horizontal to convergent in proximity to the river with a flow divide between flow to the river and underflow which discharges further downstream.

6.4.2 Application of the MODFLOW flood plain model to assess underflow

The calibrated regional MODFLOW model allows the examination of flows within each cell containing a river boundary condition. Flows across each cell face and discharge/recharge from the river were obtained. Calculations indicate the proportions of the total flow leaving the cell as discharge to the river and as underflow. The MODFLOW model gives estimates of the proportion of underflow occurring but cannot provide information on the distribution of flow in the vertical plane. For this, a detailed cross sectional model was created using FAT3D.

6.4.3 Groundwater flow to the river channel - FAT3D cross sectional model

The FAT3D code was selected to represent groundwater flow to the river through a vertical cross section (323 m wide and 100 m deep) with conditions set to be representative of profile 8 (Figure 6.5 and Section 6.8), grid reference O.S. 082913. The model was chosen to simulate flow through a multiple layer aquifer system in two-dimensions in the vertical plane, under steady state and transient conditions. The model allows fixed head boundaries to be incorporated at either edge of the model to simulate the regional head gradient and fixed head boundary conditions at the top centre of the grid to simulate the river. Boundary conditions could also be set on the bank cells for the simulation of the seepage face. The user-defined grid allowed a fine mesh size to be specified adjacent to the river to resolve the groundwater head distribution. The model allowed the representation of multiple geological units, including the riverbed, by assignment of aquifer properties on a cell by cell basis. Head data from each cell, and groundwater flows across each cell face and to each boundary condition, were used to define the groundwater flow paths to the river. The model also allowed simulation of transient river level conditions such as groundwater/surface water interactions during a river flood event, that are discussed in more detail in Section 6.10.

6.4.3.1 Boundaries and grid layout

A non-uniform, 2-dimensional grid of 100 x 100 cells was used to represent a vertical cross section 323 m wide and 100 m deep. The cell sizes in the x and z directions range from 0.3 to 100 m in the x direction and 0.1 to 5 m in the vertical z direction. Cells have a uniform thickness in the y direction of 1 m. The finest mesh size corresponds to the area immediately

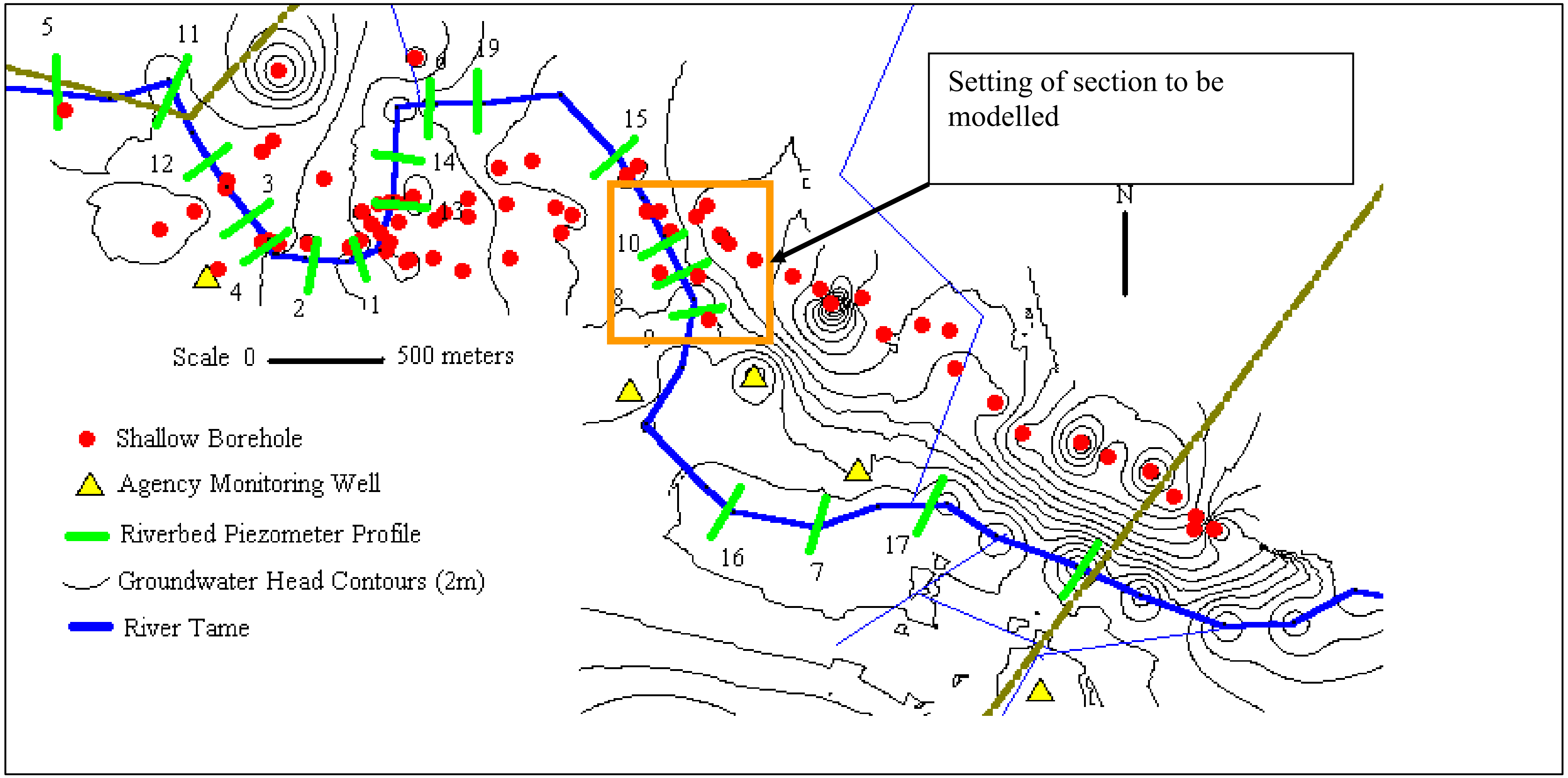


Figure 6.5 Regional setting for the FAT3D Model

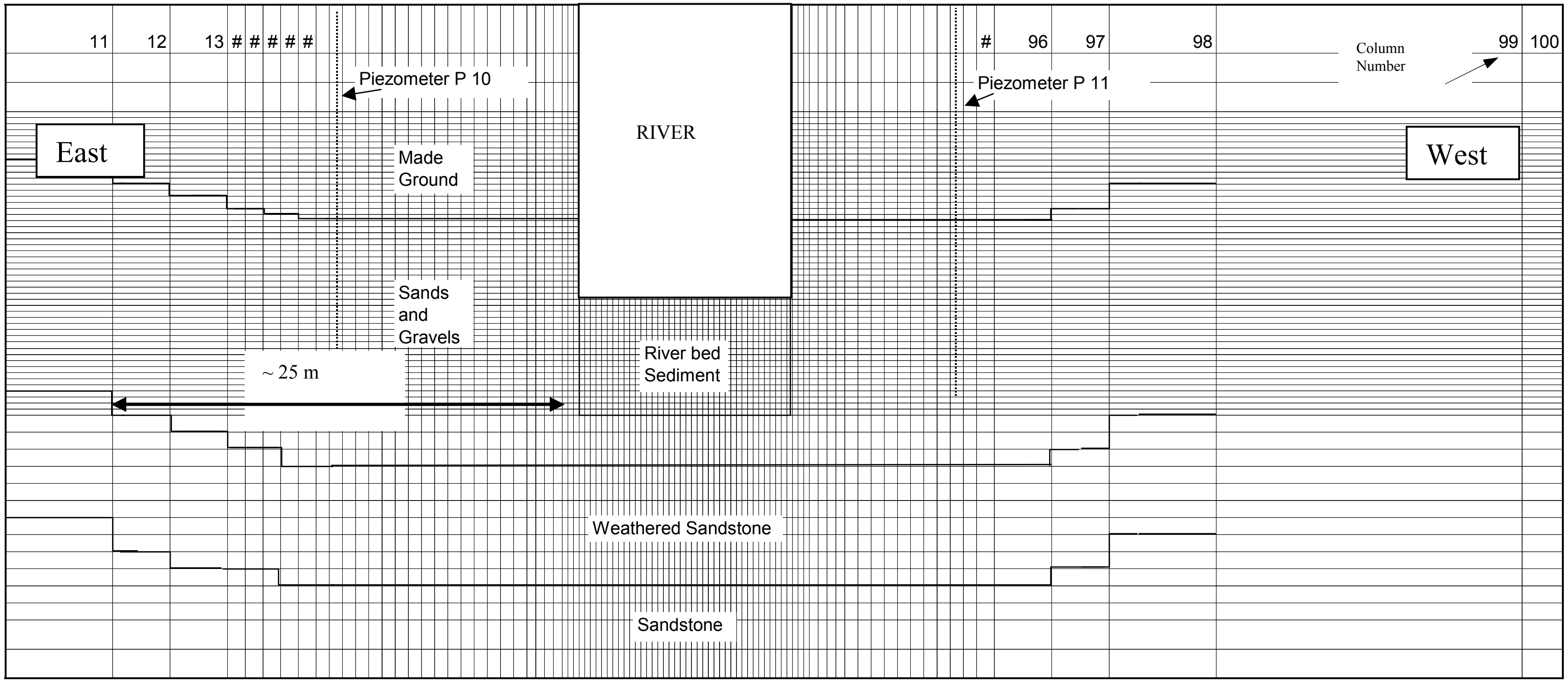


Figure 6.6 The Grid geometry adjacent to the river for the FAT3D Cross-sectional model.

adjacent to the river (Figure 6.6) and the origin corresponds to the eastern end of the section. The top elevation of the grid was set at 95 m, which is 2 m above the maximum fixed head values, at approximate ground level for the flood plain. The boundaries set for the model are as follows:

1. Fixed head boundaries at the east and west edges of the model. The eastern boundary lies 175 m from the river bank and a head value of 93.25 m.a.o.d was set, based on average water levels from borehole 19. The western boundary lies 130 m from the river bank and a head value was set by calibration (Section 6.4.3.4) at 92 m.a.o.d.
2 Fixed head boundaries for the river cells. The steady-state river was represented using fixed head cells (90.3 m.a.o.d.) across the river bed (38 cells x 0.3 m in width) and in the adjacent banks (3 cells x 0.1 m in height). Under transient conditions additional fixed head cells were added to the banks to reflect the increase in river depth.
3 Seepage cells were located in the banks above the river cells. These cells can only discharge groundwater when the groundwater head is above the cell centre elevation.
4 No flow boundary at the base of the model.

6.4.3.2 Adaptation of model to represent the unsaturated zone

The FAT3D modelling code assumes fully saturated flow and requires some adaptation to model the unsaturated zone. Field data on the location of the water table are needed to constrain the model (Section 5.8). Areas containing the unsaturated zone have been classed as no flow areas to prevent lateral flow through this zone. Tests performed under steady state conditions using a uniform geology (Kx,Kz = 5 md^{-1}) indicated a 7% reduction in flow to the river and a 7 cm rise in modelled river bank piezometer heads when the no flow zone for the unsaturated zone was incorporated. Cells were considered to contain the water table if they were intersected by straight lines projected from the eastern and western model boundaries to the nearest river bank. The elevations of the endpoints of the lines were equal to the observed steady state water levels in the adjacent riverbank piezometer at one end and the fixed head boundary conditions at the other. Cells above this level were assigned no flow conditions, as were all cells above ground level.

For transient conditions cells are assigned a specific storage coefficient. In addition, a specific yield component was set for cells that may contain the water table at some point during the event (otherwise set to zero). The field data indicated all water level fluctuations occurred within the gravel for the modelled event and only this unit was assigned a value of specific yield. For transient conditions the base of the unsaturated no flow zone must be raised. In order to estimate how much it had to be raised a transient simulation using the geology from the calibrated steady state model with specific yield set as zero was used to determine the likely maximum water levels. No flow was then assigned to levels one cell above the greatest head obtained in the riverbank observation piezometer (P10) (91.7 m.a.o.d). In all the subsequent transient simulations which incorporated values of >0 for specific yield the heads in the riverbank piezometers were >10cm below the base of the no flow zone. The model is not ideal for the simulation of transient unsaturated conditions but does provide an insight into the processes that occur during groundwater/surface water interaction.

6.4.3.3 Model Parameters

<u>Hydraulic parameters</u>

Initial model runs were undertaken with a uniform geology and more complexity was added in later runs to include five geological units identified in borehole logs representative of sandstone, weathered sandstone, sand and gravels, made ground and riverbed sediments. The geometry of the geological units was based upon sections constructed from borehole logs (Section 5.8). Each geological unit was assigned values of conductivity in the x (Kx) and z (Kz) directions. Kx values ranged from 0.1 to 10 md^{-1} and Kz values 0.1 to 5 md^{-1} (Sections 3.5 and 6.5.3.1). In general Kz values were set at 1/10th of Kx values to simulate anisotropy.

Recharge

Recharge was set to zero in the steady state model as dry weather conditions were being simulated and no data were available on other sources/sinks such as mains leakage or evapotranspiration. Recharge was also set to zero in the transient model as the flood event studied occurred after a dry weather flow period and it was thought unlikely that recharge would have reached the water table, at depths of 2m on the riverbank, during the time period (1.5 days) modelled.

Groundwater head data

Steady state groundwater head data were derived by averaging continuous logging data from piezometers (P10, P11) on the riverbanks during a dry weather flow period between 22/5/01 and 31/5/01 (Appendix 12). Transient river stage data were collected for P10 and P11 at five-minute intervals during a river flood event on the 6/3/01 and 7/3/01.

Steady state head data are used from a series of 17 piezometers within the riverbed that comprise Profile 8. The riverbed piezometer head data were collected during a low flow period on 29/08/00 and represent a single point ‘snapshot’ in time. Groundwater head data for the Eastern boundary condition were derived from dipping records for Severn Trent Borehole 19 between 1994 and 1999, as access to the borehole is no longer possible.

River level data

Steady state river level data are representative of typical dry weather flow conditions recorded from a stilling well at Profile 8 between 22/5/01 and 31/5/01. Transient river stage data were collected from the stilling well at 5 minute intervals during a river flood event on the 6/3/01 and 7/3/01.

6.4.3.4 Model Calibration

The objective of the model is to simulate the groundwater flow system at profile 8 and its interaction with the surface water. The model uses the available geological, piezometric and river discharge data from Profile 8. However, there is considerable heterogeneitity within the geology of the system at this location, with the possibility of multiple aquifer units. Also, limited data exists on the aquifer properties. Alternative combinations of aquifer properties could achieve calibration of the model, making the benefit of exact calibration limited. The model was first examined under steady state conditions to select a suitable model upon which to perform transient calibration.

Calibration of the steady state model is based on the net groundwater discharge to the river and the squared error in hydraulic head $(head_{model}\text{-}head_{observed})^2$ for the riverbank and multilevel river bed piezometers. River discharge measurements indicate groundwater inputs to the river of between 2 and 4 m^3d^{-1} per unit length of channel and the MODFLOW model estimated groundwater inputs of 3 to 5 m^3d^{-1} per unit length of channel. A variation of +/- 5 cm was deemed acceptable for the riverbank piezometers. The riverbed piezometers were considered of secondary importance in the calibration as they represent a single time point and also have an increased likelihood of survey error. A total of 0.05 cm^2 for the sum of

$(head_{model}\text{-}head_{observed})^2$ for all 17 piezometers was considered acceptable for the calibrated model.

The calibration was performed in several stages. An initial calibration was carried out on a trial and error basis with a two layer geology to obtain a suitable fixed head for the western boundary of the system for which no field data were available. Further calibration and sensitivity analyses was carried out on a trial and error basis by selecting varied blocks of cells to represent individual geological units and varying values of vertical and horizontal hydraulic conductivity for these blocks within a range suggested by field and literature data. The complexity of the geology was gradually increased from uniform to incorporate a total of five divisions. Several different geological types were required to calibrate the model effectively and the riverbed sediments exerted a significant control for head values adjacent to the river. A final calibrated model was selected for later sensitivity analyses and transient modelling on the basis that the model incorporated all five geological divisions and produced an acceptable error using conservative hydraulic parameters (Section 6.5.3.1).

Transient calibration was performed by varying the specific yield of the water table fluctuation zone and comparing the observed riverbank piezometer (P10 and P11) hydrographs with the model hydrographs. Specific yield was varied with minimum increments of 0.001 to obtain the closest fit between the modelled and observed hydrograph amplitudes. Details of the model calibration are presented in Appendix 24.

6.4.3.5 Sensitivity of parameters

The initial assessment of the sensitivity of the model was carried out indirectly during the steady state calibration process. The general sensitivity of the system to varying complexities of geology was examined and an investigation of the effect of raising and lowering the fixed heads made. For the transient conditions a systematic evaluation of sensitivity was carried out on the values of Kx, Kz and specific yield for the five geological units. A single value was changed on each occasion and represented the maximum or minimum expected value for the parameter based on data from the field and literature. The model was found to be insensitive to a 10% reduction in the grid size. The results of the sensitivity analyses are presented in more detail in the later sections of this chapter and in Appendix 24.

6.4.4 Results and discussion

Data from both the MODFLOW flood-plain model and the FAT3D channel-scale model are used in the following discussion.

The majority of the MODFLOW river cells display underflow in which a proportion of the total flow entering the cell is not discharged to the river (Figure 6.7a). This implies the existence of a flow divide beneath the river between shallow groundwater discharging to the river and deeper groundwater flowing in the direction of the regional gradient which will discharge to the river further down gradient. The proportion and amount of underflow (Figure 6.7b) is variable along the reach reflecting changes in groundwater flow paths beneath the river and changes in the river flow direction. Proportions of underflow range from 100% at the upstream end of the study reach to 10-20% further downstream reflecting increases in the head differential between the river and the aquifer downstream. The high proportion of underflow at the upstream end of the study reach may in part be related to channel engineering raising the river level in the area. Large variations in the apparent underflows between adjacent cells are a product of the coarse grid dimensions and the associated problems of representing a diagonal feature across the grid.

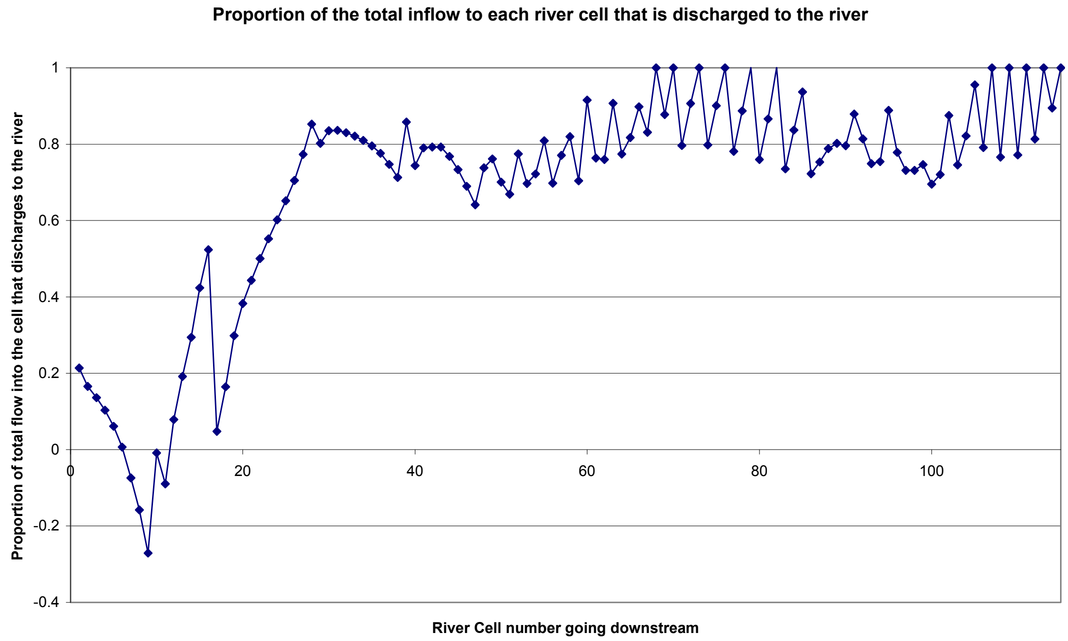


Figure 6.7a MODFLOW model - proportion of river cell inflow discharging to the river

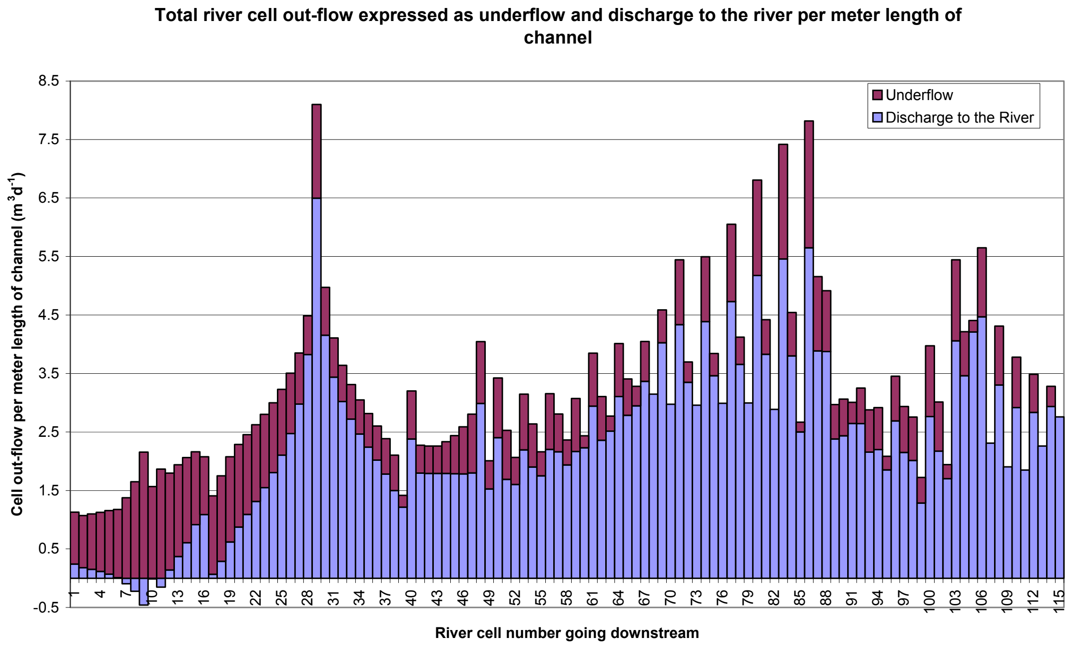


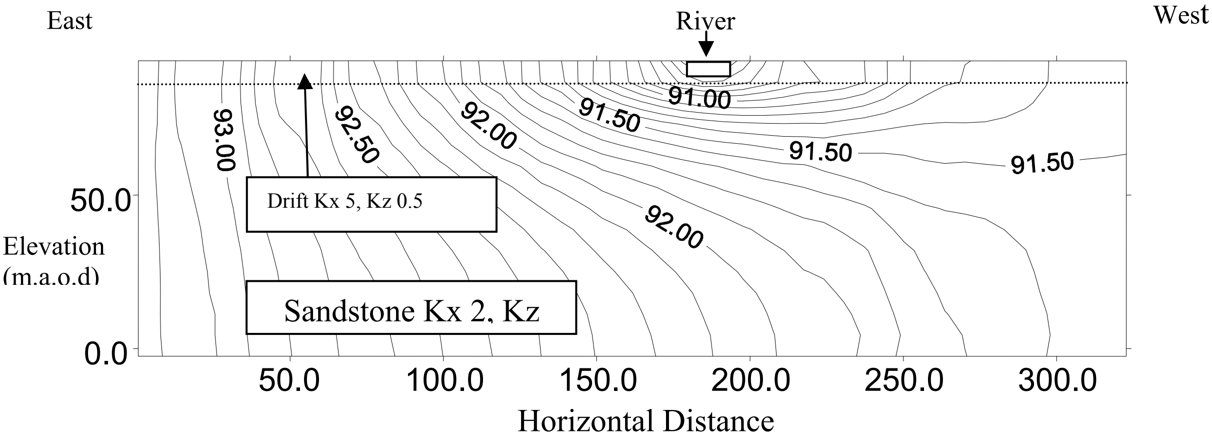
Figure 6.7b MODFLOW groundwater discharge to the river and underflow

The FAT3D model has been used to look at two-dimensional flow paths in the vertical plane at profile 8 on a section of river where regional modelling (Figure 6.4) indicates groundwater flow is nearly perpendicular to the river and flow parallel to the river can be ignored.

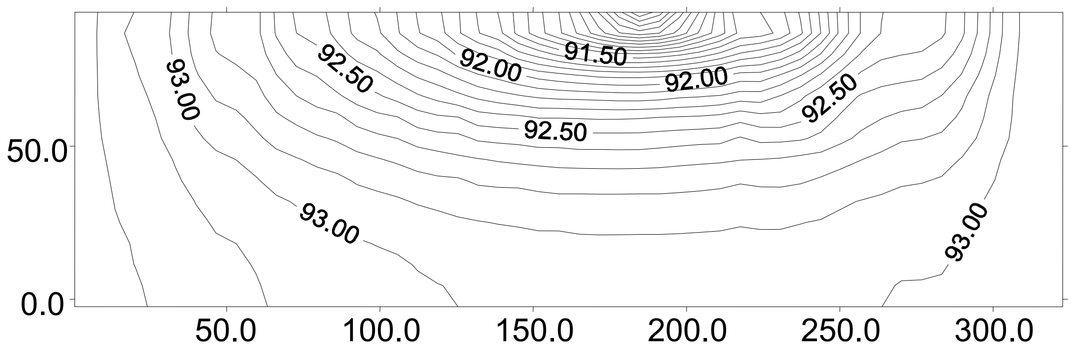
A series of steady-state model runs were carried out in which the fixed head at the western boundary was varied. The geology in each case was simulated as two distinct units comprising 5 m of superficial drift (Kx 5 md^{-1}, Kz 0.5 md^{-1}) overlying 95 m of bedrock sandstone (Kx 2 md^{-1}, Kz 0.2 md^{-1}). The head distribution was contoured (Figure 6.8 a,b,c) to visualise the flow paths and to locate the flow divide. The division of flow between groundwater discharging to the river and underflow out to the flood plain is evident at depths greater than 50 m. It is apparent that the location of the flow divide is dependent on the Western head boundary (WHB). Depths of the flow divide increase with increases to the WHB, from ~50 m when WHB is 91.5 m.a.o.d (Figure 6.8a) until all flow is directed towards the river when WHB is 93.09 m.a.o.d. (Figure 6.8b) i.e when the regional head gradient is zero.

The modelling indicates that flow through the overlying drift is primarily horizontal and therefore dominated by Kx. Flow through the underlying bedrock aquifer is in general horizontal until converging flow commences within 100 m of the river and Kz becomes increasingly important. The shallow (young) groundwater is seen to flow towards the sides of the river channel while deeper (older) groundwater flows towards the central portions of the channel.

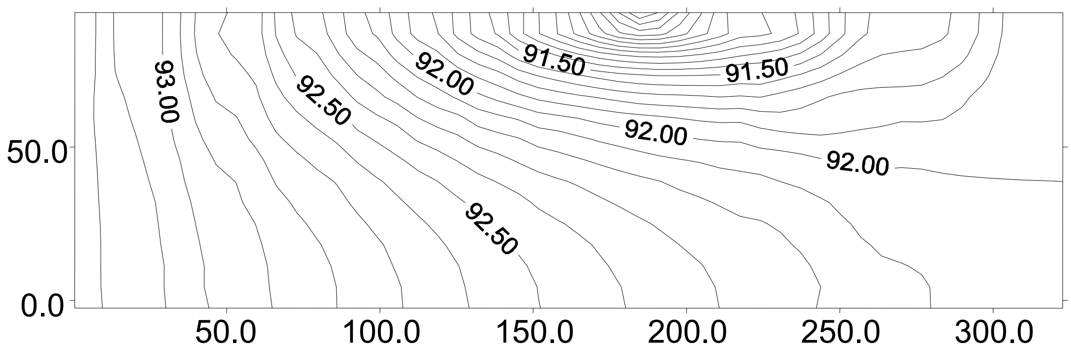
The effect of a complex four-layer geological system on flow rates across the east and west fixed head boundaries was examined. Units representing the permeable weathered sandstone



(a) Fixed Head Boundaries: East 93.25, West 91.5 (m.a.o.d)



(b) Fixed Head Boundaries: East 93.25, West 93.09 (m.a.o.d)



(c) Fixed Head Boundaries: East 93.25, West 92 (m.a.o.d)

Figure 6.8 FAT3D Model Groundwater Head Contours for Different Fixed Head Boundary Conditions and gravel formations with a combined thickness of 8 m, overlay 85 to 90 m of the less permeable bedrock sandstone. The model was run in steady state with bedrock permeability remaining constant (Kx 2 md^{-1}) and the overlying deposits with Kx values of 3 and 5 md^{-1} (Figure 6.9a) and Kx 5 and 10 md^{-1} (Figure 6.9b). The highest flow rates are seen to occur within the top high conductivity layers with flow through the bedrock aquifer slower and decreasing with depth. Velocities through the drift range from 0.18 to 0.4 md^{-1} while average velocities through the deep aquifer are <0.05 md^{-1} (effective porosity = 0.2). The highest proportion of the total flow to the river is derived from the eastern boundary with small contributions to flow from the upper 50m of the western boundary. Of the flow to the river that occurs across the western boundary >50% is derived from the superficial deposits whereas the majority of flow from the eastern boundary is derived from the bedrock sandstone. Flow is seen to occur in two directions across the western boundary, with shallow groundwater flowing towards the river and deeper groundwater (below 40 m.a.o.d) flowing away towards the flood plain.

These results may be considered in relation to contaminant discharge to the river. The groundwater quality of the bedrock aquifer on the eastern bank will provide the greatest contribution to the total discharge quality. Shallow groundwater will allow rapid transit of contaminants to the river and will provide higher proportions of the total discharge to the river with higher hydraulic conductivities of the gravel. Higher values of conductivity in the gravel will increase the contribution from the western bank. Contaminants present in older deeper groundwater derived from more distant sources will tend to discharge through the centre of the channel and contaminants in the shallow groundwater from sources more local to the river will tend to discharge closer to the channel sides. Contaminant travel velocities may differ by

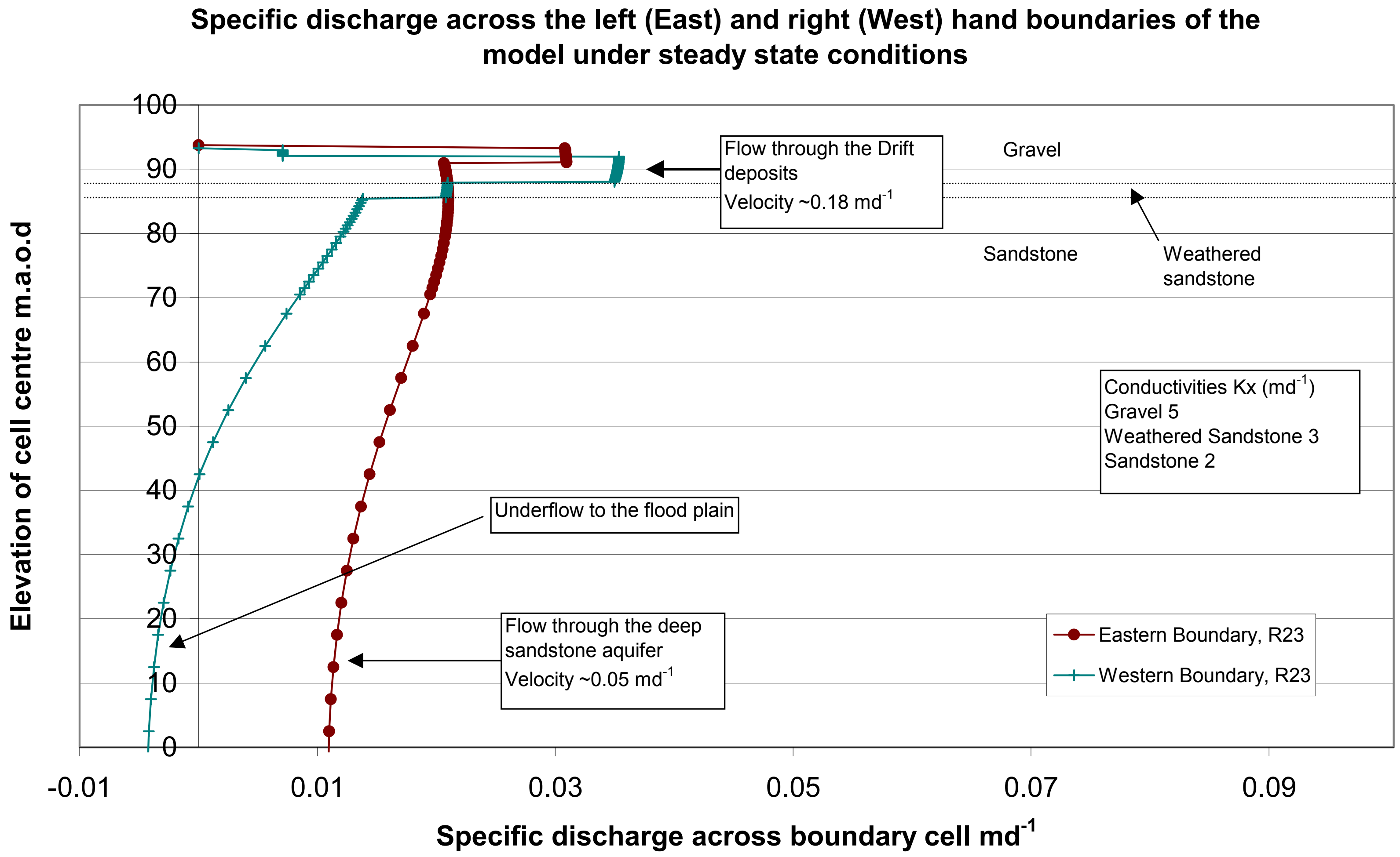


Figure 6.9a Specific discharge across the FAT3D model boundaries when gravel $Kx = 5\ md^{-1}$

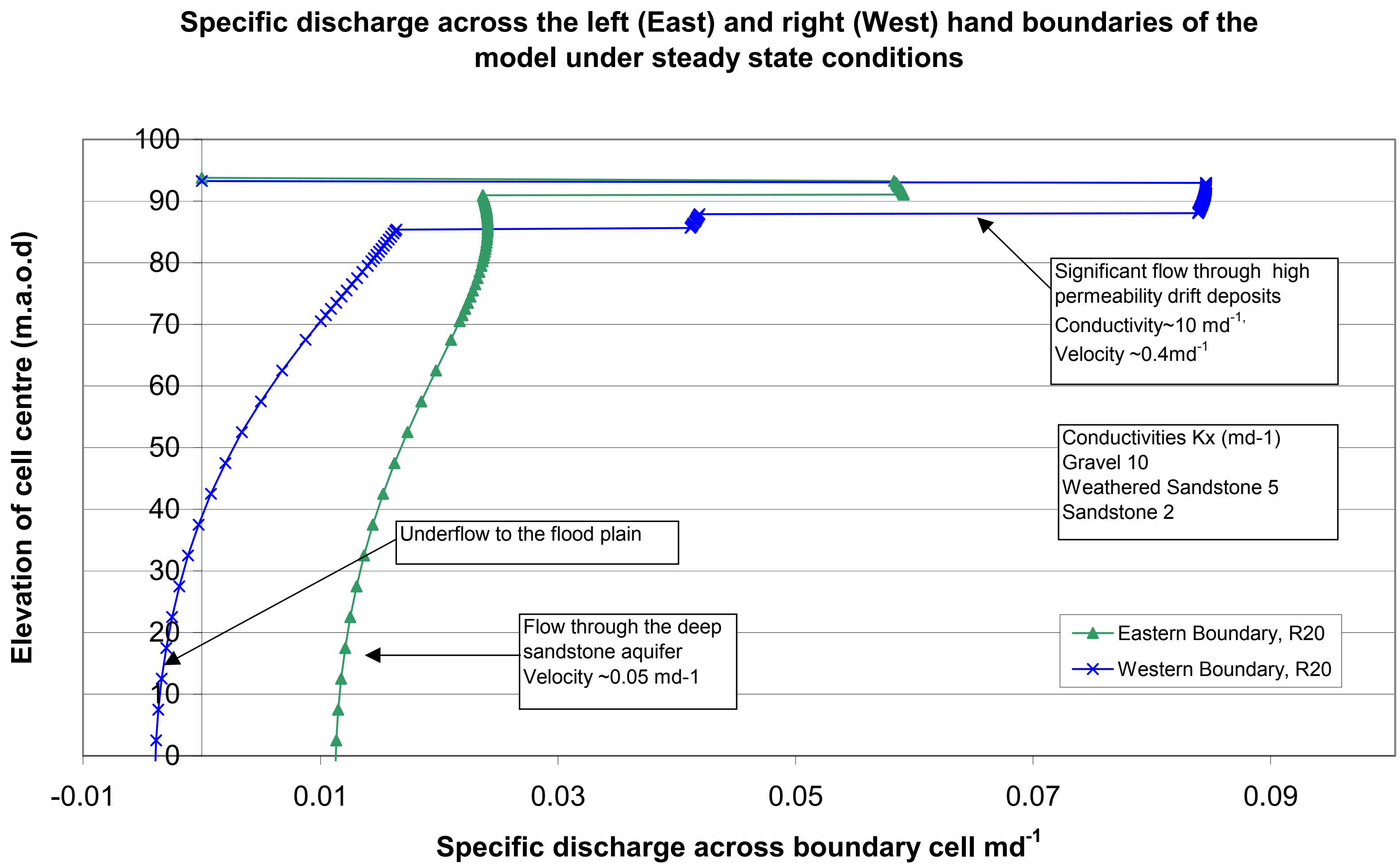


Figure 6.9b Specific discharge across the FAT3D model boundaries when gravel Kx = 10 md^{-1}

an order of magnitude between the deep and shallow groundwater. Shallow ground water will allow rapid transit of contaminants to the river with limited time for natural attenuation to occur, but will also recover rapidly on removal of the contaminant source. Deeper groundwater, with its slower velocities, will provide a long-term source of contaminants to the river. Contaminant concentrations within the shallow groundwater high flow zone (<8 m in width) are likely to be higher than those in the deeper groundwater. Contaminated groundwater below the flow divide (depths >50m) will have no impact on the river water quality in the modelled area.

The geology of the system is seen to be a significant control on groundwater flow paths, discharge rates and travel times, and therefore requires more detailed investigation.

6.5 Investigation of the geological controls on groundwater flow to the river

6.5.1 Conceptual model

Groundwater flow to the river occurs through a multiple aquifer system consisting of the bedrock aquifer and overlying superficial deposits. Flow is predominantly horizontal until flow convergence commences adjacent to the river and focuses flow to discharge through the sediments of the riverbed and banks. The riverbed materials are of particular significance in controlling the focussed groundwater discharge to the river.

6.5.2 Application of the MODFLOW flood plain model

The calibrated MODFLOW flood-plain model was used to perform sensitivity analyses of groundwater discharge to the river and groundwater head by varying the aquifer transmissivity and riverbed conductance terms. The model employs a lumped estimate of transmissivity for the geological sequence as a single unit across the model area. More detailed modelling of the geology was required to investigate the control of the individual geological formations and the FAT3D model was used for this purpose.

6.5.3 Application of the FAT3D channel scale model.

The geological controls on groundwater discharge to the river include the spatial distribution of individual geological units and their associated hydraulic parameters of Kx and Kz. The FAT3D near channel model is well constrained by head data, and the expected groundwater discharge to the river. The geology is known from borehole logs, and the model incorporates five geological units representative of sandstone, weathered sandstone, gravel, made ground and riverbed sediments. The model was used to carry out sensitivity analyses for the hydraulic parameters of Kx and Kz under both steady-state and transient conditions.

6.5.3.1 Hydraulic parameters

The conductivity values for the five geological units incorporated into the calibrated model along with the maximum and minimum values used for the sensitivity analyses are presented in Table 6.2. Initial values of Kx = 5 md^{-1} and Kz = 0.5md^{-1} for the Kidderminster Sandstone were reduced during the calibration process as additional geological units were added. Weathered sandstone was assigned a higher Kz value than the bedrock due to its likely decementation, increased fracture density and the low frequency of mudstone units. Estimates for Kx (5md^{-1}) in the sands and gravel were based on rising head tests conducted within environment agency flood defence boreholes in the Tame Valley.

Geological Unit	Parameter values (md^{-1})					
	Kx max	Kx min	Kx cal	Kz max	Kz min	Kz cal
Sandstone	5	0.5	2	1	0.1	0.2
Weathered Sandstone	5	0.5	3	2	0.1	1
Alluvial gravel	15	0.1	5	5	0.1	0.5
Made ground	20	0.05	1	2	0.005	0.1
Stream bed	15	1	5	10	0.01	0.5
max = maximum realistic value, min = minimum realistic value, cal = calibrated model value (a single value changed on each run)						

Table 6.2 Conductivity values used in the FAT3D model sensitivity analyses.

The vertical conductivity (Kz 0.5 md^{-1}) was assigned as 10% of Kx. The value of Kz is low for gravel but borehole logs indicate the occurrence of discontinuous units of clay which justify the lower value. Data on the made ground are sparse (4 values) but indicate high variability with values of conductivity 0.04, 0.03, 10, 20 md^{-1}; mean estimates of Kx 1 md^{-1} ,Kz 0.1 md^{-1} were used. A vertical conductivity for the riverbed sediments Kz 1 md^{-1} was assigned, based on the results of falling head tests but subsequently reduced during the calibration process.

6.5.3.2 Sensitivity analyses

The sensitivity of total groundwater discharge to variations in conductivity was carried out on the calibrated model under transient conditions. The conductivity of a single unit was changed on each occasion to the maximum or minimum expected value for the unit. The effect of reducing the sandstone aquifer thickness to 10 m was also examined.

6.5.4 Application of the analytical model of aquifer thickness

The analytical model provided information on the effective saturated thickness of the aquifer adjacent to the river. The model incorporated field data representative of groundwater head values at three locations within the cross section parallel to groundwater flow on the eastern bank adjacent to profile 8 (Section 5.8). Head conditions were imposed at the eastern boundary, the river boundary and on the river bank at the site of piezometer P10. The sensitivity of the model to variations in the head values was examined to see if the groundwater boundary at the river was equal to river stage alone, or contained an additional head component i.e. a seepage face. The initial model head conditions were based on mean (DWF) values for the river stage (90.3 m.a.o.d) and water levels in the piezometers P10 (90.86 m.a.o.d) and BH19 (93.25 m.a.o.d) at distances of 13.27m and 175m respectively from the river bank.

6.5.5 Results and discussion

The geological controls on groundwater flow to the river may be considered in terms of the hydraulic conductivity and the spatial distribution (thickness) of each geological unit. Modelling was used to investigate these controls under horizontal and convergent flow conditions. The control of specific yield on groundwater – surface water interactions will be considered in Section 6.10.

The regional MODFLOW modelling indicated that a range in transmissivity (T) of between 400 and 750 m^2d^{-1} is required in order to obtain realistic levels of discharge to the river of 8-16 MLd^{-1} without significantly altering the boundary conditions. These estimates of T could be low if significant abstractions are currently occurring within the model area. The values of T are higher than expected for the bedrock sandstone indicating levels of hydraulic conductivity >4 md^{-1} over a large effective aquifer thickness (>100 m) with limited impact from impermeable mudstone layers. The high values of T may incorporate a contribution from alluvial gravel on the flood plain. However, the average thickness of the gravel is 5 m and values of conductivity would need to be considerable to reduce the contribution from the bedrock aquifer. Limited field data suggest values of conductivity in the gravel are generally <20 md^{-1}.

The sensitivity analyses carried out using the MODFLOW model indicate that aquifer T is the dominant control and riverbed conductance is of secondary importance in controlling the groundwater discharge to the river. The model is most sensitive to the lower ranges of conductance (<100 md^{-2}) but field data indicate that the river bed sediments do not pose a significant barrier to groundwater discharge. Acceptable flows to the river are obtained with T = 400 m^2d^{-1} and conductance values in the 300 - 400 m^2d^{-1} range. A doubling of conductance from 360 m^2d^{-1} to 720 m^2d^{-1} (constant T = 400 m^2d^{-1}) produced an 8% increase in discharge to the river, and a tenfold increase to 3600 m^2d^{-1} produced an 18% increase in discharge to the river from 8.9 to 10.5 MLd^{-1}. In contrast to this, increases in groundwater discharge to the river are directly proportional to increases in aquifer T. The MODFLOW model indicated little impedance to flow from the riverbed sediments and a high aquifer T (>400 m^2d^{-1}), possibly incorporating a high conductivity gravel unit.

The FAT3D model indicates that flow through the overlying drift is primarily horizontal and dominated by Kx. Flow through the underlying bedrock aquifer is in general horizontal until convergent flow commences within 100 m of the river when Kz becomes increasingly important. Flow immediately beneath the river, and through the bed sediments is vertical and controlled by Kz. Average conductivity is represented by an arithmetic average of Kx where horizontal flow occurs, but by the harmonic mean of Kz when vertical flow dominates. The harmonic mean accounts for the importance of any flat-lying, low-permeability horizons, even those of only minor thickness. Clay and fine grained over-bank deposits are found in the alluvial sequence of the flood plain, and mudstone horizons occur within the Triassic Sandstone with fracture flow likely to be dominant through these units. Estimates of the vertical and horizontal K were made for the steady state model (Table 6.3). Made ground was not included as it remained unsaturated.

The effect of the low vertical conductivity in a uniform geology was examined by changing the initial Kx and Kz conditions of 2 md^{-1} to Kx 2 md^{-1} and Kz 0.2 md^{-1} . This created a 61% reduction in steady state discharge to the river, as a result of reducing vertical flow rates in the zone of convergent flow. Knowledge of the vertical conductivity of the bedrock sandstone is therefore important in constructing an accurate model of the discharge to the river.

A value of 206 m^2d^{-1} was derived for the horizontal T of the multilayer steady state FAT3D model compared with a minimum T of 400 m^2d^{-1} for the MODFLOW model. The reason for this is that groundwater head values used for the FAT3D model boundary conditions are considerably higher than the average head conditions prevailing over the entire 3,800 m

Material	Thickness m (b)	Horizontal Conductivity md^{-1} (Kx)	Vertical Conductivity md^{-1}(Kz)
Sandstone	90.25	2	0.2
Weathered Sandstone	2	3	1
Gravels	3.95	5	0.5
Total Depth	96.2	-	-
$K_{xAverage} = \sum_{1}^{n} \frac{K_n b_n}{b}$	-	2.14	-
$K_{zAverage} = \frac{b}{\sum_{1}^{n} \frac{b_n}{K_n}}$	-	-	0.209
Transmissivity m^2d^{-1}	-	206.25	-

Table 6.3 Average conductivity for the saturated thickness of the FAT3D steady state model.

MODFLOW model reach and that the calibration target used for the FAT3D model of a minimum groundwater discharge of 2 m^3d^{-1} is probably too low. MODFLOW results indicate that some of the highest groundwater discharges to the river occur in the vicinity of Profile 8 with flow rates of 4 to 5 m^3d^{-1} (Section 6.8). Values of transmissivity and conductivity derived from the FAT3D model should be regarded as minimum values.

The results of the MODFLOW model show that the riverbed poses no significant impedance to groundwater flow until the riverbed conductance term (length*width*K)/(thickness of the riverbed sediments) drops below 100 m^2d^{-1} which equates to an approximate conductivity of 0.6 md^{-1}. The FAT3D steady state model achieved calibration with a riverbed Kz conductivity of 0.5md^{-1} but considerable flow (>40%, Section 6.8) was also seen to occur horizontally through the sides of the river dependent on higher Kx values. Therefore the value of riverbed conductance of 720 m^{2}d^{-1} used in the calibrated MODFLOW model includes estimates of riverbed conductivity of 2-4 md^{-1} which incorporate values of both Kx and Kz for the riverbed sediments and the overlying drift.

The steady state FAT3D model was used to examine the effect of a series of evenly distributed impermeable obstructions simulating cobbles (dia. 30 cm) on the surface of the riverbed. This is representative of the 'armouring', by large clasts of the surface of the riverbed that occurs due to the erosion of finer material under high river velocities, which may lead to a significant reduction in riverbed conductivity. Riverbed coverage of 23% caused a 5% drop in discharge to the river. The overall importance of the riverbed sediments in controlling groundwater discharge in comparison to the other geological units of the FAT3D model was assessed during sensitivity analyses performed under transient conditions. The riverbed sediments were found not to be a significant control with a Kz maximum of 10 md^{-1} and a Kz minimum of 0.01 md^{-1} producing a 7% increase and a 10% reduction in discharge to the river respectively.

The FAT3D transient sensitivity analyses showed the dominant controls on groundwater discharge to the river in terms of conductivity to be the sandstone and gravel units. Discharge to the river was decreased by 40% with a reduction in sandstone Kx to 0.5 md^{-1}, and a fall of 37% occurred with a reduction of Kx for the gravel to 0.1 md^{-1}. Increased groundwater discharge to the river was most sensitive to an increased value of Kz to 1 md^{-1} in the sandstone that produced a 48% rise in groundwater discharge. An increase in discharge of

40% was related to a high value of Kx (15 md^{-1}) in the alluvial gravel and a rise of 35% due to an elevated value of Kx (5 md^{-1}) in the sandstone. The effect of an impermeable mudstone layer within the sandstone was simulated by reducing the unit thickness to 10 m. This caused a 48% reduction in groundwater discharge to the river and a 25 cm drop in the modelled head for piezometer P10.

The sensitivity analyses suggests that sufficient groundwater discharge to the river could be supplied under conditions of reduced conductivity or thickness in the sandstone and increased conductivity of the gravel. However the average head modelled for piezometer P10 fell by 20 cm under low Kx conditions in the sandstone. It fell by 25 cm when the sandstone thickness was reduced to 10 m and also fell by 17 cm under high Kx conditions in the gravel. This indicates that a combination of a high permeability gravel unit and a thin and/or low permeability sandstone unit is unlikely to represent the actual field conditions. Of most significance in controlling groundwater discharge under steady-state flow conditions is the transmissivity (T) of the sandstone aquifer. FAT3D results indicate a minimum T of 200 m^2d^{-1} for the sandstone. Therefore, flow must be occurring through a significant thickness of the aquifer.

The steady-state analytical model was used to derive an estimate of the saturated thickness of the aquifer based on field data. The initial value of saturated thickness derived was 1.07 m which is inconsistent with geological information that indicates sandstone to depths of greater than 100 m. A sensitivity analyses was undertaken (Figure 6.10) to investigate the impact of varying the groundwater head values to the likely maximum and minimum values. The calculated saturated thickness was most sensitive to variations in the value of the river

Sensitivity analyses of saturated thickness to different levels of seepage face (h1) and varying steepness of gradient between the observation piezometer (h) and the river, and varying distant boundary conditions (h2).

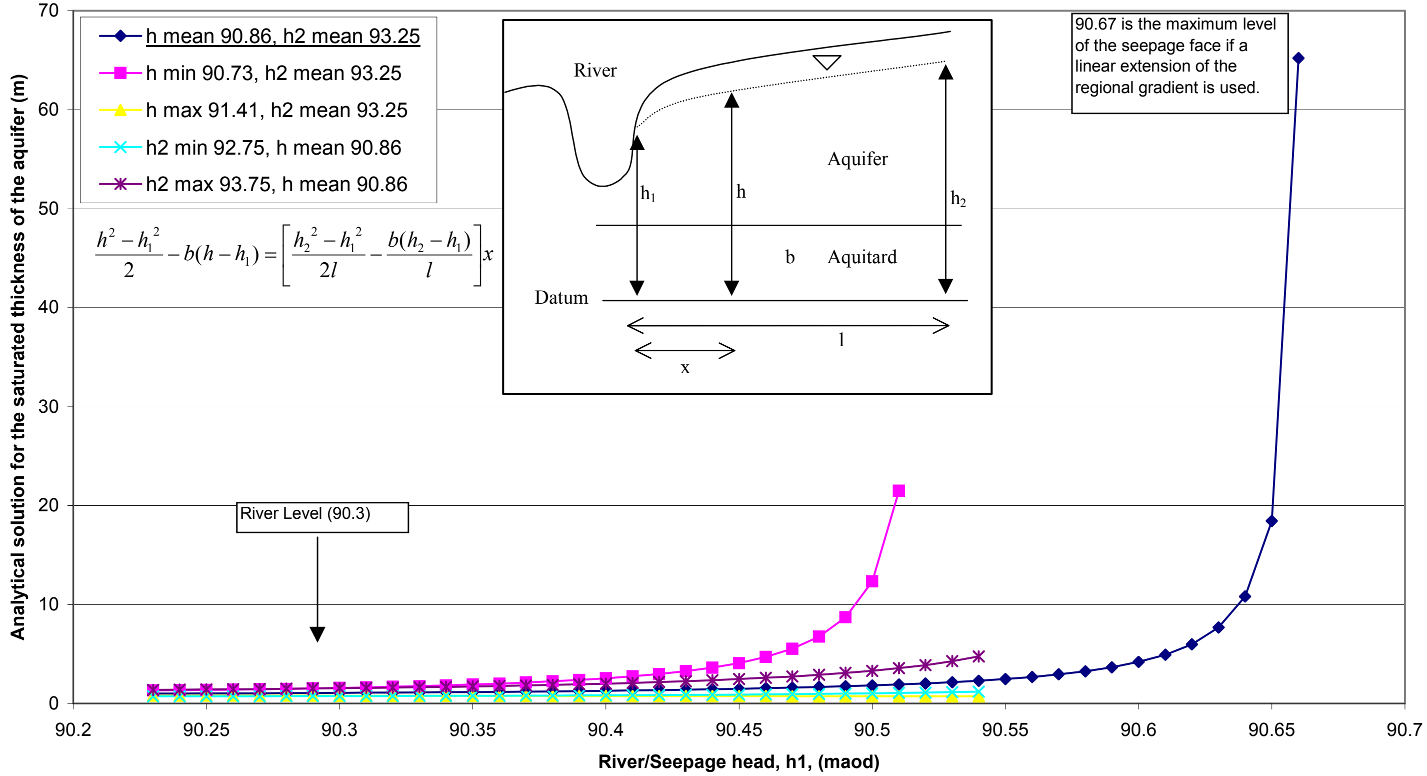


Figure 6.10 Sensitivity of the analytical solution for saturated thickness to variations in boundary conditions

boundary condition. Variations in other parameters, in conjunction with the expected dry weather flow (DWF) river stage, yielded saturated thickness values of < 3m which are still below the expected value. This implies the existence of a seepage face to increase the calculated saturated thickness to a realistic level. The maximum likely vertical expression of the seepage face is 37cm if a projection of the gradient between P10 and BH19 is used. If full connection between the different aquifer units is assumed then a saturated thickness of >10m is expected. This would require a minimum seepage face of 34 cm above the mean DWF river stage under average conditions of head in the aquifer. With a seepage face of 37cm the calculated saturated thickness is 65 m. The controls on the formation of the seepage face and groundwater flow across it are examined in more detail in the next section.

6.6 Investigation of the controls upon groundwater flow across the seepage face.

6.6.1 Conceptual model

Groundwater discharge to the river occurs through a seepage face as a result of capillarity and flow balancing when groundwater head gradients to the river are high and induce flow through the sides of the channel above the river level. Seepage may occur under steady-state conditions or following a river flood event after which ‘mounded’ groundwater and perhaps infiltrated surface water is discharged. The size of the seepage face is dependent on the conductivity values of each of the geological units through which flow to the river occurs. A high ratio of anisotropy (Kx/Kz) is likely to increase discharge through the seepage face.

The significance of contaminant flux across the seepage face is unknown and in many parts of the Tame it is difficult to measure directly the extent of the seepage face because of the presence of rock-filled support gabions. Modelling therefore provides a useful approach to understanding seepage zone processes.

6.6.2 Application of the FAT3D channel scale model

The FAT3D model was used to simulate the seepage face by incorporating seepage cells in the banks above the river cells. These cells can only discharge groundwater when the groundwater head is above the cell centre elevation. These cells were located in the riverbank to a height of 0.5 m above river level to cover the maximum likely extent of the seepage face according to field data. Under steady state conditions seepage discharge was found to occur only through the lowest seepage cell adjacent to the river cell. The model simulated the discharge occurring across the seepage boundary for varying geological conditions and boundary heads under both transient and steady-state conditions. Under transient conditions additional fixed head river cells were added to the banks to reflect the increase in river depth and the coverage of the seepage cells was maintained at 0.5 m above river level.

6.6.3 Results and discussion

Steady-state modelling indicated that for an isotropic uniform geology no discharge occurred across the seepage face when conductivity was > 2 md^{-1} . Discharge did occur when anisotropy (Kx 2 md^{-1}, Kz 0.2 md^{-1}) was introduced, amounting to <3% of total groundwater discharge to the river. The maximum specific discharge of 0.7 md^{-1} (9% of the total discharge) occurred through the seepage face when a layer of low permeability riverbed sediments was incorporated into the model. High levels of seepage discharge (6% of the total discharge) also occurred under conditions of increased gradient between the regional head boundaries and the river stage.

The specific discharge across the seepage face of the calibrated steady-state model was 0.07 md^{-1} (<1% of the total discharge). During both the steady-state and transient sensitivity analyses, discharge occurred only through the seepage cell immediately adjacent to the river boundary cells which represents a vertical extent of 10 cm. This is lower than the minimum value of 34 cm predicted by the analytical modelling, perhaps due to invalid assumptions of isotropy and longitudinal flow used in the analytical model. Under transient conditions the seepage face was absent during the rising limb of the river flood event and appeared on the falling limb, the position changing and dropping with the falling river level.

The results of the transient sensitivity analyses indicated that reduced discharge from the seepage face was sensitive to a number of factors. These included: increased and reduced Kx in the gravel, a reduction in the thickness or Kx of the sandstone, increased Kz in the river bed and a lowering of the regional fixed head boundary conditions. Maximum discharge from the seepage face was sensitive to a reduction to 0.01 md^{-1} in Kz for the river bed sediments showing a 364% increase in seepage discharge. Increased discharges by 217% and 168% result from elevated Kz values (1 md^{-1}) in the sandstone and reduced Kz values (0.1 md^{-1}) in the gravel.

The modelling indicates that seepage discharge is likely to vary along the length of the study reach. The vertical extent of the seepage face will be generally <10cm, with an upper limit of <40 cm. Seepage discharge will be at a maximum following a river flood event and is dependent on many factors, including the groundwater head gradient adjacent to the river and, most importantly, the permeability of the riverbed. Evidence of seepage may be clearly seen in sections of the Tame that are cement-lined beneath the M6 motorway. Seepage appears to form a small percentage of the total groundwater discharge to the river (in general <1%) and may be insignificant in supplying contaminant flux to the river. One exception may be the discharge of pure phase LNAPL contaminants that collect at the surface of the water table and would be most likely to discharge through the seepage face. A more detailed investigation of the distribution of groundwater flows across the river channel including the seepage face is described in the next section.

6.7 Investigation of the spatial variations in groundwater discharge to the river

6.7.1 Conceptual model

Groundwater discharges to the surface water system through the sides and base of the channel across the wetted perimeter and the seepage face. Under steady-state and transient conditions groundwater discharge is spatially variable dependent on the head gradient to the river and variations in geology. Discharge variations occur both across the channel and along the length of the reach under both steady-state and transient conditions.

Data from water quality samples and field estimates of groundwater flow are related to a single location within the channel/reach. In order to use these data in the context of the regional contaminant flux to the river it is important to understand the range and the spatial distribution of discharge likely to occur at the sample location. High concentrations of contaminant in a sample may be locally important within the hyporheic zone but a low flux rate will mean a limited impact on the surface water in terms of total mass loading. Modelling was used to investigate the range and distribution of groundwater flow across the river channel and along the reach.

6.7.2 Application of the FAT3D channel scale model

The FAT3D model was used to represent the river in vertical cross section. A horizontal row of 39 fixed head cells represented the riverbed and a vertical column of three cells at either end of the row represented the wetted river bank (cell dimensions: width 30 cm, height 10cm). The steady-state model provided data on the discharge that occurred across each boundary cell that was used to calculate specific and cumulative discharge across the channel. The effect of transient conditions on river/groundwater interactions is discussed in Section 6.10.

6.7.3 Application of the MODFLOW flood plain model

The calibrated regional MODFLOW model that was used allowed the individual interrogation of each of the 116 cells containing a river boundary condition to provide a mass balance summary. Flows across each cell face and discharge/recharge from the river were obtained and used to calculate discharge to the river per meter length of channel. The model assumed a uniform geology and therefore variations in flow along the reach were solely a result of the regional head distribution and the river geometry.

6.7.4 Results and discussion

The modelling displays a high degree of spatial variability in groundwater discharge along the reach (-0.5 to +6.5 m^3d^{-1} per metre length of reach) and across the channel (0.1 to 1.5 md^{-1} specific discharge).

The FAT3D model indicates that discharge is primarily concentrated through the sides of the channel (Figure 6.11) with discharge through the central channel bed and the seepage face being of secondary importance. For the calibrated steady-state model, 25% of the total discharge occurs within 0.3 m of each riverbank. This is a general finding, as under the same head conditions using a single isotropic media, 22% of the total discharge occurs within 0.3 m of each riverbank. This implies that the results of water-quality samples taken from the sides of the channel are more significant when calculating total contaminant flux to the river than samples from the central section of the riverbed. Also, this may explain the lower-than-expected total groundwater discharge estimates that were based on field measurements taken from piezometers within the riverbed that do not sample the high flow zones around the river banks.

The high flow through the river bank cells reflects the higher conductivity associated with horizontal flow through the bank side as opposed to the lower conductivity associated with vertical flow through the river bed. Under real conditions the vertical conductivity of the riverbed may be further restricted by the clogging of pore spaces with fine sediments when downward flow may otherwise have occurred, such as in the case of some tidal rivers where a ‘one way valve’effect is observed (Rivett, pers com). The vertical conductivity of the river bed will also be dependent to some extent on the flow conditions within the river. Low velocity conditions will facilitate the deposition of fines and a reduction of conductivity, high velocity flood conditions may lead to the stripping of fine bed sediments and an increase in conductivity.

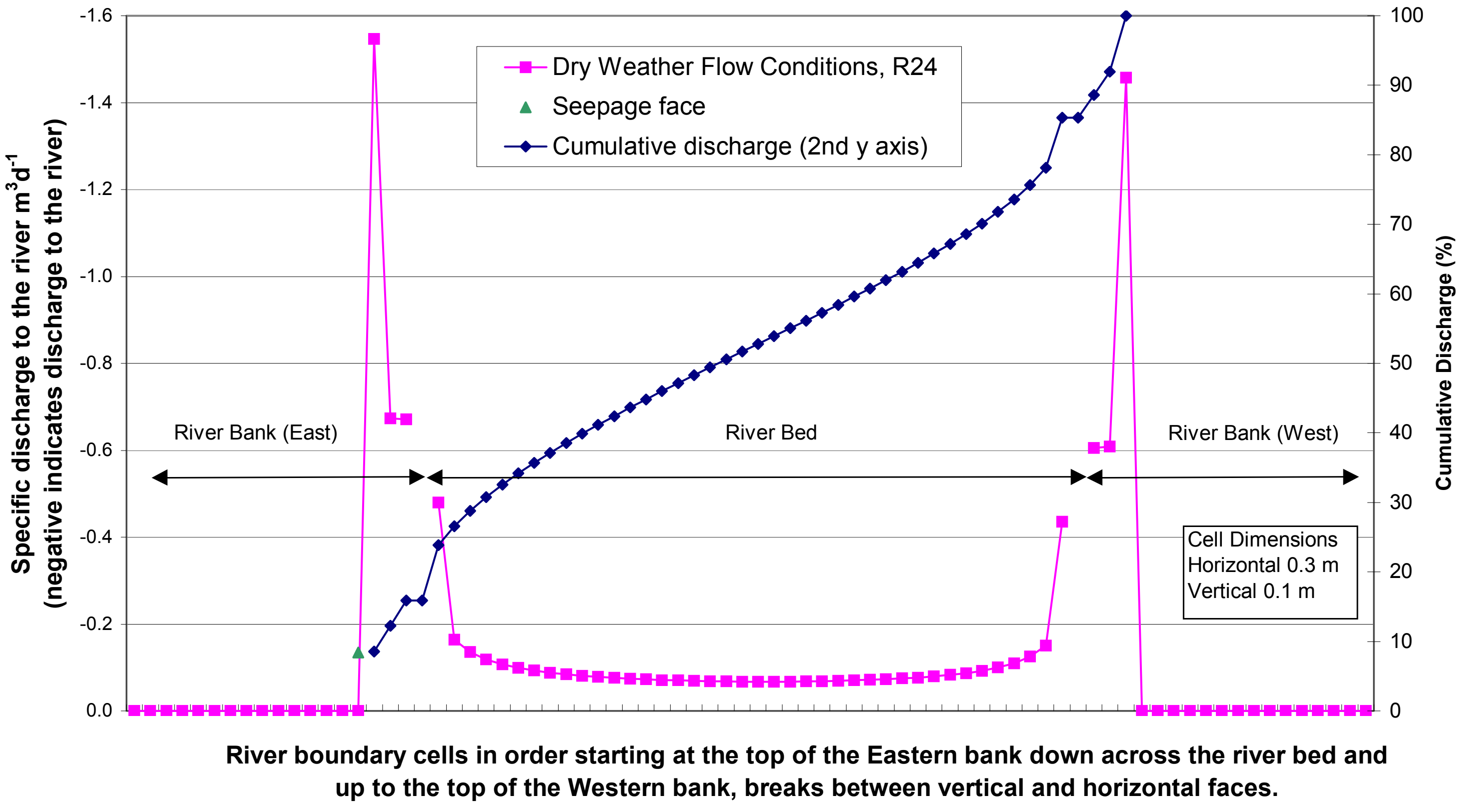


Figure 6.11 The distribution of steady state groundwater discharge across the channel.

The average specific discharge through the centre of the riverbed is $0.1 md^{-1}$. If an effective porosity of 0.2 is assumed an average linear velocity of $0.5\ md^{-1}$ results. For a riverbed thickness of 2 m there will be a contact time of four days between the groundwater and riverbed sediments. This is an important factor when considering the degree of natural attenuation that may occur during passage through the riverbed and the hyporheic zone. Residence times will be less for the higher velocity shallow groundwater entering through the riversides and consequently the potential for natural attenuation is reduced.

An investigation was undertaken on the effects of heterogeneity within the streambed resulting from obstructions such as boulders. The distribution of flow across the riverbed (Figure 6.12) shows increased variability and flow through the river banks and seepage faces is increased.

The data from the 2D cross sectional model suggests continuous discharge of groundwater across the flat-bottomed channel and no penetration of river water into the sediment. However, in reality, 3D flow is occurring, and irregularities in the river bed and the pressure distribution caused by river flow will give rise to infiltration of river water through sections of the upper river bed sediments (intergravel flow). This infiltration is likely to occur even in groundwater gaining reaches and will determine the extent of the hyporheic zone in which groundwater and surface water mixing occurs. A 3D approach is required to model this zone accurately. Considerable variation in groundwater discharge is observed along the MODFLOW model reach (Figure 6.13) ranging from -0.5 to +6.5 m^3d^{-1} per metre length of channel. This variability is a result of the geometry of the river meander in relation to the elevated heads of the valley side and the lower heads on the flood plain. Greater variation

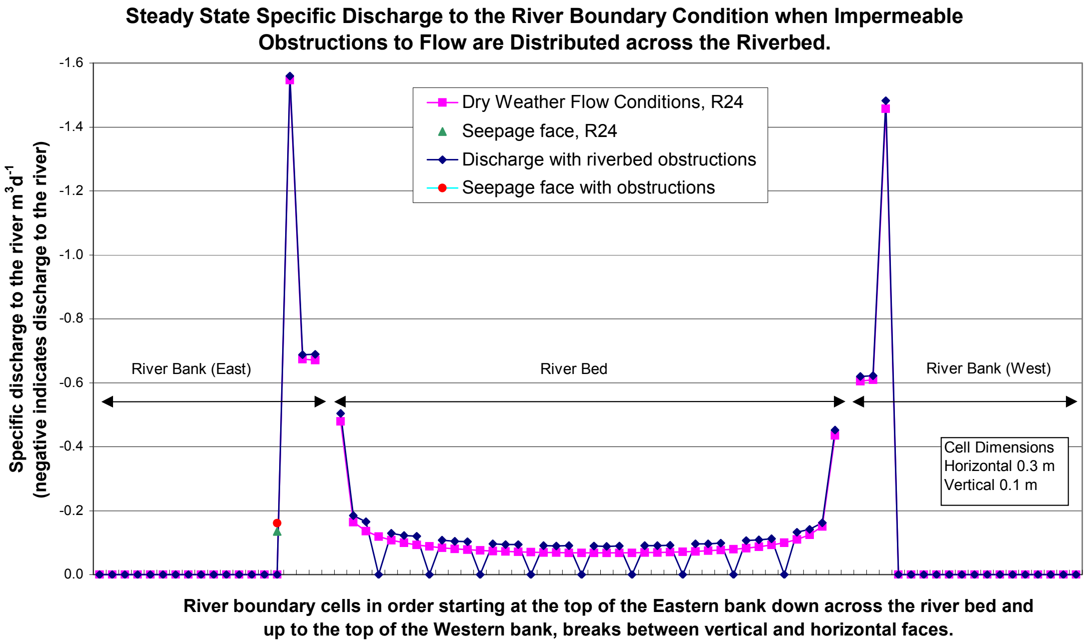


Figure 6.12 The effect of obstructions on groundwater flow across the riverbed

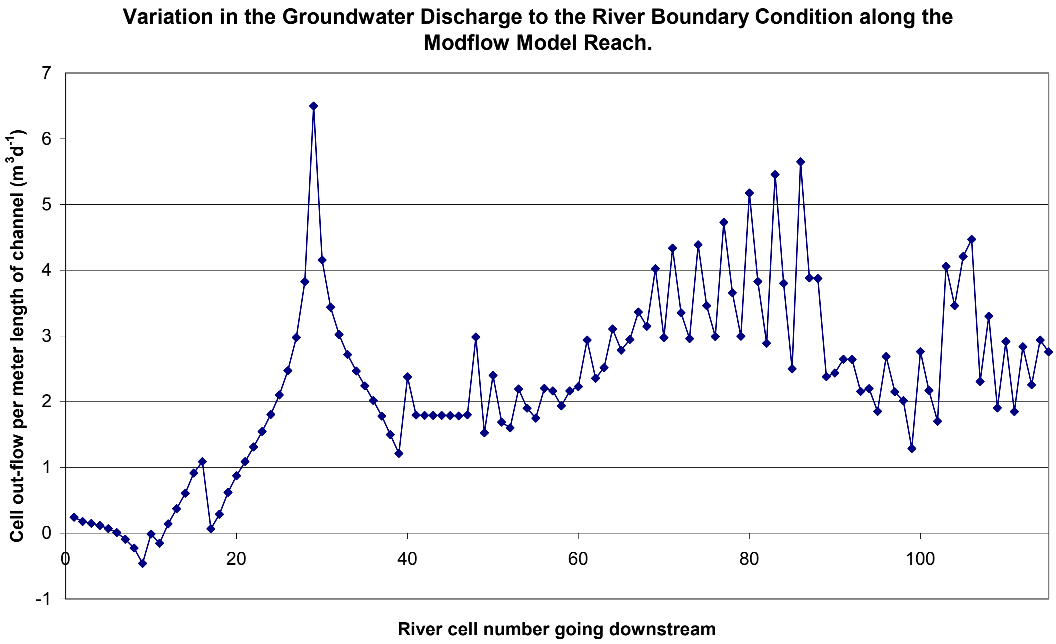


Figure 6.13 Variation in groundwater discharge to the river along the MODFLOW model reach

would result from the introduction into the model of a more realistic heterogeneous geology. The river is primarily gaining although groundwater discharge is reduced and surface water infiltration to the aquifer does occur at the upstream end of the reach.

Large variations in groundwater discharge to the river between adjacent cells (e.g. cells 71 to 85) are a product of the coarse grid dimensions and the associated problems of representing a diagonal feature across the grid. The 'diagonal step' method employed will overestimate the number of river cells per unit length. For example, in the case of the diagonal section starting from cell 50, a total of 23 cells are required to cover a straight line distance equivalent to 16 cells parallel to the grid. This represents a 44% increase in available channel length that may lead to an overestimation of net discharge to the river but an underestimation of maximum flow per unit length of channel. Nevertheless the model is still considered suitable to represent the heterogeneity of the system dependent on the geometry of the river and the regional head distribution, and a correction was not applied.

As discussed in previous sections, many different flow paths of different origin and contaminant loading culminate and discharge through a particular section of channel. The composition of the total flow to each river cell in terms of source (Figure 6.14) is highly variable along the MODFLOW reach. Different portions of the reach are dominated by groundwater (contaminant) discharge from a particular direction (bank) rather than an equal inflow from each bank.

The steady-state modelling and field data indicate that the river is currently primarily gaining groundwater but historical studies have indicated that substantial ingress of surface water to

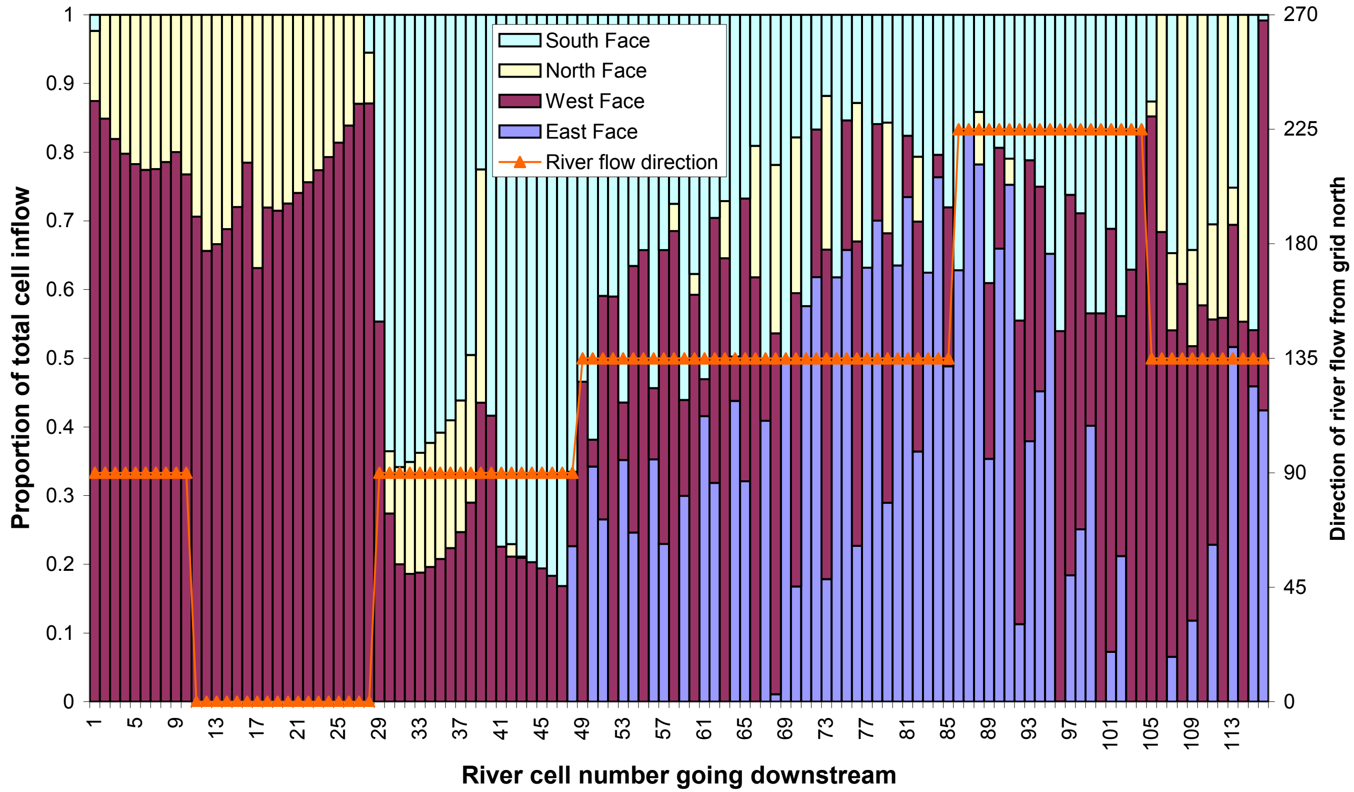


Figure 6.14 Proportions of inflow derived across each vertical face to the MODFLOW river cells

the aquifer has occurred during periods of peak abstraction. The MODFLOW model was used to investigate the effect of these abstraction wells on flow paths and surface water loss.

6.8 Investigation of the effect of abstraction wells on groundwater flow to the river.

6.8.1 Conceptual model

There is a high concentration of industry in the model area and a total of ten currently licensed industrial abstraction wells. High rates of abstraction may alter the direction and reduce the amount of contaminant flow to the river from that observed under natural conditions. Abstraction wells located close to the river may induce the infiltration of poor quality surface water into the aquifer.

6.8.2 Application of the MODFLOW flood plain model to assess abstraction.

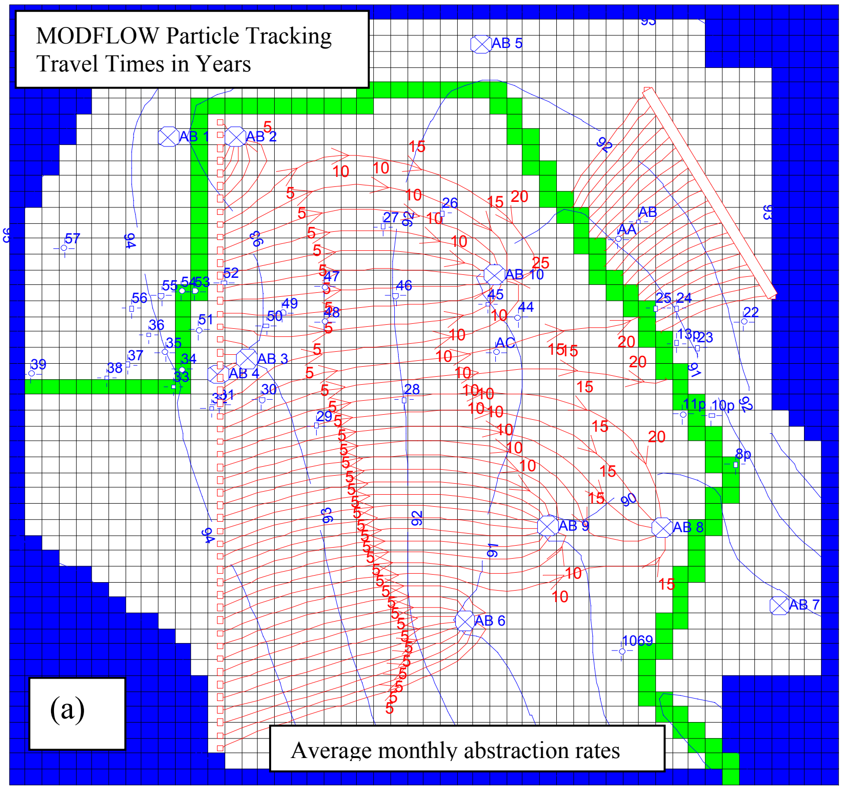
The industrial abstraction wells were modelled under steady state conditions using the calibrated MODFLOW flood plain model. Details of well locations and abstractions (Table 6.4) were obtained from Environment Agency records. (1/4/89 to 1/3/99). The model was run with particle tracking under conditions of both average and maximum abstraction rates to see the effect on contaminant flow paths and discharge to the river.

Well #	Depth (m)	Maximum Licensed Daily Discharge (m^3d^{-1})	Average Daily Discharge from Monthly Returns (m^3d^{-1})
1	75.13	1263	414
2	150	1562	746
3	61.3	450	189
4	84.3	466	234
5	45.72	32731	425
6,8,9,10	60-99	3155	575
7	91.44	546	175

Table 6.4 Borehole Abstraction Rates for the Witton area in the Tame Valley

6.8.3 Results and discussion

Particle tracking (Figure 6.15a) shows flow paths to be more complex and less linear, with a greater range in travel times than flow prior to abstraction (Section 6.4). The majority of flow across the flood plain is captured by the abstraction wells before discharge to the river may occur. This would significantly reduce contaminant loading to the river that may occur from the large number of industries located on the flood plain. At maximum abstraction rates (Figure 6.15b), particles to the northeast of the river pass beneath the river as underflow to Abstraction Well 10.



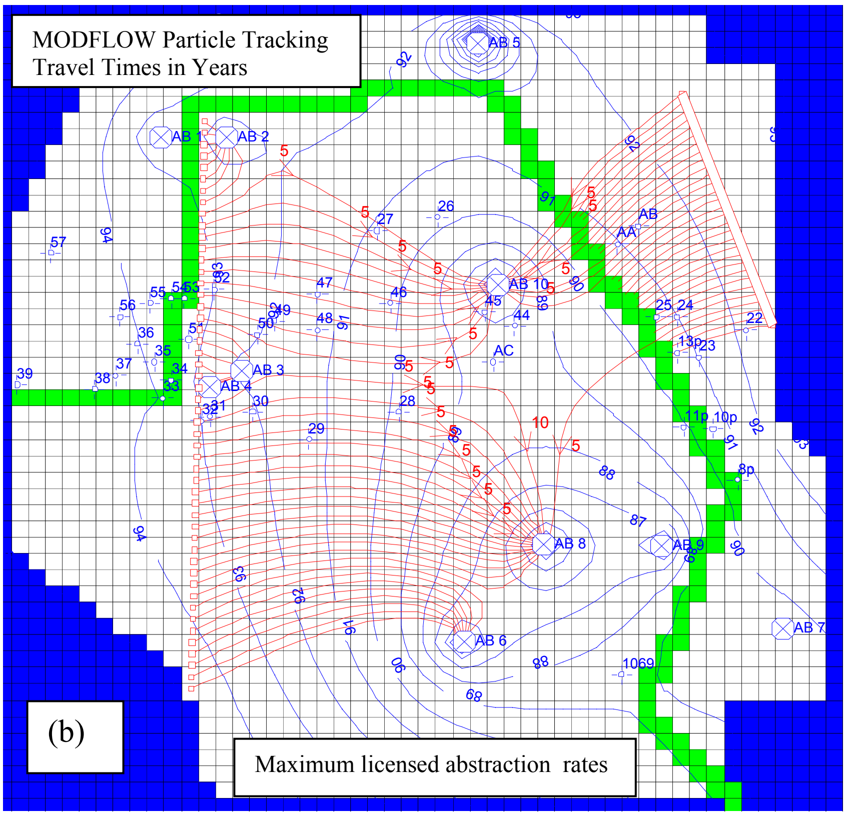


Figure 6.15 The effect of abstraction on groundwater flow across the flood plain at (a) average rates (b) maximum rates

Average abstraction rates decrease groundwater discharge to the river by 30% (Table 6.5) and maximum rates cause a 78% reduction.

Level of abstraction	Abstraction rate (m^3d^{-1})	River gain (m^3d^{-1})	River loss (m^3d^{-1})
None	0	9683	34
Average	4483	6824	381
Maximum	26907	2149	8320

Table 6.5 River gains and losses due to borehole abstractions

Under conditions of average abstraction, losses from the river are small and the river receives a net gain in flow over the reach. At maximum abstraction rates the river loses over a large proportion of its length, with a net loss in flow. If a dry weather river flow rate of 184 Mld^{-1} is used from flow gauging, then total river flow would be reduced by 5%.

After abstraction and use in industrial processes the water may be discharged to sewer but in many cases it is discharged directly to the river. This needs to be taken into account when calculating the water balance and when considering whether contaminant flux to the river is a result of the industrial process or the original contaminated groundwater. For example, in the case of abstracted water containing chlorinated solvents (Rivett, 1989) discharged to the Tame.

Historically high levels of abstraction throughout the Birmingham Aquifer caused a large drop in regional water levels resulting in reduced groundwater discharge and river flows. The impact of changes to the regional head gradient and river levels is examined in more detail in the next section.

6.9 Investigation of the effect of changes in river level and regional head.

6.9.1 Conceptual model

For a variety of reasons, changes may occur in river level or the regional groundwater head gradient towards the river. Scenarios that might produce these alterations include climate change resulting in increased or decreased recharge and river stage. A variation in groundwater abstraction rates may reduce or increase the regional groundwater levels. Sewage treatment works and industrial discharges to the river, which currently provide > 40% of flow, may alter. River levels may rise as a result of reduced flow velocities owing to channel engineering or weed growth. Both the MODFLOW and FAT3D models were used to investigate the effect on contaminant flux to the river that may result from these changes in the future.

6.9.2 Application of the Models

A sensitivity analysis of groundwater discharge to the river was undertaken. Changes in river level and the regional head were simulated by raising or lowering the fixed head conditions by a uniform amount under steady-state conditions. The changes occur in isolation whereas in reality there would be greater interdependency, but some interesting points can be made.

6.9.3 Results and discussion

The MODFLOW (Table 6.6) and the FAT3D (Table 6.7) models indicate increases of 20-38% in groundwater discharge to the river for a regional water level rise of 0.5 m. If contaminant concentrations in the groundwater remain the same, i.e. are not diluted, then the river is likely to receive a greater total contaminant mass loading derived from groundwater if increased recharge due to climate change occurs. The recovery of groundwater levels in Birmingham (Section 5.2) by several metres since the 1950s has undoubtedly led to a significant increase in groundwater discharge to the river. However, water table rebound in the Tame Valley appears to have stabilised and further increases in groundwater discharge are unlikely to result from this if current rates of abstraction are maintained.

River gain (m^3d^{-1})	Variation in river gain (%)	River loss (m^3d^{-1})	Variation in boundary heads (m)	Variation in river level (m)
9683	0	34	-	-
13364	+38	0	+0.5	-
6774	-30	798	-0.5	-
6837	-29	805	-	+0.5
13235	+37	0	-	-0.5

Table 6.6 The impact of MODFLOW boundary head variations on discharge to the river

Discharge to the river m^3d^{-1}	1.82	1.89	2.18	2.53	1.87	1.68	1.43	1.11	0.63
% variation from Run 24	0	+4	+20	+39 %	+3%	- 8%	-21%	-39%	-65%
Variation in Boundary Heads (m)	-	+0.1	+0.5	+1.0	-	-	-	-	-
Variation in river level (m)	-	-	-	-	-0.2	+0.2	+0.5	+0.8	+1.5

Table 6.7 The impact of FAT3D boundary head variations on discharge to the river

A decrease in discharge of 29% occurs when the regional water table is lowered by 0.5m . The river remains a net gaining reach, the FAT3D cross sectional model incurs no losses from the river and MODFLOW river losses are not significant (<12% of the groundwater discharge).

The raising of river levels by 0.5 m produces a similar effect on groundwater discharge to an equivalent 0.5 m drop in the regional head. To increase the river stage by even 0.2 m would require a significant input from a source such as a major sewage treatment works. For example, a simplistic calculation based on channel width 10m, flow velocity 0.5 ms^{-1} and head increase 0.2 m, would require an input of $1m^3s^{-1}$ (86 MLd^{-1}) or a large reduction in flow velocity. The most likely boundary condition to change is the regional water level.

Variations of 0.1 – 0.2 m in the river and regional boundary heads in the FAT3D model simulate the frequent (days-weeks) temporal variability likely to occur which induce a less than +/- 8 % variation in discharge to the river. High river stages were also imposed to investigate the impact of a major flood event. An increase in stage of 1.5 m reduced groundwater discharge by 65% but the river still received a net gain. This is assuming the system reacts fast enough for steady-state conditions to occur. More detailed investigations of groundwater/surface water interactions under transient river flood conditions are undertaken in the next section.

6.10 Investigation of groundwater-surface water interactions during a flood event.

6.10.1 Conceptual model

The heavily engineered nature of the Tame catchment leads to a rapid rise and fall in river levels during precipitation events. River stage flood rises of 50 cm are common and stage rises of several metres have been observed. Daily river stage fluctuations of <10cm associated with sewage treatment works discharges have been recorded. Groundwater discharge to the river is reduced on the upward limb of the flood hydrograph and increases on the downward limb with the release of bank storage. Field data indicate good connectivity between the river and the aquifer and show the occurrence of groundwater mounding with an increased river stage. The aquifer response is dependent on storage and transmissivity, and modelling is used to gain information on these parameters by calibration against field data. Modelling is also used to investigate the levels and distribution of groundwater discharge and surface water ingress that occur across the river channel during a flood event.

6.10.2 Application of the analytical model of transient groundwater-surface water interactions.

The 1-D transient analytical model was used to simulate changes in groundwater level in the riverbank (<15 m from the river) using field data on river stage fluctuations. Borehole hydrographs were modelled and compared with field data for piezometers P10, P11, P13 and P8 at Tameside Drive (River bed piezometer profiles 8, 9,10) and BH 39 Regina Drive (Riverbed piezometer profile 2). The objective of the model was to obtain a best-fit value for the S/T parameter for each piezometer based on continuously monitored (5 minute interval) head and stage data collected over a period of five to ten days.

In general, river stage data were obtained from a stilling well located on the riverbank at the minimum distance from the piezometer. The river hydrograph data used in modelling P13 was taken from a stilling well 200m downstream. A uniform adjustment to stage was calculated, based upon a simultaneous head measurement at the stilling well and the point on the bank closest to P13. This method appears to work for P13 where the river channel is almost identical to the stilling well site. However, there was difficulty in obtaining a model fit for piezometer P8 that lay 200 m downstream from the stilling well where the channel width has increased, reducing the amplitude of stage fluctuations.

The model solution assumes an initial head gradient between the river and aquifer of zero. In reality a regional head gradient does exist but is assumed to remain constant and is therefore ignored. The model calculates changes in head relative to the initial steady state level in the piezometer taken from the field data. The FAT3D model was used to verify that the amplitude and frequency of the piezometer hydrograph are independent of changes in the regional head gradient. This indicates that it is valid to compare the analytical model results of S/T for boreholes from different locations with different regional head gradient conditions. Full details of the modelling carried out are given in Appendix 22.

The 1-D analytical model provided a simple and rapid estimate of average S/T over a 5-10 day period incorporating several flood events. More detailed 2D modelling incorporating complex geology was undertaken for a single event using FAT3D.

6.10.3 Application of the FAT3D cross sectional model

The FAT3D cross sectional model was used to simulate groundwater-surface water interactions across the channel during a single flood event lasting a period of 1.2 days with the peak in stage, 0.5 m above dry weather flow conditions. The calibrated steady-state model was subjected to varying river boundary head conditions based upon field data (6-7th March 2001). A general specific storativity of 0.00001 was assigned to all geological units. In addition, the gravel unit was assigned a value of specific yield, as it was known to contain the free surface of the water table. The transient model was calibrated on a trial and error basis by varying values of specific yield in the gravel unit and comparing modelled heads with field data for piezometers P10 and P11. The model was run under steady-state conditions to obtain the initial head conditions prior to each transient run.

A sensitivity analysis was undertaken to gauge the effect of variation in hydraulic parameters on groundwater discharge to the river and hydrograph amplitude in piezometer P10. An investigation was made into the temporal variation in groundwater discharge to the river boundary cells during the flood event.

6.10.4 Results and discussion

A range of S/T values of 0.0003 to 0.0061 dm^{-2} was obtained from the analytical model (Table 6.8) from five different locations. Good repeatability of results is seen for P10 and P11, both showing a drop in S/T for the second modelled period. It can be seen by comparison with the river hydrograph (Figures 16a and b) that the P11 hydrograph displays greater peak lag times and damping of its amplitude than P10 which has a lower value of S/T . There is an order of magnitude of difference between the S/T value for P10 and those of P11, P13, P8 which are all located along the same 400m reach of the river. This difference may be explained by the occurrence of a 30 cm thick clay-rich unit noted in the geological log for P10, which may locally confine the sand and gravel units generally found in the borehole logs. The clay unit does not, however, prevent good connectivity with the river.

The FAT3D model returned values of specific yield in the gravel of 0.005 and 0.03 for calibrations against P10 and P11 respectively (Figure 6.17). This heterogeneity may represent local conditions only, but it does make the model assumption of uniformity in the gravel

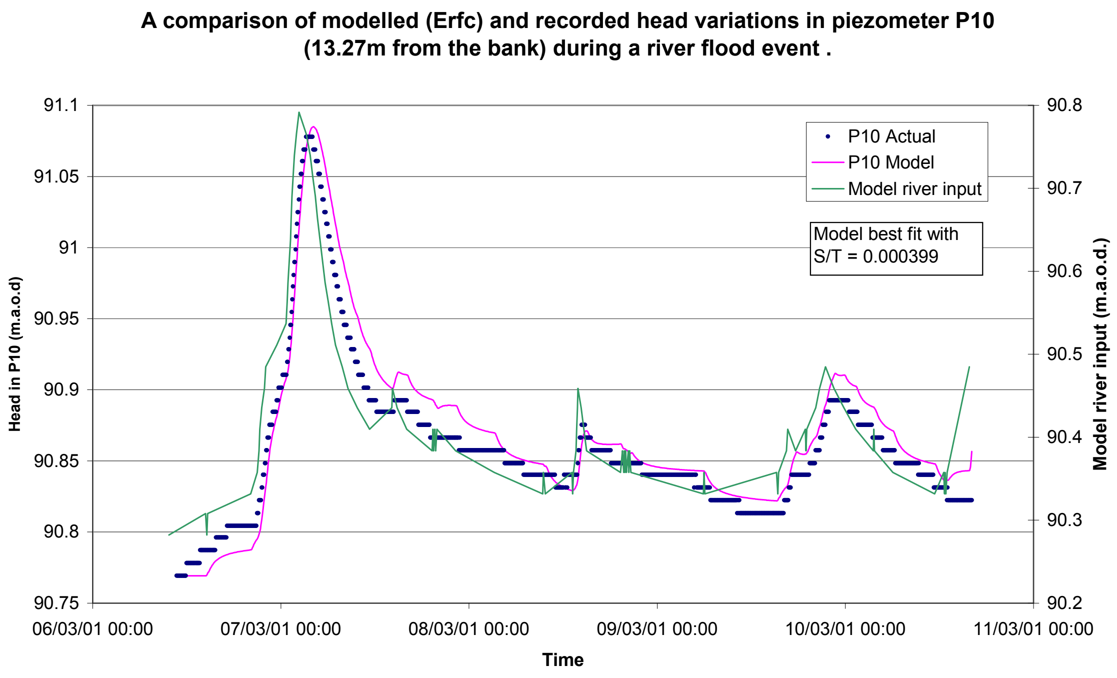


Figure 6.16a Results of the transient analytical modelling of the hydrograph of Piezometer P10 (6/3/01)

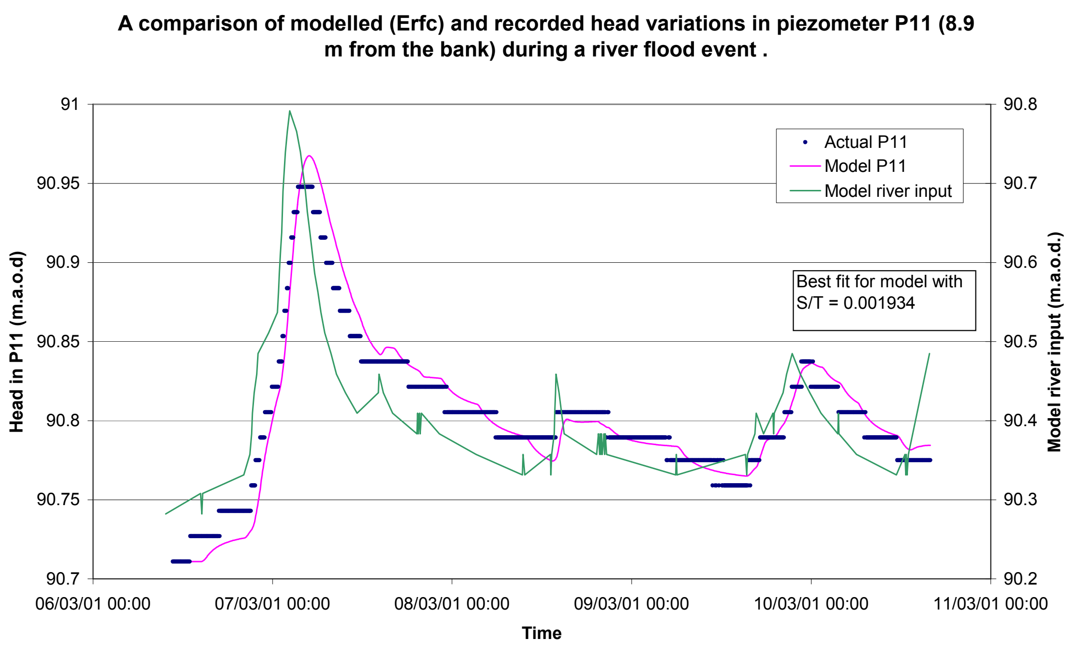


Figure 6.16b Results of the transient analytical modelling of the hydrograph of Piezometer P11 (6/3/01)

across the river doubtful. Subsequent modelling and sensitivity analyses were based on the calibration against P10 with specific yield = 0.005.

Piezometer	Distance from river (m)	Period Modelled	Best fit value of S/T
P10	13.27	6/3/01 – 10/3/01	0.000399
P10	13.27	11/5/01 – 22/5/01	0.000321
P11	8.9	6/3/01 – 10/3/01	0.001934
P11	8.9	11/5/01 – 22/5/01	0.001069
P13	9	9/6/01 – 18/6/01	0.001457
P8	13.25	27/6/01 – 3/7/01	0.004 – poor fit
BH39	13.96	3/7/01 – 12/7/01	0.006092

Table 6.8 Values of S/T derived for the transient analytical model.

The FAT3D transient calibration was performed by varying specific yield (Sy) alone, as transmissivity had been set previously in the steady-state calibration. It was possible to obtain the same calibration hydrograph for higher values of Sy if Kx and Kz were also increased by the same factor. Any possible increase in Sy is limited by the expected groundwater discharge to the river. The values of Sy derived in the calibration are therefore thought to be representative of the actual field parameters.

Using the results of the analytical modelling to derive a meaningful value of storage is problematic. Straight use of values of horizontal transmissivity from the MODFLOW (450 m^2d^{-1}) and FAT3D (206 m^2d^{-1}) models yields storage values well in excess of those derived

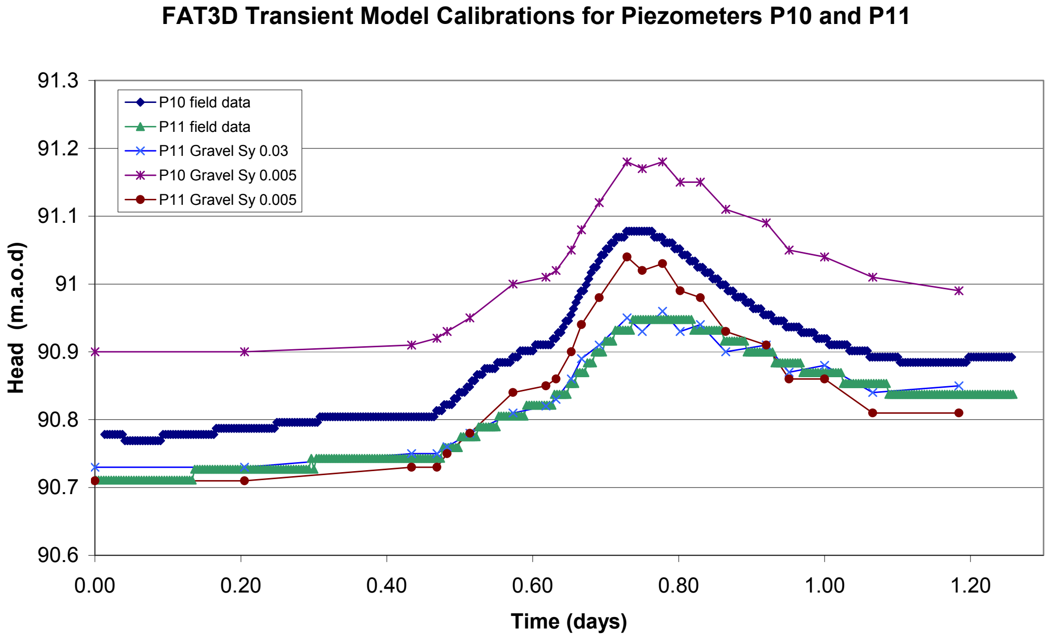


Figure 6.17 FAT3D Transient calibration against piezometers P10 and P11 hydrographs (6/3/01).

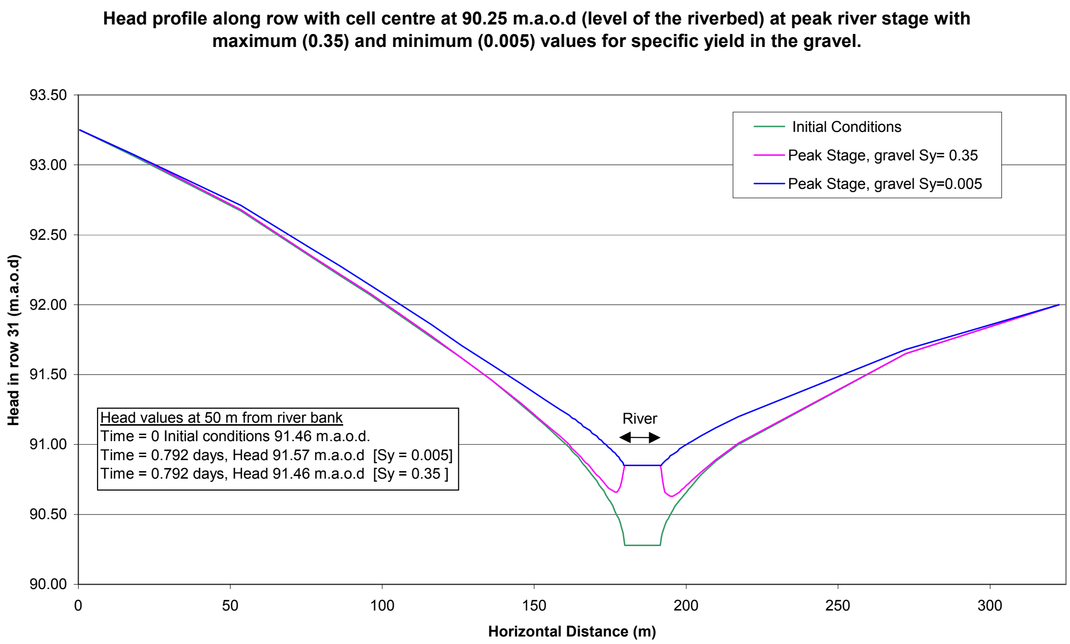


Figure 6.18 Water table response to the flood peak for different specific yields in the gravel.

by the FAT3D model. For P11, estimates of storage from the analytical solution would be 0.68 to 0.31 (T = 450 – 206 m^2d^{-1}) versus the FAT3D estimate of Sy=0.03. The difference results from an overestimation of T used to derive the analytical storage values. Pressure changes generated by the river may be assumed to propagate in a straight line from the bottom corner of the river channel to the centre of the piezometer screen. The centre of the screen for piezometer P10 is 13.27 m in horizontal distance and 2.5 m in vertical distance below the eastern corner of the riverbed (gradient 19%). The centre of the screen for piezometer P11 is 8.9 m horizontal distance and 2 m vertical distance below the western corner of the riverbed (gradient 22%). There is therefore a significant vertical component of T to consider which is already incorporated within the FAT3D model, which simulates convergent flow and anisotropy. An order of magnitude of difference exists between the average values of Kx (2.14 md^{-1}) and Kz (0.209 md^{-1}) calculated for the FAT3D model (Section 6.6). An order of magnitude reduction in the value of T used to estimate storage values for the analytical model would give greater consistency with the FAT3D storage values. While the analytical model provided estimates of S/T it required estimates of T containing a vertical component to calculate meaningful storage values. The model would be better suited to obtain storage values under confined conditions, and at greater distances from the river where the vertical component of T would be less significant.

The results of the FAT3D sensitivity analyses indicated that the specific yield of the gravel was the dominant control of groundwater-surface water interaction under transient conditions. An increase in the Sy of the gravel from 0.005 to 0.35 caused an 83% decrease in the net groundwater discharge for the event and maximum losses from the river of 0.73 m^3d^{-1} over a short (40 minute) time period. A reduction in the sandstone thickness to 10 m and an increase in Kz for the riverbed sediments to 10 md^{-1} produced minor river losses of 0.003 m^3d^{-1} and 0.0009 m^3d^{-1} respectively over a short (40 minute) time period. Losses from the river are prevented by the rapid response of the aquifer to rises in river stage that occur under conditions of low Sy.

High specific yield in the gravel greatly reduces the amplitude of the groundwater fluctuations and decreases the distance from the river at which the influence of flood events may be observed. An increase in water levels of 11 cm (Figure 6.18) can be seen in response to the flood peak at a distance of 50 m from the river when gravel Sy = 0.005, but any response is limited to <25m when gravel Sy = 0.35 . The importance of this tidal zone in terms of contaminant flux to the river and natural attenuation processes has not been determined.

River water begins to infiltrate the aquifer when the river stage increases to 25 cm above DWF under conditions of high Sy (0.35) in the gravel (Figure 6.19). Under conditions of low Sy (0.005) in the gravel, no river-water losses occur and groundwater is effluent to the river throughout the transient event. When river losses do occur they are through the river banks and surface water does not penetrate the bed of the river (Figure 6.20). In reality, a hyporheic zone of groundwater mixing will occur in the riverbed due to variations in the river head along the channel, and this zone will extend further into the river banks during flood events. Discharge through the river bank is seen to return (Figure 6.19) to a new lower level of steady-state discharge under conditions of higher river stage within 10 hours of the flood peak passing. The return from bank storage of infiltrated surface water and shallow groundwater occurs on the downward limb of the flood hydrograph through the river banks across a vertical face of up to 90 cm, including a transient seepage face.

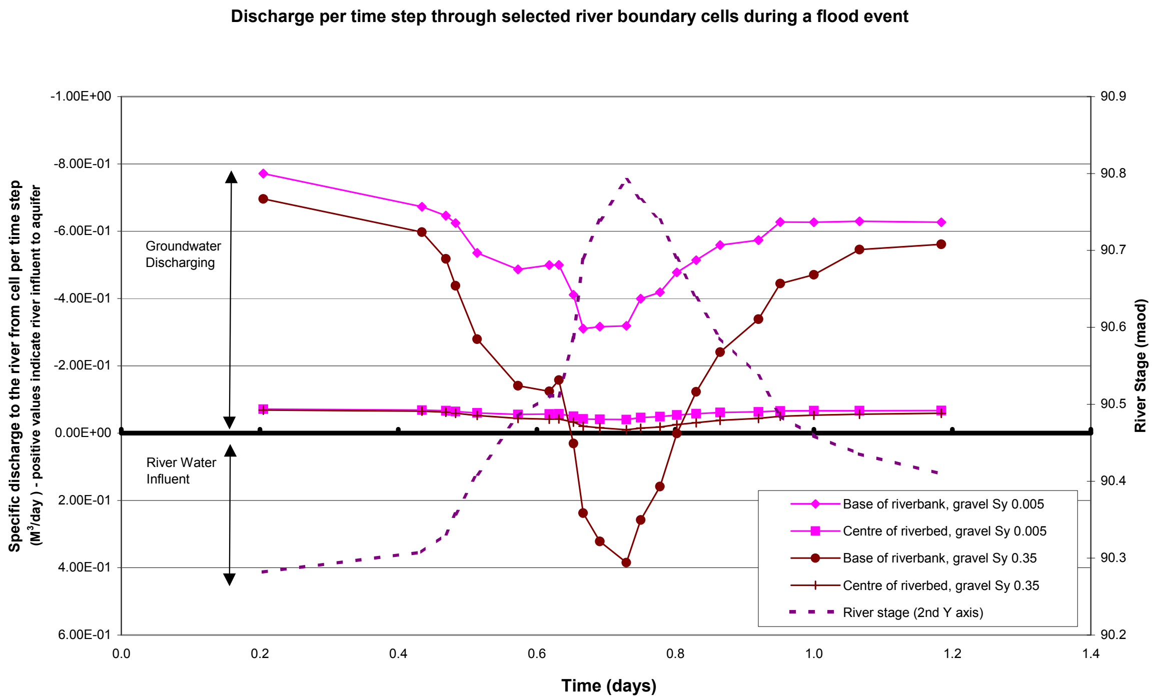


Figure 6.19 Groundwater discharge through the riverbed and riverbank during the course of a flood event

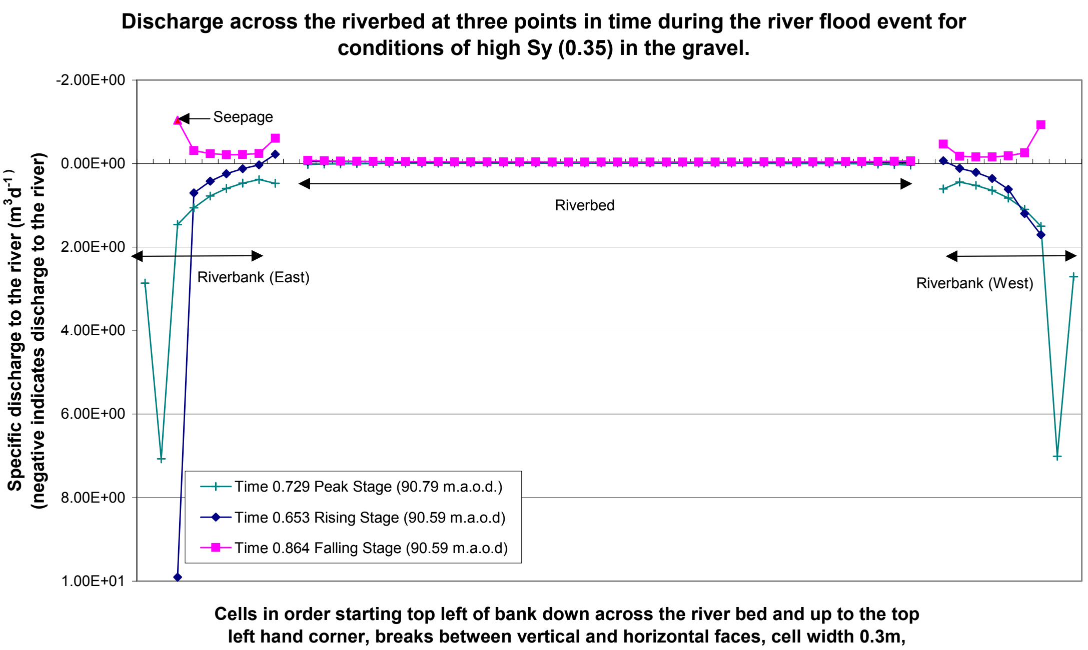


Figure 6.20 The spatial distribution of groundwater discharge across the riverbed during a flood event

The value of the FAT3D specific yield parameter derived for the piezometer P11 (0.03) is lower than expected for an unconfined sand and gravel aquifer which usually have specific yields > 0.15. There is no evidence of any horizons within the geological log that would reduce the storage. An underestimation of Kx and Kz in the model may have caused an underestimation in Sy but would not alone be sufficient to explain the low value. A likely explanation is the effect of the capillary zone in reducing available storage adjacent to the free water surface. Other evidence of the effect of unsaturated zone processes is found in the variation in aquifer response to similar river flood events (Section 5.7) which must reflect a variation in S/T. Different values of S/T for P10 and P11 were calculated by the analytical model over two different time periods (Table 6.8) reflecting changes in the moisture content profile which effects storage and flow processes above the water table. The variation in S/T for P10 implies that the clay layer does not fully confine the aquifer as in that case S/T would remain constant.

The transient analytical model indicates that a single value of S/T will not serve to fit the modelled and observed data between events or for the entire duration of a single event (Figure 16a and b). The deviation of the model is most notable in the tails of the falling limbs of the flood peaks where model values do not decline as rapidly as actual values indicating an over estimation of S/T. The model also over-estimates lag times for peak arrivals which is an over estimation of S/T. Maximum peak heights exceed actual values, indicating an underestimation of S/T . The initial rise of the flood peak occurs more slowly in the model than in the actual data, indicating an overestimation of S/T . Transmissivity is unlikely to vary greatly, owing to an increase in saturated thickness which is small compared with the total. Small variations in S/T may result from changes in geology across the fluctuation zone but these would not vary in time only with groundwater level. The most likely explanation is that the S/T parameter varies over time as a function of unsaturated flow and differences in the available storage as the water table fluctuates. An additional factor may be that increased water levels and mass loading on the river bed contributes to a reduction in the elastic storage. The unsaturated zone is seen to be a major control on the groundwater and surface water interaction adjacent to the river and further modeling was undertaken to investigate the processes associated with water table fluctuation using the code UNSAT.

6.11 Investigate the control of unsaturated flow processes and the capillary fringe on the fluctuation of the water table in response to changes in river stage.

6.11.1 Conceptual model

Variations in river stage cause a response in groundwater levels as observed in shallow piezometers on the riverbank. The aquifer is unconfined and the observed fluctuations represent variations in the level of the water table, the position at which pore pressure is equal to atmospheric pressure. Above this level water is held under tension in pore spaces and as grain coatings by the effect of soil-water attraction. The negative pressure created by the soil-water (capillary) attraction draws water upward from the water table to form the capillary fringe. This is a saturated zone with the upper limit defined as the point at which the gravity head of the supported water column above the water table is equal to the capillary suction. Above this point, the gravity head of the column of water will be greater than the suction and gravity drainage will result.

The height of the capillary fringe is a function of the diameter of the pore space and the pore throat. The narrower the pore diameter the greater the column of water supported. Thus clay will have a more extensive capillary fringe than gravel. Sediments display a degree of heterogeneity that leads to variations in capillary rise. The upper limit of the saturated zone represents the level of capillary rise for the widest pore spaces; capillary rise will occur above this level for smaller diameter pores. Moisture content generally decreases with height above the saturated zone, although it does depend to some extent on the recharge history and the geology.

An increase in pressure in the aquifer results in upward flow and a rise in the water table. Initial movement of the water table will occur subject to saturated conductivity flow rates through the capillary fringe. Conductivity then declines because of air entrapped within the pore spaces being infiltrated. Unsaturated flow will occur above, and in advance of, the water table as a result of the redistribution of moisture by capillary attraction subject to unsaturated conductivity. The rate of upward movement of the water table is determined by both saturated and unsaturated flow processes and the available storage to be filled. The high moisture content around the capillary fringe will lead to a lower available storage capacity and more rapid water table movement than sections higher in the profile with lower moisture content. This will lead to variability in the rate of change of the level of the water table that will not be solely dependent on the forcing head. This is observed as variations in the modelled S/T values obtained from monitoring piezometers.

A decrease in pressure in the aquifer on the falling limb of the river hydrograph results in downward flow and a drop in the water table. Flow is by gravity drainage controlled by unsaturated conductivity that is dependent on moisture content. The vertical distribution of moisture content is a major control on rates of water table movement and is dependent on the previous wetting and drying history. For example, a second river flood event may occur before completion of gravity drainage from a previous event, or surface recharge may occur. Differences in this moisture profile between events will lead to different water table responses. A modelling code (UNSAT) that incorporates unsaturated flow was used to examine the processes controlling the fluctuation of the water table in an unconfined aquifer.

6.11.2 Modelling of the unsaturated zone using the UNSAT code

The objective of the model was to investigate the control of unsaturated flow processes and the capillary fringe on the fluctuation of the water table in response to changes in river stage. The UNSAT code enabled the construction of a 1-D model to simulate the rise and fall of the water table within the unsaturated zone in response to a river flood event. This was used to investigate:

(i) The effect of different geology on the vertical distribution of moisture content.

(ii) The effect of the variation in the vertical distribution of moisture content on the observed storage during the course of a forcing head event.

(iii) The sensitivity of the observed storage to different wetting and drying histories.

(iv) The sensitivity of the observed storage to different rates of change in the forcing head.

The code calculates values of soil moisture content, soil pressure and unsaturated conductivity in each cell for different geology types based on the empirical Van Genuchten equations (Van Genuchten, 1980) and simulates both saturated and unsaturated flow under transient conditions. A variable pressure boundary condition may be applied at the base of the model to induce upward flow or downward gravity drainage simulating changes in the level of the water table. The model provides information on the flow across the boundary condition that may be used in conjunction with the change in applied head to calculate storage values.

6.11.2.1 Boundaries and grid layout

The 1-D model comprised a 2 m column of 100 cells with boundary conditions applied at the top and bottom. A no flow condition was assigned as the upper boundary. The lower boundary at the base of the column was assigned a pressure head that was held either constant or varied sinusoidally over time.

6.11.2.2 Procedure adopted, starting conditions and modelling periods

The model was used to investigate the occurrence of the capillary fringe under stable conditions for sand and clay - the two extremes of the materials likely to occur adjacent to the river. In the initial experiment, conditions for the column were set as dry with residual moisture content. The model was run for a period of 10 days with the lower boundary condition set as zero pressure to wet the column. A second experiment was carried out in which a saturated column was allowed to drain for 10 days with a pressure of zero applied at the base.

To simulate the aquifer response to river level fluctuations, a pressure head varying in the form of a sine function was applied at the base. The amplitude and wavelength were varied for different model runs over the ranges seen in field observations (wavelength 0.5-2 days, amplitude 5-100cm) and the function varied over time steps of 1/50 days. Initial moisture conditions for the model were set, based upon the moisture profile after 10 days wetting. Cumulative inflow to the model was sensitive to the initial conditions, and the model was run through at least 10 cycles of the sin function until stable conditions of flow across the boundary were reached.

6.11.2.3 Calculation of apparent and global specific yield

In order to investigate the variations in storage that occur during a forcing head cycle it was necessary to calculate a value representative of storage (apparent storage) for each time step in the forcing cycle. The apparent storage was calculated based upon the definition of specific yield, that is the volume of water released over a given time period per unit horizontal area per unit head change. The flow across the lower boundary of the model was divided by the change in the forcing head on a per time step basis. A global specific yield was calculated equal to the cumulative inflow divided by the total head change per quarter of the sin forcing head cycle.

6.11.2.4 Model Parameters

The geology was varied between runs and the column was set to represent a single geological type on each occasion. Van Genuchten parameters for unsaturated flow were obtained (Anderson, 1990, Digges la Touche, 1998) for the soil types clay, sand and loam to simulate the range of conditions that are likely to occur in the Tame Valley. Details of the results and parameters used are given in Appendix 25.

6.11.3 Results and discussion

Both the sand and the clay columns display capillary rise when wetted from dry at the base of the column (Figure 6.21), the clay column coming close to saturation after the ten-day wetting period. The drained clay profile is saturated over the entire 2 m section and would perhaps be saturated to the surface of the riverbank which lies only 3 m above the water table. The clay has very limited available storage ranging from 0 to 0.02

The sand displays a capillary fringe and a reduction in storage over the bottom 30 cm (wetted profile) to 60 cm (drained profile), the most significant reduction to 0.066 (16% of possible) occurring over the bottom 10 cm.

The drained moisture profiles are not the same as the wetted profiles indicating that one or both experiments had not reached equilibrium after ten days. Under field conditions it is unlikely that a period of longer than ten days of steady-state conditions will occur and therefore the near water table zone is never likely to reach a state of equilibrium. This means that conditions will be different for each flood event depending on the recent wetting and drying history, and the response of the river bank water table will be slightly different for each event. The most likely moisture content profiles for the study reach will lie between the two extremes of sand and clay, with the low modelled values of storage indicating a considerable clay component within the gravel. A considerable (30 cm to 2 m) saturated /partially saturated zone exists above the water table and little is known about contaminant transport and attenuation within this zone. Horizontal flow rates may be assumed to be low due to soil-water attraction and primary contaminant movement would occur through diffusion. Oxygen contents in this zone are high with large water/atmosphere contact areas and conditions ideal for aerobic bacteria which may aid in contaminant attenuation processes.

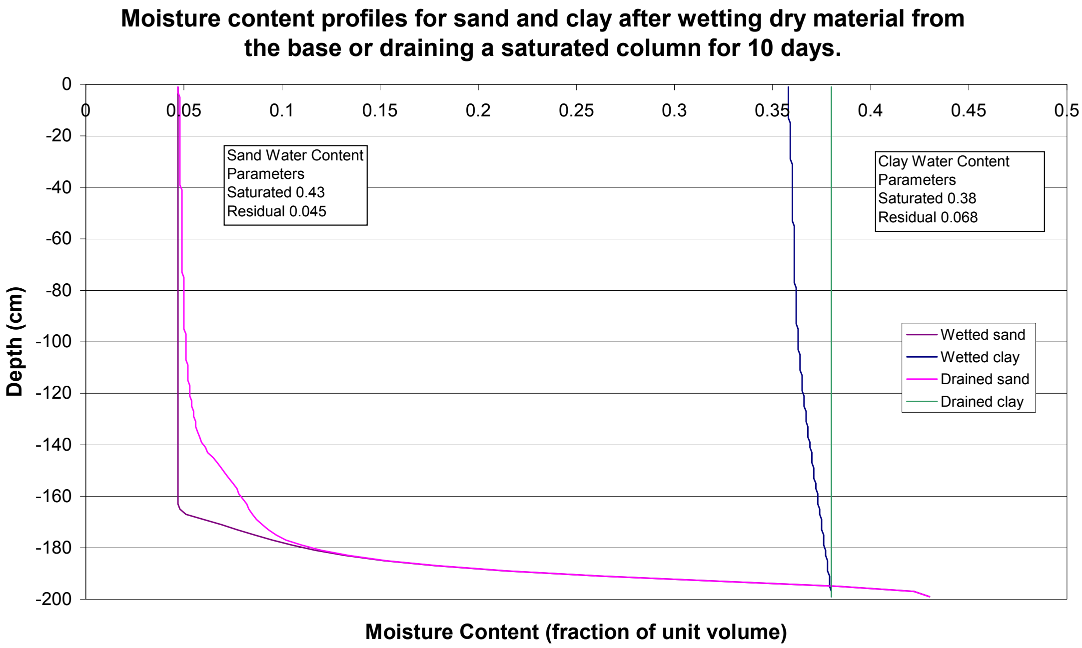


Figure 6.21 Moisture content profiles for sand and clay

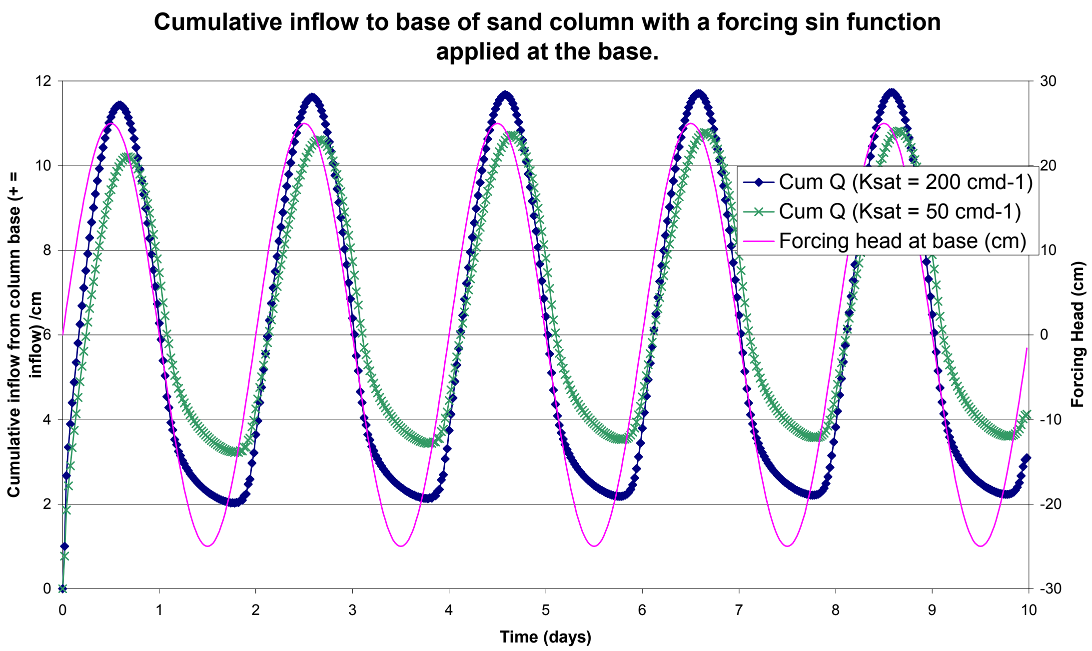


Figure 6.22 Cumulative inflow to the base of a sand column with a sin variation in the applied head.

For sand under conditions of a sin variation in the applied pressure head, cumulative inflow across the base of the column is seen (Figure 6.22) to vary non-uniformly and displays a phase difference with the forcing head function. A greater inflow occurs during the first cycle as initial conditions require wetting before more stable conditions are reached in the following cycles, with cumulative inflow remaining above zero. The minimum level of the cumulative inflow is seen to rise between each cycle indicating a continued slow draw of water by capillary action. The cumulative inflow is less when a lower saturated conductivity (Ksat) is assigned to the sand - a 75% reduction of Ksat from 200 cmd^{-1} to 50 cmd^{-1} produces a 30% reduction in the range of the cycle of cumulative flow. This indicates that both saturated and unsaturated flow processes are occurring. The cumulative inflow curve displays asymmetry in the downward half of each cycle where unsaturated flow and gravity drainage dominate.

The moisture content profile of the sand and clay columns vary during the course of a cycle (Figure 6.23 a and b). The formation of drainage fronts increases the variability of the available storage during the cycle. The full 25 cm rise in the water table occurs in both the sand and the clay moisture profiles.

The range of variation (Figure 6.24) in apparent specific yield (aSy) was greatest in the sand and least in the clay. The variations in Sy are non-linear and are heavily influenced by the phase difference between the forcing head function and the cumulative flow. The falling limb of the head function produces a generally lower Sy than the rising limb. The lowest values of aSy in any one cycle occur during the initial head increase and the first phase of drainage. The aSy values calculated for the first cycle are considerably higher than those of the following

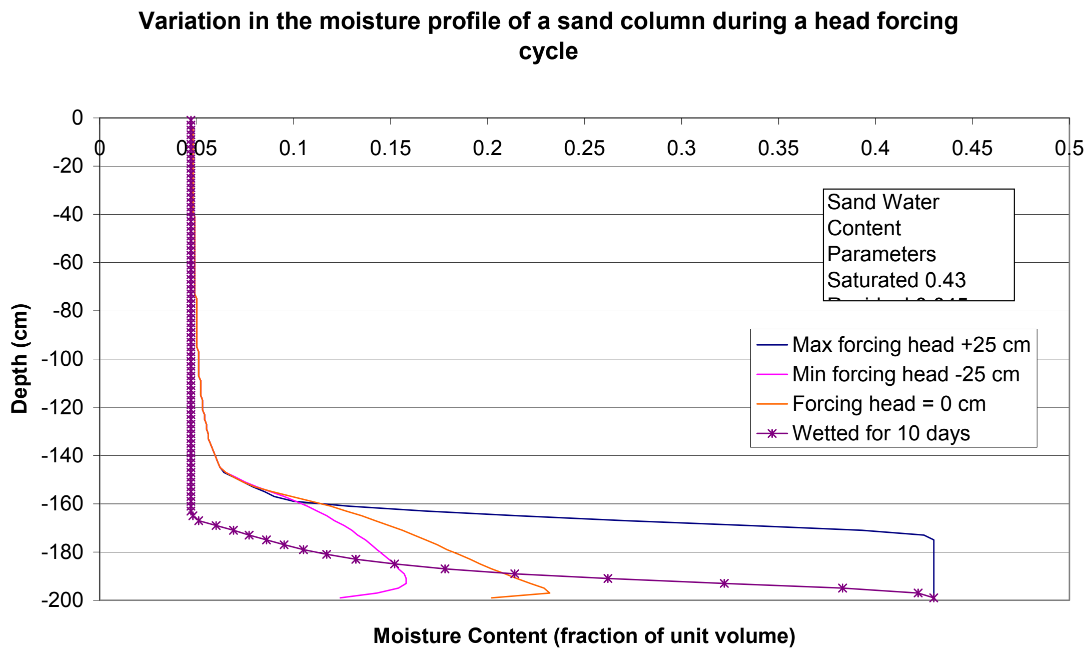


Figure 6.23a Variation in the moisture profile of a sand column during a head forcing cycle.

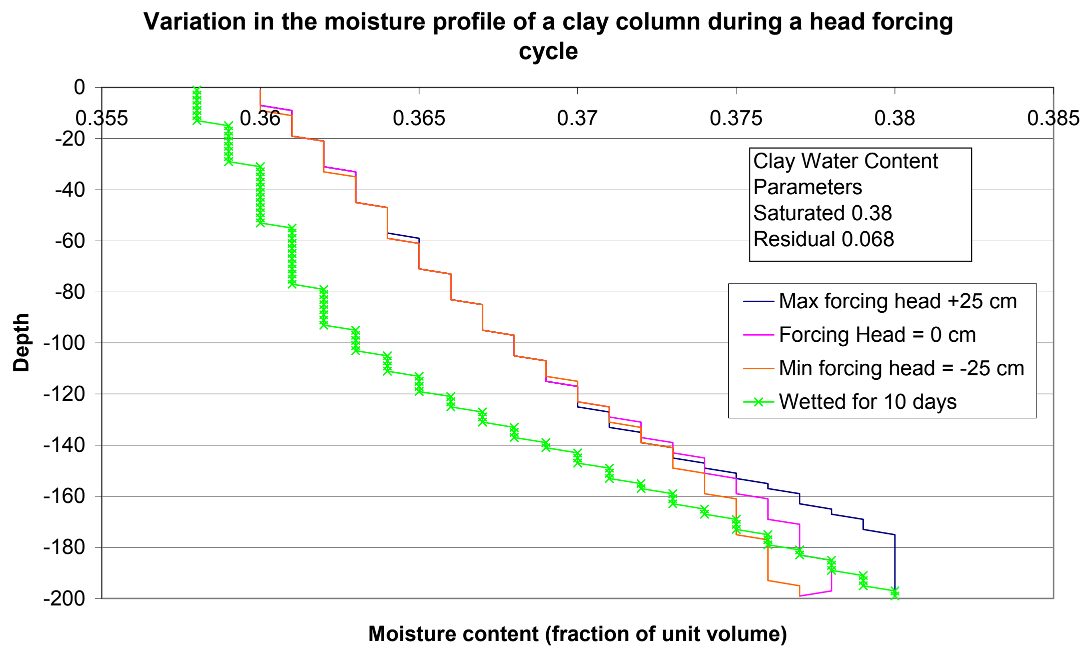


Figure 6.23b Variation in the moisture profile of a clay column during a head forcing cycle.

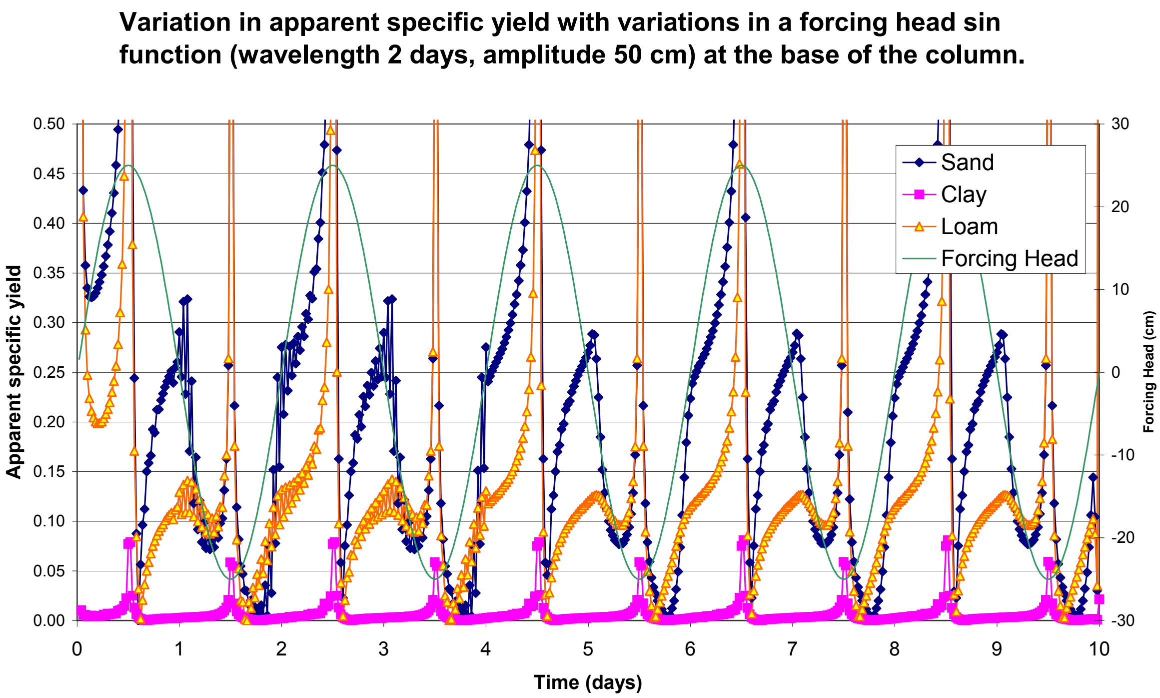


Figure 6.24 Variation in the UNSAT model apparent specific yield with changes in the forcing head

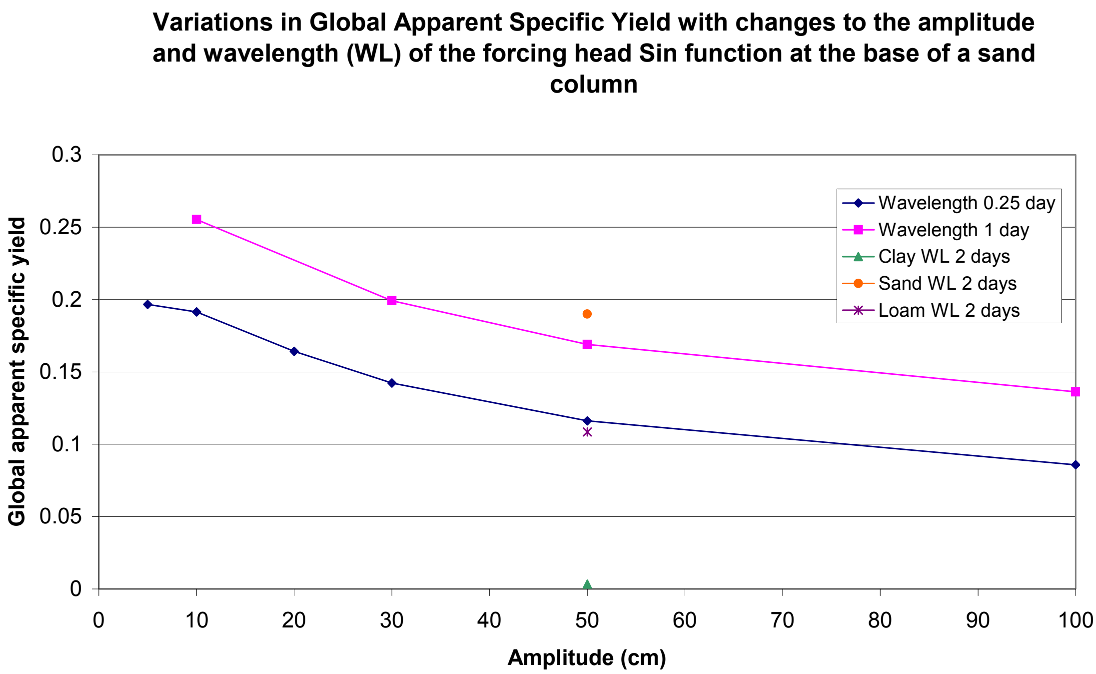


Figure 6.25 Variations in global specific yield with changes in the amplitude and wavelength of the forcing head

cycles owing to the initial ‘wetting up’, and highlight the importance of the initial conditions and the drying history.

The apparent specific yield can not be directly related to the actual specific yield as aSy values of > 0.5 are obviously incorrect. Global estimates of specific yield calculated using the total change in head and cumulative inflow over the quarter cycle were sand 0.19 (max possible 0.43), loam 0.11(max possible 0.25), clay 0.005 (max possible 0.38). The apparent specific yield and global specific yield are both dependent on the forcing head function applied. An increase in wavelength or a decrease in amplitude both result in increased estimates of aSy and global specific yield (Figure 6.25). The apparent specific yield is time dependent and governed by flow rates in the unsaturated zone. These flow rates determine the time taken to saturate/desaturate the column by a height equivalent to the step change in the forcing head. This time may be greater than the time interval over which the change in forcing head occurs which would lead to a decrease in the apparent storage. Downward flow rates of unsaturated gravity drainage determine the rate of change of storage prior to the next re-saturation. Lower frequency head cycles increase drainage time and lower amplitudes decrease the vertical extent to be drained between maximum and minimum forcing head.

The storage capacity and conductivity of the unsaturated zone is controlled by the vertical distribution of the moisture content and this is dependent on the geology and the wetting and drying history. The response of the water table to a forcing head event is dependent on the unsaturated storage capacity and conductivity, which will vary according to the relative position of the water table within the vertical moisture content profile. The water table response will vary between events and will be dependent on the rate of change in the river forcing head, and on the previous wetting and drying history. In general higher storage values might be expected after a long dry period with a stable water table. Low amplitude and large wavelength river stage events will be less easy to identify in river bank observation piezometers because of the larger damping effect associated with higher apparent storage.

The parameter S/T derived from the transient modelling work indicated lower than expected values of storage which varied within and between events. The UNSAT model demonstrates that these observations are likely to be a result of the complex behaviour of the unsaturated zone. The UNSAT estimate for the global specific yield of a sand was 0.19 (Figure 6.25) for a forcing event (amplitude 50 cm, wavelength two days) similar to that used in the FAT3D model. This estimate of specific yield is 49% lower than the total available storage (saturated – residual moisture content) of 0.385 for the sand. The FAT3D model estimated specific yields of 0.03 to 0.005. Even including the effect of the capillary zone and unsaturated flow processes, these are significantly lower than would be expected for a pure sand and imply a considerable clay content in the riverbank material. However, the clay content appears not to be so high as to reduce the hydraulic conductivity below the 5 md^{-1} derived in the model calibration. Therefore additional reasons are also likely to contribute to the low estimates of Sy in the FAT3D model such as an underestimation of transmissivity and incorrect distribution of the geological units. The response of the riverbank observation piezometers may also incorporate components of both confined and unconfined storage. The system is complex and not fully understood and requires further investigation involving a 3D model that incorporates the geology, the river and unsaturated zone processes.

6.12 Conclusions

Particle tracking indicates that in some cases groundwater contaminant plumes within a meander may flow away from the river beneath the flood plain before discharging to the river further downstream. This may considerably increase contaminant travel times and the chance for natural attenuation to occur. Flow beneath the flood plane is primarily horizontal to within approximately 100 m of the river when the vertical component becomes increasingly important, as flows with a variety of origins converge and discharge to the river. The younger groundwater discharges through the sides of the channel and the older groundwater through the centre. The modelling indicated the existence of a flow divide between shallow groundwater that discharges directly to the river and deeper groundwater that may flow over a considerable distance before discharging to the river further downstream.

Groundwater flows to the river occur through a complex geology comprising made ground, alluvial gravel, weathered and unweathered sandstone and the riverbed sediments. In agreement with the field data, the modelling indicated that the riverbed sediments did not significantly restrict groundwater discharge to the river. Borehole evidence (Figure 5.25a) indicates that on the edges of the flood plain the gravel receives groundwater flow from the underlying sandstone. The FAT3D model indicates that flow velocities through the alluvial gravels (<0.4 md^{-1}) are an order of magnitude higher than through the underlying sandstone. Estimated transmissivities from the modelling, based on field measurements of baseflow accretion, range from 200 to 400 m^{2}d^{-1}. These relatively high transmissivities indicate that the thin (5 m) gravel unit is highly permeable and/or groundwater flow occurs through a considerable thickness of the less permeable bedrock sandstone which may have mudstone layers that restrict but not prevent upward flow.

The formation of the seepage face was found to be most likely when horizontal was significantly higher than vertical hydraulic conductivity. The modelling predicted seepage amounting to <1% of the total groundwater discharge to the study reach across a seepage face of between 10 to 40 cm vertical extent. Results of the modelling indicated the development of a transient seepage face following the river flood waters as they subside.

The modelling predicts large variability in groundwater discharge along the reach (-0.5 to +6.5 m^3d^{-1} per metre length of reach) and across the channel (0.1 to 1.5 md^{-1} specific discharge), with a large proportion (~25%) of the total discharge occurring within 0.3 m of each bank. This variability is controlled by the geometry of the river, the groundwater head distribution and the geology. The specific discharge of groundwater through the centre of the channel is less than through the sides and the longer residence times will allow more time for natural attenuation to occur. The MODFLOW model of the flood plane and meander indicates that groundwater is primarily effluent to the river, with only minor recharge of surface water to the aquifer, perhaps associated with channel engineering. The MODFLOW model predicted that average levels of abstraction decrease groundwater discharge to the river by 30%. The maximum licensed abstraction rates caused a 78% reduction in groundwater discharge and resulted in a net loss (5%) of surface water across the reach. The MODFLOW and the FAT3D models indicate increases of 20-38% in groundwater discharge to the river for a regional water level rise of 0.5 m.

A range of S/T values of 0.0003 to 0.0061 was obtained after calibration of the transient analytical model against five observation piezometers. The range in S/T reflects the considerable geological heterogeneity that exists over short distances of <30 m. The results of the FAT3D sensitivity analyses indicated that the specific yield of the gravel was the dominant control of groundwater/surface water interaction under transient conditions. A low specific yield caused a rapid response in groundwater levels to rises in river stage and prevented surface water loss to the riverbanks. For the 50cm flood event modelled, no surface water losses were observed when specific yield was less than 0.35. Groundwater discharge was reduced on the upward limb of the river hydrograph and increased on the downward limb, with the formation of a transient seepage face. The transient analytical model indicates that a single value of S/T will not serve to fit the modelled and observed data, either between events or for the entire duration of a single event. In addition, the estimates of specific yield in the gravel from the FAT3D model (Sy = 0.03 for P11) were lower than expected. The influence of the unsaturated zone is the most likely explanation for the variability and low storage estimates.

The UNSAT model of the unsaturated zone predicted that the capillary zone extends through the entire 2 m column of clay, after wetting at the base for ten days, with low available storage of between 0 to 0.02. The capillary fringe for sand extended over 30 cm (wetted profile) to 60 cm (drained profile) after a ten-day period, and storage was reduced to 0.066, 16% of the total possible over the basal 10 cm. The response to wetting of the riverbank material adjacent to the piezometers is likely to be between that of sand and clay. The model predicted that under field conditions it is unlikely that the near water table zone on the river bank will reach a state of equilibrium and that the soil moisture profile will be different for each flood event.

The UNSAT model predicted that during the course of a simulated flood event, cumulative inflow across the base (i.e the initial water table) of a column of sand will vary non-uniformly. The cumulative inflow curve displays asymmetry on the downward limb where unsaturated flow and gravity drainage dominate. The lowest estimates of specific yield occur during the initial head increase and the first phase of drainage. For a simulated flood event (amplitude 50 cm, wavelength two days) the UNSAT model estimate for the global specific yield of a sand was 0.19 which is 49% lower than the total available storage. The response of the water table to a forcing event is dependent on the unsaturated storage capacity and conductivity which will vary according to the relative position of the water table within the vertical moisture content profile. The water table response will vary between events depending on both the rate of change in the river forcing head, and the previous wetting and drying history.

A good conceptual model of groundwater flow in the study area was developed during the modelling exercise, enabling predictions to be made about the likely distribution of groundwater flow. This was necessary in order to interpret the results of the water-quality sampling which are discussed in the following chapters.

CHAPTER 7. GROUNDWATER - SURFACE WATER QUALITY INTERACTIONS.

Results are presented from the water quality sampling of surface waters, shallow piezometers (<20m), deep abstraction wells (20-100 m) and riverbed piezometers (<2m) taken over a two-year period. The surface water quality data covers the 24 km study reach with the main focus on the 7.4 km of river overlying the Birmingham unconfined aquifer. Groundwater data were collected to determine the quality contribution of waters discharging from the aquifer and overlying drift deposits to the river system. Results of sampling undertaken in the large ($9km^2$) undeveloped area of Sutton Park are presented as an indication of the natural quality of waters derived from the Triassic Sandstone.

The results of the summer 2001 groundwater sampling program are summarised (inorganic - Tables 7.1, 7.2 and 7.3, organic-Tables 7.5 and 7.6 in Section 7.7, and Appendix 20) and discussed with respect to the variations observed across the groundwater and surface water systems. Data from the spring 2001, autumn 2001, summer 2000 and autumn 2000 sampling programs are included to illustrate any temporal changes observed in groundwater and surface water concentrations. Mean values are quoted including none detected results assigned the value of the detection limit.

Water Quality Data From The Tame Valley 2001 - Cations																		
	Ca	K	Mg	Na	Sr	Mn	Si	Fe	Ba	Al	Pb	Hg	Cd	Cr	As	Cu	Zn	Ni
Units	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l
Det Limit	< 0.02	<0.02	<0.02	<0.02	<0.02	< 0.02	<0.02	< 0.03	< 0.02	< 0.3	<0.4	<0.01	<0.1	<0.5	<1	<0.5	<10	<5
Abstraction Wells																		
Mean	79.02	4.69	23.06	23.82	0.36	0.08	6.54	0.31	0.12	< 0.3	0.7	0.06	0.1	128.9	4	5.9	46	10
Median	83.25	4.21	21.41	21.62	0.13	0.21	6.58	0.16	0.08		0.6	0.07	0.2	1.7	4	3.8	14	42
max	151.40	14.70	49.88	55.56	2.93	0.39	9.78	2.65	0.63	0.0	3.5	0.17	0.6	1600.0	31	19.7	220	74
min	15.32	1.68	3.83	4.21	0.02	0.02	2.84	0.03	0.02	0.0	0.4	0.02	0.1	0.7	1	1.1	5	10
N > Det	14	14	14	14	14	5	14	6	14	0	5	11	4	13	8	14	14	2
N < Det	0	0	0	0	0	9	0	8	0	14	9	3	10	1	6	0	0	12
N	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14
Shallow Groundwater																		
Mean	136.60	11.63	17.95	70.95	0.34	1.49	7.34	0.18	0.09	0.7	16.5	0.03	4.4	2.7	2	13.0	54	18
Median	115.90	7.69	16.20	33.32	0.25	0.39	6.30	0.13	0.07	5.4	10.5	0.03	0.2	1.8	2	7.4	17	16
Maximum	462.10	44.38	47.89	513.10	1.42	11.49	27.85	0.64	0.37	10.2	145.0	0.23	80.6	15.3	8	69.1	451	143
Minimum	24.77	1.84	3.36	6.71	0.07	0.02	2.14	0.03	0.03	0.7	0.4	0.01	0.1	0.5	1	1.0	6	5
N > Det	20	20	20	20	20	18	20	18	20	2	15	9	12	16	8	19	19	12
N < Det	0	0	0	0	0	2	0	2	0	18	5	11	8	4	12	1	1	8
N	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20
Riverbed Piezometers																		
Mean	147.13	14.52	21.76	90.57	0.49	3.15	10.31	1.41	0.11	4.3	6.2	0.03	0.6	2.4	3	11.9	81	44
Median	135.25	15.61	19.98	61.92	0.45	2.33	7.50	0.24	0.09	29.9	2.0	0.02	0.6	1.8	3	5.6	37	27
Maximum	482.40	31.21	83.23	704.20	1.95	11.06	33.08	24.06	0.52	33.0	158.0	0.19	3.4	20.4	33	96.9	429	304
Minimum	43.36	2.63	2.87	4.13	0.04	0.02	1.82	0.02	0.05	0.5	0.4	0.01	0.1	0.6	1	1.3	9	6
N > Det	54	54	54	54	54	54	54	52	54	11	34	36	29	35	30	47	44	39
N < Det	0	0	0	0	0	0	0	2	0	43	13	11	18	12	17	0	3	8
N	54	54	54	54	54	54	54	54	54	54	47	47	47	47	47	47	47	47
Surface Water																		
Mean	104.11	19.82	22.58	108.87	0.45	0.02	3.89	0.03	0.03	< 0.3	0.6	0.02	<0.1	1.3	3	12.4	58	56
Median	103.20	16.34	23.54	102.40	0.47	0.12	3.56	0.07	0.03		0.7	0.03		1.6	3	12.1	33	55
Maximum	129.30	44.63	29.07	190.70	0.54	0.12	5.98	0.11	0.04	0.0	0.9	0.05	0.0	1.9	3	13.6	112	63
Minimum	36.73	9.23	8.02	65.27	0.17	0.07	1.29	0.02	0.01	0.0	0.5	0.01	0.0	1.4	2	11.5	28	50
N > Det	66	66	66	66	66	3	66	23	66	0	2	2	0	2	3	3	3	3
N < Det	0	0	0	0	0	63	0	43	0	66	1	1	3	1	0	0	0	0
N	66	66	66	66	66	66	66	66	66	66	3	3	3	3	3	3	3	3

Table 7.1 Water quality data from the Tame Valley 2001 - Cations

Water Quality Data From Sutton Park 2001 - Cations																		
	Ca	K	Mg	Na	Sr	Mn	Si	Fe	Ba	Al	Pb	Hg	Cd	Cr	As	Cu	Zn	Ni
Units	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l	μg/l
Det Limit	< 0.02	<0.02	<0.02	<0.02	<0.02	< 0.02	<0.02	< 0.03	< 0.02	< 0.3	<0.4	<0.01	<0.1	<0.5	<1	<0.5	<10	<5
Shallow Groundwater																		
Mean	46.30	10.59	6.77	15.93	0.09	0.84	6.18	2.90	0.13	0.6	<0.4	<0.01	<0.1	<0.5	2	2.0	14	<5
Median	43.59	10.27	7.17	13.21	0.08	1.26	6.20	5.79	0.07	2.5					2	2.0	14	
Maximum	82.13	18.38	9.54	28.25	0.16	1.45	9.07	11.02	0.31	2.5	0.0	0.00	0.0	0.0	2	2.0	14	0
Minimum	15.90	3.44	3.21	9.03	0.06	0.64	3.25	0.56	0.05	2.5	0.0	0.00	0.0	0.0	2	2.0	14	0
N > Det	4	4	4	4	4	3	4	2	4	1	0	0	0	0	1	1	1	0
N < Det	0	0	0	0	0	1	0	2	0	3	1	1	1	1	0	0	0	1
N	4	4	4	4	4	4	4	4	4	4	1	1	1	1	1	1	1	1
Riverbed Piezometers																		
Mean	43.66	4.61	5.11	9.59	0.05	0.16	5.42	0.66	0.19	0.5	5.6	0.02	0.4	1.7	1	10.5	22	11
Median	43.28	3.51	3.50	7.40	0.04	0.13	5.32	0.31	0.10	0.7	6.3	0.02	0.2	1.9	2	6.0	25	15
Maximum	95.34	11.47	15.03	24.87	0.10	0.62	7.57	3.67	0.65	1.4	12.6	0.06	2.8	4.7	3	52.7	65	33
Minimum	3.70	1.46	1.76	4.72	0.03	0.02	3.76	0.02	0.06	0.5	0.8	0.01	0.2	1.4	1	0.6	11	7
N > Det	12	12	12	12	12	10	12	12	12	5	8	7	4	6	2	8	6	4
N < Det	0	0	0	0	0	2	0	0	0	7	1	2	5	3	7	1	3	5
N	12	12	12	12	12	12	12	12	12	12	9	9	9	9	9	9	9	9
Spring 1	61.06	3.05	2.39	7.38	0.04	< 0.02	5.34	< 0.02	0.22	< 0.3	1.0	0.01	0.1	0.8	<1	2.5	<10	<5
Spring 2	44.35	3.60	4.78	7.05	0.04	< 0.02	5.03	0.02	0.10	< 0.3								
Surface Water																		
Mean	36.62	3.53	4.89	10.56	0.05	0.04	4.32	0.93	0.11	< 0.3	7.8	0.05	0.5	4.3	<1	5.6	14	5
Median	39.30	3.31	3.56	12.17	0.04	0.05	4.85	0.46	0.11		11.5	0.04	1.3	2.6		8.2	15	5
Maximum	42.20	4.83	7.62	13.00	0.07	0.06	5.30	2.23	0.13	0.0	12.3	0.09	1.3	7.8	0	10.6	19	5
Minimum	28.37	2.44	3.48	6.52	0.03	0.02	2.81	0.10	0.10	0.0	10.6	0.02	1.3	2.4	0	5.8	9	5
N > Det	3	3	3	3	3	3	3	3	3	0	2	3	1	3	0	2	3	1
N < Det	0	0	0	0	0	0	0	0	0	3	1	0	2	0	3	1	0	2
N	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3

Table 7.2 Summary of water quality data from Sutton Park 2001 - Cations

Water Quality Data From The Tame Valley And Sutton Park 2001 – Anions and Field Measurements											
	pH	D.O.	Temp	EC	E_h	Alkalinity	F	Cl	NO_3	SO_4	P
Units		mg/l	C	µS/cm	mV	mg/l $CaCO_3$	mg/l	mg/l	mg/l	mg/l	mg/l
Det Limit	0.1	0.5	0.1	1	1	1	<0.05	<0.05	<0.05	<0.05	<0.05
Abstraction Wells											
Mean	6.8	8.9	14.1	656	456	102	0.1	55.0	45.6	140.9	0.1
Median	6.9	10.0	13.7	717	463	109		46.1	58.7	135.8	
Maximum	7.5	11.0	20.1	1287	514	180	0.0	121.9	129.7	444.3	0.0
Minimum	5.9	3.0	10.7	144	361	32	0.0	9.7	10.8	20.6	0.0
N > Det	14	14	14	14	13	14	0	13	11	13	0
N < Det	0	0	0	0	0	0	14	0	3	1	14
N	14	14	14	14	13	14	14	13	14	14	14
Shallow Groundwater											
Mean	7.0	6.1	15.1	1042	444	241	5.1	64.6	38.8	141.5	2.1
Median	7.0	7.0	14.9	737	441	200	0.1	37.9	31.3	98.1	0.1
Maximum	8.2	10.0	23.4	6330	561	600	70.8	279.7	158.4	452.8	14.6
Minimum	5.9	1.0	11.9	166	331	109	0.1	0.1	0.1	0.1	0.1
N > Det	20	17	20	20	17	18	20	20	20	20	20
N < Det	0	0	0	0	0	0	0	0	0	0	0
N	20	17	20	20	17	18	20	20	20	20	20
Riverbed Piezometers											
Mean	7.0	3.3	20.1	976	298	304	22.9	148.2	54.2	209.0	2.5
Median	7.0	3.5	20.3	825	333	299	1.3	59.2	21.6	157.7	2.1
Maximum	8.1	6.0	31.0	5580	498	648	197.6	1872.9	233.0	590.7	6.3
Minimum	5.4	1.0	0.6	296	78	65	0.0	9.4	2.8	45.5	1.4
N > Det	56	55	56	58	54	50	60	60	60	60	60
N < Det	0	0	0	0	0	1	0	0	0	0	0
N	56	55	56	58	54	51	60	60	60	60	60
Surface Water											
Mean	7.6	6.0	12.0	723	317	310	1.3	164.5	40.9	166.1	3.4
Median	7.5	6.0	13.9	757	326	330	1.3	161.5	40.6	172.3	4.4
Maximum	8.1	11.0	23.0	875	369	344	2.8	223.9	67.0	241.2	10.9
Minimum	7.4	1.0	6.0	633	257	174	0.0	0.8	0.0	0.5	0.0
N > Det	27	2	60	60	3	27	67	67	67	67	67
N < Det	0	0	0	0	0	0	0	0	0	0	0
N	27	2	60	60	3	27	67	67	67	67	67
SUTTON PARK											
Shallow Groundwater											
Mean	6.0	5.3	12.4	423	337	65	0.1	25.5	18.8	91.1	0.1
Median	6.0	5.0	12.9	413	326	56		23.0	11.8	91.7	
Maximum	7.0	8.0	13.6	651	419	144	0.0	38.9	51.9	160.0	0.0
Minimum	4.9	3.0	10.1	217	276	5	0.0	17.2	11.6	20.9	0.0
N > Det	4	4	4	4	4	4	0	4	3	4	0
N < Det	0	0	0	0	0	0	4	0	1	0	4
N	4	4	4	4	4	4	4	4	4	4	4
Riverbed Piezometers											
Mean	6.6	3.2	15.8	253	329	138	0.6	23.7	18.6	48.7	2.2
Median	6.7	3.0	15.5	240	380	150	0.4	16.0	13.2	49.5	2.2
Maximum	7.6	9.0	20.8	422	499	217	1.9	68.4	55.5	103.6	2.7
Minimum	4.8	1.5	12.7	121	27	0	0.1	10.1	4.4	11.3	1.8
N > Det	12	12	11	12	10	6	12	12	12	12	12
N < Det	0	0	0	0	0	0	0	0	0	0	0
N	12	12	11	12	10	6	12	12	12	12	12
Spring 1	7.0	6.0	13	248	410	143.00	2.0	14.39	35.87	41.32	2.26
Spring 2	6.8		10	185			0.4	16.54	10.33	48.46	2.29
Surface Water											
Mean	7.1	6.8	13.3	200	344	80	0.7	25.2	15.1	43.4	2.3
Median	7.1	6.0	12.2	190	344	80	0.6	24.9	8.3	39.4	2.4
Maximum	7.4	8.5	16.8	235	369	107	1.4	35.3	29.5	57.3	2.5
Minimum	7.0	6.0	11.0	175	319	53	0.1	15.3	7.6	33.6	2.0
N > Det	3	3	3	3	2	2	3	3	3	3	3
N < Det	0	0	0	0	0	0	0	0	0	0	0
N	3	3	3	3	2	2	3	3	3	3	3

Table 7.3 Water Quality Data From The Tame Valley And Sutton Park 2001 – Anions and Field Measurements

7.1 Indications of water quality in the groundwater and surface water systems from electrical conductivity measurements.

Electrical conductivity (EC) is proportional to the total dissolved ionic mass. Elevated levels of EC (Table 7.3) in the urban shallow groundwater (mean 1042 μScm^{-1}, median 737 μScm^{-1}) relative to the rural levels at Sutton Park (mean 423 μScm^{-1}, median 413 μScm^{-1}) are indicative of widespread contamination beneath Birmingham. However the use of EC as a measure of water quality is limited as it does not reflect organic quality and is representative only of total dissolved mass, regardless of the relative toxicity of the components. In surface water, elevated levels of chloride from water treatment processes may raise the EC without significantly decreasing the water quality.

Deep urban groundwater from the abstraction wells appears to be less contaminated than the shallow groundwater showing generally lower values of EC (mean 668 μScm^{-1}, median 703 μScm^{-1}). EC values are also markedly different for the surface waters of the River Tame (mean 723 μScm^{-1}, median 757 μScm^{-1}) compared with the streams of Sutton Park (mean 200 μScm^{-1}, median 190 μScm^{-1}).

The mean EC value of 976 μScm^{-1} (median 875 μScm^{-1}) for the riverbed piezometers (Table 7.3) is considerably higher than for the surface waters of the Tame implying that they are likely to decrease the water quality of the river. However, this may be partly due to an upward bias in the riverbed mean due to high levels (maximum 5580 μScm^{-1}) detected at one locality. The values of EC in the surface water for different sampling periods show no conclusive increase (Figure 7.1) with distance downstream from profile 5,

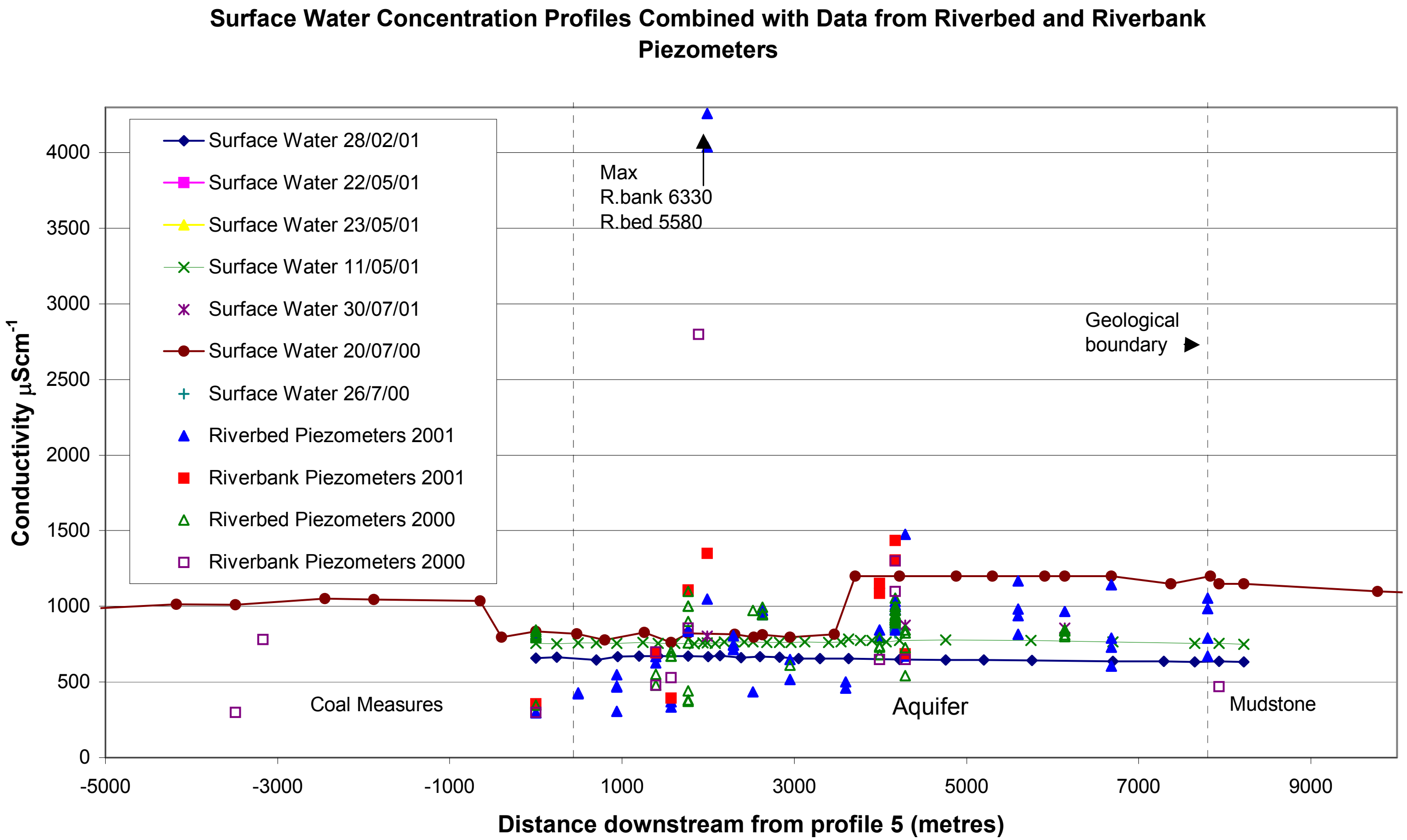


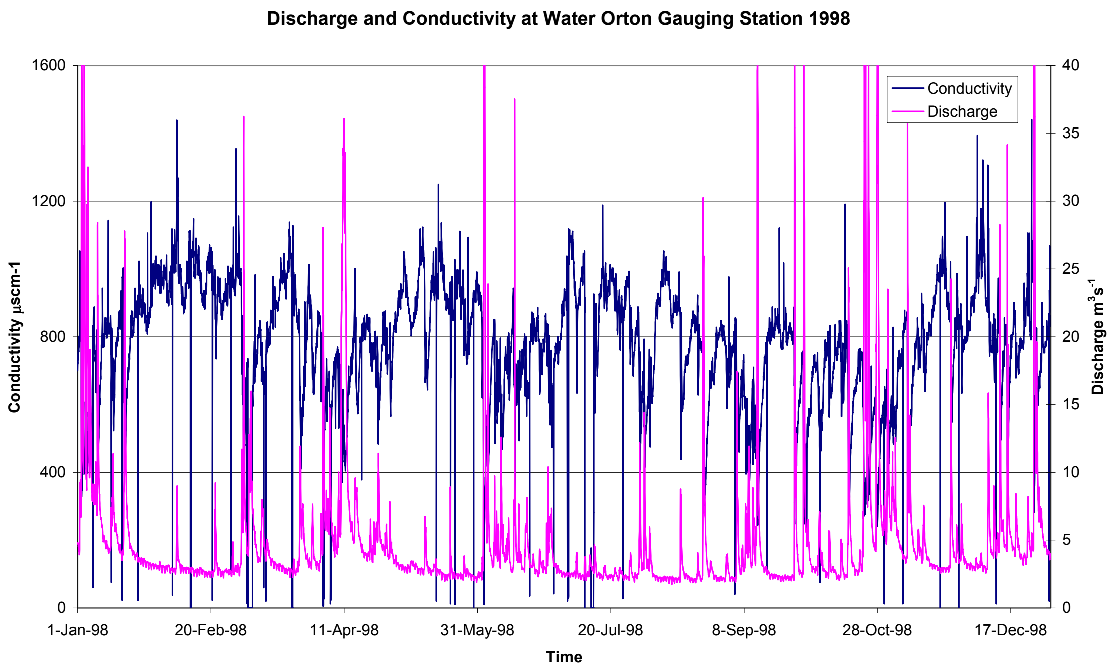
Figure 7.1 Longitudinal profile of groundwater and surface water conductivity.

(the furthest upstream of the riverbed profiles). The riverbed piezometers and the shallow piezometers located on the river banks display a considerable contrast with the EC measured in the surface water at the same location.

A general upward trend in EC values is observed in the riverbed piezometers downstream across the aquifer which coincides with a change in land use, from parkland and residential areas (first 4 riverbed profiles) to more industrial areas. Adjacent samples in the surface water profile for the Tame taken on consecutive days 19-20/7/00 show two large step changes in the EC of 30% (796 to 1035 μScm^{-1}) and 47% (815 to 1200 μScm^{-1}). These are associated with temporal changes in the sewage treatment works (STW) discharges and different sampling times. By comparison, hourly sampling at Profile 8 over an eight hour period on 26/7/00 showed an increase through the day in surface water EC of only 14% (825 to 941 μScm^{-1}). This daily variation makes it difficult to interpret the surface water profiles and relate them to the discharge of contaminated groundwater. There is no conclusive evidence in the surface water profile of any impact from the high EC plume located in the riverbed (Figure 7.1).

Data on EC and flow in the River Tame for 1998 from the gauging station at Water Orton (Figure 7.2 a and b) show considerable (50-200 μScm^{-1}) daily and weekly fluctuations in EC. These are only partly explained by variations in STW discharge and probably indicate multiple sources of flux to the river. Rainfall events are sometimes seen to produce a rapid initial peak in EC reaching values of over 1,500 μScm^{-1} associated with the first flush of urban run-off and overflow discharges from combined sewer overflows (CSOs). Levels of EC then fall due to dilution by the subsequent drainage of less contaminated run-off.

(a)



(b)

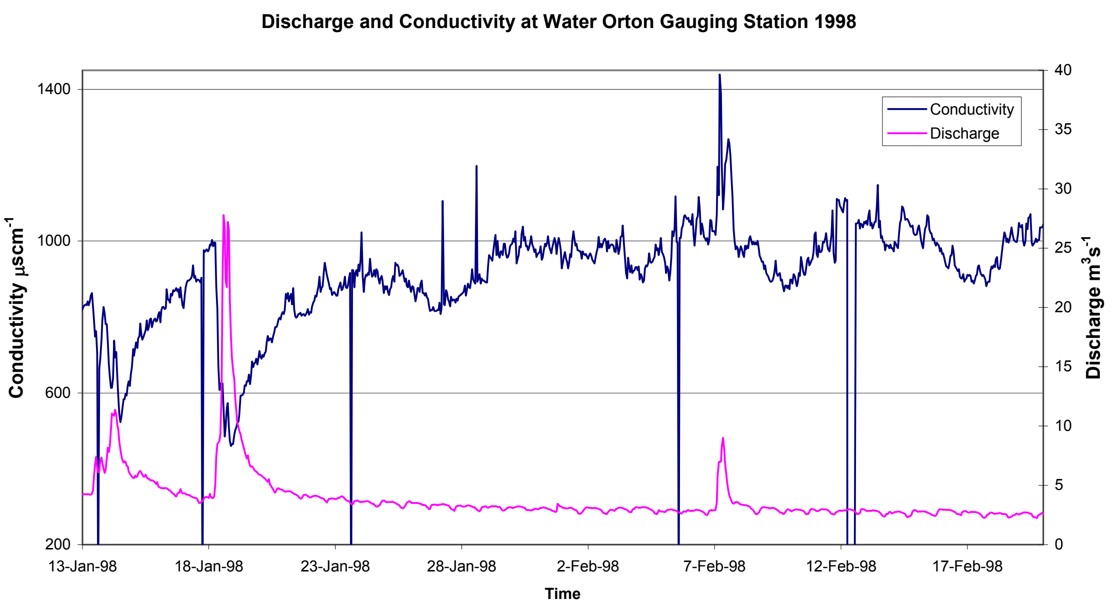


Figure 7.2 The relationship between conductivity and discharge at Water Orton, for (a) the entire year, 1998 (b) winter baseflow January – February.

Underlying the fluctuations due to anthropogenic inputs and rainfall is a general progressive increase in EC during the baseflow recession. This indicates that low levels of baseflow are associated with poorer surface water quality. For example, EC increases from 900 μScm^{-1}on 23/1/98 to 1000 μScm^{-1} on 20/2/98 as baseflow declines (Figure 7.2b). Thus, if EC increases as the groundwater contribution to surface water decreases, and if anthropogenic inputs are assumed to remain relatively constant through baseflow periods, the implication is that, on the catchment scale (400 km^2), groundwater discharge improves the quality of the river by diluting anthropogenic inputs. The results from the riverbed piezometers indicate a higher EC in the groundwater but these results are specific to discharge from the Birmingham Aquifer whereas the gauging station data reflect the larger catchment scale.

The level of industrial discharges will limit the extent to which groundwater affects the quality of surface water, but even without industrial discharges, the levels of EC of surface water in urban areas are still likely to remain higher than in rural areas.

The poorest levels of water quality occur in the river during the initial phase of a flood event. Previous workers (O'Connor, 1976, Pinder et al.,1969) have found that groundwater forms a significant component of flow in a natural catchment during these events (up to 80%, Sklash et al., 1979). Although it is generally considered most important in upland catchments the effect of groundwater dilution may be significant in mitigating the effects of first flush run-off and CSO discharges in the Tame.

7.2 The pH and redox environment

Table 7.3 indicates a mean pH for the deep, shallow and riverbed urban groundwater system of 7.0 (median 7.0). A minimum value for pH of 5.4 detected in the riverbed at Profile 8 is believed to be associated with a discharging contaminant plume. The pH of the groundwater system is dominated by the dissolution of calcite cement from the sandstone that buffers the pH by the formation of HCO_3^- ions. This has maintained near neutral levels of pH despite acidification of the groundwater (Ford et al., 1992) associated with the leakage of industrial acids, N-species transformations, and hydrocarbon oxidation which have either released hydrogen ions or increased the content of carbon dioxide.

The surface water pH ranges between 7.4 and 8.1 with a mean of 7.6 and displays significant daily fluctuations. Agency data from monthly water quality sampling at Water Orton between 1989 and 1996 show a range in pH of between 7.1 and 8.7. Data from the continuous monitoring of pH were available for Water Orton during July 1998 and the maximum pH range observed in a single day was from 7.3 to 8.1 over a nine-hour period. The rural catchment displays a mean pH of 6 in the shallow groundwater and 6.6 in the riverbed associated with acidic peat bogs in the valley. These conditions are perhaps representative of the original environment in the Tame Valley where peat deposits have been observed at several locations (Powell et al., 2000). The acidic conditions prevalent within the bogs may have contributed, along with recent anthropogenic acidification, to the formation of a shallow (<20m) zone of decementation (Ford et al., 1992) across the aquifer as a result of calcite dissolution.

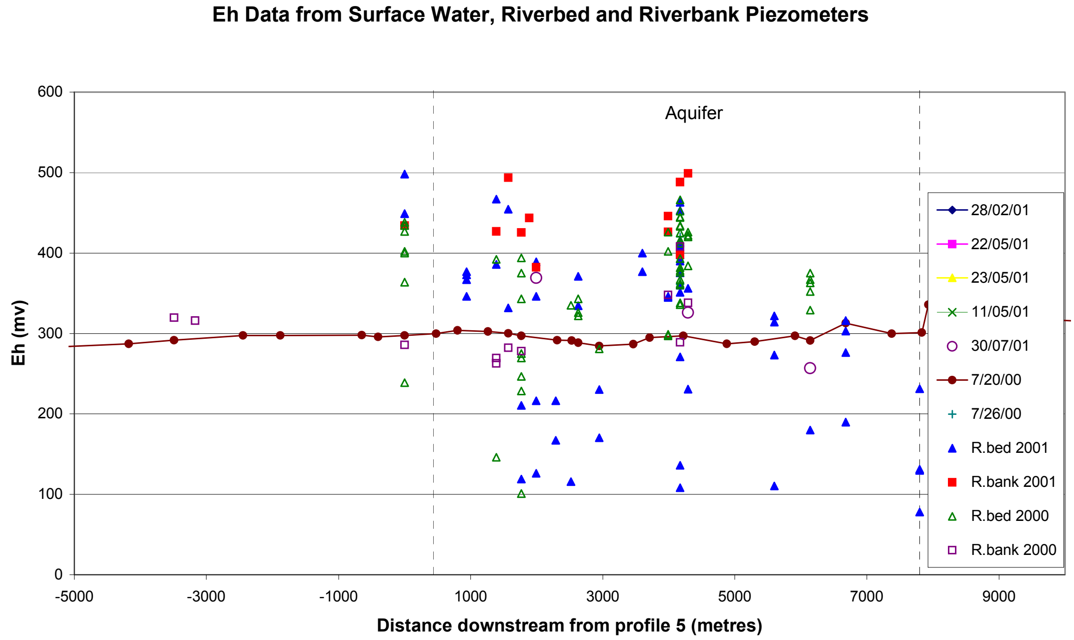
Oxygen levels in the urban groundwater drop from a mean of 8.4 mgl^{-1} (median 9 mgl^{-1}) in the deep to a mean of 6.1mgl^{-1} (median 7 mgl^{-1}) in the shallow. This, perhaps, reflects oxygen consumption related to microbial oxidation of shallow hydrocarbon and ammonium contamination as the recent recharge water would be expected to contain more dissolved oxygen than observed. The main aquifer comprises red bed sandstones with hematite and iron oxyhydroxide grain coatings that has been extensively oxidised under previous geologic conditions. There are unlikely to be large amounts of original organic material or sulphides that would consume significant amounts of oxygen and it is reasonable, therefore, to expect high levels of oxygen at depth. However, oxygen levels from the abstraction wells are higher than the mean value of 5.6 mgl^{-1} recorded from previous sampling (Ford, 1990), perhaps indicating exposure to the atmosphere between abstraction and sampling. The riverbed sediments have a low mean oxygen content of 3 mgl^{-1} (median 3.5 mgl^{-1}) with anoxic conditions observed in several locations. The drop in oxygen levels is associated with microbial activity in the riverbed sediments where a supply of organic compounds from the surface water is readily available for degradation. Oxygen levels observed in Sutton Park follow a similar trend, with the shallow groundwater having a mean of 5.3 mgl^{-1}, dropping to a mean of 3.2 mgl^{-1} in the riverbed. Data on oxygen content from the 2001 surface water survey is limited (n=3, mean 6 mgl^{-1}) but Agency data from monthly spot sampling between 1993 and 1996 indicate a fall in dissolved oxygen across the aquifer from mean 10.1 mgl^{-1} at Profile 5 (upstream) to 9.1 mgl^{-1} at Profile 18 (downstream). The Agency data show fluctuation in oxygen levels between the 5th percentile and the 95thpercentile of 6.8 to 12.7 mgl^{-1}, with a minimum of 1.7 mgl^{-1} reported at the downstream end reflecting high biological oxygen demand.

Redox conditions are consistent with the oxygen levels. The aquifer is generally oxidising with mean Eh 456 mv (median 463mv) in the deep aquifer, falling slightly to 444 mv (median 441 mv) in the shallow. More reducing conditions (mean 298 mv, median 333 mv) coincide with microbial activity and the drop in oxygen levels across the riverbed. Surface water data show a mean Eh of 317 mv (median 326 mv). Considerable variation in both the oxygen content and the redox potential (Figure 7.3a and b) is observed within the riverbed piezometers along the reach and within the same lateral profile. Previous workers (Conant, 2000) have attributed this to differences in groundwater discharge velocities and residence times within the sediments, low oxygen content corresponding to long residence times.

Information from five multilevel piezometer profiles (Figures 7.4 a, b and c) shows the often complex vertical zonation in redox conditions that occurs across the riverbed. The western, eastern and central multilevel piezometers at Profile 8 were sampled on more than one occasion and display considerable temporal variability which will be discussed in more detail in Section 7.6. Mixing with surface water appears to occur across the top 40cm to 20 cm of the riverbed as evidenced in some profiles by increasing levels of pH and dissolved oxygen (D.O.) approaching surface water conditions. However, there is great variability in the data and profile 2 shows a decrease in oxygen content towards the surface, with considerable (microbial) oxygen consumption taking place between 30cm and 70 cm depth.

Solid-phase iron and manganese oxides are present throughout the aquifer and the solubilities of both are sensitive to redox conditions. Variations in dissolved concentrations coincide with changing conditions across the aquifer system. Manganese shows increasing mean concentrations from 0.08 mgl^{-1} in the deep, 1.49 mgl^{-1} in the shallow to 3.15 mgl^{-1} in the

(a)



(b)

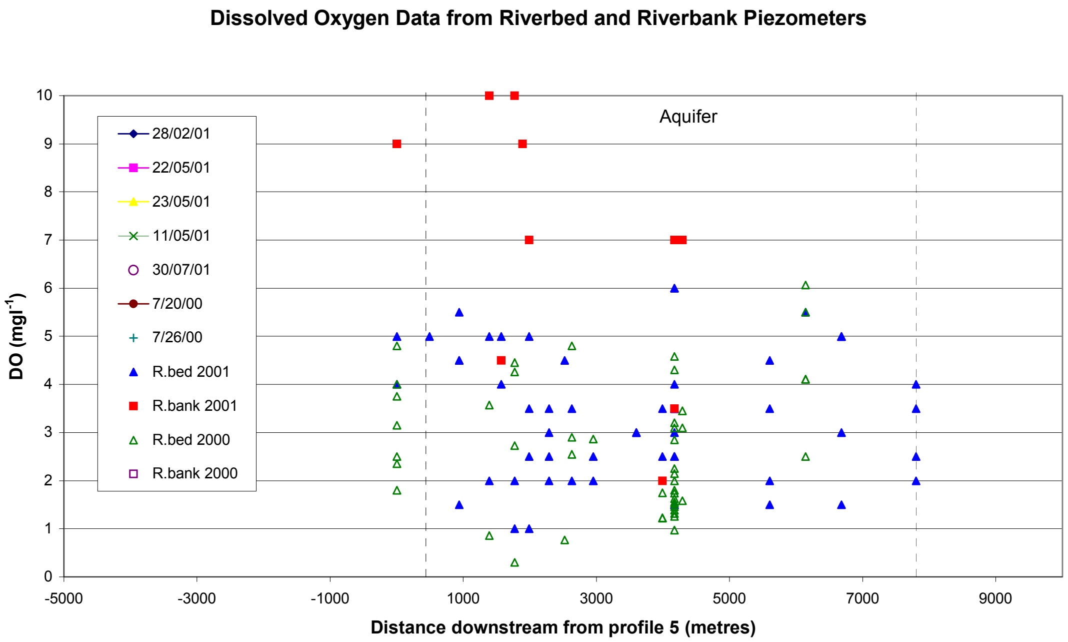
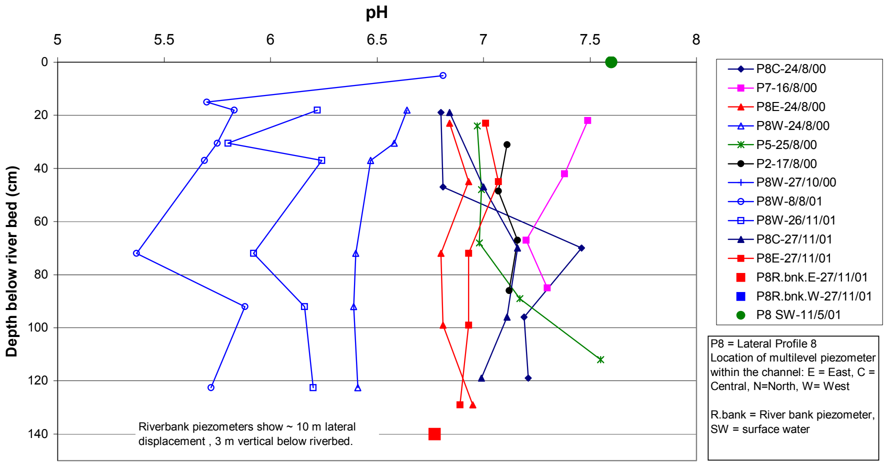


Figure 7.3 Longitudinal profile of surface water and groundwater (a) Eh, (b) D.O.

(a)



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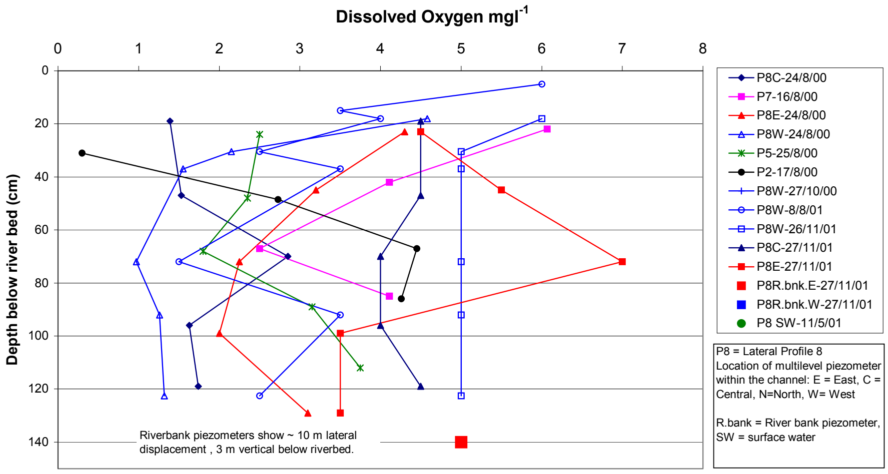


(c)

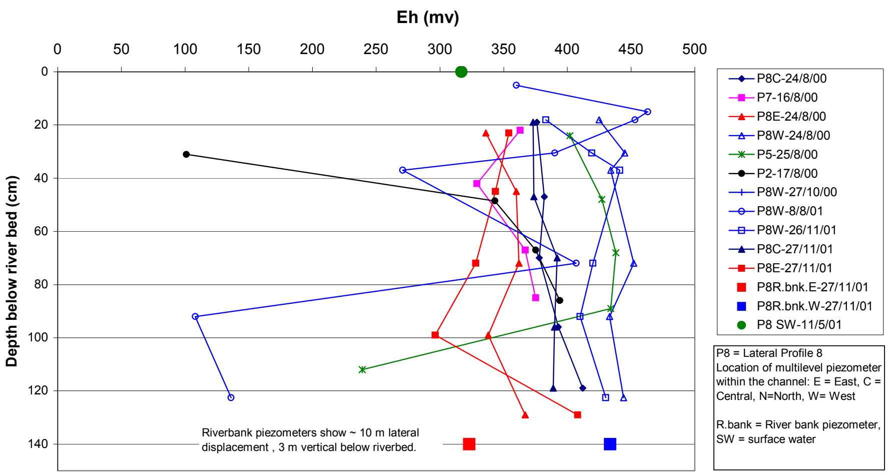


Figure 7.4 Multilevel piezometer profiles within the riverbed of (a) pH (b) D.O. (c) Eh

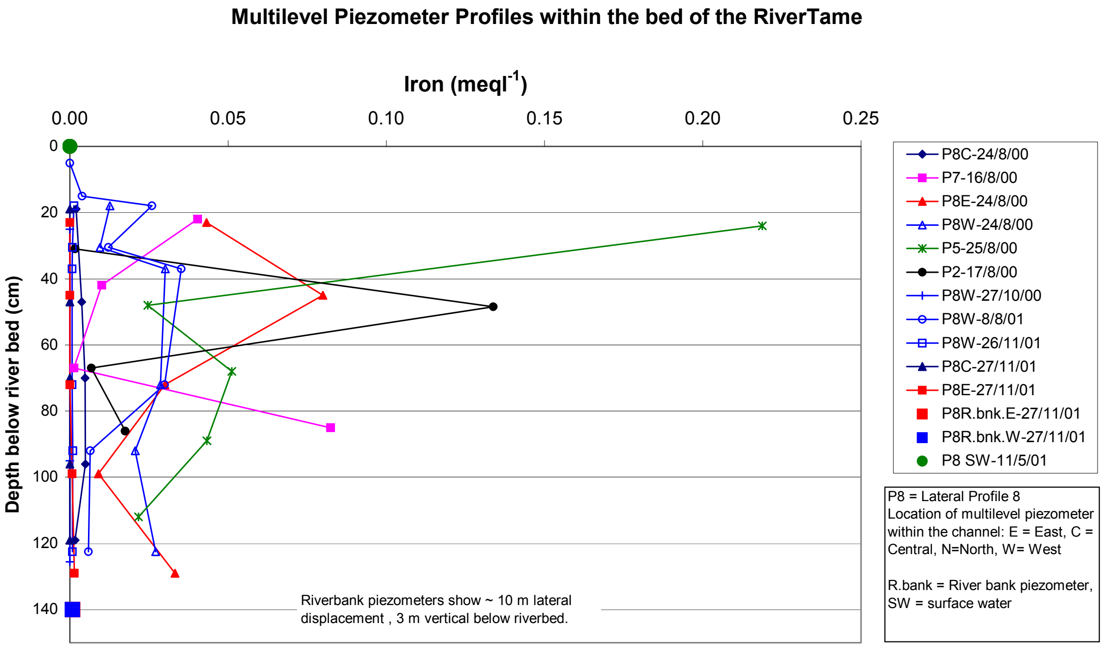
riverbed, implying conditions of decreasing pH and Eh. Iron shows mean concentrations of 0.31 mgl^{-1} (median 0.16 mgl^{-1}) in the deep, 0.18 mgl^{-1} (median 0.13 mgl^{-1}) in the shallow and 1.41 mgl^{-1} (median 0.24 mgl^{-1}) in the riverbed, also implying more reducing and lower pH conditions higher in the system. Manganese has a greater solubility (as Mn^{2+}) than iron (Fe^{2+}) at higher levels of Eh which may explain the higher concentrations. Both are likely to precipitate ($Fe(OH)_3$, $MnCO_3/MnO_2$) with the increase in pH. This is likely to occur during mixing with the surface water and evidence of this can be seen in the multilevel profiles (Figures 7.5 a and b). Other workers (Harvey et al., 1998) have indicated the importance of manganese and iron in the co-precipitation and removal of heavy metals from groundwater.

The abrupt change in redox conditions to a more oxidising and higher pH environment across the upper section of the riverbed, combined with elevated levels of microbial activity, may result in significant changes to the dissolved mass flux from the groundwater to the river.

7.3 Anion Hydrochemistry

The total anion content increases upwards through the groundwater system (Table 7.3) showing higher levels of calcite dissolution and chloride, sulphate and nitrate contamination in the shallow and riverbed piezometers. Elevated concentrations of anions and other contaminants provide evidence of modern recharge in all 14 of the abstraction wells with the exception of a deep (154 m) well beneath the city centre which has chloride, nitrate and sulphate levels of <10 mgl^{-1}. The modern (<200years) recharge has been introduced at depth by high drawdowns during the historical periods of peak abstraction. The impact of high levels of localised contamination is seen in the shallow and riverbed piezometers where mean values are considerably higher than the median, while the abstraction well and surface water

(a)



(b)

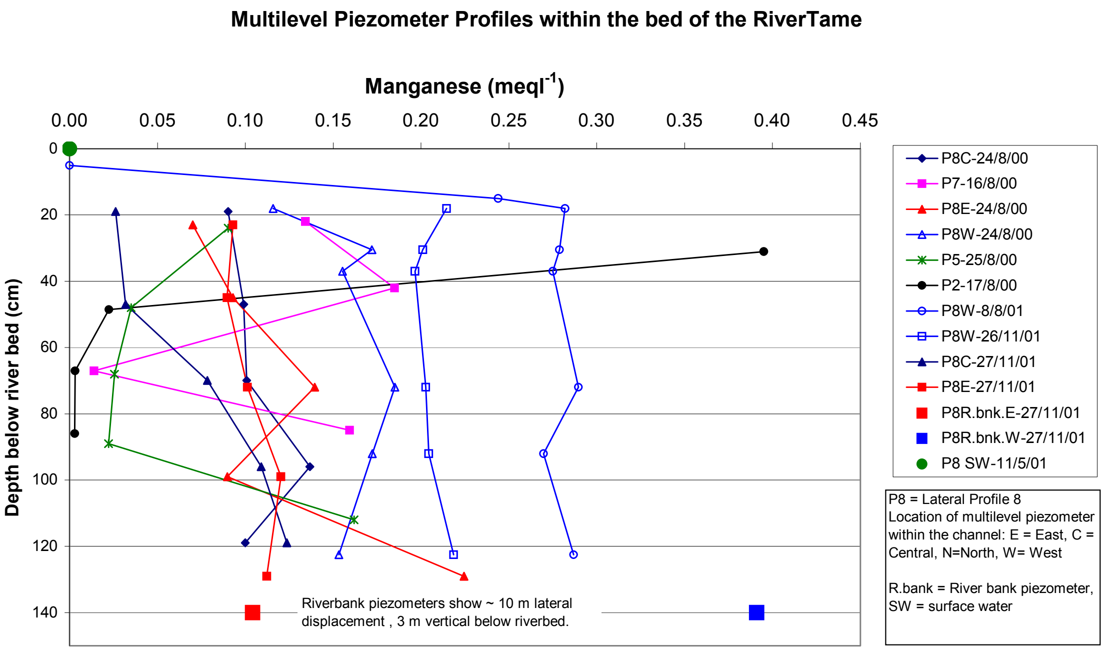


Figure 7.5 Concentration profiles from multilevel piezometer within the riverbed of (a) Fe (b) Mn

samples show only limited variation. This reflects the nature of the sampling, the abstraction and surface water samples giving an integrated sample from a large well-mixed volume whereas the piezometers reflect localised heterogeneity.

A comparison of riverbed and surface water quality (Figure 7.6) shows comparable values for the major anions, the median riverbed value being slightly lower than the surface water and the mean value being slightly higher, perhaps indicating localised impact from contaminant plume discharge. Based upon the similarity in mean riverbed and surface water anion concentrations, and the large dilution effect in the river, the groundwater anion composition will not normally alter the surface water quality significantly. However, should the upstream surface water quality improve, perhaps moving towards levels observed in Sutton Park, then groundwater will be a limiting factor on the maximum level of improvement. Sutton Park shows much lower anion concentrations in all sample types than observed in the Tame Valley, although these are still likely to be elevated above 'natural' levels owing to atmospheric fallout from Birmingham which lies within 10 km in the direction of the prevailing wind.

Bicarbonate is the major anion in the shallow groundwater and surface water but in the deeper abstraction wells sulphate is the dominant species (mean HCO_3^- 2 $meql^{-1}$, mean SO_4^{-2} 2.9 $meql^{-1}$). Calculations indicate that all the abstraction wells were undersaturated with respect to calcite, with saturation indices ranging from –0.3 to –2.8. This is not what would be expected for old groundwater at depth and indicates a source of relatively rapid modern recharge. The modern waters may have had limited time in contact with calcite and/or have mixed in the well with waters from other sources leading to the under saturation of the new solution with respect to calcite. Other reasons such as a common ion effect due to the presence of SO_4 or

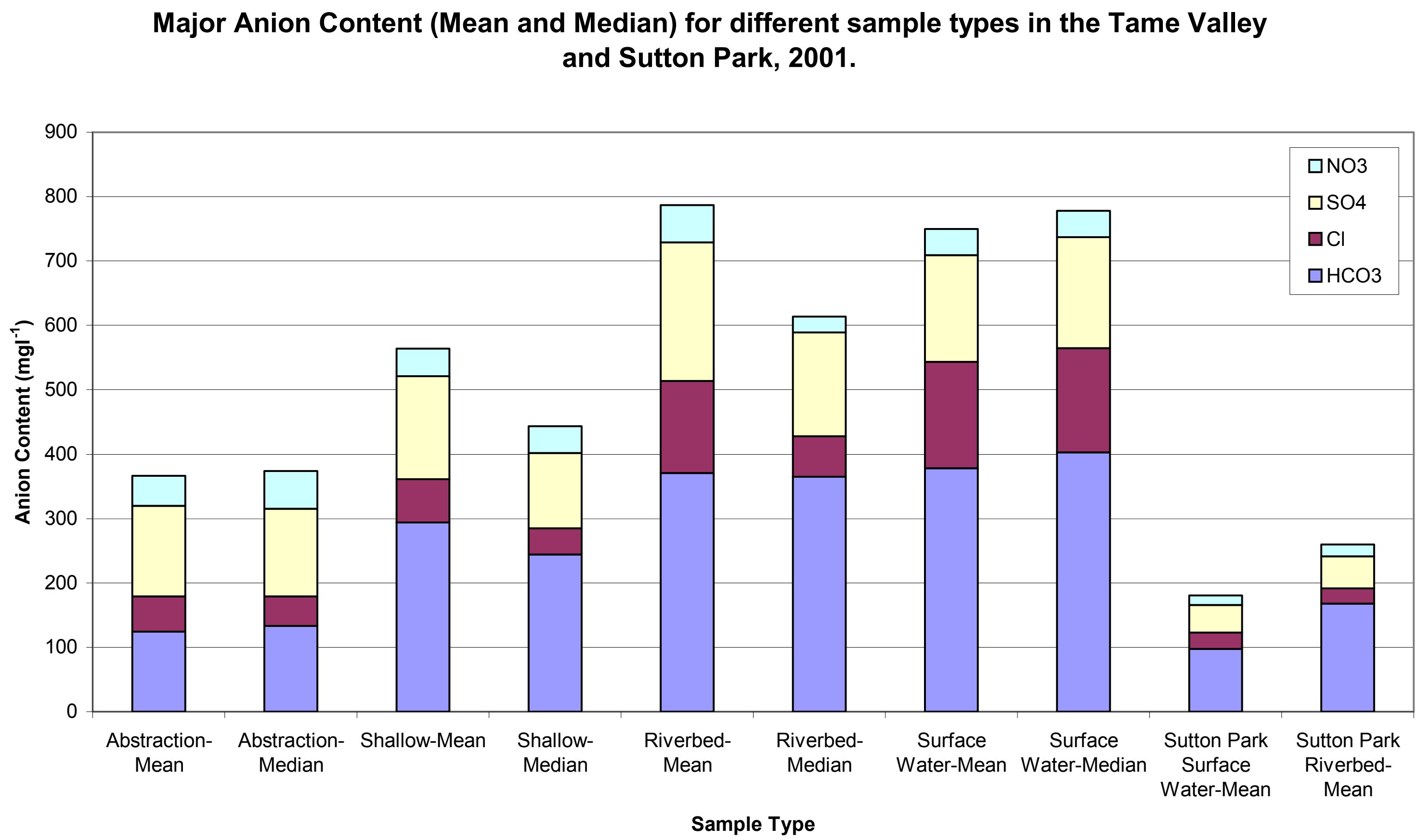


Figure 7.6 The major anion content of different sample types

errors in the pH measurements may have influenced the results. The deep uncontaminated borehole beneath the city centre had a saturation index of –0.8 and a pH of 7.38. These values represent a downward temporal trend, being lower than the results obtained by previous workers (Table 7.4). An upward trend in EC was seen from 290 μScm to 356 μScm between 1988 and 2001. This may indicate increased contamination at depth but the results from the other determinands do not support this conclusively.

The results of shallow groundwater sampling showed general saturation with respect to calcite, with an average HCO_3^- content of 294 mgl^{-1}. The piezometers were in general situated on, or adjacent to, open land and would be expected to receive considerable recharge through the soil zone with a high $ppCO_2$ content. The riverbed piezometers showed a mixture of saturation and undersaturation with respect to calcite and the surface water was saturated with respect to calcite. The source of the recharge to the abstraction wells is of significance when considering the active groundwater flow system in the Tame Valley that will ultimately discharge to the river. The lack of any substantial contribution by old calcite saturated water (i.e. the undersaturation) at depth (<100m) to the abstraction wells implies the dominance of modern recharge in the active groundwater system in the Tame Valley. The use of isotope studies to determine the age of the groundwater discharging across the lateral riverbed piezometer profiles would provide useful information but are beyond the scope of this study.

Sulphate is present throughout the system, attaining levels greater than the 250 mgl^{-1} standard for drinking water in two of the 14 abstraction wells, four of the 20 shallow piezometers and 18 of the 60 riverbed piezometers. The sulphate source is primarily anthropogenic, with heavy atmospheric loading and contributions from industrial, domestic and construction waste and

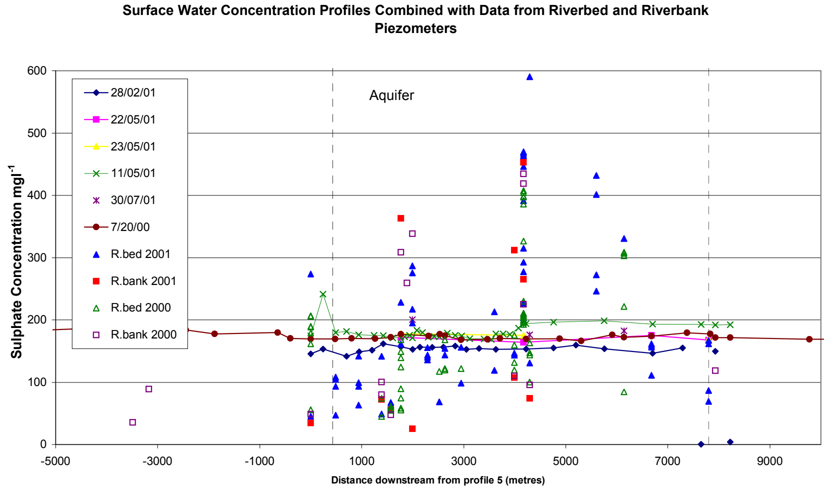
Previous Worker	Sample year	Reference No.	Ca (mgl^{-1})	Cl (mgl^{-1})	Na (mgl^{-1})	NO_3 (mgl^{-1})	Conductivity (µScm)	Alkalinity (mgl^{-1} $CaCO_3$)	pH	Calcite Saturation Index
Jackson (1981)	1979	BH51	35	9	7.4	5.3		131	8	0.06
Ford (1990)	1988	BH30	31.3	11.9	7	6.6	290	131	8.05	0.12
Hughes (1998)	1995	BH30	30.6	16.3	6.5	6.4	310	134	7.61	-0.35
Taylor (1998)	1998	BH16	33.8	22	7.64	3.2				
Shepherd (2002)	2001	5016	31.55	<10	7.24	<10	356	97	7.38	-0.77

Table 7.4 Temporal variation in levels of selected determinands from the deep borehole beneath the city centre

industrial processes. A natural component may be derived from the dissolution of anhydrite and gypsum from the Upper Bromsgrove Formation which has also been reported in the confined aquifer (Ford, 1990). The longitudinal profile (Figure 7.7a) shows background groundwater concentrations ranging from 50 to 200 mgl^{-1} with the occurrence of several high concentration plumes discharging to the river (e.g. riverbed data with a maximum of 590 mgl^{-1}). Some of the riverbed piezometer profiles display a large variation in sulphate concentration across a single lateral profile indicating different sources to the groundwater discharging from either bank. Similarities are seen at some riverbed profiles between the sulphate content in the riverbank piezometer and those in the riverbed, indicating shallow groundwater discharge. However, the riverbank piezometer is seldom representative of the water quality across the entire profile. The five multi-level piezometer profiles (Figure 7.8a) showed limited variation in concentration across the riverbed and did not provide compelling evidence for microbial reduction of sulphate in this zone. The surface water concentration profiles show no obvious correlation with the discharging groundwater quality probably because of dilution effects and temporal variations in the surface water.

Nitrate levels are elevated throughout the system and exceed the drinking water limits of 50 mgl^{-1} in seven of 14 abstraction wells, five of 20 shallow piezometers and 20 of the 60 riverbed piezometers. High background levels of nitrate are reported by previous workers (Ford, 1990) throughout the aquifer and may reflect loading of the aquifer over a period of several hundred years. Sources include land disturbance, atmospheric fall out, sewer leakage, fertilizer, industrial processes and the use of urea as a de-icing agent on the elevated section of the M6 which follows the course of the Tame across much of the aquifer. The longitudinal concentration profile (Figure 7.7b) shows several high concentration plumes (maximum 232

(a)



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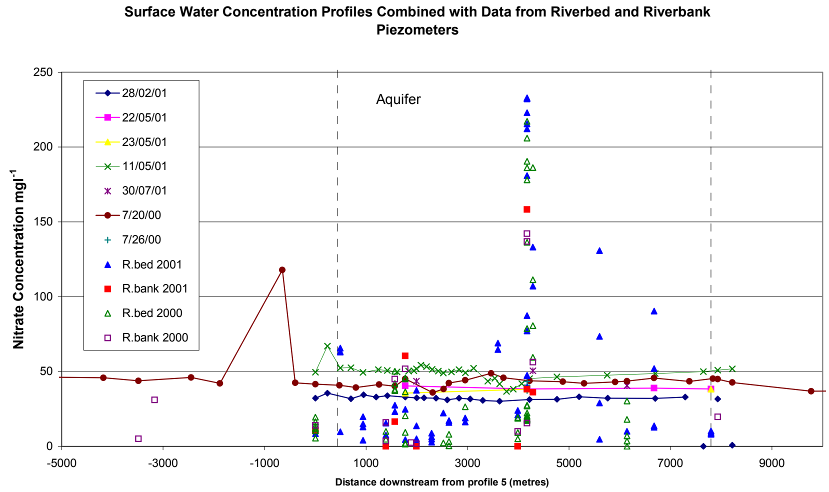

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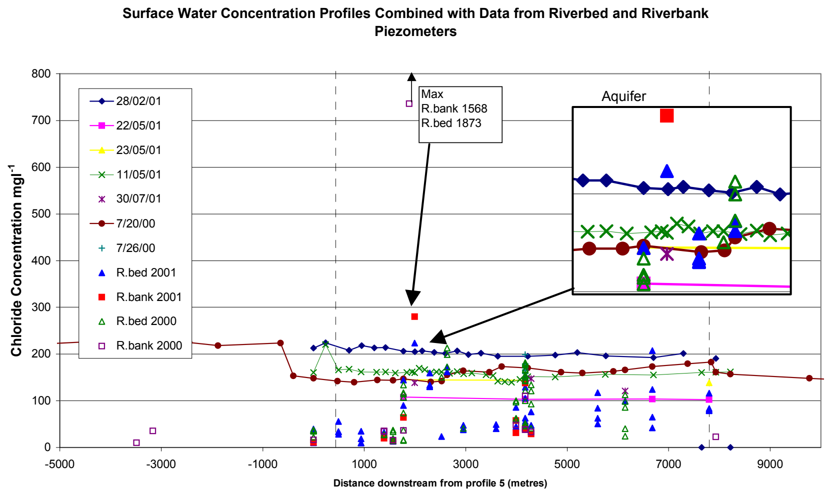


Figure 7.7 Longitudinal concentration profiles for (a) Sulphate (b) Nitrate (c) Chloride.

(a)

(b)

(c)

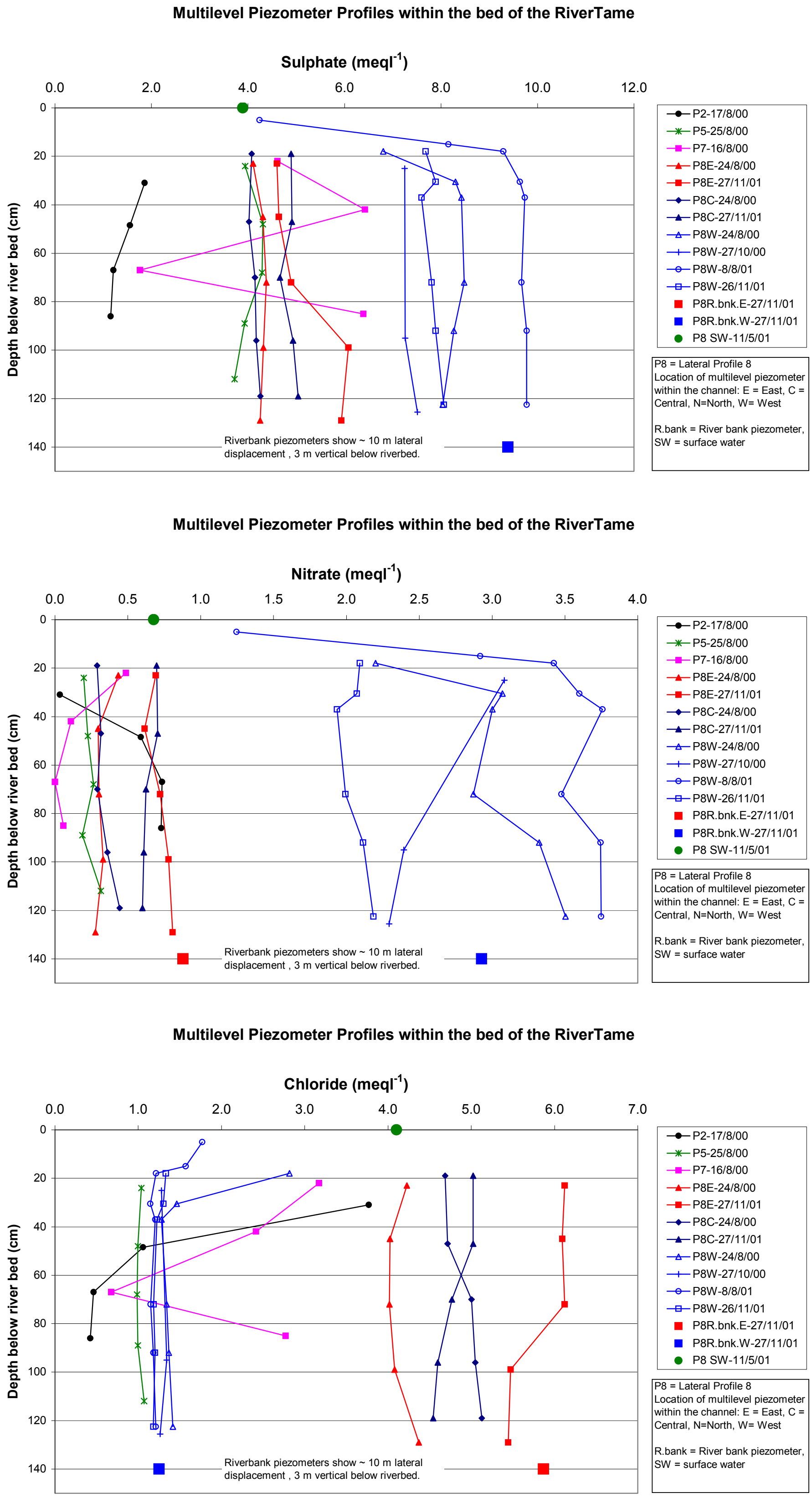


Figure 7.8 Concentration profiles from multilevel piezometers within the riverbed, of (a) Sulphate (b) Nitrate (c) Chloride.

mgl^{-1} in the riverbed), but with generally low 10-20 mgl^{-1} background levels. These background levels are lower than might be expected given the high level of local contaminant sources and may reflect microbial reduction. Evidence for reduction has been found in the multilevel riverbed piezometers (Figure 7.8b). Profile 2 displays a rapid decline in nitrate levels to below detection limits within 30cm of the riverbed, coincident with anaerobic conditions and an Eh of 100 mv. A small subset of samples from the three multi-level riverbed piezometers at profile 8 were analysed for nitrite which was detected in all the samples with maximum values of 0.33 mgl^{-1} recorded.

Chloride concentration shows higher mean values than medians throughout the system reflecting locally high levels of contamination with maximum values of 1,873 mgl^{-1} measured in the riverbed piezometers. Median values varied from 46 mgl^{-1} in the abstraction wells to 41 mgl^{-1} in the shallow groundwater and 63 mgl^{-1} in the riverbed piezometers. Four of the deep abstraction wells displayed low chloride values of between 10 to 26 mgl^{-1}, implying an older uncontaminated source of groundwater, and current background levels from all waters in Sutton Park averaged 25 mgl^{-1} . The widespread more elevated levels of chloride found generally in the groundwater are a result of anthropogenic contamination from industry and urban sources including a high loading from winter road salting, leaking mains and sewers, and the degradation of chlorinated organic compounds. The surface water shows significantly higher levels of chloride (median 162 mgl^{-1}) than the groundwater as a result of sewage inputs and the subsequent discharge to the river (>30% of DWF). Previous workers (Jackson, 1981, Ford, 1990) have indicated that high chloride concentrations in ground waters abstracted from the Tame Valley may have resulted from the inflow of surface water into the aquifer during periods of peak historical abstraction. The three abstraction wells sampled within 500 m of the river during the current survey did not show significantly higher concentrations (41, 46, 104 mgl^{-1}) than the other 11 wells sampled across the aquifer but the data set is too limited to draw any conclusions from this.

The longitudinal profile (Figure 7.6c) shows background levels of chloride increasing from 20 to 80 mgl^{-1} across the aquifer perhaps associated with changing land use from parkland to industrial (Figure 4.2). A high concentration plume of fairly limited lateral extent was detected in the riverbank (1,568 mgl^{-1}) and riverbed piezometers (1,873 mgl^{-1}) of one profile but was not detected in profiles 250m upstream or downstream. Surface water sampling across the plume at 100 m intervals detected a rise in surface water chloride concentrations from 160 to 169 mgl^{-1} in 100 m on 11/5/01, and from 205 to 207 mgl^{-1} on 28/2/01. However, these changes do lie within the range of analytical error and the surface water concentration profile is difficult to interpret because of variations over time in the chemical flux coming downstream. Repeat surface water sampling on 26/7/00 at profile 8 showed a rise in chloride concentrations from 150 to 200 mgl^{-1} in 6 hours. The variation in STW discharges also explains the sharp change in the chloride concentrations (223 to 153 mgl^{-1}) observed in surface water sampling between the 19th and 20th of July 2000. The effect of dilution from the waters of the River Rea, which does not contain any STW discharges, is visible at the downstream end of the aquifer on the 19/7/00. Consecutive samples were taken within 15 minutes on either side of the confluence and a fall in chloride concentration from 183 to 161 mgl^{-1} was observed.

The high chloride concentrations in the river (median 162 mgl^{-1}) compared with those generally found in the groundwater (median 59 mgl^{-1}) provide a useful tracer to establish the depth of surface water penetration into the riverbed. Data from the multilevel riverbed piezometers generally show an increase in chloride concentrations over the upper 20cm to 60 cm of the riverbed which is interpreted as an upward flow of groundwater mixing with surface water (Figure 7.8c). The degree of mixing was calculated from the ratio of the concentrations of chloride in the surface water and the groundwater at depth. Profile 2 has >95% surface water at 35 cm depth while in some other profiles no mixing with surface water is apparent at depths of just 20 cm . Temporal variation in the extent of the mixing zone is also apparent, with a decrease from 40 cm (24/8/00) to 20 cm (8/8/01) in the western multilevel piezometer at Profile 8. The extent of the hyporheic zone is dependent on local groundwater heads and discharge rates and river stage, gradient and bedforms. The deepest penetration of surface water occurs at Profiles 2 and 7 which are narrow, engineered sections of channel, with river depths >70cm and low predicted groundwater discharge based on Darcy flow calculations. Conversely, Profiles 5 and 8 have wide channels (~12m) with DWF river depths of ~30 cm and high predicted groundwater discharge. Profiles 2 and 8 are within the river meander modelled in Section 7.4.2, (Figure 7.1), with modelled discharge rates of 0.2 and 3.7 m^3d^{-1} per metre length of channel respectively. A proportion of the total volume of surface water flowing along the study reach will pass through the hyporheic zone at some point. This provides an important environment in which microbially mediated reactions, sorption and precipitation may take place that could considerably modify the surface water quality. The residence time of the surface water within the hyporheic zone is unknown but in Profile 2 it is long enough for an anoxic environment to develop in which nitrate from both groundwater and surface water sources is being reduced.

The greatest contrast between surface water and groundwater concentrations was observed for fluoride in which levels of 198 mgl^{-1} were measured in the riverbed discharging to the surface water with a mean concentration of 1.3 mgl^{-1}. The plume was not detected in riverbed profiles 200 m upstream or downstream. The effect of the plume on surface water concentrations was difficult to observe with sample spacings > 400m. A series of closely spaced samples (11/5/01) detected a rise in fluoride concentrations from 1.3 to 1.7 mgl^{-1} over a 270 m length of reach across the plume discharge area (Figure 7.9). This may be associated with the plume but also lies within the range of analytical error and so any relationship is uncertain. The surface water profiles also appear to be subject to fluctuations of fluoride concentration, such as the 6 mgl^{-1} peak only observed on the 20/7/00, that are associated with pulse type discharges coming downstream from an unknown source.

7.4 Major Cation Hydrochemistry

The total concentration of cations in the groundwater is greater at shallower depths as a result of higher levels of mineral dissolution and weathering and anthropogenic sources at the surface (Figure 7.10). The minimum observed levels of Ca 32 mgl^{-1}, Na 7 mgl^{-1}, Mg 16 mgl^{-1}, K 2 mgl^{-1} were sampled from old pre-industrial water within the deep borehole below the city centre. The groundwater discharging to the Tame has a high mineral content, containing on average a four times greater amount of dissolved cations than water discharging through the riverbed in Sutton Park. Under present conditions, groundwater concentrations of the major cations are similar to those in the river and will cause limited variation to the overall surface water concentrations except in localised areas of the riverbed associated with contaminant plume discharge. Calcium, magnesium and sodium are all below the UK limits for drinking water and have no detrimental effect on the surface water quality. Potassium is the only major

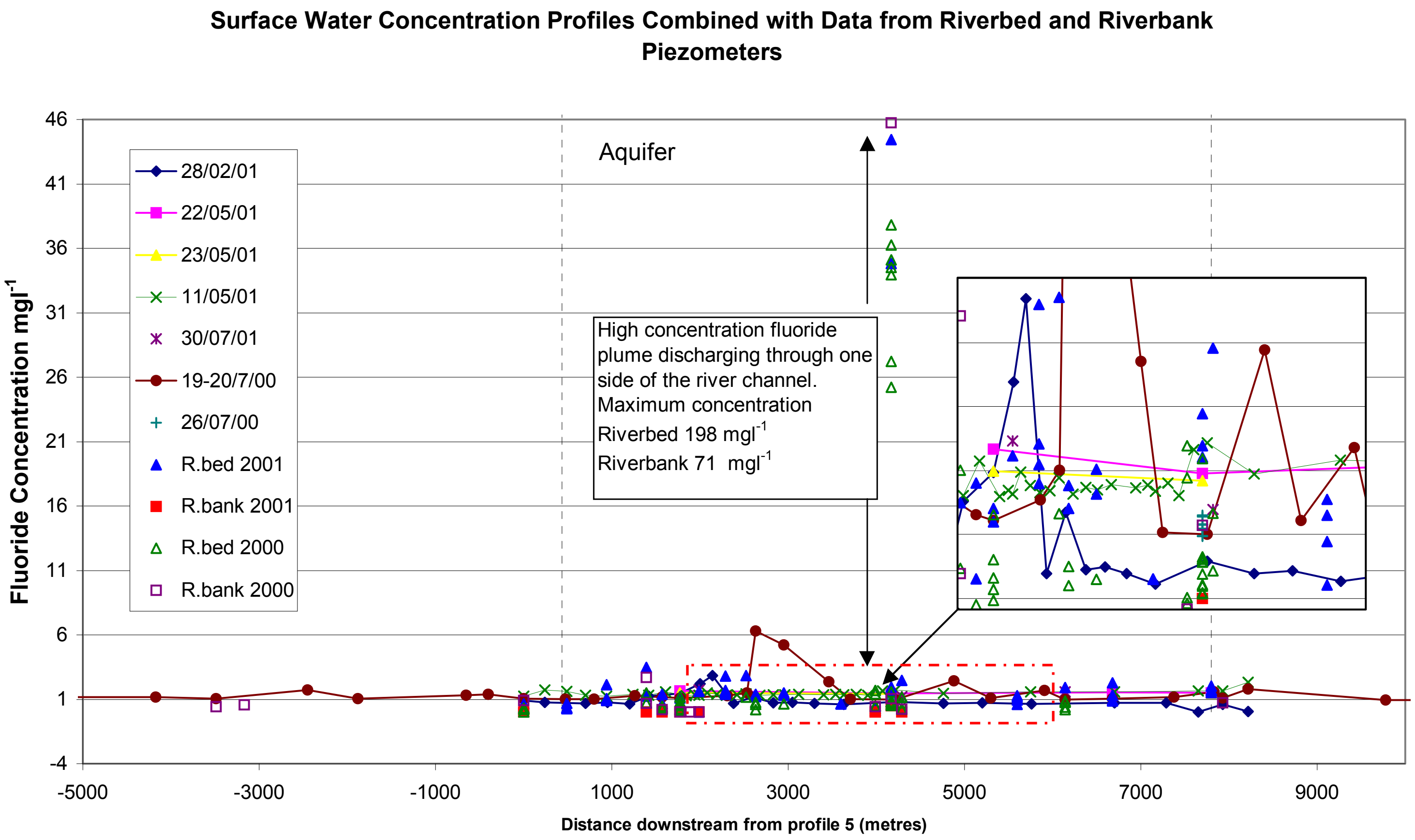


Figure 7.9 Longitudinal concentration profile for fluoride in surface water and groundwater.

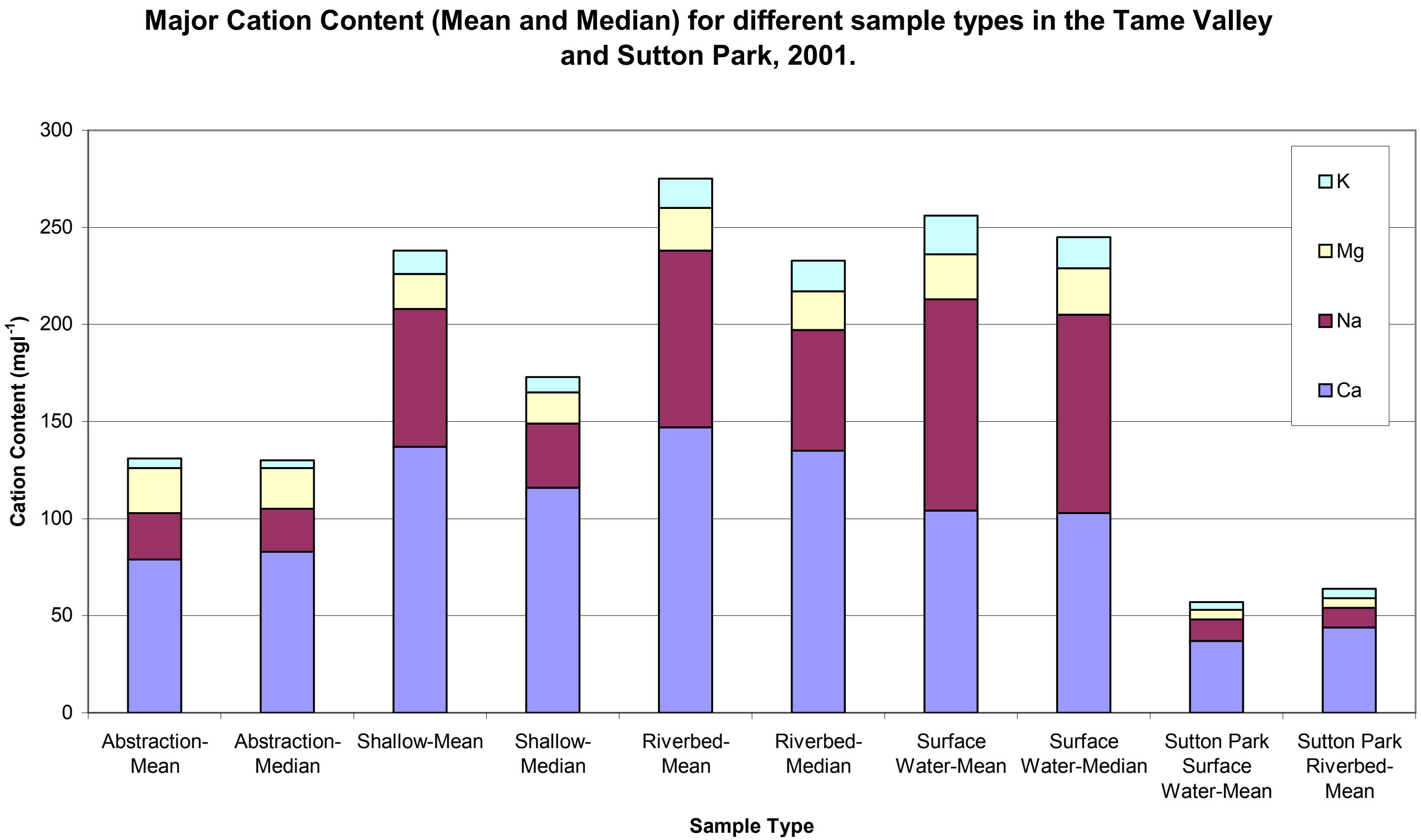
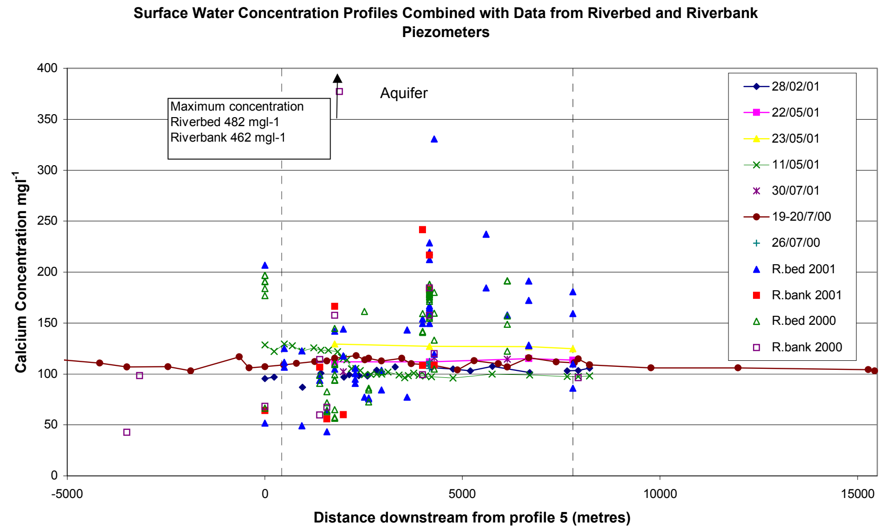


Figure 7.10 Summary of the major cation content of each different sample type.

cation that approaches the drinking water PCV of 12 mgl^{-1} which it regularly exceeds in the shallow (mean 11.6 mgl^{-1}) and riverbed piezometers (mean 14.5 mgl^{-1}) and surface water (mean 19.8 mgl^{-1}).

Calcium is the dominant ion in the groundwater system, increasing from a mean of 79 mgl^{-1} in the abstraction wells to 137 mgl^{-1} in shallow groundwater and 147 mgl^{-1} in the river bed. This is a result of increased calcite dissolution and contaminant inputs, most likely from $CaSO_4$ that is found extensively in building waste and made ground. The calcium concentration in the riverbed piezometers generally increases downstream across the aquifer from levels of ~ 60 mgl^{-1} in the parkland (0 km) to ~150 mgl^{-1} immediately downstream from industrialised areas (Figure 7.11a). It is difficult to say whether this change is simply related to changing land use and increasing anthropogenic sources or whether changes in the underlying hydrogeology along the reach are also having an impact. Progressing downstream the river first flows over the Upper Coal Measures and then across the Kidderminster Formation, which increases in thickness downstream to the contact with the Wildmoor Formation and then the Bromsgrove Formation. In addition, the bedrock is overlain by an extensive alluvial aquifer that covers much of the flood plain to depths of 5 m . Due to a limited sample distribution, previous work on abstraction well data was unable to conclude that there was any distinctive hydrochemistry characterising the sandstone aquifer sub-units. However, the alluvium is expected to have a distinctive hydrochemistry with a lower calcite content than the sandstone. The low calcium waters observed at the upstream end of the aquifer are under-saturated with respect to calcite, and may represent a local fast-flow system through the thin sandstone and overlying alluvial gravel. Increasing calcium content may indicate a rising proportion of discharge from the underlying sandstone in addition to the

(a)



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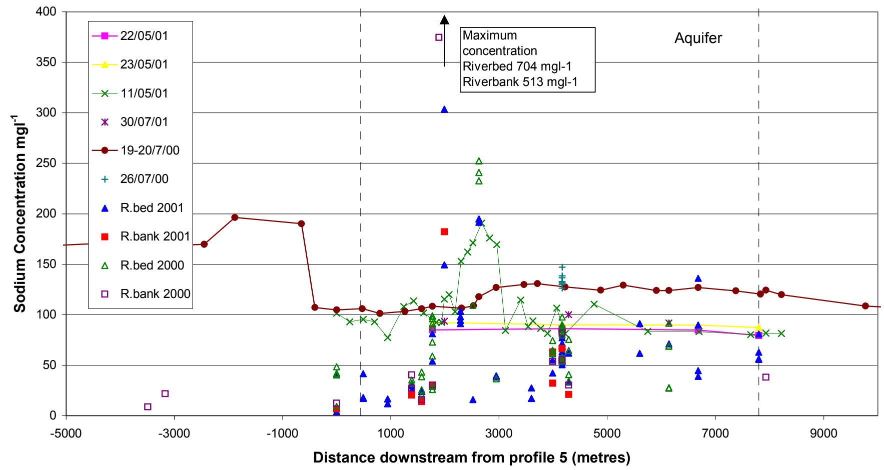

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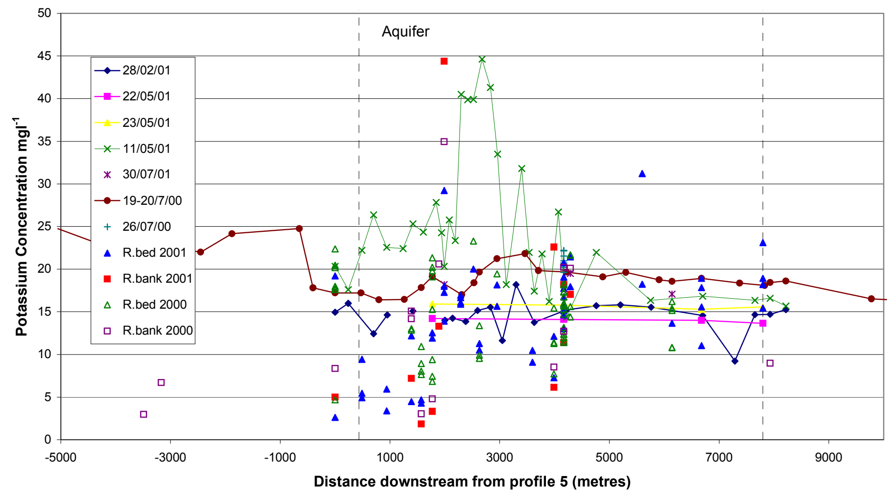


Figure 7.11 Longitudinal concentration profiles for (a) Ca (b) Na (c) K.

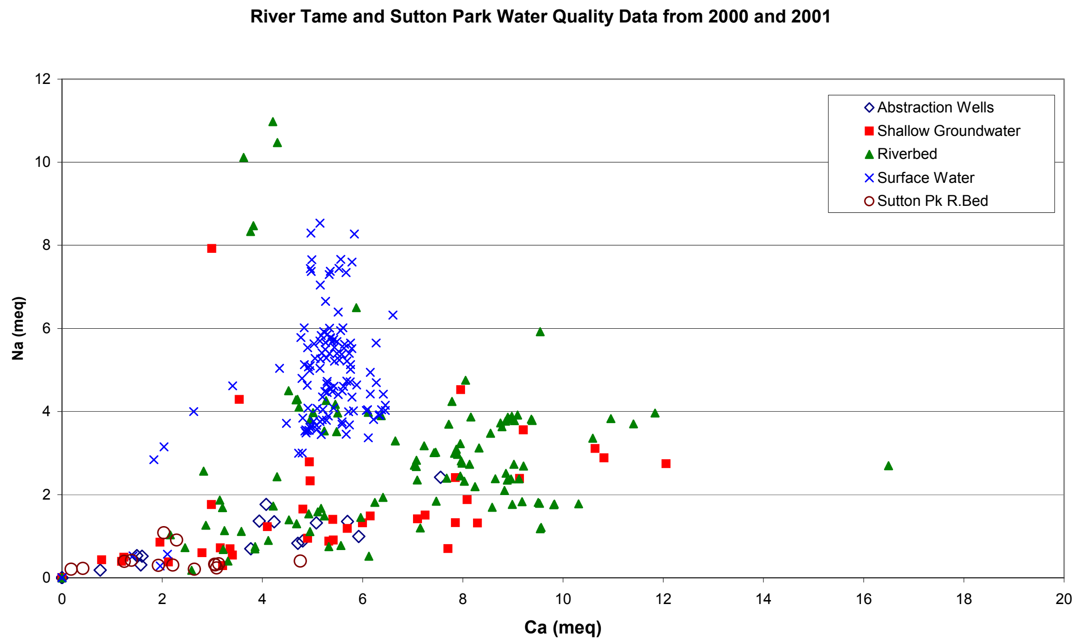
anthropogenic sources. Variations in upstream discharges cause large temporal fluctuations (>20 mgl^{-1} in 1.5 hours, 11/5/01) in surface water calcium concentrations that mask any change that may occur due to groundwater discharge, either from the Triassic Sandstone aquifer, or from the Carboniferous strata.

The quality of waters discharging from opposite banks of the river may differ markedly, as in the case of Profile 5 (at 0 km, Figure 7.11a) located in parkland on the Carboniferous-aquifer boundary. High levels of calcium were detected discharging from the northern bank and low calcium concentrations from the south. The high calcium levels were associated with elevated levels of trichloroethene (TCE) that were unexpected in the park land. An industrial site thought to be the source was located from aerial photographs <500 m up gradient from the river. Many cases were identified in which elevated calcium was associated with other contaminants, within acidic plumes from industrial sites, or as leachate from made ground.

Sodium displays a similar trend to calcium, increasing in concentration from a mean of 23.8 mgl^{-1} in the abstraction wells to 70.9 mgl^{-1} in the shallow and 90.6 mgl^{-1} in the riverbed piezometers. The increase is associated with anthropogenic sources such as NaCl which is widely applied as a winter road de-icing treatment. Locally high levels have been associated with industrial sites, with a maximum recorded level of 704 mgl^{-1} in the riverbed. Surface water concentrations (mean 108.8 mgl^{-1}) exceed groundwater values as a result of the extensive use of NaCl in the treatment of sewage which is subsequently discharged to the river. An increase in the sodium content of the riverbed piezometers is observed across the aquifer (Figure 7.11b), probably related to changes in land use. Any change in the surface water concentration profile due to the groundwater input was obscured by temporal variations in the upstream flux. Magnesium and potassium (Figure 7.11c) display a similar increasing trend across the aquifer.

Good evidence of the imported origin of the surface water is seen by the contrast with the groundwater for the Na, Ca, Cl and SO_4 data (Figure 7.12a and b). A relationship is apparent in all the types of urban groundwater sampled between the concentrations of sodium and calcium in the approximate milli-equivalent ratio 1:4 (Figure 7.12a). This may be due to the fact that contamination by both NaCl and $CaSO_4$ often occur together at a site and therefore higher levels of contamination have a proportionate increase in both Na and Ca concentrations. The ratio of chloride to sulphate shows a similar relationship (Figure 7.12b) but with a higher degree of scatter between the points. The closer grouping of data points for the calcium/sodium relationship may indicate that additional controls such as ion-exchange are taking place. The Na:Ca ratios show a fairly distinct divide between samples of groundwater and surface water with little overlap, implying a limited extent to the groundwater/surface water mixing zone. A greater overlap is observed between groundwater and surface water concentrations for chloride and sulphate. This may be coincidental, or it may imply that surface water influent to the groundwater system has undergone a decrease of the Na/Ca ratio via ionic exchange, with the relatively conservative Cl/SO_4 ratio remaining unchanged. However, the large heterogeneity in the system, and in the sample types and depths, introduces a high degree of uncertainty and makes it difficult to draw any valid conclusions on this.

(a)



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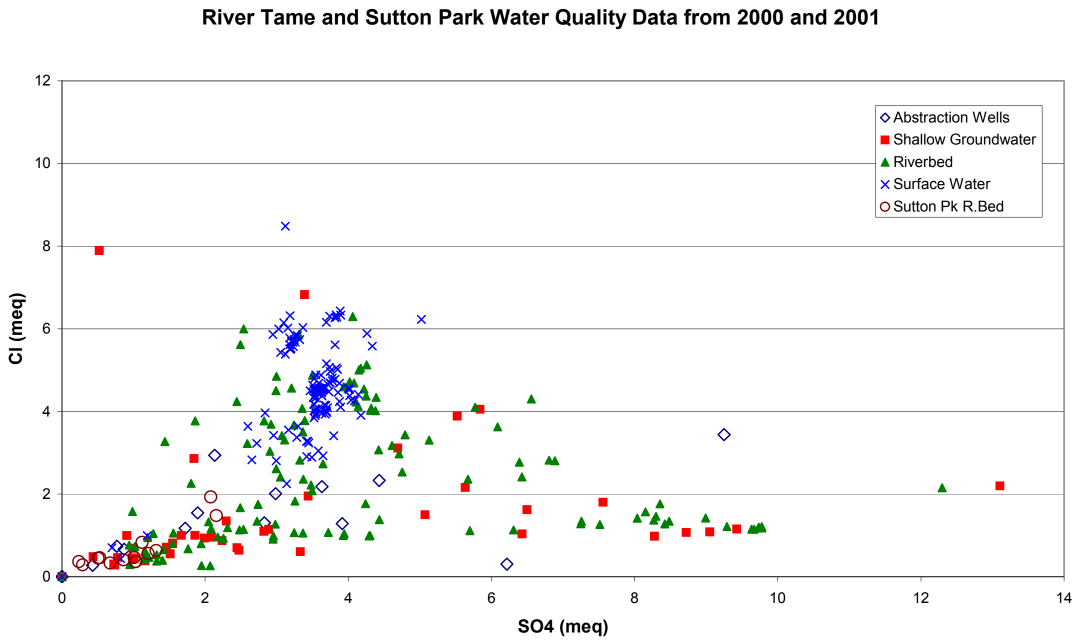


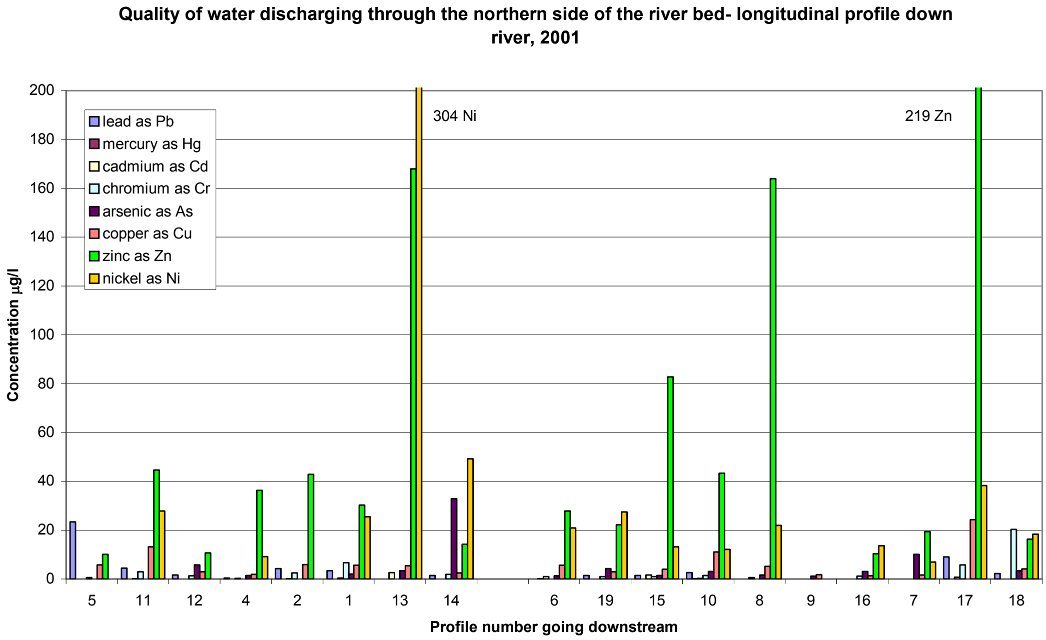
Figure 7.12 Comparison of hydrochemical data for (a) Na and Ca (b) Cl and SO_4

7.5 Toxic Metals

Multiple sources of minor and heavy metals are present across the aquifer associated with industrial manufacturing and metal working industries. Despite this, metal concentrations in the groundwater remain low, apart from localised high level areas. This is mainly attributed to the large adsorption capacity of the iron oxyhydroxide grain coatings within the aquifer which has limited the solubility and transport of the minor and heavy metals (Ford, 1990). Metals concentrations from the deep borehole beneath the city centre thought to be representative of natural background levels were Hg 0.2, Cr 1, As 5, Cu 3, Zn 5, Fe 36, - μgl^{-1}, with Pb, Ni, Mn, Cd, Al below detection limits. Levels of metals concentrations are generally greater at shallower depth in the groundwater system. Between the abstraction wells and the riverbed piezometers mean concentrations increase: from 6 to 12 μgl^{-1} for Cu, 46 to 81 μgl^{-1} for Zn and from 10 to 44 μgl^{-1} for Ni. Mean values in the surface water (from 3 samples) are Cu 12 μgl^{-1}, Zn 58 μgl^{-1} and Ni 56 μgl^{-1}. It is believed that the groundwater is unlikely to make a noticeable impact on the surface water concentrations under present conditions. However, locally high levels of metals are observed in groundwater discharging through the riverbed. Maximum values from the riverbed piezometers in micrograms per litre were Mn 11000, Fe 24000, Al 33000, Pb 158, Hg 0.2, Cd 3, Cr 20, As 33, Cu 97, Zn 429, and Ni 304 μgl^{-1}. High levels of cadmium (80 μgl^{-1}) were detected in the riverbank piezometer at profile 13, in association with the maximum value of 3 μgl^{-1} found in the riverbed.

The distribution of heavy metal concentrations in the groundwater discharging through the riverbed displays large heterogeneity both along the reach and between the opposing banks (Figure 7.13 a and b). Profiles 11 and 12 are located where the study reach is bordered by

(a)



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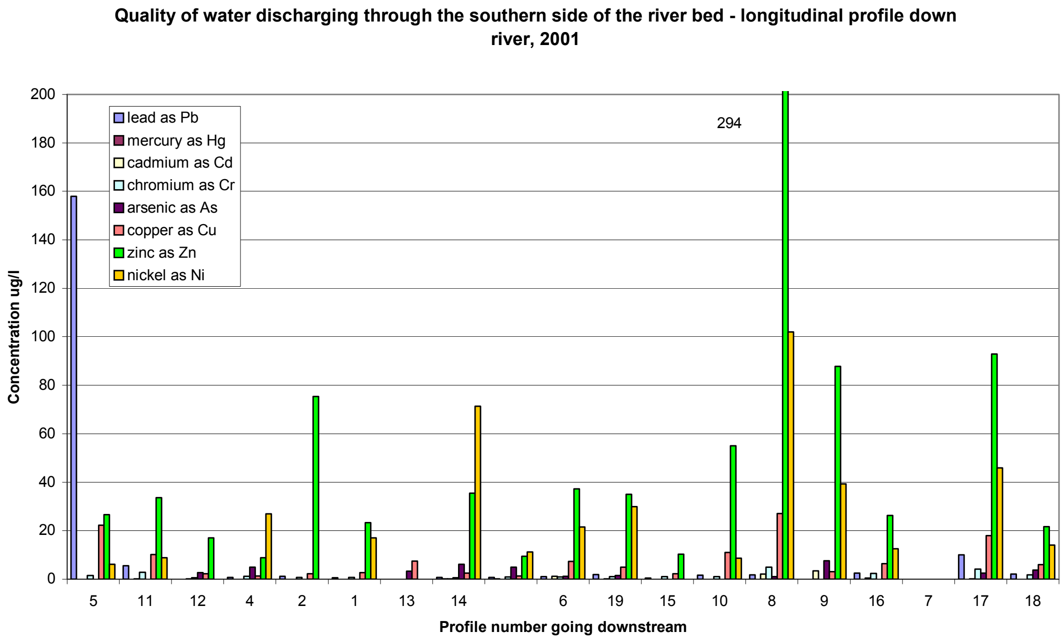


Figure 7.13 Heavy metal concentrations in groundwater discharging through the (a) North and (b) South banks of the study reach.

parkland to > 500 m from the river and may be considered to give background urban metal concentrations. Similar low levels of metal concentration across the aquifer in more industrialised areas indicate that the impact of industry on the levels of heavy metals in the groundwater is perhaps lower than expected. Taking into account both sides of the river nine of the 18 profiles did not display concentrations above the background. Movement of heavy metals across the groundwater-surface water interface was investigated at a particular site by the use of multilevel sampling techniques. The results of this work will be discussed in the next section in conjunction with other water quality data from the same plume.

7.6 Discharge across the groundwater-surface water interface of a multi-component contaminant plume

Groundwater plumes located using the riverbed piezometers were sometimes found to contain a cocktail of several different contaminant types, both inorganic and organic, that may be associated with metals-related industry. Such was the case for Profile 8, where an acidic plume was found discharging from one bank, with elevated levels of nickel, zinc, copper, aluminium, fluoride, sulphate, nitrate, TCE and TCA. A combination of data is presented to show on a local scale the processes occurring during the discharge of the contaminant plume across the riverbed. The interpretation of the multilevel data is based on upward vertical flow as indicated by the 2D flow modelling of Profile 8, and field measurements that indicate heads of 1-4 cm in all the piezometers relative to DWF river levels. However, there is a degree of uncertainty as heterogeneity within the sediments and the bed forms may lead to some degree of lateral flow both inwards from the bank and down river.

High concentrations of fluoride and aluminium form a plume that discharges over a discrete area on the western side of the channel (Figure 7.14). The area of discharge extends 4 m from the riverbank with the highest concentrations occurring closest to the riverbank. Water of different composition was observed discharging across the central and eastern sections of the river reflecting the convergence of flow paths of different origins and travel times (Figure 7.15a to f). The samples within each profile display relatively uniform composition, reinforcing the assumption of upward vertical flow. Some variations are apparent in the shallower piezometers due to mixing with the surface water, e.g. the increase in the chloride content in the shallowest sample on the left bank.

The ratio of the major determinands varies between the three multilevel piezometers across the channel (Figure 7.16a,b,c). A clear difference in water chemistry is visible between the western profile and the central and eastern profiles which are similar. The content of Mg, NO_3 and SO_4 is higher and Na and Cl lower in the western profile compared with the eastern and central profiles. Similarities exist between the riverbank piezometers and the groundwater discharging through the adjacent bank, but the water types are not identical. This may reflect mixing between shallow and deep groundwater in the riverbed samples or heterogeneity in the contaminant concentrations. The content of Ca, Mg, NO_3, SO_4 (Figure 7.16b,c) shows a general decrease from higher values in the western riverbank piezometer to lower values in the eastern riverbed piezometers, with the central riverbed piezometers possibly representing a mixture between these two end members. The data on Na and Cl are more difficult to interpret. Flow modelling indicates the central profile to be sampling the discharge of older, deeper groundwater, and the hydrochemical similarities with the eastern profile may represent a high degree of mixing with the shallower groundwater. Flow modelling also indicates the

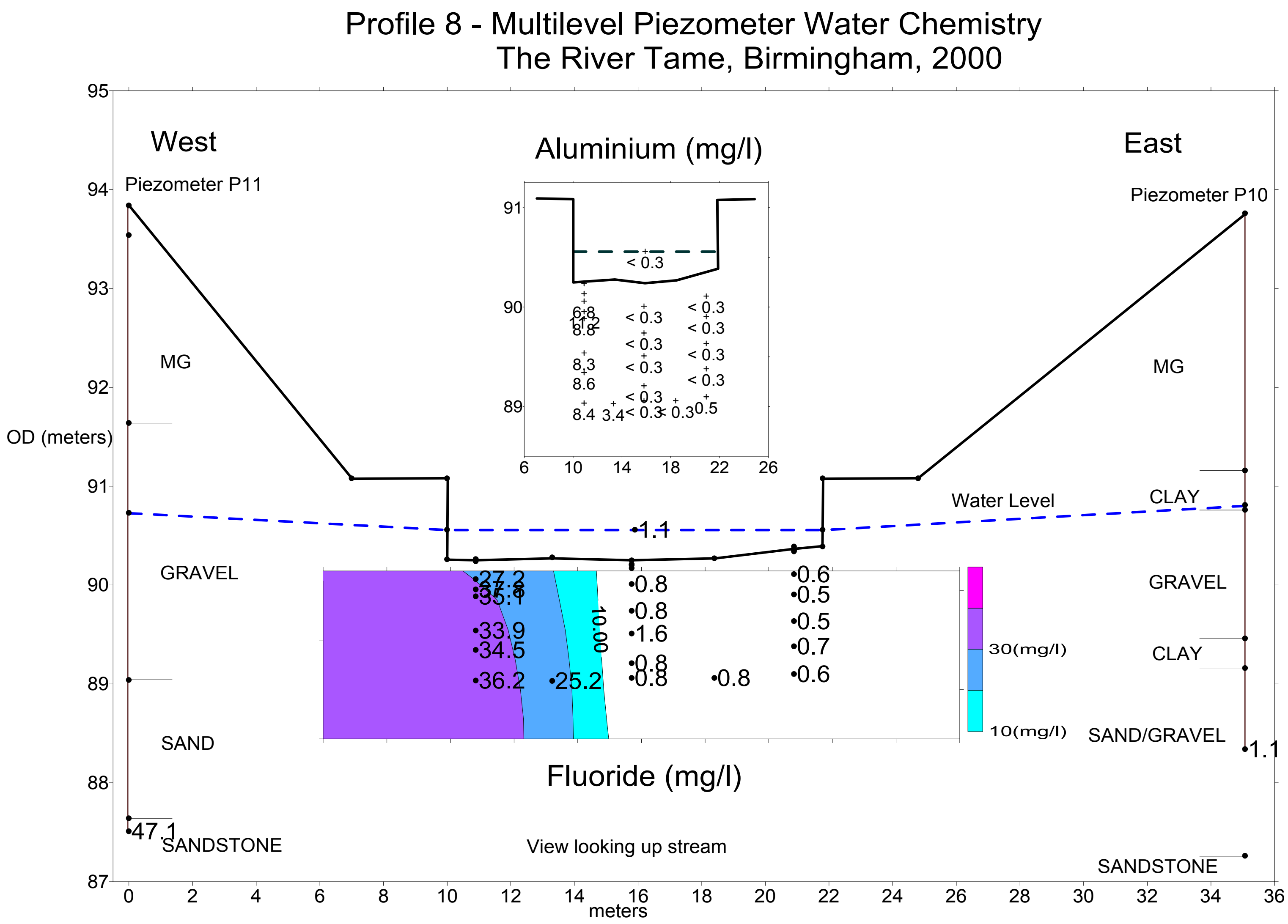


Figure 7.14 Cross section through a groundwater plume containing Al and F that is discharging to the river.

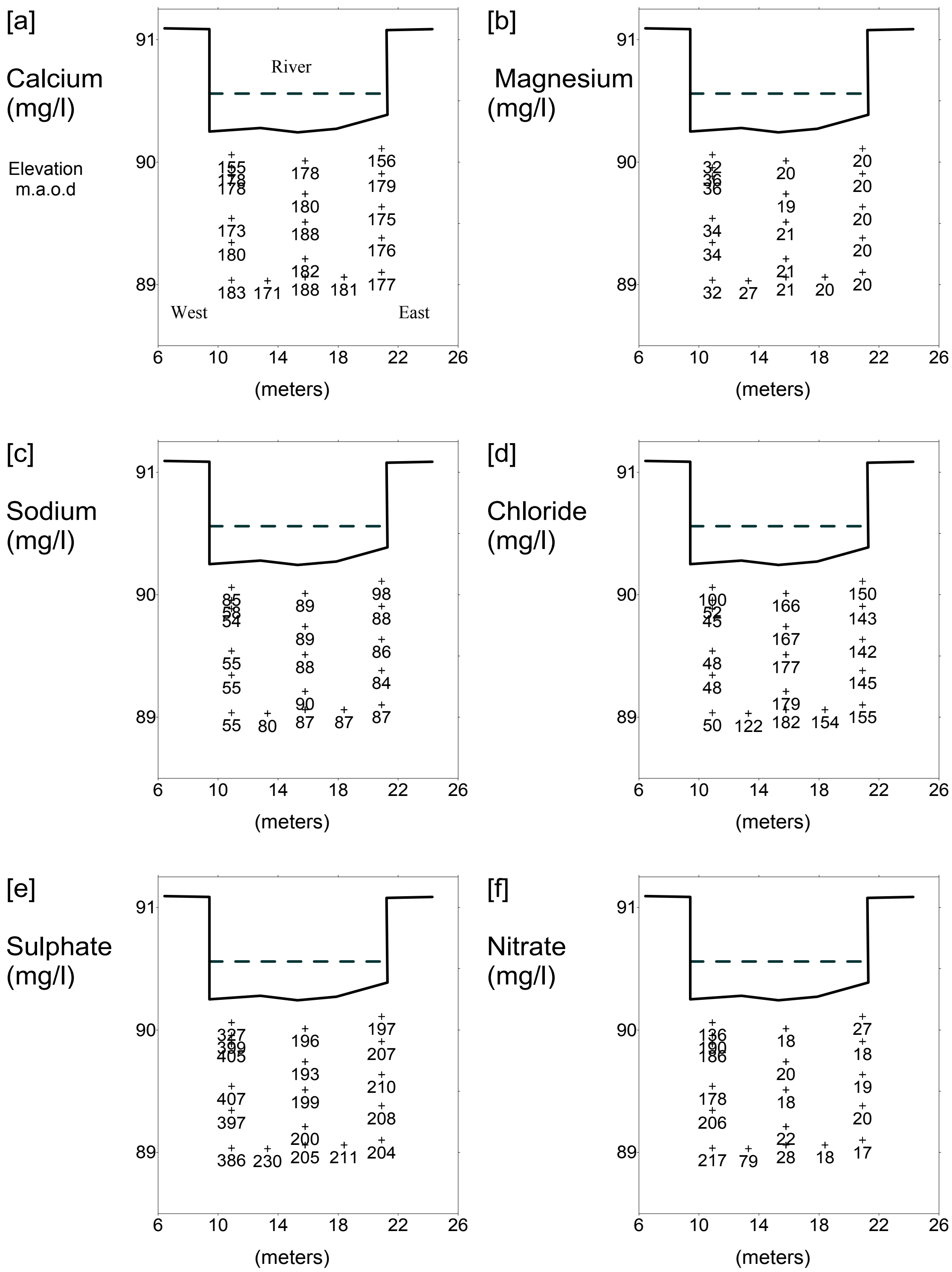


Figure 7.15 Profile 8 water quality data cross sections, summer 2001, for (a) Ca (b) Mg (c) Na (d) Cl (e) SO_4 (f) NO_3.

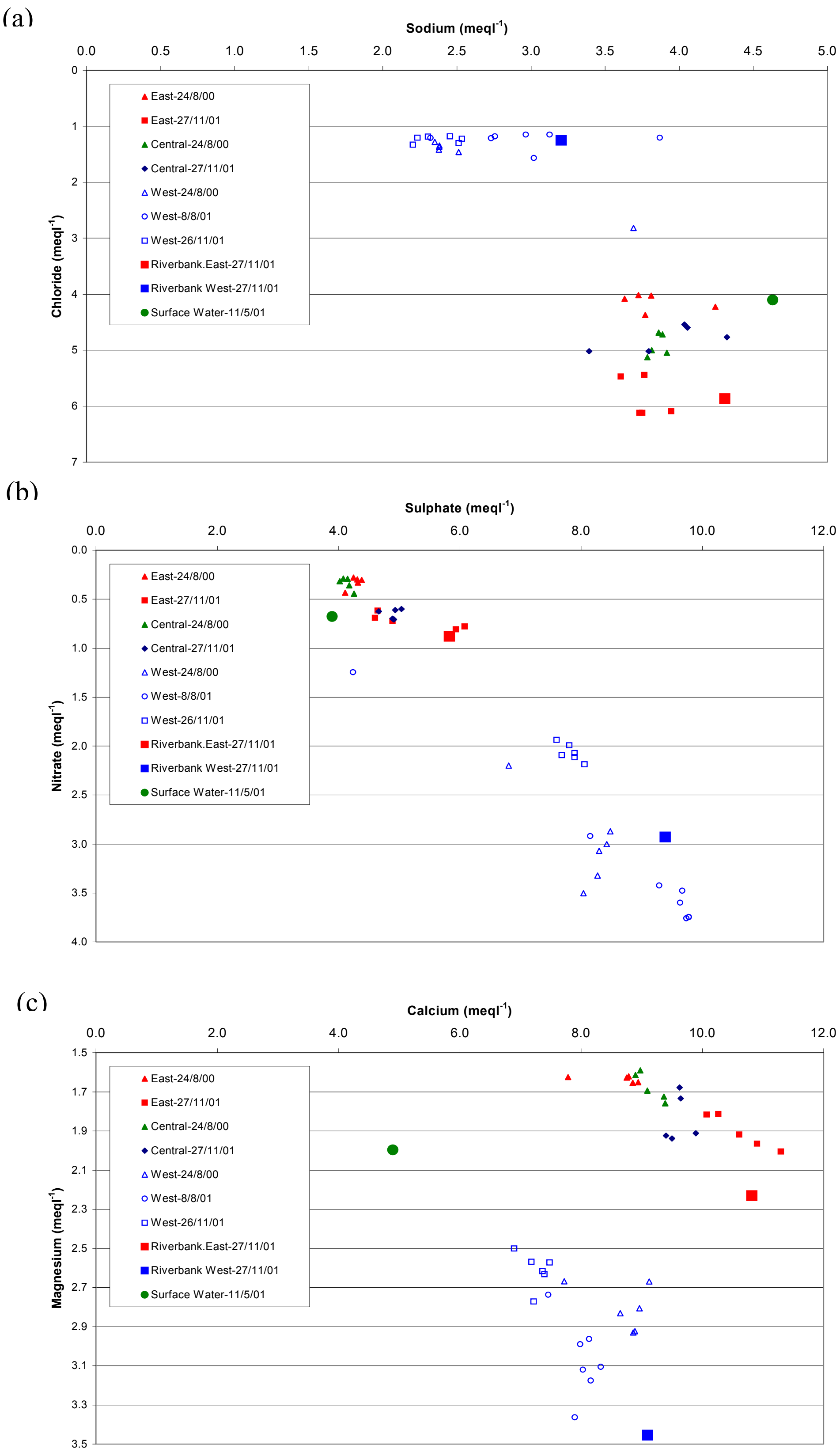


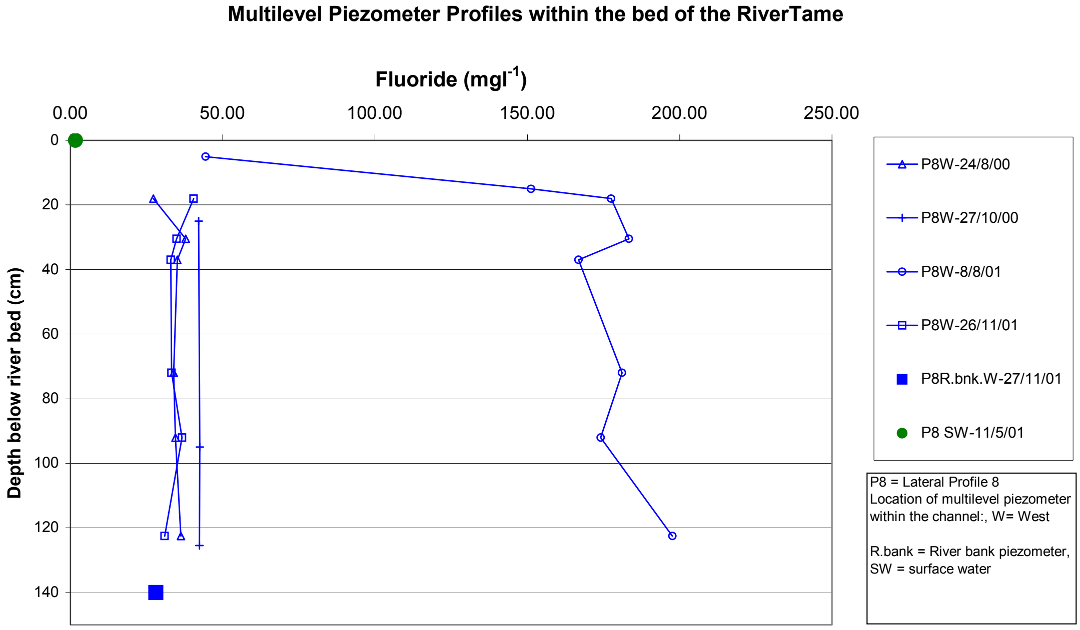
Figure 7.16 Multilevel piezometer data from Profile 8 showing concentrations of (a) Na and Cl (b) NO_3 and SO_4 (c) Mg and Ca.

greatest proportion of the groundwater that is discharging is coming from the eastern side of the channel. Evidence of this may be the limited extent of the fluoride plume on the west compared to the dominance of the similar water types from the east.

All the different water types display a degree of temporal variability, the central profile remaining the most constant. Temporal variations in the plume are significant and are most marked in the changes of fluoride and aluminium concentrations (Figure 7.17 a and b). These vary from a mean of F 35 mgl^{-1}, Al 8 mgl^{-1} (summer 2000) to F 180 mgl^{-1}, Al 32 mgl^{-1} (summer 2001). A change is not detected in the area of discharge (Figure 7.18 a) which remains between 3.3 and 5.8 m from the riverbank (2nd and 3rd sampling points). Geochemical modelling (Tellam, pers. comm) indicates the aluminium to be soluble as a complex with fluoride. The solubility of aluminium is pH dependent and the high concentrations detected in the summer of 2001 are coincident with a decrease in pH to more acidic conditions from ~6.4 to ~5.8. The source of the fluoride may be from the use of hydrofluoric acid or the mineral cryolite (Na_3AlF_6) as a flux within the metals industry. The aluminium may also be derived from industrial sources or released from the ubiquitous aluminium silicate minerals.

The discharge area of the plumes containing sulphate, nitrate and heavy metals show a similar distribution to the aluminium and fluoride. Chloride (Figure 7.18 b) shows an inverse correlation and copper (Figure 7.18 c) a positive correlation with the aluminium. Data collected on the 8/8/01 from the western multilevel profile are presented (Figure 7.19a to f) to illustrate the processes that may alter contaminant concentrations within the groundwater during vertical flow across the groundwater – surface water interface. The groundwater flows into the profile at 1.25 m depth

(a)



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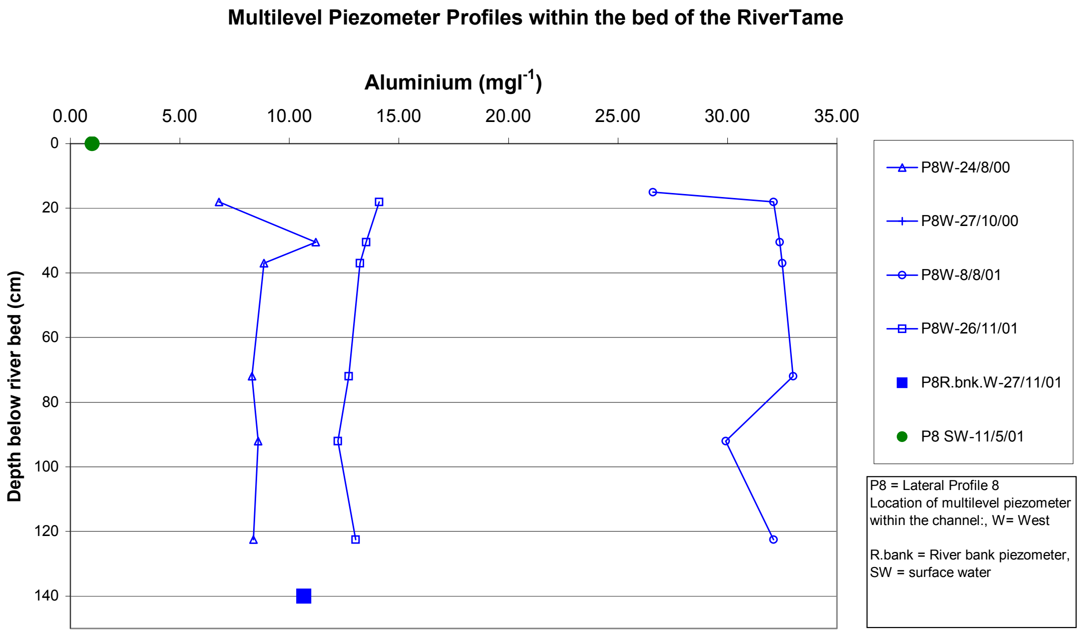


Figure 7.17 Vertical concentration profiles for (a) F and (b) Al at Profile 8

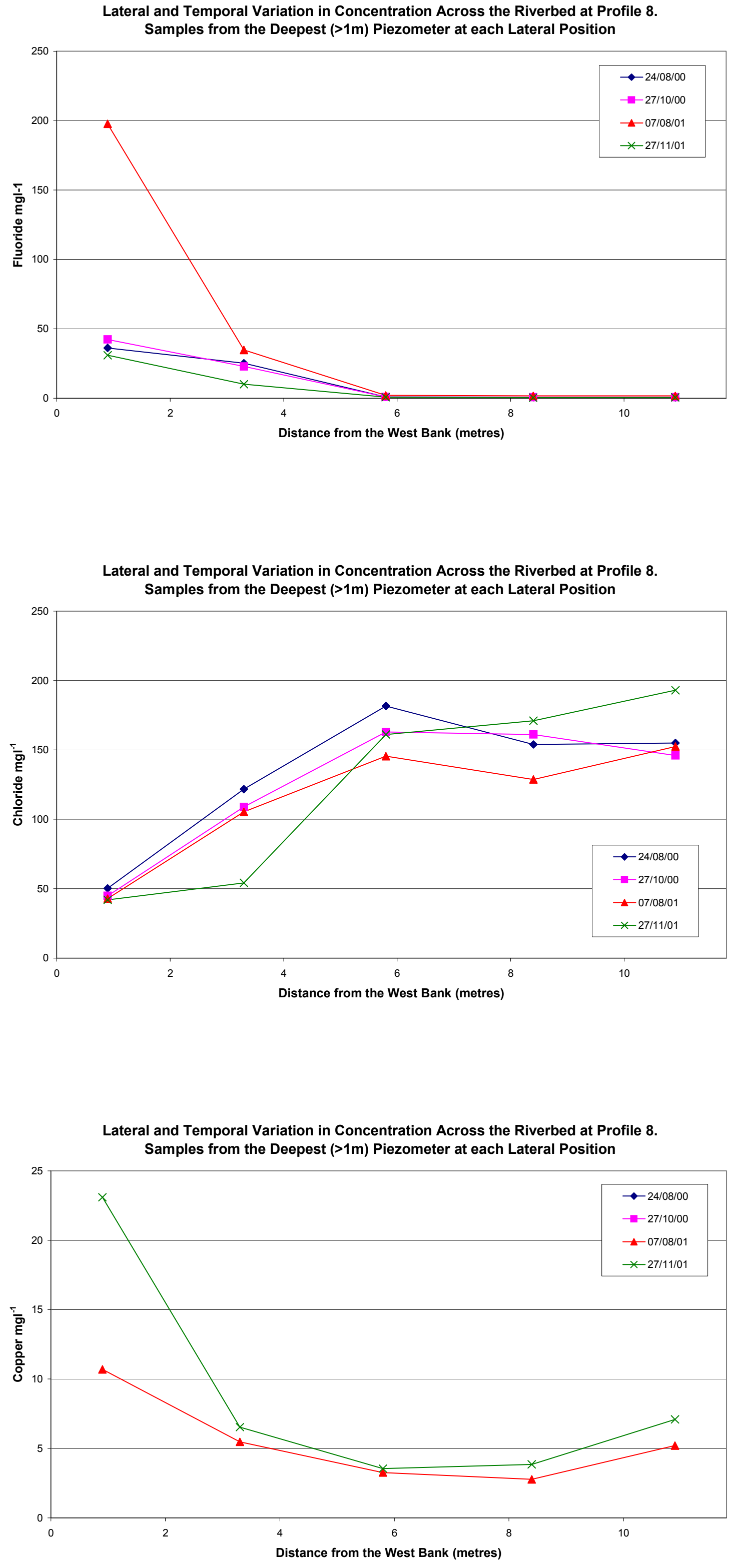


Figure 7.18 Temporal variation in the lateral concentration distribution of (a) F (b) Cl (c) Cu.

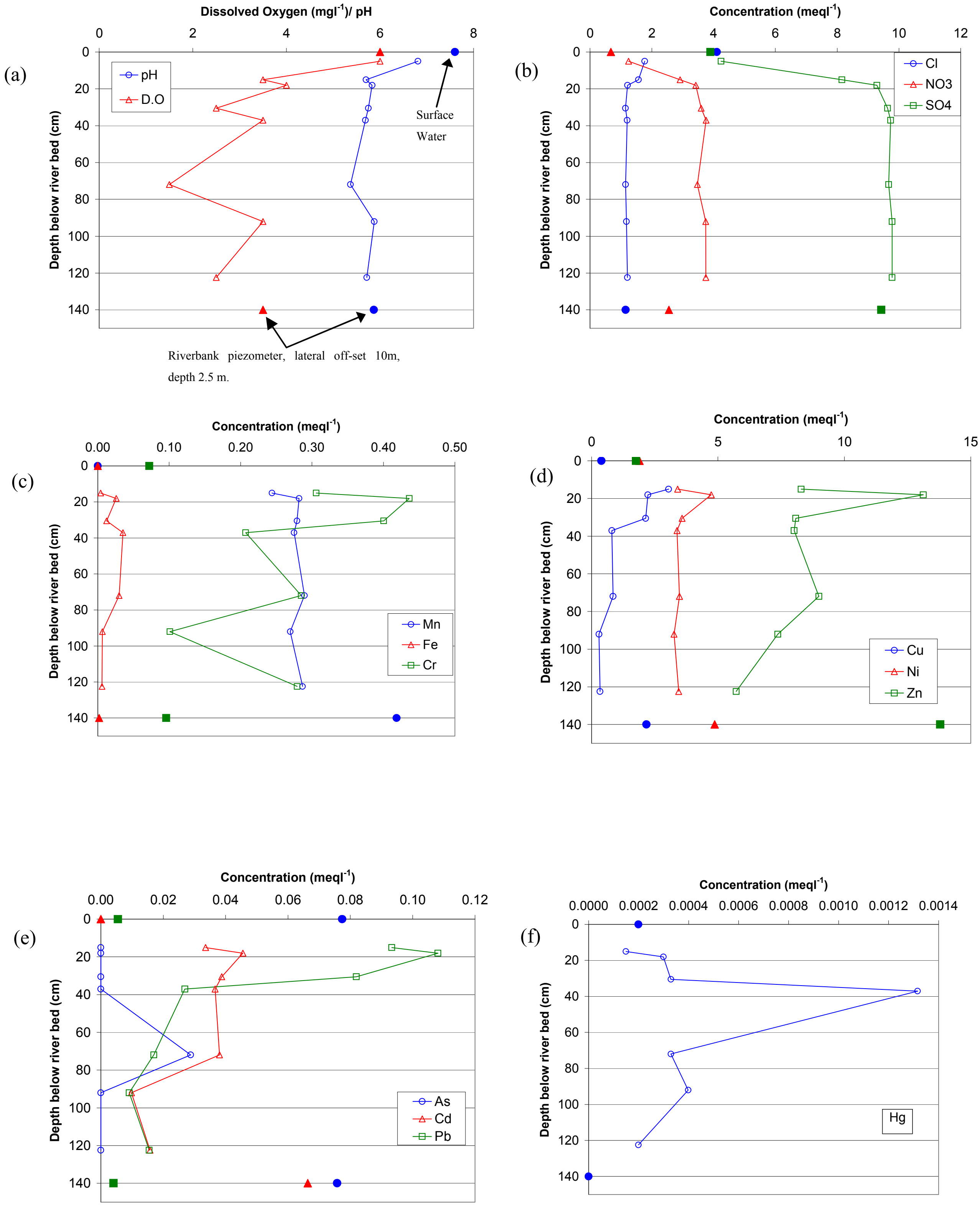


Figure 7.19 Water quality results (8/8/01) from the western multilevel riverbed piezometer at Profile 8.

with a dissolved oxygen (DO) content of 2.5 mgl^{-1} which increases rapidly to 6 mgl^{-1} across the top 30 cm due to mixing and diffusion from the oxygenated surface water. The pH of the influent groundwater is 5.7 which exhibits minor fluctuations (<0.3) before increasing rapidly to 6.8 within 5 cm of the riverbed, approaching the surface water value of 7.6. Chloride values remain fairly constant throughout the profile at 1.2 $meql^{-1}$ before increasing across the upper 18 cm indicating mixing with the surface water. The chloride concentration indicates a 13% content of surface water mixed with the groundwater at a depth of 15 cm . Sulphate concentration remains relatively constant at 9.8 $meql^{-1}$ until mixing with surface water occurs and there is no evidence of sulphate reduction. Influent groundwater has nitrate concentrations of 3.8 $meql^{-1}$ which decreases at 72 cm to 3.5 $meql^{-1}$ and then consistently from 3.8 $meql^{-1}$ at 37 cm to 1.2 $meql^{-1}$ at 5cm below the riverbed. Dilution by surface water (13%) may be responsible for some of the decrease but other factors such as microbial reduction within the top 40 cm of the riverbed must also be at work. Low levels (<0.004 $meql^{-1}$) of nitrite support this (only analysed for 27/11/01 samples). Oxygen levels do not indicate the aneorobic conditions usually required for nitrate reduction. However, anerobic conditions may have occurred across a relatively small depth interval and then been mixed with oxygenated surface waters. Further work is required to obtain better sample resolution across the top 20 cm of riverbed. Manganese shows a relatively uniform concentration of 0.29 $meql^{-1}$ across the profile until it drops to 0.24 $meql^{-1}$ at 15 cm due to dilution with surface water and/or precipitation due to increasing pH. Previous workers (Harvey et al., 1998) have indicated that co-precipitation of heavy metals may occur, with manganese providing an important sink at the groundwater-surface water interface. Iron shows an increase from influent levels of 0.006 $meql^{-1}$ to 0.035 $meql^{-1}$ at 37 cm returning to low concentrations of 0.004 $meql^{-1}$ at 15 cm depth. This indicates dissolution of iron oxide/hydroxide coatings on the sediments and then re-precipitation upon mixing with surface water and an increased pH.

All the heavy metals exhibit similar trends in concentration over depth (Figure 7.19c,d,e,f), with influent groundwater levels increasing and then generally falling in the groundwater surface water mixing zone. The depth interval over which the changes occur vary between the metals: copper, nickel, zinc and lead concentrations rise from 40 cm and start dropping from 15 cm depth with the exception of copper which continues to rise. Cadmium shows a rise from 72 cm and mercury concentrations peak at 37 cm before dropping steadily towards the surface water interface. The rises in concentrations are likely to be due to either disolution or de-sorption from the riverbed sediments. Both controlling processes are dependent on the pH and redox conditions which appear to be similar, both at depth and in the zone of increased metal concentration. This suggests that a source of solid/sorbed material is present that does not exist at depth, perhaps a layer of metal-rich sediment washed down from industrial activities upstream. High values for mean total copper (42 μgl^{-1}) zinc (139 μgl^{-1}) and nickel (126 μgl^{-1}) values have been recorded in monthly Environment Agency surface water sampling at the downstream boundary of the aquifer; approximately 50% of the copper was in particulate form (Appendix 2).

The drop in concentration values that takes place for Zn, Ni, and Cr, between depths of 18 cm and 15 cm is too great to be explained by a dilution of 13%, and sorption or precipitation reactions are the likely explanation. Under pH conditions of 6.8 at 15 cm depth, copper, nickel and zinc are likely to remain soluble despite the increasing oxygen content. Chromium as Cr^{3+} is slightly mobile under acidic conditions and with a rise in pH and a change to oxidising conditions, is likely to convert to the very soluble Cr^{6+}. Therefore, it can be deduced that sorption is the likely explanation for the drop in concentrations. Sorption may occur onto clays, organic material and iron and manganese oxyhydroxide coatings present within the sediment. The sorption is pH dependent and the pH_{pzc} (pH point of zero charge) for the iron oxyhydroxide goethite is 7.3, above which pH metals are rapidly adsorbed. This point lies within the pH transition across the groundwater/surface water interface suggesting that goethite sorption maybe a significant control on metal flux to the river. The acidic plume appears to be mobilising metals from the sediments and transporting them to the groundwater/surface water interface where they are rapidly sorbed onto grain coatings. The metals may then re-enter the river as the coatings on fine particles that are stripped from the riverbed into suspension during high flow events

7.7 Organic Water Quality

The urban groundwater and surface water systems show widespread contamination by volatile organic compounds (VOCs) though generally at a low level (Table 7.5).

The highest levels of contamination were associated with the abstraction wells, which contained the maximum total VOC content of 879 μgl^{-1}. This is in contrast to the inorganic data that show lower levels of contamination at depth. The most frequently detected contaminants (Table 7.6, Figure 7.20) were derived from dense nonaqueous-phase liquids (DNAPLs) and the physical-chemical properties of these compounds may explain the high levels of contamination at depth. Driven by the density contrast and often low viscosity in relation to water, the pure phase DNAPLs may have penetrated far below the water table into the aquifer to form a long-term source of dissolved phase DNAPL over a considerable vertical extent. An additional reason for the high levels of VOCs detected in the abstraction wells is that many are associated with industries that use VOCs on site. The use of these compounds is specialised and generally restricted to these locations rather than the more widespread distribution of the inorganic contaminant sources such as chloride.

Sample Type	Number of samples	Samples in which VOCs were detected (%)	Samples in which total VOC content < 10 μgl^{-1} (%).	Mean Total VOC content (μgl^{-1})	Maximum Total VOC content (μgl^{-1})
Abstraction Wells	14	79	50	114	879
Shallow Groundwater	20	60	95	4	70
Riverbed Piezometers	44	82	79	13	210
Surface Water	3	100	67	8	11

Table 7.5 The level of total VOC contamination in the aquifer (analysis was for a range of 52 VOCs)

The most frequently detected compounds were Trichloroethene (TCE), cis 1,2-Dichloroethene (DCE), Tetrachloroethene (PCE), Trichloromethane (TCM, chloroform), 1,1-Dichloroethane (DCA), and 1,1,1-Trichloroethane (TCA). This is in agreement with results from the sampling of abstraction wells in the Birmingham Aquifer undertaken by previous workers (Rivett, 1990 and Taylor et al., 1999). Both TCE and TCA are widely used for metal degreasing by industry in the Tame Valley and the primary use of PCE is in dry cleaning. TCM is a by-product of water treatment processes and may indicate leakage from water mains or sewer pipes. TCM

Water Quality Data from the Tame Valley, 2001	Detection Limits µg/l	Total > Det	Abstraction Wells						Shallow Groundwater						Riverbed Piezometers						Surface Water					
			Mean *	max	min	Median	N > Det	N	Mean *	max	min	Median	N > Det	N	Mean *	max	min	Median	N > Det	N	Mean *	max	min	Median	N > Det	N
Analysed at the University of Birmingham																										
Trichloroethene (TCE)	<0.1	70					0	0	5.9	11.6	0.1	5.9	2	2	16.9	50.5	18.9	24.3	4	7	1.8	6.7	0.3	1.4	64	64
Trichloromethane (TCM)	<0.1	66					0	0	1.0	1.9	1.9	1.9	1	2	0.5	1.5	0.1	0.7	4	7	1.3	7.3	0.2	0.9	61	63
Tetrachloroethene (PCE)	<0.1	66					0	0	0.1	0.2	0.2	0.2	1	2	0.1	0.2	0.1	0.1	2	7	2.3	20.0	0.2	1.7	63	64
Tetrachloromethane (CTC)	<0.1	55					0	0	0.1	0.0	0.0		0	2	0.1	0.0	0.0		0	7	0.1	0.5	0.0	0.1	55	62
Trichloroethane (TCA)	<0.1	50					0	0	4.1	8.0	8.0	8.0	1	2	7.1	15.0	7.8	13.4	4	7	0.3	1.2	0.0	0.3	45	63
Analysed at the Environment Agency Laboratory in Leeds																										
Trichloroethene (TCE)	<0.1	47	36.0	340	0.6	10.1	9	14	1.5	28.0	0.1	0.5	4	20	10.7	64.0	0.1	0.6	31	44	1.4	2.7	0.7	0.8	3	3
cis-1,2-Dichloroethene (DCE)	<0.2	35	9.0	110	0.8	5.8	5	14	0.6	6.8	0.8	3.8	2	20	2.6	24.0	0.3	1.9	25	44	1.4	2.3	0.6	1.3	3	3
Tetrachloroethene (PCE)	<0.1	27	53.2	700	0.1	14.0	5	14	0.2	0.8	0.1	0.4	4	20	0.3	2.8	0.1	0.2	16	44	0.6	0.8	0.8	0.8	2	3
Trichloromethane (TCM)	<0.2	25	1.5	13.0	0.3	0.6	7	14	1.7	9.2	0.2	4.6	5	20	1.5	11.0	0.3	3.1	12	44	0.8	0.8	0.8	0.8	1	3
1,1-Dichloroethane (DCA)	<0.2	22	0.2	0.3	0.2	0.3	2	14	0.5	5.0	0.5	2.7	2	20	1.3	25.0	0.2	0.5	18	44					0	3
1,1,1-Trichloroethane (TCA)	<0.2	16	0.3	1.2	0.4	0.5	4	14	1.3	22.0	22.0	22.0	1	20	9.2	110.	0.3	44.0	10	44	0.2	0.3	0.3	0.3	1	3
Bromo-dichloromethane	<0.2	12	0.6	4.0	0.6	1.0	3	14	0.3	2.2	0.3	0.5	3	20	0.3	0.9	0.6	0.7	6	44					0	3
Vinyl Chloride	<0.2	10	0.8	1.9	0.2	1.1	2	3	0.5	0.7	0.7	0.7	1	2	0.6	1.0	0.8	0.9	7	11					0	0
1,1-Dichloroethene (DCE)	<0.3	10	0.3	0.9	0.9	0.9	1	14	0.3	1.1	1.1	1.1	1	20	0.6	8.6	0.4	1.5	8	44					0	3
trans-1,2-Dichloroethene (DCE)	<0.2	8	0.2	0.8	0.8	0.8	1	14	0.2	0.4	0.4	0.4	1	20	0.3	3.0	0.3	0.3	6	44					0	3
MTBE	<0.3	4					0	14					0	20	0.3	0.5	0.3	0.4	4	44					0	3
1,2-Dichloroethane (DCA)	<1	4	1.0	1.2	0.3	0.7	2	14					0	20	1.0	0.6	0.2	0.4	2	44					0	3
1,4-Dichlorobenzene	<0.2	3					0	14					0	20	0.2	0.6	0.5	0.5	3	44					0	3
Tetrachloromethane (CTC)	<0.2	2	0.3	0.6	0.5	0.6	2	14					0	20					0	44					0	3
Toluene	<4	1					0	14				0.3	1	20					0	44					0	3
1,2-Dichlorobenzene	<0.3	1					0	14					0	20	0.3	0.6	0.6	0.6	1	44					0	3
Benzene	<0.1	1	0.1	0.2	0.2	0.2	1	14					0	20					0	44					0	3
1,1,1,2-Tetrachloroethane	<0.2	1	0.2	0.2	0.2	0.2	1	14					0	20					0	44					0	3
1,1,2-Trichloroethane (TCA)	<0.2	1	0.2	0.4	0.4	0.4	1	14					0	20					0	44					0	3
Chlorobenzene	<0.4	1	0.4	0.9	0.9	0.9	1	14					0	20					0	44					0	3
Dichloromethane	<2	1					0	14					0	20					0	44	4.4	9.2	9.2	9.2	1	3

Not Detected									
1,1,1,2-Tetrachloroethane	<0.2	1,3,5-Trimethylbenzene	<10	Bromobenzene	<0.4	Dibromomethane	<0.2	n-Butylbenzene	<0.2
1,2,3-Trichloropropane	<0.2	1,3-Dichlorobenzene	<0.2	Bromochloromethane	<0.3	Ethenylbenzene(styrene)	<0.1	n-Propylbenzene	<0.1
1,2,4-Trimethylbenzene	<10	1,3-Dichloropropane	<0.2	Bromoform	<0.2	Ethylbenzene	<0.1	O-Xylene	<0.1
1,2-Dibromo-3-Chloropropane	<0.6	2-Chlorotoluene	<0.1	Bromomethane	nr	Iso-propylbenzene	<0.1	sec-Butylbenzene	<0.1
1,2-Dibromoethane	<0.2	2,2-Dichloropropane	<0.4	Chloro-dibromomethane	<0.2	M-P-Xylene	<0.3	tert-Butylbenzene	<0.2
1,2-Dichloropropane	nr	4-Chlorotoluene	<0.1	cis1,3-Dichloropropene	<0.2	Naphthalene	<100	Trichlorofluoromethane	<0.3
		4-Isopropyltoluene	<0.1						

*Mean includes none detects equal to the detection limit

Table 7.6 Summary of organic water quality data from the Tame Valley, 2001.

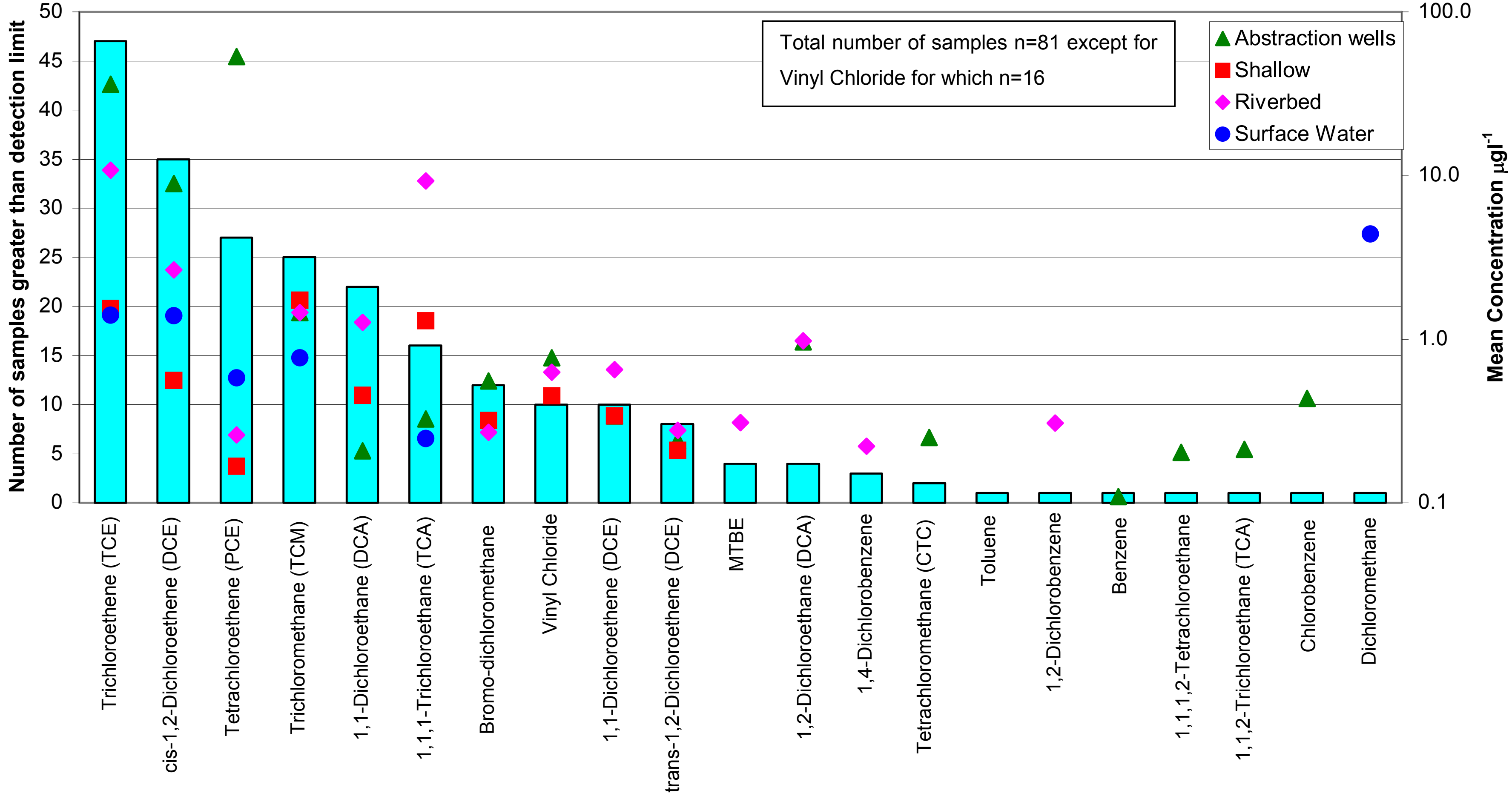


Figure 7.20 Frequency of VOC detection and mean concentrations in groundwater and surface water in the Tame Valley, 2001. (Analysed at the Environment Agency Laboratory in Leeds).

results from the interaction of chlorine with humic substances, which may be present due to the origins of the public supply from Welsh upland sources. The compound cis 1,2-DCE has no industrial use and is formed following the breakdown of TCE through biodegradation. The compound 1,1-DCA has some industrial use as a solvent and is also a breakdown product of 1,1,1-TCA.

Many of the compounds detected are regarded as toxic carcinogens and EC drinking water guide levels have been set at 1 μgl^{-1} for TCE, TCA, PCE and TCM and UK drinking water standards for TCE and TCM at 30 μgl^{-1}. The UK drinking water limits for TCE are exceeded four times (maximum 340 μgl^{-1}) in the abstraction wells including two of the three wells within 500 m of the river. TCE and TCA only exceeded 30 μgl^{-1} in two of the riverbed piezometer profiles. These lay within 120 m of each other and were most likely within the same plume. The only significant shallow groundwater VOC concentrations (TCE 28 μgl^{-1}, TCE 22 μgl^{-1}) were also found in this area. The maximum concentration of PCE (700 μgl^{-1}) occurred in an abstraction well within 100 m of the river. However, only trace amounts of PCE were detected in the two adjacent riverbed piezometer profiles. There may be several reasons for this:

1. the plume is narrow and discharges between the 2 profiles which are 300 m apart;
2. continued abstraction restricts contaminant movement off site;
3. the plume is at depth in a slow moving/stagnant groundwater system that is not discharging to the river; and
4. the direction of plume migration is away from the river in this area across the adjacent flood plain as indicated by the numerical flow modelling.

The only other VOC that occurred at a high level was the PCE/TCE biodegradation product cis 1,2-DCE, with a maximum concentration of 110 μgl^{-1} detected in the same abstraction well as the maximum PCE content. The other VOCs occurred infrequently and at low levels.

There is little data available on the toxicity of VOCs to benthic organisms living in the hyproheic zone but if the environmental quality standards of 10 μgl^{-1} are applied then the discharge of VOCs will have limited effect on the hyporheic zone and even less after dilution in the river. However, if lower limits (~1 μgl^{-1}) are imposed then the impact is considerably more widespread. A compound that may have considerable toxicity even at low levels is vinyl chloride, which forms as the end member of the breakdown of TCE and PCE. Analysis for this compound is difficult and expensive and was undertaken for only 16 samples in areas of TCE and PCE contamination. Vinyl chloride was detected in 10 samples (maximum value of 1.9μgl^{-1}) and its occurrence at low levels is widespread, associated with the biodegradation of TCE and PCE, as indicated by the common occurrence of cis-1,2 DCE.

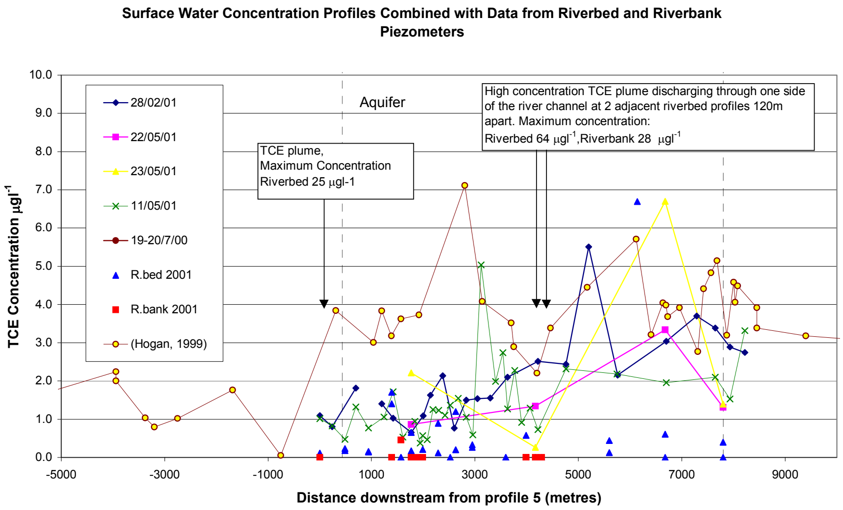
Chlorinated solvents are frequently detected in the groundwater due to their widespread use, relatively high solubility and resistance to natural attenuation (NA). Other groups of organic compounds such as BTEX and PAHs have similar or higher levels of usage (e.g. from petroleum fuels, gas works hydrocarbons, road run off) but were either infrequent or not detected, showing a greater susceptibility to NA. Napthalene is a common PAH but was not detected in any sample. This is likely due to it being a large molecule with a low solubility that fairly readily sorbs to the aquifer material. The BTEX compounds have multiple sources associated with petrol stations and industrial usage but benzene and toluene were detected only once and ethylbenzene and xylene not at all. The most likely reason for the limited detection of these compounds is the ready occurrence of biodegradation. Field studies at the Borden site in Canada have shown significant natural attenuation of plumes, eg. BTEX (Barker et al., 1987) and PAHs (King et al., 1999), and studies of a large number of plumes in the US ‘plumathon’ survey showed the average length of a BTEX plume to be <100 m due to rapid biodegradation (Newell et al.,1998). The fuel additive MTBE was detected in four of the riverbed samples, with no trace of any BTEX compounds which must have been removed by natural attenuation. The river is only likely to be at significant risk from BTEX within 100 m of the bank. The pure phase BTEX compounds are LNAPLs and will float on the surface of the water table. Over a short distance the dissolved phase plume will not reach a significant depth and will discharge through the river bank. Therefore, none of the piezometers currently installed in the riverbed is likely to detect it. At one location during the research pure phase LNAPL from a spill <100 m away was observed discharging through the bank side and seepage face but was not detected in later (five months) sampling from nearby riverbed piezometers.

Surface water sampling detected low levels of VOCs in all samples, TCE and cis 1,2-DCE were always present, TCM and PCE were detected in the majority of samples and trace amounts of TCA and CTC were often present. Apart from dichloromethane other VOCs were absent though only three broad-scan VOC analyses were conducted upon the surface water samples. The volatile nature of the compounds and their tendency to partition from water to air phase implies a limited residence in the surface water. Concentration profiles were obtained along the reach to identify whether inputs of VOCs (perhaps from groundwater) were maintained across the aquifer. The main VOCs (TCE, DCE, TCA, PCE, TCM) detected in the river were found at higher levels within the groundwater which may be a significant source for the surface water contamination. No information is available on pipe discharge concentrations across the aquifer but it is thought that some discharges comprise contaminated groundwater abstracted for use in industrial processes and eventually discharged to the river.

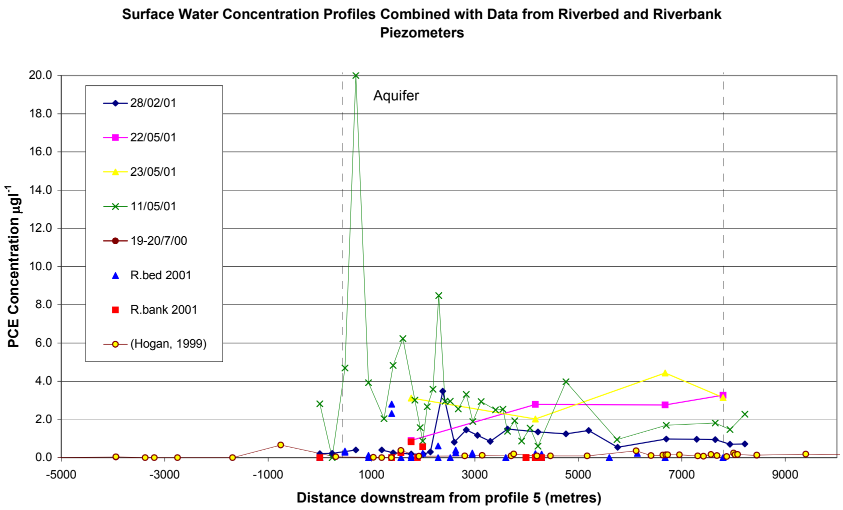
The surface water concentration profile for TCE (Figure 7.21a) is complex and shows considerable variation between different sampling periods. It is not possible to relate high levels of TCE in the groundwater at a particular location directly with local increases in surface water concentrations. However, levels of TCE are maintained or increased across the aquifer despite volatilisation and this indicates continuing input from what are believed to be groundwater sources. There is perhaps a general increase for all the sampling periods in the TCE surface water concentrations from 4 km onwards which may correspond to the changes in land use and hydrogeological factors discussed for the inorganic water quality. There is evidence to suggest that some of the TCE is being derived from intermittent sources, probably pipe discharges. An example of this was the large peak detected on 11/5/01 at 3 km which was not detected in any of the previous sampling. A similar high degree of variability is seen in both the PCE and TCM surface water profiles (Figure 7.21 b and c). Reasons for the variability between and within sampling periods may be a combination of factors:

1. changes in pipe and groundwater plume discharge rates and concentrations;
2. variation in river flow;
3. different temperatures and surface water mixing will alter rates of volatilisation;
4. the quantities detected are low and may be subject to large percentage errors in the analyses; and
5. variation in the type and quantity of suspended particulate matter may alter sorption rates.

(a)



(b)



(c)

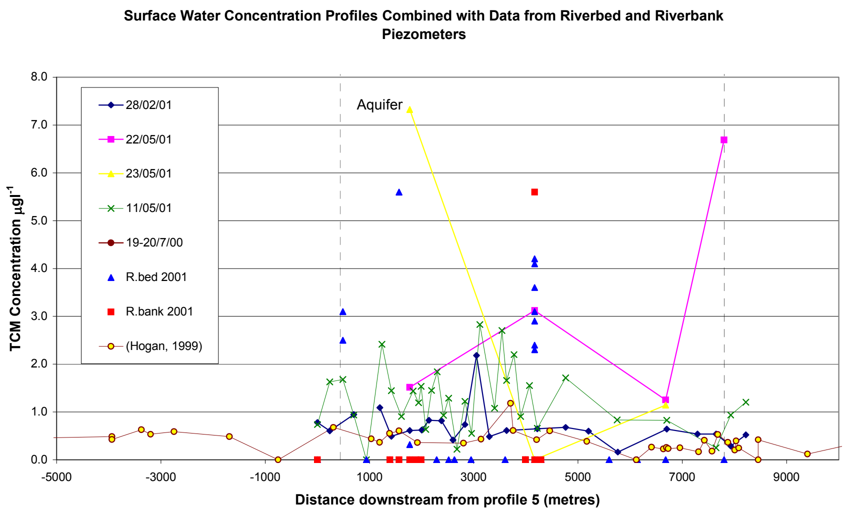
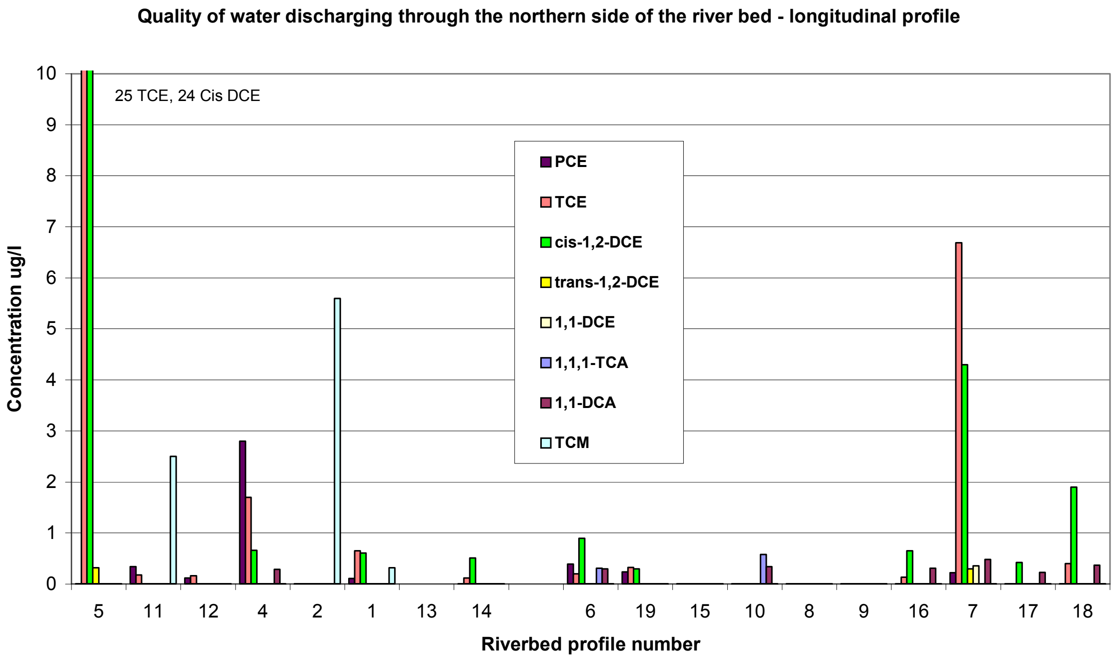


Figure 7.21 Longitudinal concentration profiles for (a) TCE (b) PCE (c) TCM.

The surface water concentration profiles imply that there may have been input across the aquifer of a number of VOCs that are found in the groundwater, but beyond that any more detailed interpretation was not attempted. The most useful indication of the impact of groundwater quality on the surface water system and the hyporheic zone is provided by the riverbed piezometer profiles. A variety of different contaminants discharge at each location along the reach with a marked contrast in the water qualities discharging from the northern (Figure 7.22a) and southern (Figure 7.22b) banks. A marked contrast is observed between the VOC concentrations discharging from opposite banks (eg profiles 5,8,9, and 7) showing that, in some areas, one side of the riverbed may be strongly contaminated whilst the other side is not. The largest plume with the highest concentrations can be seen discharging from the southern bank at profiles 8 and 9 which are 120 m apart; the plume is not detected in profile 10 which lies 180 m upstream. Profiles 8 and 9 are also sites of high levels of inorganic contamination, including high concentrations of sulphate, and fluoride at Profile 8. Variations in the ratio of TCE to TCA are observed between the two profiles, probably indicating multiple sources for the plume.

Chloroform was detected on both sides of the river at Profiles 11 and 2 and this may indicate groundwater from one side of the river discharging across the entire width of the channel. The inorganic quality data are also similar for both piezometers and support this possibility. Profile 11 lies on a meander bend with flow likely to occur from north to south across the flood plain. Profile 2 lies in a diverted section of deep river channel with the highest topography closest to the northern bank. Alternatively, a source of TCM such as a water main may be present on both banks.

(a)



(b)

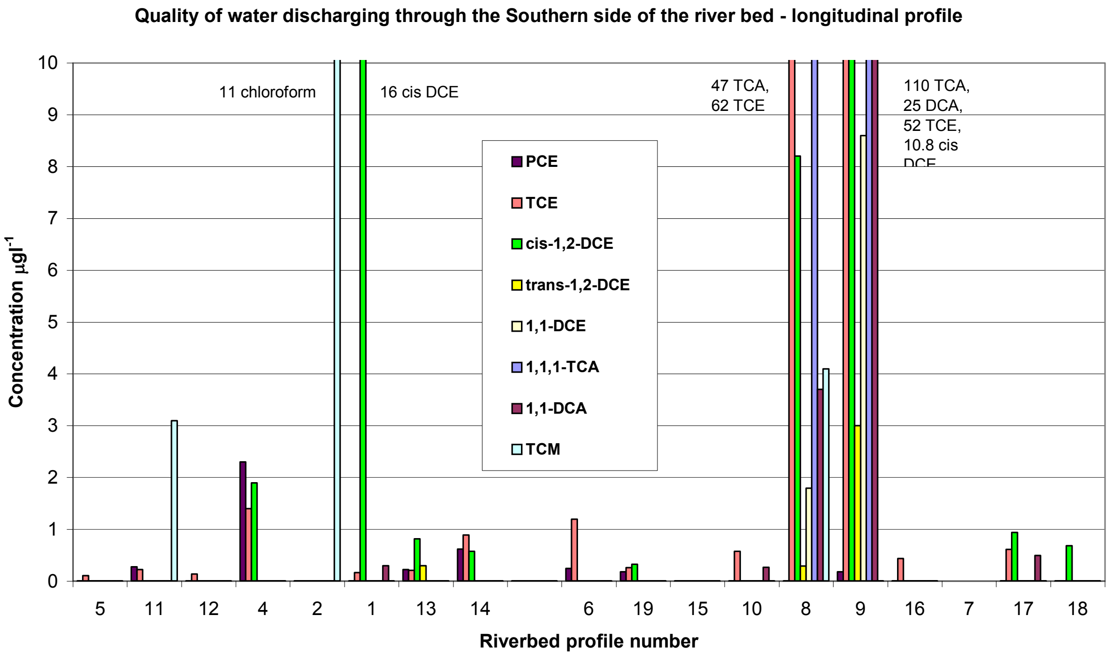


Figure 7.22 Chlorinated solvent concentrations in groundwater discharging through the (a) North and (b) South banks of the study reach.

The Environment Agency has undertaken a survey for 56 pesticides, herbicides and selected trace organic compounds in water supply rivers, for the Midland Region (Environment Agency, 1998). The pesticides mecoprop and isoproturon and the organic compounds nonyl phenol, chloroform and trichloroethene were identified in the sewage treatment discharges from Ray Hall and Minworth in the upper Tame. In addition to these compounds MCPA was identified in the surface water at Lea Marston, a distance of 10 km downstream of the study area. No surface water sampling was conducted within the upper Tame in this survey. No work has been done on the groundwater contribution of pesticides to the upper Tame, but they may be present in the groundwater owing to the extensive use of amenity and industrial pesticides in major conurbations.

7.8 Biodegradation and transport of chlorinated solvents across the groundwater/surface water interface

The degradation of PCE, TCE and 1,1,1-TCA is indicated by the presence of cis 1,2-DCE and 1,1-DCA within the groundwater. The rate of abiotic transformation is generally low and most reactions are probably a result of microbially mediated reductive hydrogenolysis (Fetter, 1999). The reduction follows a pathway with decreasing chlorination from PCE to TCE to DCE to vinyl chloride to ethene (Figure 7.23), and from TCA to DCA to chloroethane to ethane. Cis 1,2-DCE is only produced as a product of biodegradation and the ratio of TCE/DCE provides an indication of the extent to which biodegradation has occurred. The average TCE/DCE ratio varied from 62 in the abstraction wells to 2 in the shallow groundwater and 4 in the riverbed piezometers. In the case where either only TCE or only DCE was detected the ratio was calculated using the detection limit for the non-detected compound. Average values were taken for the north and south bank of each profile to remove

Tetrachloroethene

PCE

Trichloroethene

TCE

Dichloroethene

Cis 1,2 DCE

(other forms are rare).

Vinyl

Chloride

Ethene

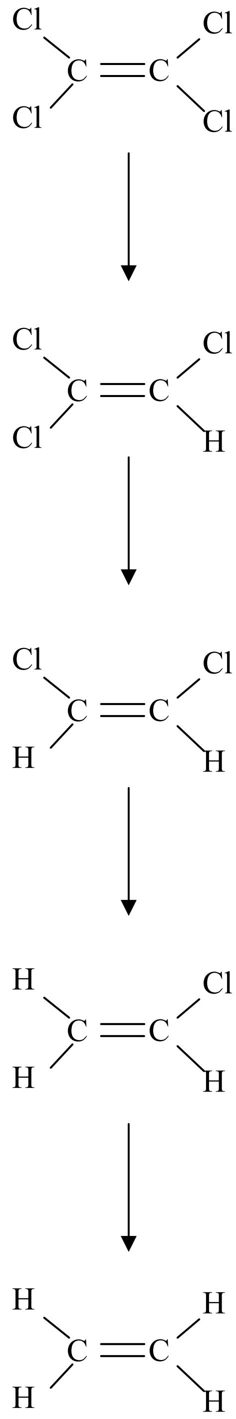


Figure 7.23 Biodegradation pathway (anaerobic dechlorination) for PCE/TCE (Fetter, 1999).

the bias resulting from multilevel samples collected within the same plume. These ratios suggest the abstraction wells are close to the DNAPL source where, despite the occurrence of biodegradation (as evidenced by some high levels of cis 1,2-DCE), concentrations of TCE remain high due to continued dissolution from the pure phase DNAPL. Further away from the source, the TCE/DCE ratio drops as TCE is removed by biodegradation. This shows that natural attenuation does occur within the aquifer but not at sufficient levels to remove all the VOCs prior to discharge to the river.

The low concentration of PCE in groundwater discharging from the southern bank (Figure 7.24) indicates limited sources and/or biodegradation of the parent compound to TCE. The low TCE/DCE ratio (0.01) at Profile 1 shows that nearly complete biodegradation of the original TCE has occurred, with concentrations of DCE at 16 μgl^{-1} and TCE at 0.17 μgl^{-1}. By contrast, the high ratio and concentrations of TCE at Profiles 8 and 9 imply rapid transit times from a nearby source. However, as anaerobic biodegradation for dechlorination requires a specific set of physical-chemical conditions there is no straightforward relationship between biodegradation and travel time from the source. Previous work (Lorah et al., 1999, and Fetter, 1999) indicates that hydrogenolysis occurs under anaerobic reducing (sulphate reducing to methanogenic) conditions which are not the general conditions throughout the aquifer. It may be that high levels of microbial activity close to the source, perhaps related to the degradation of mono-aromatic (BTEX) compounds, consumes the oxygen producing localised anaerobic zones. Reduction may occur in these zones before the plume travels further from the source and mixes with more oxygenated groundwater containing less dissolved carbon, at which time dechlorination ceases.

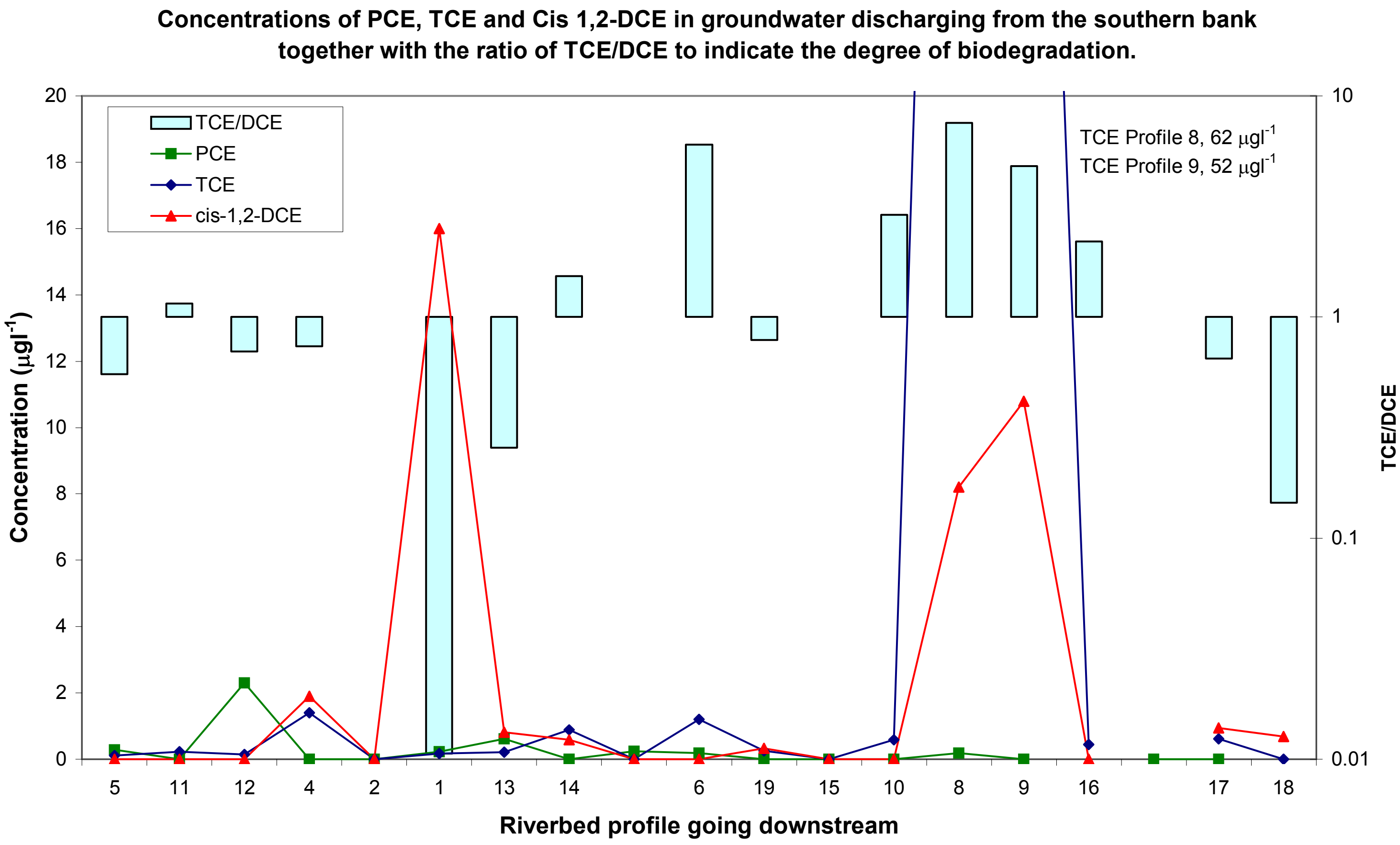


Figure 7.24 The ratio of TCE to Cis 1,2- DCE as an indication of biodegradation.

The groundwater/surface water interface provides an environment rich in dissolved organic carbon and other nutrients suitable for sustained microbial activity, which may lead to reducing conditions conducive to biodegradation. Previous work (Conant, 2000) has recorded dramatic reductions in PCE concentration (from 3700 μgl^{-1} to < 50 μgl^{-1}) in a plume over a 15 cm vertical interval of riverbed in areas of low to moderate groundwater discharge rates. Biodegradation reactions themselves often occur rapidly, but time may be required for microbial activity to consume oxygen and develop a suitable anaerobic and reducing environment before the groundwater discharges to the river. Reducing conditions have been recorded at several locations within the Tame riverbed but often the oxygen content is too high and sulphate reducing conditions have not developed. Flow velocities are thought to be generally rapid (>50 cmd^{-1}) through the sand and gravel of the riverbed with a limited residence time over which degradation may occur. However, considerable heterogeneity in flow exists both across the channel and along the reach and it is likely that low flow zones exist that are conducive to biodegradation. Modelling indicates that the highest flow rates occur immediately adjacent to, and through, the riverbank. Flow from these areas is derived from local shallow groundwater that flows at relatively high velocity through the drift deposits. The chance for biodegradation to occur in plumes derived from sources local to the river therefore appears to be considerably lower than for sources that are more distant.

Sorption of the hydrophobic contaminants onto organic matter will retard transport rates and will increase the residence time during which microbial action can occur. The results of the f_{OC} analyses performed on 15 riverbed core samples gave an organic carbon content ranging from 0.08% to 9.18% with a mean of 1.12% (median 0.37%). These values are similar to those found by analyses (Shepherd, 2002) of the flood plain alluvium (mean 1.2%, n=2) and glacial sand and gravel deposits (mean 0.4%, n=3). The FOC values obtained from the analyses of aquifer material (Shepherd, 2002) are substantially lower with a mean value of 0.049%. It can be inferred, therefore, that a considerable difference in contaminant transport rates exists between the unconsolidated and the bedrock aquifers. Groundwater velocities may be higher but sorption and retardation will be greater in the permeable alluvial deposits compared with the sandstone.

The retardation factor may be calculated for PCE, the most sorbing of the chlorinated solvents, in order to estimate the residence time within the riverbed environment.

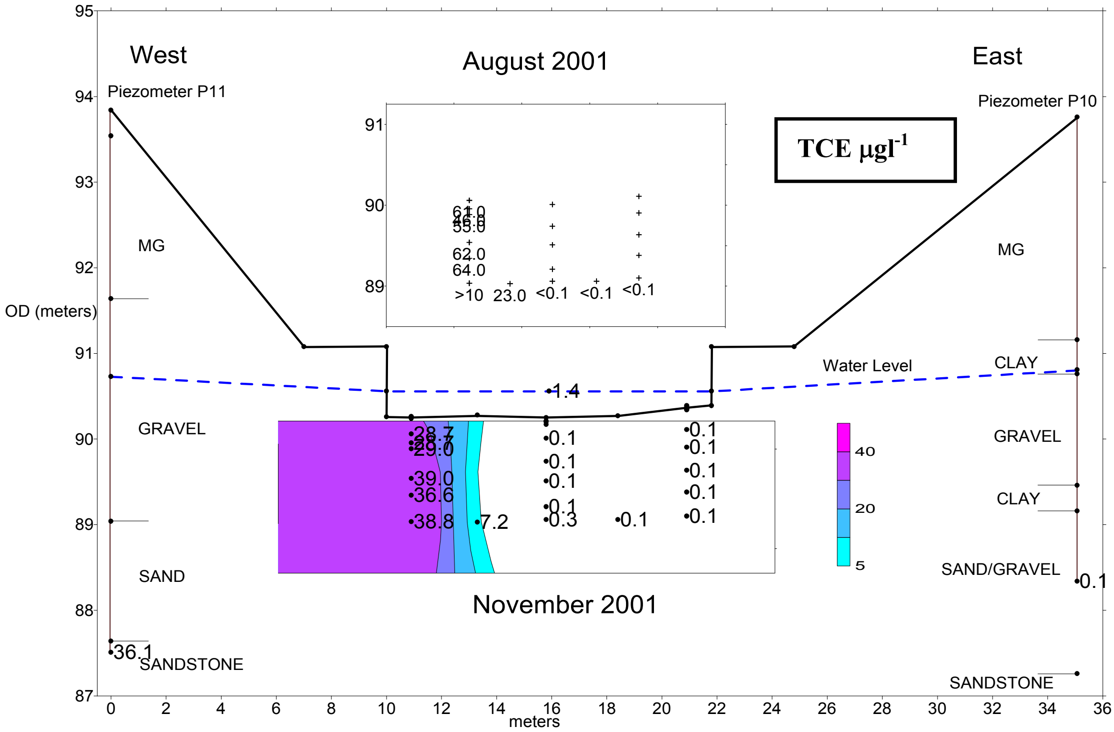
$R = V_L/V_S = 1+(Pb/n)*Kd$

R = Retardation factor	[-]
Pb = dry bulk density, 1.72 gcm^{-3} (Moylett, 2001)	[ML^{-3}]
n = porosity, 0.2	[-]
V_L = linear groundwater velocity	[LT^{-1}]
V_S = velocity of solute	[LT^{-1}]
$K_d = FOC*K_{oc}$	[$M^{-1}L^3$]
FOC = fraction of organic carbon	[-]
Koc = organic carbon partition coefficient,	[$M^{-1}L^3$]

The Koc value for linear sorption of PCE was estimated as 588 lkg^{-1} for aquifer material (Shepherd, 2002) which is considerably higher than the US Environmental Protection Agency (EPA) estimate of 265 and is related to the type of organic material present. Using the higher value of Koc a mean retardation factor of 58 may be calculated for the riverbed , ranging from 5 to 465 depending on the fraction of organic carbon. Estimates of linear discharge velocities from the field measurements and modelling indicate maximum velocities >5 md^{-1} (effective porosity 0.2), most likely through the channel sides, with discharge through the central riverbed ~50 cmd^{-1}. The mean residence time for PCE across a vertical 2m section of riverbed may be as high as 232 days, with extremes ranging from 2 to 1860 days depending on groundwater velocity and f_{OC} content. This has implications regarding whether biodegradation has time to occur within the riverbed prior to contaminant discharge.

The plume detected at Profile 8 discharges across a 4 m section of the riverbed (Figure 7.25 a and b) showing a similar distribution to the elevated levels of fluoride, sulphate and nitrate. The main components of the plume are TCE and 1,1-TCA with minor amounts of 1,2-DCE, 1,1-DCE, 1,1-DCA, TCM PCE, VC, and bromodichloride. TCE and TCA have similar lateral and vertical (Figure 7.26 a) distributions within the channel and show a considerable temporal variation in concentration, dropping by half between the summer and winter levels. This fall in concentration was also detected in the inorganic quality data with fluoride, sulphate and nitrate following a similar trend between summer and autumn in both 2000 and 2001 (Figure 7.8). This may be a seasonal trend related to the changes in the recharge rate and groundwater levels. These may affect the level of dilution within the plume and groundwater flow rates, and flow direction through the source zone. However, the contaminant travel time from the source zone to the river is unknown, making it difficult to relate the sampling time directly to any temporal variability in the source. Another explanation for the temporal variability may be that the direction of plume migration changes the high concentrations of the plume centre being sampled in the summer, but the lower concentration fringes of the plume being sampled

Profile 8 - Multilevel Piezometer Water Chemistry



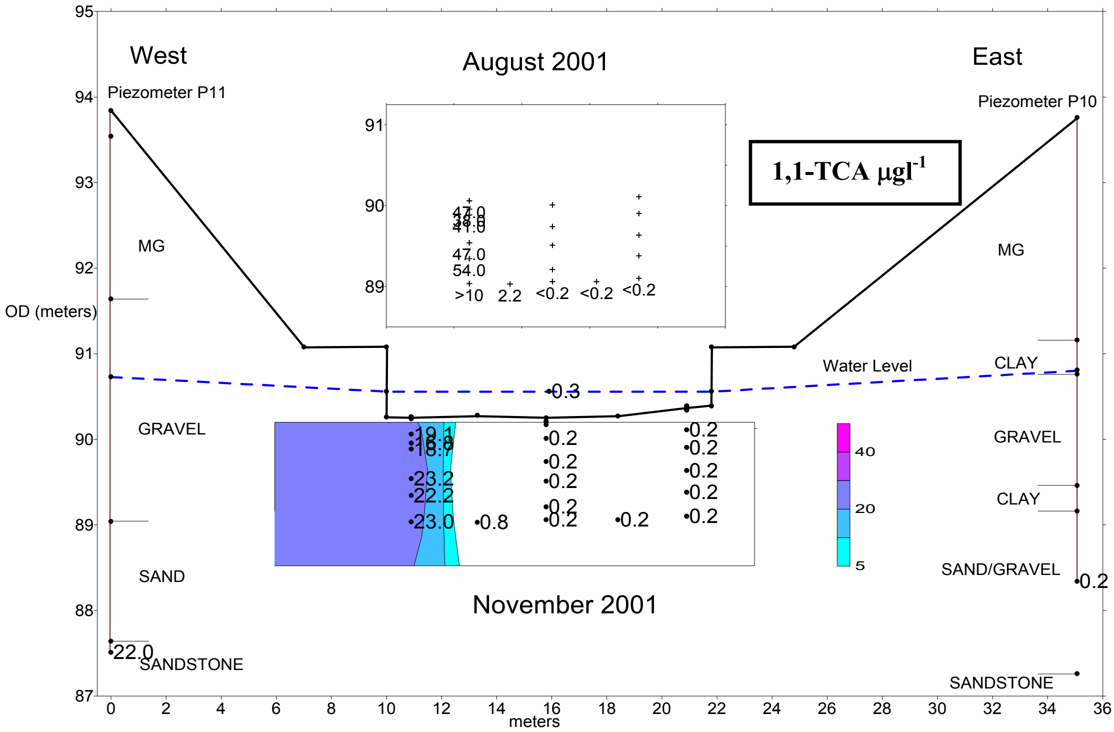


Figure 7.25 Cross section through a groundwater plume containing (a) TCE (b) 1,1,1 TCA.

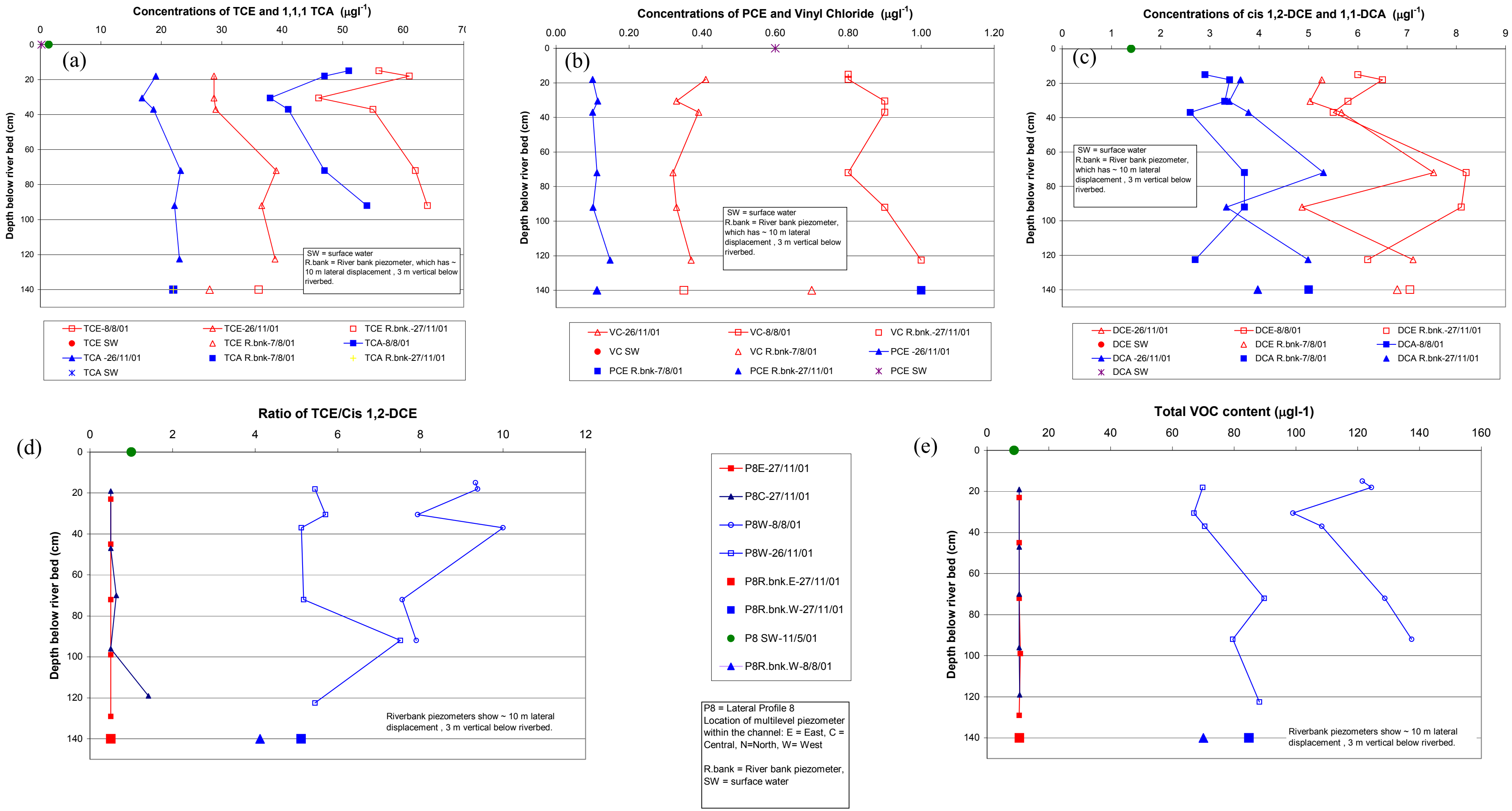


Figure 7.26 Concentration profiles from the multilevel piezometers at Profile 8; from the western profile (a) TCE and TCA (b) PCE and VC (c) DCE and DCA; from across the whole lateral profile (d) is the ratio of TCE/ cis 1,2-DCE (e) total VOC content.

in the autumn. The source may also be receiving additional mass inputs, perhaps through leakage from an industrial process, but this would require increased inputs for all the organic and inorganic contaminants together. This is unlikely from a single industrial process. A more likely explanation would be leakage from a mixed waste dump or sewer.

The variations can not be explained by dilution alone as the ratios of TCE to TCA change from 1.2 in the summer to 1.6 in the autumn. Levels of VC (Figure 7.26 b) follow a similar pattern to the TCE and TCA but the decrease in cis 1,2-DCE is small and the levels of 1,1-DCA increase between the summer and autumn samples (Figure 7.26 c). PCE was detected in only trace amounts and does not appear to be derived from the same source as the other VOCs. PCE is mainly used for dry cleaning whereas TCE and TCA are primarily used by the metals industry. Modelling of groundwater flow through the flood plain indicated a large source area for the discharge at Profile 8 which contains many current and historical industrial sites. It is possible that a high concentration source of PCE ($700 \mu g l^{-1}$) observed in an abstraction well > 1km away may be the source as PCE is not usually associated with the industries in the immediate vicinity of Profile 8.

The presence of VC, DCE, and DCA demonstrates that biodegradation has occurred en route to the riverbed. Moreover, a change in the ratio of TCE/DCE (Figure 7.26 d) from approximately 8 in the summer to 6 in the autumn indicates that less biodegradation has occurred in the summer samples. This may be due to changes in the redox conditions or high concentrations of the inorganic determinands inhibiting the biodegradation.

There is very limited evidence that biodegradation is occurring within the riverbed. The total VOC content (Figure 7.26e) does show a general decrease in the shallower samples from 80 μgl^{-1} to 70 μgl^{-1} in the autumn and 137 μgl^{-1} to 121 μgl^{-1} in the summer (between 92 cm and 18 cm below the riverbed). However, there is no rise in the levels of daughter products such as VC, DCE and DCA, and the ratio of TCE to DCE shows no obvious fall. The riverbed environment does not appear to provide the necessary reducing conditions to initiate the microbial breakdown processes. Oxygen levels are high (3 mgl^{-1}) and neither sulphate nor nitrate appear to be reduced below a depth of 20 cm (Figure 7.19). However, both sulphate and nitrate concentrations appear to decrease by a greater amount than predicted by simple dilution with surface water based on chloride concentrations. The increase in chloride concentrations at 15 cm indicates a surface water content of 12.5% and a surface water content of 19% at 5 cm. At these mixing ratios, if the sulphate is conservative, the concentration should be 8.6 $meql^{-1}$ at 15cm (actual 8.15 $meql^{-1}$) and 8.24 $meql^{-1}$ at 5 cm (actual 4.24 $meql^{-1}$). This implies that the sulphate is not conservative (i.e. is being reduced) from 0 to 15 cm depth below the riverbed. This calculation is not wholly reliable as there may have been some analytical error and because surface water concentrations are known to vary over time. If biodegradation does occur in this profile it lies within the surface water-groundwater mixing profile within the top 15 cm of the riverbed. There is insufficient VOC sample coverage in this area to provide any further information.

7.9 Comparison of results with general toxicity standards

The results of the water quality sampling were compared against the U.K. Environmental Quality Standards (EQS) for freshwater. These comprise a set of general toxicity standards which indicate levels harmful to aquatic life (Table 7.7). The U.K. Drinking Water

Water Quality Tame Valley 2001				Abstraction Wells			Shallow Groundwater			Riverbed Piezometers			Surface Water		
Determinand	Units	DWS	EQS	N samples	Failed DWS	Failed EQS	N samples	Failed DWS	Failed EQS	N samples	Failed DWS	Failed EQS	N samples	Failed DWS	Failed EQS
EC	uS/cm	1500		14			20	1		58	3		60		
F	mg/l	1.5		14			20	2		60	26		67	17	
Cl	mg/l	250	250	14			20	1	1	60	3	3	67		
NO_3	mg/l	50		14	7		20	5		60	20		67	17	
SO_4	mg/l	250	400	14	2	1	20	4	1	60	18	9	67		
Ca	mg/l	250		14			20	1		54	3		66		
K	mg/l	12		14	1		20	6		54	38		66	64	
Mg	mg/l	50		14			20			54	2		66		
Na	mg/l	200	170	14			20	1	2	54	3	5	66		
Mn	mg/l	0.05		14	3		20	12		54	49		66	3	
Fe	mg/l	0.2	1	14	3	2	20	7		54	27	1	66		
Ba	mg/l	1		14			20			54			66		
Al	mg/l	0.2		14			20	2		54	11		66		
Pb	µg/l	10	20-250	14			20	8		47	3		3		
Hg	µg/l	1	1	14			20			47			3		
Cd	µg/l	5	5	14			20	1	1	47			3		
Cr	µg/l	50	50-250	14	2	2	20			47			3		
As	µg/l	10	50	14	1		20			47	2		3		
Cu	µg/l	2000	28	14			20		3	47		3	3		
Zn	µg/l	5000	125-500	14		2	20			47			3		
Ni	µg/l	50	200	14	1		20	1		47	11	1	3	2	
1,1,1-TCA	µg/l		100	14			20			44		1	45		
1,1-DCA	µg/l	3	10	14			20	1		44	5	1	3		
TCM	µg/l	200	12	14		1	20			44			63		
cis-1,2-DCE	µg/l	50		14	1		20			44			3		
PCE	µg/l	10	10	14	3	3	20			44			64	1	1
TCE	µg/l	10	10	14	5	5	20	1	1	44	9	9	64		
VC	µg/l	0.5		3	1		20	1		11	7		0		

EQS = UK Environmental quality standards for sensitive species of fish at hardness >250 mgl-1 as $CaCO_3$.

DWS = UK Drinking Water Standards

Table 7.7 Comparison of data against environmental quality and drinking water standards.

Standards (DWS) were also used, as not all determinants were covered by the EQS and surface water in the River Trent downstream of the Tame is being considered for public supply. The DWS are generally more stringent than the EQS with the exception of certain contaminants such as Cu and TCM to which aquatic life is more susceptible. The EQS are generally applicable to sensitive species of fish and may not be entirely appropriate to assess the chronic affects of discharging plumes on biota within the riverbed as little research has been done on this.

Within the surface water, only one EQS failure was recorded which was for a PCE value of 20 μgl^{-1} and multiple failures against DWS were recorded for K, NO_3, F, Mn and Ni. The groundwater in the riverbed contains higher mean concentrations for all these determinants with the exception of K. Therefore groundwater should be considered as an important control on water quality in the river. The number of EQS failures increases significantly in the riverbed piezometers indicating the surface water sampling is not adequate to identify groundwater contaminant discharge to the river. EQS failures occurred in the riverbed piezometers for SO_4, Cl, Na, Fe, Cu, Ni, TCA, DCA, and TCE, the most frequent of which were for TCE and SO_4. Additional failures occurred against the DWS for F, NO_3, Ca, K, Mg, Mn, Al, Pb, As, and vinyl chloride. The failures often occur for several determinants at one site and are associated with localised discharge areas of high concentration. A large number of DWS failures are observed for Fe and Mn which are widely distributed in oxide form and become mobile under reducing and acidic conditions which often occur in the riverbed environment. However, on mixing with surface water, the change in pH and redox conditions is likely to remove these determinants from solution. The majority of DWS failures for aluminium are associated with a single plume in which the aluminium is mobile as a complex with fluoride.

The shallow piezometers show a similar pattern to the riverbed piezometers with additional failures for Pb and Cd. Many of the failures for lead were situated in residential areas adjacent to busy roads and the source of the lead may be associated with the combustion of leaded fuel. The lower concentrations of lead in the riverbed piezometers implies restricted mobility for the lead (perhaps due sorption) or dilution.

The abstraction wells have EQS failures for SO_4, Fe, Cr, Zn, TCM, PCE, TCE and additional failures against the DWS of NO_3, K, Mn, As, Ni, DCE and VC. The most widespread contamination is by NO_3 and TCE which would be significant if the abstraction wells were ever to be used for drinking-water supply.

7.10 Concluding Discussion

The urban groundwater quality data from the Tame Valley showed evidence of widespread anthropogenic contamination by the presence of industrial volatile organic compounds (primarily chlorinated solvents), heavy metals, and high concentrations of calcium, sodium, aluminium, fluoride, chloride, sulphate and nitrate. The survey did not analyse for other organics such as pesticides, semivolatiles and non-volatiles. The urban data showed considerably higher levels of electrical conductivity, major cations and anions than the natural background data set from Sutton Park and the deep borehole beneath Birmingham city centre. However, it was recognised that Sutton Park lies at the head of the catchment and is representative of a local, relatively rapid, groundwater flow system (as evidenced by the general under-saturation with respect to calcite), rather than the larger, more complex and perhaps slower systems operating in the Tame Valley. Acidic conditions were observed in the peat bogs of Sutton Park and this has increased the mobility of the heavy metals such that values of copper in the riverbed piezometers are comparable with those in the Tame Valley.

The results indicate a conceptual model for groundwater contamination in the Birmingham Aquifer that incorporates both diffuse and multiple point sources that generate contaminant plumes which, subject to natural attenuation, discharge to the river system or abstraction wells. Diffuse pollution by low levels of many contaminants was observed in the shallow and deep groundwater throughout the Tame Valley associated with mains and sewer leakage (Cl, NO_3), widespread usage (Cl as a road de-icer) and atmospheric fall-out (SO_4). Other source areas of more limited spatial extent are associated with current and historical land use. The riverbed piezometer profiles showed a general increasing trend in some contaminant concentrations across the aquifer associated with changes in land use from parkland and residential to more industrialised areas. For example, the minimum concentrations of sulphate in the riverbed profiles rises from 45 mgl^{-1} to 69 mgl^{-1} across the study reach (Figure 7.7a). The contaminant point sources may have considerable variability in size ranging from large areas (100's m^2) of made ground to localised areas (10's m^2) associated with spills and leakage from industrial processes. The resultant plumes may follow a flow path up to several kilometres in length with a considerable vertical component through both fractures and matrix to the final discharge point. The final location and extent of the plume discharge area to the river is dependent on the groundwater/surface water interactions and the degree of lateral dispersion. The average spacing between the riverbed piezometer profiles was ~ 400m and so it is likely that a number of plumes were not detected by this coarse sample spacing.

However, several high concentration inorganic and organic contaminant plumes were identified, often containing elevated levels of several contaminants in association with each other. The two major inorganic plumes identified had high concentrations in the riverbed piezometers of:

(1) 1873 Cl mgl^{-1}, 275 mgl^{-1} SO_4, 414 mgl^{-1} Ca, 704 mgl^{-1} Na, 2.7 μgl^{-1} Cd, 3.5 μgl^{-1} As, 304 μgl^{-1} Ni

(2) 198 mgl^{-1} F, 233 mgl^{-1} NO_3, 470 mgl^{-1} SO_4, 33 mgl^{-1} Al, 11 μgl^{-1} Pb, 97 μgl^{-1} Cu, 294 μgl^{-1} Zn, 139 μgl^{-1} Ni.

A major organic contaminant plume was identified across two piezometer profiles spaced 120 m apart with maximum concentrations of 62 μgl^{-1} TCE, 110 μgl^{-1} TCA in the riverbed.

The concentrations of contaminant that finally discharge to a river will be modified by dilution and natural attenuation processes within the aquifer and the riverbed, such as redox reactions, sorption and biodegradation. The data provide evidence for the occurrence of all these processes in the Tame Valley, including the biodegradation of TCE to cis 1,2-DCE, nitrate reduction, and the sorption and mobilisation of metals across the groundwater/surface water interface owing to gradients in the redox conditions. However, natural attenuation is not sufficient to remove all traces of contaminants from the discharging groundwater. Further work is required to investigate whether enhanced natural attenuation in the riverbed is more important than attenuation processes in the main aquifer.

Most persistent contaminants will eventually discharge to the river or abstraction wells which receive 59 % and 41% of groundwater flow respectively under current conditions. Natural attenuation will decrease contaminant concentrations but this is partially dependent on travel times, and the least amount of attenuation will occur for contaminants with sources close to the river. The high concentration plumes detected in this study may have sources within 100 m of the river where the protection afforded by the unsaturated zone is least.

Mixing of groundwater from different origins may occur in the focussed discharge zone around the river, and this would cause dilution of contaminant concentrations. In some instances, a substantial variation was observed in the composition of waters discharging from opposite banks, and this together with further variations in concentrations observed across the riverbed may reflect the culmination of other water types from a variety of flow paths. The vertical extent of the active groundwater system is unknown but modelling indicates that the deeper groundwater does not discharge to the river. The older groundwater from depth would be expected to discharge through the middle of the channel and younger, shallower groundwater, through the riverbanks, but the chemical data set is insufficient to confirm this. The similarities in water quality between the shallow and the riverbed piezometers are greater than between the abstraction wells and the riverbed piezometers, implying that the majority of groundwater discharging to the river is of relatively shallow origin.

Interaction between groundwater and surface waters was observed in multilevel piezometers using the concentration gradient between high levels of chloride in the surface water and usually lower levels in the groundwater. The maximum observed depth of surface water penetration into the riverbed was <60 cm though generally much less, and accords with the regional head gradient towards the river. Evidence for nitrate and sulphate reduction was observed in this mixing zone at some locations. The depth of this mixing zone was seen to vary over time. High levels of temporal variability were also observed in the contaminant concentrations within the discharging groundwater, perhaps associated with changes in the source, levels of natural attenuation, recharge and groundwater flow, and interaction with the surface water.

It is difficult to say how representative the riverbed piezometer sampling results are to the overall level of contaminant discharge to the Tame. The large heterogeneity of the system means that small variations in the position of the piezometer are likely to produce significant variations in concentration. Sample spacing, both along and across the river channel, needs to be closer to characterise the system fully. Significant changes in contaminant concentrations may also occur in the vertical plane including the upper 30 cm of the riverbed which is difficult to sample. It is also very likely that some contaminant plumes were not detected and that, even within the plumes that were, localised zones of high concentration were missed. Nevertheless, despite these limitations, the riverbed data yield a valuable insight into the processes and contaminant concentrations occurring across the riverbed and on the larger scale of the study reach.

The results of the surface water sampling under dry weather baseflow conditions were complex and difficult to interpret as the river displays considerable temporal variation in terms of both flow and mass flux. There are several time scales apparent in these variations: on a seasonal basis with changes in baseflow, and over a daily cycle determined by pipe discharges primarily from the sewage treatment works. In addition to this temporal variation there is the spatial variation of discharge accretion that occurs progressively downstream. All these factors need to be considered when comparing surface water results, even over relatively short distances.

For reasons of time and money, the data set used in this study has not been exhaustive, and the toxicity standards that have been applied have had their limitations. Nevertheless, a number of broad conclusions can be drawn, not least, that the quality of groundwater discharging into the Tame is better than might have been expected, despite Birmingham's long industrial history and the intensive urbanisation that has taken place. Sampling has shown that in many places in the study area, the quality of the groundwater is better than the quality of the surface water, and that in these areas the groundwater may be contributing to improved river quality and helping to sustain the riverbed ecology.

In other areas, it is clear that contaminant plumes discharging through the riverbed are having a detrimental effect on life within the riverbed although this is generally mitigated by dilution of the plume within the main surface water column.

It has been established that the levels of all the determinants used to gauge groundwater quality are above natural background levels. Although these elevated levels in the groundwater are having only a very limited impact on the existing quality of surface water, they could impede future efforts to improve surface water quality (e.g. if all pipe discharges into the river were to cease).

The most important determinants for surface water quality (EQS), particularly in the hyporheic zone, appear to be SO_4, Cu, Ni, TCE and VC because of their widespread occurrence and/or the stringent toxicity standards that apply to them. These toxicity standards are extremely useful in assessing the impact on the ecology in specific areas, but to manage the overall river quality effectively and to assess any attenuation processes that may be taking place a calculation of contaminant flux is required. Some of the high concentration plumes identified may have only a limited discharge, while other lower concentrations may discharge at a greater level and so provide a more significant mass input to the surface water. The mass flux to the river from the groundwater will be considered in Chapter 8.

CHAPTER 8. ESTIMATION OF GROUNDWATER FLUX TO THE RIVER TAME

The objective of this chapter is to estimate the dissolved chemical mass loading to the river from the groundwater discharging across the 7.4 km reach overlying the Birmingham Aquifer. The methods used included:

i) continuous measurements of electrical conductivity (EC) and discharge to derive temporal variations in surface water mass flux at Water Orton gauging station which result from changes in groundwater discharge;

ii) the use of baseflow analyses and mean riverbed sample concentrations to derive an average mass flux to the river from the groundwater;

iii) combined surface water sampling and discharge measurements to determine the increase in mass flux within the surface water across the study reach; and

iv) mass flux calculations for some of the individual contaminant plumes identified.

The calculation of mass flux requires measurement of both concentration and discharge. The marked heterogeneity of the system introduces significant uncertainty to both which is additive. The determination of groundwater discharge is not trivial. Increases in discharge along a river may be measured directly by flow gauging but in an urban setting it is difficult to separate the groundwater component reliably. Discharge may be calculated using hydraulic conductivity and head measurements taken from piezometers in the riverbed, but these values often show high variability over a small distance and provide point source information only. Groundwater modelling may be used to provide discharge estimates on a wider scale but is often constrained by limited available data on the subsurface hydrogeology. For studies on individual plumes, extensive investigation is required to determine the spatial distribution of discharge and concentration across the river channel.

There are uncertainties associated with selecting the appropriate values of concentration to be used in the flux calculation. The riverbed piezometers may have sufficient sample density and distribution to be representative of the overall quality of the groundwater discharging but further work would be required to confirm this. Surface water sample concentrations are subject to variability that may result from changes in upstream pipe discharges rather than the effects of discharging groundwater. Considerable temporal variability was observed in several of the groundwater plumes that requires regular repeat sampling to characterise them fully. Many of the determinants are not conservative and will be modified by attenuation processes such as biodegradation, redox reactions, sorption and volatilisation within the surface water column and the riverbed.

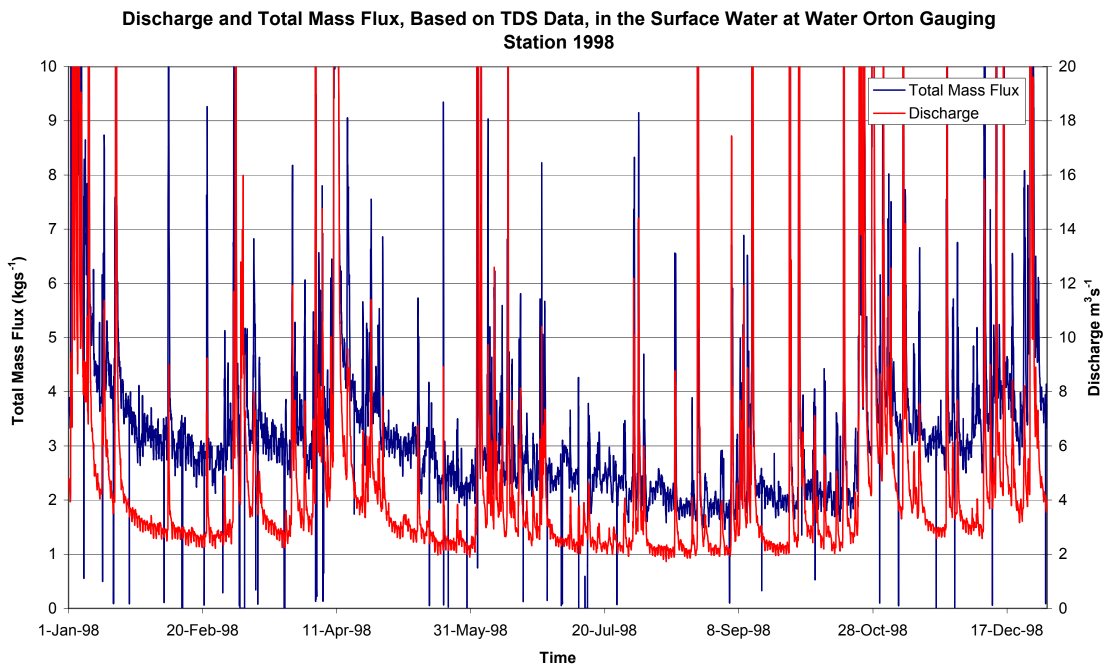
However, despite these uncertainties, mass flux estimates can be made from the River Tame data set using a variety of methods.

8.1 Electrical conductivity as an estimate of mass flux within the river.

Continuous monitoring data (15 minute intervals) were obtained for surface water electrical conductivity (EC) and discharge from the Water Orton gauging station at the downstream end of the study reach for 1998. If EC is assumed to be proportional to the concentration of the solution then the total mass of (ionic) dissolved solids (TDS) may be estimated. This does not account for the non-ionic inorganic and organic compounds but the results of the surface water sampling indicate the mass of these to be <1% of the total. The results of the 2001 surface water sampling were used to calibrate total dissolved solids (mgl^{-1}) against EC (μscm^{-1}). From these data, a factor of 1.09 was derived to convert EC to TDS for the measurements taken at Water Orton. These were then multiplied by the river discharge to derive the total mass flux (Appendix 17). There was a significant spread in the data used for the calibration that indicated a possible range in conversion factors of between 0.75 and 1.69. The relationship between EC and TDS is dependent on the composition of the solution, and changes in the ratio of NaCl to $CaCO_3$ resulting from variations in sewage treatment works (STW) discharges may explain the range in the conversion factor. Further work is required to remove this uncertainty from the flux calculations.

The maximum mass fluxes observed in the surface water (<20 kgs^{-1}) are related to rainfall events as calculated from a combination of surface run off, combined sewer overflow and increased groundwater discharge (Figure 8.1a and b). Daily variations are apparent in the mass flux owing to pipe end discharges from the sewage treatment works and other industrial and urban sources. Seasonal variations are also apparent and related to changes in the baseflow, the lowest baseflow and mass flux occurring in August. Mass flux in the surface water under baseflow conditions ranged from 3.44 kgs^{-1} on 21st January to 1.83 kgs^{-1} on 14th August, 2001. If the change in baseflow from 3.89 m^3s^{-1} in January to 1.90 m^3s^{-1} in August is related solely to groundwater discharge, then the dissolved mass within the groundwater can be calculated. The change in discharge divided by the change in mass flux gives dissolved solids within the groundwater of 809 mgl^{-1}, compared with the mean value of 1106 mgl^{-1} from

(a)



(b)

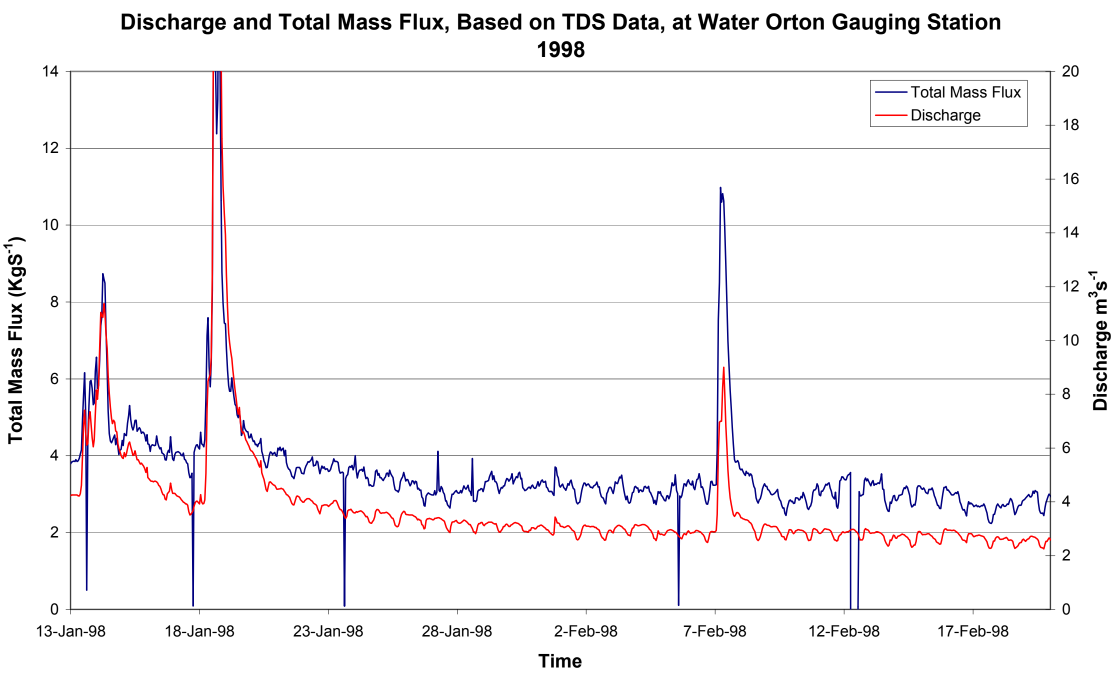


Figure 8.1 Mass flux in the surface water at Water Orton derived from TDS data for (a) the total year of 1998 (b) 13/2/98 to 19/2/98.

the riverbed piezometer sampling (Chapter 7, Tables 7.1 and 7.3). The results indicate lower quality groundwater discharging through the riverbed across the Birmingham aquifer (measured in the piezometers) compared with the gauging station estimate that is representative of groundwater across the entire catchment. The TDS content of the surface water at Water Orton is higher than the groundwater (809 mgl^{-1}), and rose from 884 mgl^{-1} to 965 mgl^{-1} over the period 21/1/98 to 14/8/98. This implies that on the catchment scale the groundwater is diluting the surface water.

During high baseflow conditions (21/1 to 19/2/98) a 30-day recession period (Figure 8.1b) showed a fall in the daily minimum mass flux from 3.44 kgs^{-1} to 2.44 kgs^{-1} and a drop in discharge from 3.89 m^3s^{-1} to 2.25 m^3s^{-1}. The TDS in the groundwater was calculated as 610 mgl^{-1}, lower than the estimate of 809 mgl^{-1}over the full range in baseflow conditions. This difference may be related to variation in the STW discharges or it may indicate that the average TDS content of the groundwater increases with a decrease in baseflow. The answer to why the TDS content of the groundwater should change is not straightforward and is dependent on mass flux from a complex system. One explanation is that groundwater under high baseflow conditions may contain a greater proportion of recent recharge than under low flow conditions when the proportion of groundwater with longer residence times (and higher TDS) increases. Groundwater modelling (Section 6.2) indicated that the permeable alluvial flood plain deposits are an important control on the groundwater system and provide a rapid flow path for the shallow groundwater (comprising a large component of recent recharge) to the river. Computer modelling of contaminant flux through the aquifer and flood plain deposits incorporating recharge is required in order to understand the system better.

Separation of the hydrograph data during flood events to determine the groundwater component was not possible without knowledge of that proportion of the mass flux originating from run off and sewer overflow.

8.2 Mass flux calculation from baseflow analyses and riverbed piezometer data

The baseflow analyses of gauging station data from 1999 (Section 5.3.2) was used in conjunction with the results of riverbed piezometer sampling undertaken in 2001 to estimate mass flux to the river from the 7.4 km study reach (Table 8.1). The estimate used the maximum, minimum and mean daily discharge rates under baseflow conditions and the total discharge for the year. These values were multiplied by the mean concentration for the determinant measured in the riverbed. Some of the determinants (e.g. Cl, NO_3, SO_4) show means that are considerably higher than the median values, reflecting the occurrence of low frequency high concentration contaminant plumes, hence the use of the mean value may overestimate the mass flux. On a local scale, considerable variation in both riverbed concentrations and estimated groundwater flows occurs across the aquifer. As discussed in the previous chapter a general increase is observed in riverbed concentrations across the aquifer, and therefore an associated increase in mass flux is also likely. The variation may be related to increasing industrial land use and may be observed in the distributions of Cl, NO_3, SO_4 concentrations in the riverbed piezometers (Figure 7.7). However, for the purposes of the mass flux calculation the aquifer will be considered as a single unit.

The groundwater that discharges directly to the 7.4 km study reach (mean baseflow 19.2 Mld^{-1} , Section 5.3.2) may comprise 26-59% of the total aquifer discharge depending on how much tributary flow is fed by groundwater from the aquifer (dry weather tributary flow is estimated as 42.6 Mld^{-1}, Section 5.3.1). Between 18% and 41% of groundwater from the

Determinant	Mean Groundwater Concentration*[1] mgl^{-1}	Median Groundwater Concentration*[1] mgl^{-1}	Groundwater baseflow discharge from the (7.4 km)study reach *[2]			
			Maximum m^3d^{-1}	Mean m^3d^{-1}	Minimum m^3d^{-1}	Years Total (1999) m^3yr^{-1}
			40700	19240	12580	8861345
			Mass Flux from the Groundwater (calculated from mean groundwater concentrations)			
			Maximum kgd^{-1}	Mean kgd^{-1}	Minimum kgd^{-1}	Years Total $kgyr^{-1}$
HCO_3^-	371	299	15093	7135	4665	3286063
F	23	1	926	438	286	201714
Cl	143	62	5835	2759	1804	1270514
NO_3	58	24	2363	1117	730	514509
SO_4	215	162	8746	4135	2703	1904280
Ca	147	135	5988	2831	1851	1303781
K	15	16	591	279	183	128641
Mg	22	20	886	419	274	192837
Na	91	62	3686	1743	1139	802574
Mn	3	2	128	61	40	27928
Fe	1	0	57	27	18	12503
Ba	0.11	0.09	4.5	2.1	1.4	988
Al	4.3	29.9	177	83	55	38449
Pb	0.0062	0.0020	0.25	0.12	0.08	55
Hg	0.00003	0.00002	0.0013	0.00061	0.00040	0.3
Cd	0.00065	0.00062	0.026	0.012	0.008	6
Cr	0.0024	0.0018	0.098	0.046	0.030	21
As	0.0031	0.0032	0.126	0.060	0.039	27
Cu	0.012	0.006	0.486	0.23	0.150	106
Zn	0.081	0.037	3.30	1.56	1.02	719
Ni	0.044	0.027	1.79	0.85	0.55	390
TCE	0.0107	0.0006	0.44	0.21	0.14	95
cis-1,2-DCE	0.0026	0.0019	0.108	0.051	0.033	23
PCE	0.0003	0.0002	0.0106	0.0050	0.0033	2
TCM	0.0015	0.0031	0.059	0.028	0.018	13
1,1-DCA	0.0013	0.0005	0.052	0.024	0.016	11
1,1,1-TCA	0.0092	0.0440	0.38	0.18	0.12	82

*[1] Values taken from riverbed piezometer sampling 2001.

*[2] Values from baseflow analyses of gauging station data 1999.

Table 8.1 Estimated geochemical mass flux from the groundwater to the study reach.

aquifer is abstracted (13.3 Mld^{-1}). The mean values for total dissolved mass in the riverbed piezometer and abstraction well data from the 2001 sampling (Section 7.1, Tables 7.1 and 7.3) were 1101 mgl^{-1} and 504 mgl^{-1} respectively. Therefore, in terms of the total groundwater chemical mass flux, 76-91% goes to the river and only 9-24% to the abstraction wells. Of the total groundwater flux, some comprises relatively recent recharge affected by anthropogenic pollution and the remainder consists of older groundwater with a composition reflecting previous recharge history and mineral dissolution. The relative contribution to the surface water flux from the different water types is unknown. The calculation of mass flux through the aquifer is highly complex but some estimates of current mass loading via recharge can be made for comparison with the estimates of groundwater flux to surface water. The current recharge to the aquifer is estimated to comprise 52% rainfall run-off and 48% from other sources such as mains and sewer leakage (Section 6.4.2.2). A recent survey of roof run-off in a residential area of Birmingham (Harris,2002) and from a study on pollutant flux to the aquifer (Thomas, 2001) provide data on the likely composition of recharge water (Table 8.2). The residential roof run-off is generally representative of atmospheric loading, while the general urban and industrial categories reflect the higher concentrations observed in storm drains associated with different land uses. Many of the concentrations in rainfall from the roof run-off survey are one to two orders of magnitude lower than the groundwater discharging to the riverbed, showing the extensive input from natural and anthropogenic sources. However, the run-off concentrations for the metals Cu, Pb, Zn are similar to, or greater than, the concentrations in the riverbed. Given the sources in addition to the run-off, this implies attenuation is occurring within the aquifer.

		Cl	NO3	SO4	Ca	K	Mg	Na	Zn	Pb	Cu	pH	EC
Residential roof run-off [*1]	Concentration (mgl^{-1})	1.9	2.3	2.2	0.7	0.6	0.15	1.4	0.03			5.1	25 μscm^{-1}
	Mass loading ($kgyr^{-1}$) [*3]	71382	86966	84914	25899	21730	5717	52925	1153			-	-
General urban run off [*2]	Concentration (mgl^{-1})	15	4						0.21	0.165	0.04		
	Mass loading ($kgyr^{-1}$) [*3]	577209	153922						8081	6349	1539		
Industrial run-off [*2]	Concentration (mgl^{-1})	148	8.4						1.063	0.115	0.032		
	Mass loading from industry ($kgyr^{-1}$) [*4]	1139026	64647						8181	885	246		

*1 Concentrations from Harris (2002)

*2 Concentrations from Thomas (2001)

*3 Based on recharge to the aquifer from rainfall of 38268 m^3yr^{-1}

*4 Based on 20% industrial land use

Table 8.2 Estimated compositions of recharge water and mass loading to the unconfined Birmingham Aquifer.

An estimate of mass loading from rainfall run-off may be calculated for the unconfined aquifer using a surface area of 1.063 x 10^8 m^2 and an average recharge from rainfall of 0.00036 md^{-1}. This does not include the recharge of 0.00033 md^{-1} estimated to come from other sources such as leaking water mains (Section 6.4.2.2). The estimate of pollutant mass flux from industry is based on a land use of 20% in the industrial/commercial category (Thomas, 2001). The estimated groundwater mass flux to the river for Cl was 1,271 tonnes per $year^{-1}$ (Table 8.1) which may be compared with an atmospheric loading (residential roof top run-off) to the aquifer of 71 ty^{-1}. A combined estimate of Cl mass flux for urban and industrial recharge to the aquifer was ~ 1,700 ty^{-1}. In addition to this an average amount of 8,781 ty^{-1} of NaCl (5,326 ty^{-1} as Cl) is applied in the Birmingham area during winter road salting (Jenkins, 1995). The estimated groundwater mass flux to the river for SO_4 was 1,904 ty^{-1} and for NO_3 515 ty^{-1} compared with an atmospheric loading to the aquifer of 85 and 87 ty^{-1} respectively. The combined estimate for urban and industrial recharge to the aquifer for NO_3 was 219 ty^{-1}. This compares with estimates by Ford (1994) for the atmospheric fallout on the Birmingham area for SO_4 of 275 $ty^{-1}$and for NO_3 160 ty^{-1}.The fluxes to the river are considerably higher than the contributions from run-off suggesting considerable input from other sources such as leaking sewers and water mains, contaminated land and mineral dissolution.

The mass flux of the chlorinated solvents to the river is limited despite the large mass estimated to be resident in the aquifer (Rivett, 1989). For TCE a total mass flux of 95 kgy^{-1} equates to 65 litres, ~1/3 of a drum, and for TCA 82 kgy^{-1} equates to 61 litres, ~1/3 drum, and PCE 2 kgy^{-1} equates to 1.2 litres, ~ 1 lemonade bottle. Calculations from the sampling of abstraction wells indicated loss of TCE from the aquifer of 1,500 kgy^{-1} (1987) and 160 kgy^{-1}(1990) (Taylor et al.,1999). The lower value in 1990 was the result of the more restricted availability of sampling wells; many of the highly contaminated abstractions sampled by Rivett et al. (1990) were not available. These values are much greater than the estimates for the mass flux to the river despite the mean groundwater baseflow discharge to the river of 19.2 Mld^{-1} being greater than 13.3 Mld^{-1} discharge to the abstraction wells (1999 data). This indicates either that the chlorinated solvents do not reach the river in large quantities, or that the riverbed sampling failed to locate some of the significant plumes. Biodegradation is responsible for the removal of some of the TCE, with the mass flux of 23 kgy^{-1} cis 1,2-DCE, likely a biodegradation product of PCE/TCE, being equivalent to the breakdown of 31 kgy^{-1} TCE. This is equal to 25% of the total concentration of TCE detected discharging to the river (inclusive of the cis 1,2-DCE) and implies that some 25% of the original dissolved mass of TCE in the groundwater flowing towards the river has degraded prior to discharge.

The yearly mass flux of heavy metals to the river from the groundwater is low and given the large dilution effect there is unlikely to be a significant effect on the surface water quality downstream. The greatest mass flux estimate was for Zn (719 kgy^{-1}) followed in descending order by Ni (390 kgy^{-1}), Cu (106 kgy^{-1}), Pb (55 kgy^{-1}), As (27 kgy^{-1}), Cr (21 kgy^{-1}), Cd (6 kgy^{-1}), and Hg (0.3 kgy^{-1}). Heavy metals are often found at high concentrations in contaminated land and are widespread at lower levels in urban soils. For example, concentrations of Zn detected in a soil survey of the Wolverhampton area (Bridge et al., 1997) ranged from 27 to 2,853 ppm with a mean value of 488 ppm. However, results from the Tame valley indicate that, if present, heavy metals in contaminated soils are having only a limited impact on the groundwater discharging to the river.

The bias on the mean and median values introduced by high concentrations at some localities is particularly evident for F (Table 8.1). The plume detected at Profile 8 has contributed to the mean concentration of F being 23 times greater than the median. This introduces uncertainty into the calculated mass flux which may be over- estimated by the use of the mean value. A similar overestimate may also have occurred for Al, the mean concentration being 4.3 mgl^{-1} when 80% of the samples were below the detection limit of 0.3 mgl^{-1} (Table 8.1). The median value for Al of 29.9 mgl^{-1} represents only the subset of samples above the detection limit and does not indicate an underestimation of the mean.

8.3 Mass flux estimates from surface water sampling and discharge measurements

A series of surface water samples were taken on 2/6/00, 22/5/01 and 23/5/01 at points across the study reach in combination with river discharge measurements (Figure 8.2). The total mass in the river at each point was calculated by multiplying the discharge by the concentration. For each determinand, the average increase in mass flux per metre length of channel was then divided by the average increase in discharge per metre of channel to derive the concentration of the influent water (Table 8.3). The results were compared against the mean concentrations found from the riverbed piezometer sampling. A mass flux was calculated over the 7.4 km study reach assuming all the influent water to be groundwater.

The results display considerable variability, showing the very complex nature of mass flux in the river. The discharge measurements taken on the 2/6/00 show high variability and may not be suitable for detailed consideration. The discharge measurements taken on the 22nd and 23/5/01 are consistent (Figure 8.3) but comparison with the influent water compositions shows some large variations between the determinants. Differences range from 2-6 % for Na, K, Cl to 34-38% for Mg, SO_4, and NO_3. Some of the differences may be due to errors in the

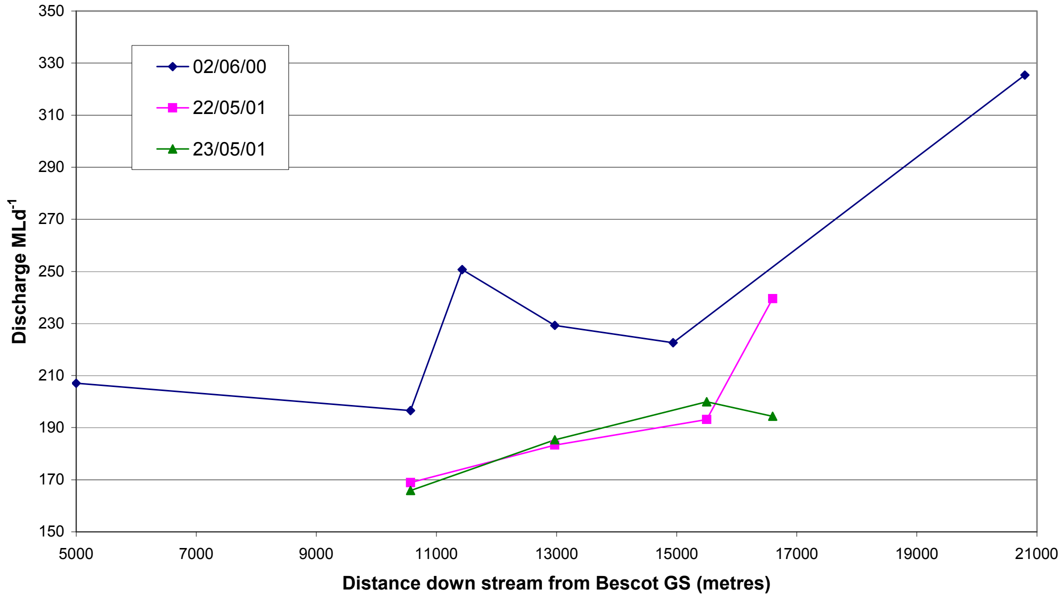


Figure 8.2 River discharge measurements taken in conjunction with water quality sampling across the study reach.

Determinant	Mean groundwater concentration (Riverbed Piezometers) mgl^{-1}	Date Sampled					
		2/6/00	22/5/01	23/5/01	2/6/00	22/5/01	23/5/01
		Increase in discharge across the 7.4 km study reach (m^3)			53391	73013	38193
		Estimated concentration of the influent water (mgl^{-1})			Mass flux to the river that occurs across the 7.4 km study reach (kgd^{-1})		
F	23	3	1	1	182	85	43
Cl	143	53	89	84	2842	6510	3196
NO3	58	15	34	46	777	2477	1757
SO4	215	75	170	105	4007	12378	3999
Ca	147	73	121	104	3921	8862	3965
K	15	4	12	13	211	904	494
Mg	22	11	23	15	562	1692	582
Na	91	22	67	65	1199	4883	2492
Sr	0.49	0.1	0.4	0.2	8	32	8
Si	10.3	2.8	-0.9	-3.8	150	-69	-145
Ba	0.11	0.048	0.001	0.019	3	0.1	1
TCM	0.0015		0.015	-0.039		1.1	-1.5
TCA	0.0092		0.001	-0.002		0.1	-0.1
TCE	0.0107		0.005	0.014		0.4	0.5
PCE	0.0003		0.009	0.009		0.7	0.3

Table 8.3 The geochemical mass flux to the study reach estimated from combined surface water sampling and discharge measurements.

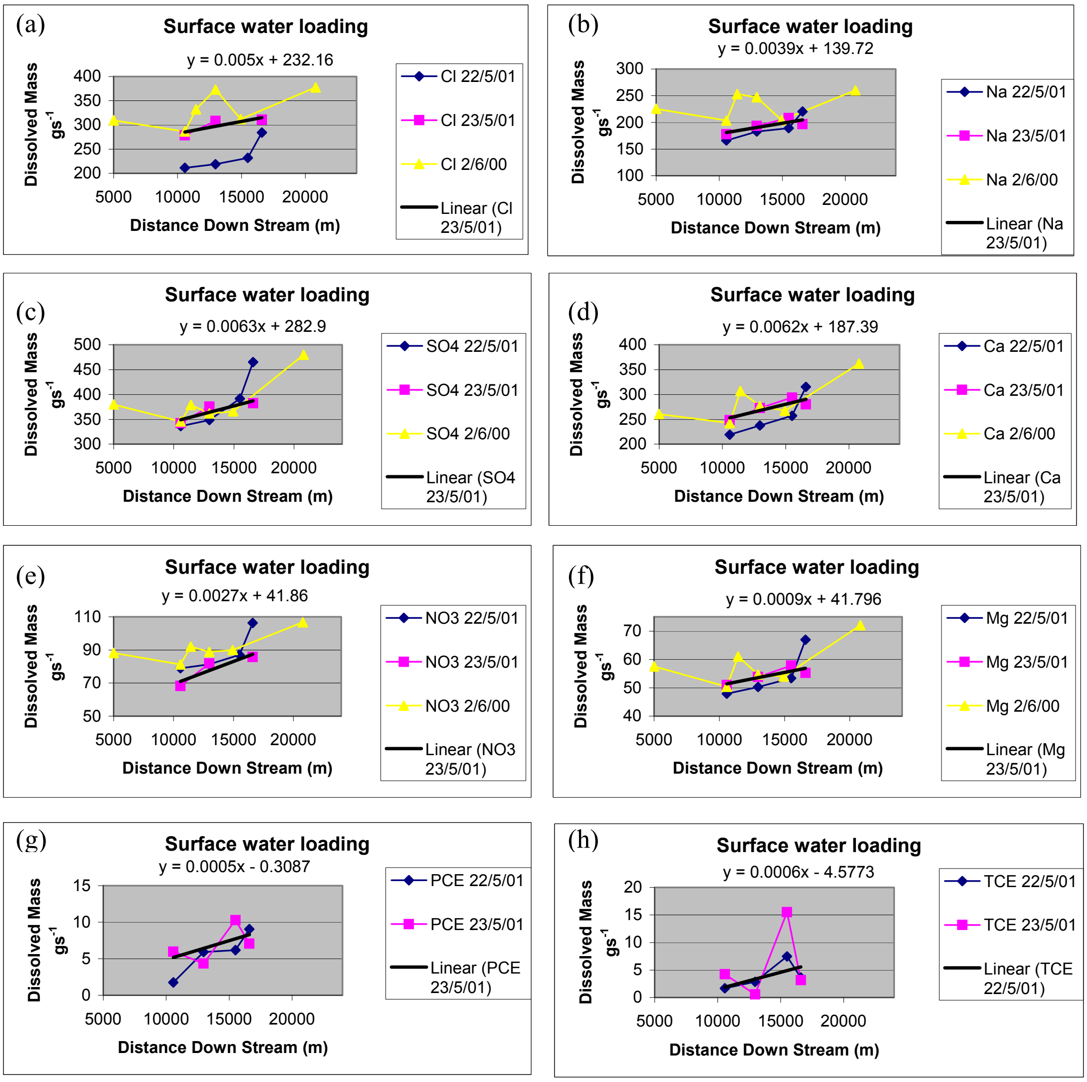


Figure 8.3 Estimates of the change in surface water mass loading across the aquifer for (a) Cl (b) Na (c) SO_4 (d) Ca (e) NO_3 (f) Mg (g) PCE (h) TCE.

analyses or the discharge measurements but some are likely due to variations in the flux coming downstream. The sewage treatment works and other pipe end discharges produce different daily variations in mass flux for each determinand. This was observed during the hourly sampling (over an eight-hour period) undertaken at Profile 8 on 26/7/00, with the range in concentration for each determinant being different, Cl 33%, Na 17%, SO_4 4%.

For each different sampling period nearly all of the influent water showed concentrations less than the mean concentrations from the riverbed piezometers. This indicates one of the following possibilities:

- water of lower concentration is influent to the river in addition to groundwater;
- a decline in the dissolved mass flux coming downstream occurs during the course of the days sampling;
- non-conservative behaviour of the solutes;
- overestimation of the groundwater concentrations from the riverbed sampling.

Approximately 27% (10.3 Mld^{-1}) of the 38.2 Mld^{-1} increase in discharge recorded across the study reach on 23/5/01 is estimated to comprise water from tributaries (58%) and pipe discharges (42%). No actual measured data are available for the sample days in terms of the concentration or discharge of the tributary or pipe discharges, although none carries any high concentration STW discharges. To obtain an estimate of the maximum likely groundwater concentrations, the surface water inflows (27% of total inflow) were assumed to have the same composition as the surface water from Sutton Park. Recalculation of the concentrations of the groundwater component (63% of the total inflow) brought the concentrations for the majority of the determinands to levels similar to the mean riverbed concentrations with the exception of Si, Cl, F and SO_4. These remained at lower levels, perhaps indicating that the riverbed piezometer sample mean may be an overestimate for F, Cl and SO_4 in the groundwater. The high mean concentration of F from the riverbed piezometers of 23 mgl^{-1} is biased upwards by the samples from the high concentration plume. It is probably more appropriate to use the median value of 1 mgl^{-1} for the groundwater concentration which agrees well with the predicted values from the discharge measuring of 1-3 mgl^{-1}. There is evidence of non-conservative behaviour by Si which shows a decrease in mass flux across the aquifer maintaining a concentration that varies by < 1mgl^{-1}. This may be due to sorption or precipitation. With the restrictions on the data set (i.e. the limited number of multilevel samples and temporal variability in surface water concentrations), it is difficult to reach any definite conclusion about the level of nitrate reduction that may occur within the river and the hyporheic zone. The positive increase in mass flux suggests attenuation is limited and that a substantial amount of nitrate is reaching the surface water system along the reach.

The chlorinated solvent aqueous concentrations will be subject to attenuation via loss to the vapour phase from the surface water. The low or negative predicted concentrations of the influent water indicate loss of TCM and TCA on the 23/5/01 and 22/5/01. Predicted TCE concentrations for the influent water range from 6 μgl^{-1} (53%) below, to 3 μgl^{-1} (31%) higher than the mean riverbed estimate of 10.7 3 μgl^{-1}. Further data on the rates of TCE sorption in the surface water system and loss to atmosphere would be required before any predictions as to the actual concentrations of influent water can be made. The increase in mass flux along the reach indicates the rate of inflow of TCE exceeds the rate of vapour loss, but this is not the case for TCA and TCM. PCE shows consistently higher estimated concentrations for the influent water (0.009 μgl^{-1}) than the field data (0.003 μgl^{-1}) suggests, indicating unknown sources from either pipe discharge or groundwater. The calculated mass flux of TCE ranged between 0.4 and 0.5 kgd^{-1} and was higher than the range 0.4 to 0.1 kgd^{-1} estimated from the baseflow analyses (Table 8.1) which incorporates no attenuation. This implies additional sources of TCE or an underestimation by the baseflow-riverbed piezometer method of calculation. Using the maximum daily flux of 0.54 kgd^{-1} a total mass flux of at least 197 kg of TCE (~2/3 drum) enters the river across the study reach annually.

The combined sampling and discharge measurements did not, unfortunately, incorporate detection limits sufficiently low to detect toxic metals. However, data were available from Environment Agency bi-monthly surface water sampling, which were used to estimate flux in the surface water (Appendix 2). There were two sampling points located on either side of the unconfined aquifer, Perry Barr upstream and Saltley downstream. The sample data were screened to remove results from wet weather days which were counted as those with >3mm rainfall on the day of sampling. This screening was designed to remove the effect of run-off and sewer overflow so that any increase in mass flux would be more representative of a groundwater source. The metals data are primarily for total content and so include the contribution from suspended particulate matter as well as the dissolved phase. The difference between the Cu total and the dissolved values indicates that approximately half of the copper is in particulate form.

For the entire data set (Table 8.4) a decrease in concentration is observed for each determinand between Perry Barr and Saltley, implying dilution by water of lower concentration, settling of particulates or natural attenuation. This is also true for the dry weather data with the exception of Cr which shows a slight increase in concentration. The

Location	Distance from Bescot GS (km)	Mean Concentrations from Environment Agency Bi-Weekly Surface Water Sampling (1993 – 1996)								
		NH_4 mgl^{-1}	Cd (total) μgl^{-1}	Cu (total) μgl^{-1}	Cu (dissolved) μgl^{-1}	Ni (total) μgl^{-1}	Cr (total) μgl^{-1}	Pb (total) μgl^{-1}	Zn (total) μgl^{-1}	Tot. Hard As $CaCO_3$ mgl^{-1}
		Mean concentrations inclusive of all samples								
PERRY BARR	10.3	1.76	0.43	52.3	23.9	145.2	13.2	13.1	164	382
SALTLEY	17.7	1.45	0.32	42.1	19.3	125.6	12.4	9.2	139	334
		Mean concentrations for dry weather flow only (<3 mm rainfall that day)								
PERRY BARR		1.85	0.39	50.0	24.7	154.8	11.4	7.7	144	402
SALTLEY		1.6	0.3	42.1	19.6	134.9	12.4	6.8	136	363
	Discharge MLd^{-1}	Mass Flux gd^{-1} Estimated using discharges from baseflow analyses and dry weather flow concentrations								
PERRY BARR	217.5	403000	84	10871	5374	33675	2470	1677	31369	87328000
SALTLEY	266	424000	83	11196	5225	35888	3293	1796	36197	96533000
Difference in mass flux		21000	-1	326	-149	2214	824	119	4828	9205000
	Discharge MLd^{-1}	Mass Flux gd^{-1} Estimated using discharges on 22/5/01 as typical baseflow and dry weather flow concentrations								
PERRY BARR	166	307000	64	8297	4101	25701	1885	1280	23942	66650000
SALTLEY	194	309000	60	8166	3811	26174	2402	1310	26399	70404000
Difference in mass flux		2000	-4	-131	-291	473	517	30	2458	3753000

Table 8.4 The mass flux of heavy metals in the surface water estimated from Environment Agency sample data.

changes in concentration could also be due to systematic error - if, for example, the sample points were regularly visited but at times corresponding to different stages in the daily sewage discharge cycle as it propagated down river. The dry-weather data show a fall in mean concentration at Perry Barr for Cd, $Cu_{(total)}$, Cr, Pb and Zn showing that, despite the increased dilution caused by wet weather, the mass flux is greater, because wet weather inputs and increased flow velocities carry more suspended material. The concentrations of NH_4, $Cu_{(dissolved)}$, Ni and total hardness increase because of less dilution from run-off. Similar trends are observed between the Saltley composite and dry weather averages but the change is not as great.

The discharge used to calculate the mass flux was derived using mean values for baseflow discharge at Bescot plus known mean discharges. In addition to this the average baseflow increment of 3.6 m^3d^{-1} per metre of channel was added over the 10.3 km from Bescot to Perry Barr and the further 7.4 km to Saltley. For comparative purposes, readings from the 22/5/01 discharge measurements were also used to calculate the mass flux. Both sets of discharge values used show a similar increase of 17-22% but the mass flux is much lower for the 22/5/01 and it is unknown which, if either, is more appropriate. The mass flux reveals a complex story that is not immediately obvious from consideration of the concentration data alone. Some of the determinands using the 22/5/01 discharge data, Cd and $Cu_{(dissolved)}$ and $Cu_{(total)}$, exhibit an overall reduction in mass flux which may be due to sorption or settling of particulates. NH_4 shows an increase of between 2 and 21 kgd^{-1} even though some losses will have occurred by the ready oxidation of NH_4 to NO_3. The most significant increases in metal mass flux are for Zn (2.5-5 kgd^{-1}, Ni 0.5-2 kgd^{-1} and Cr 0.5-0.8 kgd^{-1}). It is not known whether the additional mass flux is in dissolved form but it is not unreasonable to assume that groundwater is a significant contributor. The values for Zn and Ni are in a similar range to those derived from the baseflow and riverbed piezometer data (Table 8.1) but the mass flux of Cr is considerably higher than the 0.03-0.1 kgd^{-1} calculated from the riverbed data. This suggests either a significant groundwater source not detected in the riverbed sampling or that Cr enters as a surface water or pipe end discharge either in dissolved or particulate form.

As discussed in the previous chapter, the possibility exists that acidic groundwater discharge may mobilise contaminants from within the metal rich riverbed sediments. The mass loading appears to be low enough not to have a significant adverse impact on the overall surface water quality under current conditions. For example, the maximum mass flux for zinc of 5 kgd^{-1}, if added to a typical dry weather flow of 166 Mld^{-1} at Perry Barr would produce a rise in concentration of only 0.03 mgl^{-1}. The study of mass flux has so far focussed on a general contaminant contribution across the study reach. It would be beneficial to consider the possible contributions from some of the individual contaminant plumes identified and to assess whether any remediation is worth attempting on an individual plume basis.

8.4 Mass flux from individual contaminant plumes.

The calculation of mass flux from a discharging groundwater plume requires specific information on:

- the spatial extent of the plume discharge area;
- the distribution of contaminant concentration values within the plume;
- the spatial distribution of groundwater discharge rates to the surface water mixing zone across the plume;
- the impact of retardation processes such as sorption on the mass flux;

- the control of natural attenuation on the mass flux;
- an indication of the temporal variability of all the parameters.

To obtain this information, from what is usually a very heterogeneous system, is very time and data intensive. The data set from the riverbed survey is not sufficient to allow anything more than a simplified calculation for the one major organic and two inorganic plumes identified, with the contaminants considered as conservative. Selecting the correct value for plume discharge is problematic; the Darcy specific discharge calculated for individual riverbed piezometers (Section 5.6.1) showed a high degree of variability (–0.01 to 2.53 md^{-1}) even within the same vertical or horizontal profile. Groundwater modelling of flow to the river meander bend on which the plumes were located showed a range in discharge of –0.5 to 6.5 $m^3\ d^{-1}$ per metre of channel. The mass flux for each plume was calculated for a discharge of 1 m^3d^{-1} per metre of channel ignoring the lateral extent of plume discharge across the channel. This discharge was compared with the values derived based on the field estimates of Darcy flow and radial flow where available, and the discharge predicted by the groundwater modelling for the 36 m^2 cell in which the profile was located. The percentage of the mean total mass flux to the river represented by the plume was calculated and the effect the plume would have on surface water concentrations under a typical baseflow of 166 Mld^{-1} was estimated.

The first of the two major inorganic plumes (Profile 13) showed elevated concentrations of contaminants across the entire (~10 m wide) lateral profile of four riverbed piezometers with EC values >1000 μscm^{-1}. High concentrations were also found in the piezometer located on the eastern bank and no piezometer was available on the western bank.

Estimate of mass flux from the plume detected in Profile 13							
	Cl	SO_4	Ca	Na	Cd	As	Ni
Maximum concentration in the plume (mgl^{-1})	1873	275	414	704	0.0027	0.0035	0.304
Mass flux (kgd^{-1})	486980	71500	107640	183040	0.702	0.91	79.04
Change in surface water concentration *1 (mgl^{-1})	2.93	0.43	0.65	1.10	0.000004	0.000005	0.0005
Percentage of mean daily mass flux to the 7.4 km study reach *2	12.7	1.2	2.7	7.6	3.5	1.1	6.6
*1 Under typical flow conditions of 166 Mld^{-1} and a total plume discharge of 260 m^3d^{-1}. *2 Based on mass flux calculated from baseflow analyses and riverbed piezometer samples.							

Table 8.5 The estimated contaminant mass flux from the plume identified at Profile 13

It is not straightforward to identify a source area for the contamination, particularly in the shallow (<9m) riverbank piezometer. The groundwater modelling indicated an easterly regional flow direction across the river, which flows south to north in this location. However localised shallow flow in a westerly direction towards the river may occur on the eastern bank though an accurate survey of river level is not available to confirm this. The source of the contamination could therefore come from either or both banks. There was no evidence of the plume in the riverbed profiles either 220 m upstream or 300 m downstream and the plume was hence estimated to discharge over a channel length of 260 m in the absence of closer piezometer data. The maximum concentrations of the significant contaminants were used to calculate the mass flux to the river (Table 8.5) at a discharge rate of 1 m^3d^{-1} per metre of channel.

The plume may provide a significant proportion (>5%) of the total mass flux to the surface water for Cl, Na and Ni and should therefore warrant further investigation although the actual elevation of surface water concentrations is small. The most significant impact to the surface water will come from the high concentration of Cl which could account for 13% of the total mean mass flux of chloride to the river calculated from baseflow analyses and mean riverbed concentrations. Also it could account for 7-17% of the chloride load calculated from the combined discharge measurements and surface water sampling. The variability in upstream pipe discharges and analytical errors make it difficult to distinguish any impact from the plume on the longitudinal surface water sampling profiles. However, the rise in Cl concentrations from 160 to 169 mgl^{-1} in 100 metres on 11/5/01 may perhaps be an indication of the plume discharge.

The second major inorganic plume was located discharging from one bank of the river at Profile 8, where elevated concentrations of contaminants, in particular F, were detected in two of the riverbed piezometers extending ~4 m across a channel width of 12 m . High concentrations were also found in the piezometer located on the western bank. Regional and local flow modelling (Sections 6.4 and 6.5) and field measurements show flow to the river occurring from both banks putting the source of the contamination on the western bank. Average flow estimates for the entire profile were Darcy flow 10.7 m^3d^{-1}, Radial flow 1 m^3d^{-1}, and regional groundwater flow modelling of 3-5 m^3d^{-1}. The high levels of F were not detected in the profiles 180 m upstream or 120 m downstream. A plume discharge length to the river of

150 m was used, together with the maximum detected concentrations in the plume, and a discharge of 1 m^3d^{-1} per metre length of channel to calculate the impact on the surface water (Table 8.6).

Estimate of mass flux from the plume detected in Profile 8								
	F	NO_3	SO_4	Al	Pb	Cu	Zn	Ni
Maximum concentration in the plume (mgl^{-1})	198	233	470	33	0.011	0.097	0.294	0.139
Mass flux (kgd^{-1})	29700	34950	70500	4950	1.65	14.55	44.1	20.85
Change in surface water concentration ^{*1} (mgl^{-1})	0.18	0.21	0.42	0.03	0.00001	0.00009	0.0003	0.0001
Percentage of mean daily mass flux to the 7.4 km study reach ^{*2}	4.9	2.3	1.2	4.3	1.0	4.9	2.0	1.7
^{*1} Under typical flow conditions of 166 Mld^{-1} and a total plume discharge of 260 m^3d^{-1}. ^{*2} Based on mass flux calculated from baseflow analyses and riverbed piezometer samples.								

Table 8.6 The estimated contaminant mass flux from the plume identified at Profile 8.

The mass flux to the surface water is significant for both F and Cu, amounting to ~ 5% of the total mass flux to the study reach. If higher values of discharge are used, such as indicated by the modelling, then the contribution to the NO_3, Al, Zn, Ni surface water loading also becomes significant. The source of the Cu, Zn, and Ni in this plume may be from the riverbed sediments themselves where the discharge of acidic groundwater has mobilised the metals (Section 7.5) and other acidic plumes may cause a similar effect. The F concentration used was the maximum value from four different sampling events. The concentrations from the other events were much lower (~ 20% of the maximum) demonstrating the significant temporal variation in mass flux to the river. At the maximum concentrations, F would produce a change in surface water concentration of 0.18 mgl^{-1} which would be undetectable through the ‘noise’ in the surface water concentration profiles. A rise in fluoride concentrations from 1.3 to 1.7 mgl^{-1} was detected during surface water sampling downstream across the plume (on 11/5/01) but it is unclear whether this is related to the plume discharge.

The major organic contaminant plume was also identified in Profile 8 with a similar lateral extent across the riverbed. It was not identified in the riverbed profile 180 m upstream but was identified 120 m downstream. The next riverbed profile was > 1 km downstream and so an estimate of 90 m was made for the continuation of the plume downstream from Profile 9. A plume discharge length to the river of 300 m was used together with the maximum detected concentrations in the plume and a discharge of 1 m^3d^{-1} per metre length of channel to calculate the impact on the surface water (Table 8.7).

The plume provides a significant contribution to the total surface water mass loading for all of these compounds, in particular TCA and DCA, but changes in the surface water concentrations resulting from the plume discharge are unlikely to be detectable.

Estimate of mass flux from the plume detected in Profiles 8 and 9				
	TCE	1,1,1-TCA	1,1-DCA	Cis 1,2-DCE
Maximum concentration in the plume (mgl^{-1})	0.052	0.11	0.025	0.011
Mass flux (kgd^{-1})	15.6	33	7.5	3.3
Change in surface water concentration *1 (mgl^{-1})	0.0001	0.0002	0.00005	0.00002
Percentage of mean daily mass flux to the 7.4 km study reach *2	5.2	16.5	25.0	3.3
*1 Under typical flow conditions of 166 Mld^{-1} and a total plume discharge of 260 m^3d^{-1}. *2 Based on mass flux calculated from baseflow analyses and riverbed piezometer samples.				

Table 8.7 Mass flux of organic contaminants from the plume detected in Profiles 8 and 9.

8.5 Concluding Discussion

The Birmingham Aquifer receives a significant pollutant flux from contaminated land and other urban sources and, under current mean baseflow conditions, 59% of the total groundwater discharge from the aquifer ends up in the River Tame (41% abstracted). Considerable temporal variability was observed in the surface water mass flux coming downstream and the groundwater mass flux from individual contaminant plumes. The lowest mass flux within the river occurs in the summer under low baseflow conditions and the maximum mass fluxes in the river are associated with rainfall events that appear to be a limiting factor on river quality. The impact of the groundwater contaminant flux on the surface water is limited by the effect of dilution and natural attenuation processes. Natural attenuation may be particularly important for the chlorinated solvents and heavy metals which display concentrations in the discharging groundwater which are lower than other data such as abstraction well sampling and estimates of recharge loading suggest. Further investigation is required to determine if the most significant natural attenuation occurs within the source zone, the aquifer or the riverbed.

Agreement between the different methods of mass flux estimation varied between determinands but values were generally within one order of magnitude of each other. For example, annual mass flux calculated from baseflow analyses and riverbed piezometer data for Ca, Cl, TCE and Zn were 1,300,000, 1,270,000, 95 and 719 kgy^{-1} respectively. This compares with 1,450,000 (Ca), 1,170,000 (Cl) and 183 (TCE) kgy^{-1} calculated from the combined surface water sampling and discharge measurements. The mass flux of Zn estimated from the regular Environment Agency surface water sampling ranged between 897 and 1,762 kgy^{-1}. The calculations are based on limited data sets and further work is required to assess the requirements for sample density. Some of the contaminants (e.g. F, TCA) may receive a significant contribution to the total mass flux from a single plume while other contamination (e.g. NO_3) is more diffuse. Very detailed investigations are required for individual plumes to obtain an accurate estimate of mass flux. However, based on limited calculations, none of the plumes by itself will raise surface water concentrations above the EQS limits and the principal impact of the plume will be on the ecology within the specific discharge zone where locally high concentrations may exist prior to dilution by surface water.

All the methods of groundwater flux estimation should be used in conjunction, as none in isolation is sufficient to assess the impact of the groundwater on the surface water quality in an urban setting. Direct measurement of changes in discharge and concentration along the river provides mass flux in the surface water but includes inputs from sources other than the groundwater and is susceptible to the temporal variations in pipe discharges. Riverbed piezometers directly measure the concentrations in the groundwater discharging to the river and are useful for delineating localised areas of plume discharge which surface water sampling does not. However, the mass flux calculation relies on estimates of groundwater flow rather than direct measurements, and a wide range in estimated flows may be calculated depending on the method used (Chapters 6 and 7). River baseflow and groundwater discharge display considerable temporal variation through the year and it is necessary to take this into account by using continuous discharge monitoring. This may be combined with continuous EC measurements so as to estimate the variability in surface water mass flux.

Ideally, continuous discharge and EC data would be available from the upstream and downstream boundary of the aquifer and from the major surface discharges such as tributaries and sewage treatment works. Regular surface water sampling could be conducted at the gauged points and converted directly to a surface water chemical mass flux. The increase in surface water mass flux at points across the aquifer could be determined more accurately from the combined sampling and discharge measurements if the temporal variations in mass flux entering the upstream limit of the study reach were known. These measurements could be combined with a surface water tracer test (Kimball et al., 2002) to reduce the uncertainty in the results. The riverbed piezometer monitoring network should be extended to identify any narrow, high concentration, plumes that may have been missed, and to provide more reliable mean groundwater concentrations. Geostatistical analyses should be performed to determine the optimum sample density. Further investigation is necessary on the sensitivity of the measured concentration and flow (chemical mass flux) to the lateral position of the piezometer within the channel. The greatest groundwater discharge is likely to occur close to the riverbank and comprises shallow perhaps more contaminated groundwater than older water discharging through the centre of the channel and this is significant to the mass flux estimation.

The contribution to surface water mass flux from the Birmingham Aquifer is estimated as 6.7% of the total dissolved mass flux in the surface water at Water Orton downstream of the study area. This estimate is based upon a mean baseflow discharge of 308 Mld^{-1} at Water Orton, a mean groundwater discharge of 19 Mld^{-1} from the aquifer and TDS in the surface water, and groundwater of 1,014 mgl^{-1} and 1,101 mgl^{-1} respectively. The effect of discharge from the aquifer is therefore relatively minor owing to the dilution effect and the fact that the TDS content of the water moving downstream is already high. The impact of discharge from the aquifer is dependent on conditions within the overlying river system. The Tame rises in an urban catchment and receives numerous discharges from sewage treatment works and other industry that significantly increases contaminant loading in the river before it reaches the Birmingham Aquifer. However if the aquifer were located immediately downstream of a rural catchment then the impact would be more significant. For example, using a surface water TDS content of 245 mgl^{-1} from Sutton Park, the contribution of the aquifer to total mass flux at Water Orton would increase to 23%.

Calculations of the groundwater contribution to surface water mass flux using gauging station data on discharge and EC indicate that on the catchment scale groundwater (mean estimated TDS 809 mgl^{-1}) is diluting the surface water (mean estimated TDS 924 mgl^{-1}) implying an improvement of surface water quality. Results of the 2001 water quality survey show mean EC (976 μScm^{-1}) in the riverbed piezometers across the 7.4 km reach overlying the Birmingham aquifer to be higher than in the surface waters (2001 mean is 723 μScm^{-1} and 848 μScm^{-1} for 1998 gauging station data at Water Orton). This indicates the groundwater discharging from the aquifer is detrimental to the surface water because it contains a higher TDS content than the catchment average. However, owing to dilution this has a limited impact at the catchment scale.

The groundwater does contribute to an increase in the surface water mass flux for all determinants across the aquifer, but no conclusive rise in surface water concentrations for any of the inorganic determinands was observed. An increase in the surface water concentrations of the chlorinated solvents, primarily TCE, was observed across the aquifer though at low levels (<10 μgl^{-1}). The most significant contributions in terms of estimated mass flux are, for anions 3,300 ty^{-1} HCO_3^- and 1,900 ty^{-1} SO_4, cations 1,300 ty^{-1} Ca, heavy metals 719 kgy^{-1} Zn and chlorinated solvents 95 kgy^{-1} TCE. As discussed in the previous chapter the distribution of contaminant concentrations in the riverbed piezometers is variable and many of the determinands often display lower concentrations in the groundwater than in the surface water. However, high concentrations (plumes) detected at a few localities tend to give the mean groundwater concentration an upward bias. All available guideline limits are set in terms of concentrations rather than mass flux which may not be ideal when comparing the impact of individual contaminant plumes on the surface water system. A high concentration, low-flux plume may be important locally to ecology within the plume discharge area, but a low concentration high-flux plume may have a more significant impact on the general surface water system.

Based upon the limited data set it would seem that the groundwater discharge from the Birmingham Aquifer does not significantly reduce the current surface water quality. In some cases the groundwater may improve surface water quality by diluting sewage effluent discharges and upstream sources of Cu, Zn, Ni. The Tame is at present categorised within the worst classes E/F (poor/bad) under the General Quality Assessment (Environment Agency, 1999), primarily due to excessive levels of biological oxygen demand (B.O.D) and ammonium from pipe end discharges. Should these pipe end discharges be halted the river quality is likely to improve although the minimum baseflow (180 Mld^{-1}, Section 6.3.2) at the downstream end of the study reach may be reduced by 50%. Groundwater discharge from the aquifer will supply a contaminant loading to the river from urban sources but not at levels sufficient to break the EQS limits, except in localised areas of the riverbed where high concentration contaminant plumes are discharging.

CHAPTER 9. CONCLUSIONS AND FURTHER WORK

9.1 Introduction

The purpose of the Birmingham Aquifer – River Tame case study was to investigate the impact of contaminated land and groundwater on urban river systems. The research had the following objectives.

1. To characterise and quantify the contribution of groundwater-derived contaminants to the surface-water quality of an urban river at the subcatchment scale.
2. To investigate the physical and chemical processes controlling contaminant flux across the groundwater/surface-water interface.
3. To investigate the processes controlling the temporal and spatial variations in groundwater flow and contaminant flux to the river.
4. To develop suitable monitoring methods to quantify contaminant flux to the river.

These objectives were, for the most part, achieved successfully.

A large database was amassed concerning water quality and levels, obtained by sampling surface waters, using shallow piezometers in the riverbed and across the aquifer, supplemented by data from deeper abstraction and monitoring wells. Temperature probe surveys and sediment coring of the riverbed were undertaken. Surface water flows were investigated using archived gauging station data from the Environment Agency and discharge measurements along the river. Computer modelling was used to investigate groundwater flow to the river at several scales during dry weather flow and during a river flood event. Analysis of the chemical quality (and flow) data sets was undertaken to discern trends, quantify mass fluxes and understand physical-chemical controlling processes.

9.2 Conclusions

Detailed conclusions specific to each aspect of the work have already been summarised in the relevant chapters (6,7,8,9). Key conclusions arising from the study and their generic relevance are summarised for each of the study objectives below.

Objective 1. To characterise and quantify the contribution of groundwater-derived contaminants to the surface water quality of an urban river at the subcatchment scale.

Groundwater comprises up to 60% of the dry-weather flow at the downstream end of the study reach. The remainder is derived from pipe-end discharges which contain a high proportion of imported water from outside the catchment. At the downstream end of the study reach an estimated 6% of the mean dry weather flow and 6.7% of the surface water geochemical (inorganic) mass flux is derived from the Birmingham Aquifer and the overlying drift deposits.

Urban groundwater beneath Birmingham, particularly the shallow groundwater, is contaminated relative to the natural background levels as measured in Sutton Park. However, the estimated contaminant mass flux from the groundwater to the river was not sufficient to cause a significant reduction in the current surface water quality based on the U.K. environmental quality standards for freshwater. In some cases the groundwater improved the surface-water quality by diluting sewage effluent and industrial pipe discharges. The impact of the groundwater on the river was found to be scale dependent. High concentration contaminant plumes were observed within the riverbed which may have a localised impact on the ecology, but the large dilution within the river limited the impact on the overall surface-water quality. There is some evidence that natural attenuation within the aquifer and/or the riverbed limits the quantity of organic and inorganic contaminants entering the river by the groundwater pathway.

Objective 2. To investigate the physical and chemical processes controlling contaminant flux across the groundwater/surface water interface.

Surface water was observed to penetrate the riverbed and mix with the groundwater in the hyporheic zone to depths of up to 60 cm . Rapid changes in pH, Eh and dissolved oxygen content occurred across the groundwater/surface water interface leading to precipitation or sorption of some inorganic contaminants, probably to iron oxyhydroxide grain coatings. Zonation of some processes occurred within the interface over a limited (tens of centimetres) vertical extent. Dissolved oxygen levels were generally lower in the riverbed than the groundwater or surface water, probably due to microbial activity, and there was evidence of nitrate and sulphate reduction in some localities from both the groundwater and the surface water. There was no conclusive proof of the biodegradation of organic contaminants such as chlorinated solvents within the riverbed, but the presence of daughter compounds indicated that biodegradation of PCE/TCE had occurred at least somewhere in the groundwater system. Natural attenuation within the riverbed does not occur in all situations and may be limited by rapid rates of groundwater discharge through the more permeable bed sediments.

The groundwater computer modelling and sediment coring indicate that the riverbed sediments do not limit the groundwater discharge to the river. During river flood events the groundwater system was observed to react quickly, preventing the uptake of surface water as bank storage. The rate of change in groundwater levels in unconfined material within the riverbank was partly dependent on the extent of the capillary fringe and the soil moisture history and is subject to unsaturated flow processes. The dammed groundwater released when the river stage drops contributes to the falling limb of the hydrograph. Urban river conditions are often at their most extreme during wet weather in terms of poor water quality and high flow velocities, and an understanding of the groundwater contribution to these events is important.

Objective 3. To investigate the processes controlling the temporal and spatial variations in groundwater flow and contaminant flux to the river.

The river baseflow and the groundwater contribution to it followed a generally exponential decay between recharge events, with the lowest flows occurring during the summer months. The surface water quality (in terms of total dissolved solids) deteriorated during these low-flow periods, with less groundwater dilution of the poor quality pipe-end discharges that the river receives. An average of 30% of the river flow at the downstream end of the study reach comprises sewage effluent which displayed considerable variation in discharge levels during a single day.

The groundwater contribution to the river from the Birmingham Aquifer has increased and apparently stabilised since the cessation of historically high levels of abstraction (75 Mld^{-1},

1945) which caused significant draw-down and the probable recharge of the aquifer by surface water. If current abstractions were to cease (average rate of 13 Mld^{-1}), the groundwater discharge to the surface water could potentially increase by up to 70%. Furthermore, if pipe-end discharges were also to be reduced, the contribution of contaminated groundwater to the river would become more significant.

Large temporal variations in contaminant concentrations were observed within one of the discharging groundwater plumes. These may be related to changes in the source term, geochemical transformation and/or other factors such as the levels of recharge. The maximum mass fluxes within the river are associated with short term (<2 days) rainfall events that appear to be the limiting factor on river quality.

Groundwater discharge to the river is variable along and across the channel, depending on the situation of the river within the regional groundwater flow system. The highest groundwater discharge to the river is usually concentrated through the sides of the channel, discharge through the central channel and seepage face being of secondary importance. The anisotropy in the hydraulic conductivity of the material adjacent to the channel has a significant effect on the distribution of groundwater discharge to the channel.

The River Tame is the discharge point for converging groundwater flows which may have a wide range of travel times, origins and contaminant levels. This will have contributed to the substantial lateral variations in water quality within the riverbed that were observed. When estimating contaminant flux to the river, the variation in groundwater concentrations and flow must be considered when using point-source sampling methods such as riverbed piezometers.

A strong association was observed between land use and the distribution of contaminants within the groundwater. Specific contaminant plumes were identified in the riverbed associated with nearby industry or contaminated land e.g. TCE associated with the metal-working industry. Other contaminants were more diffuse and widespread e.g. chloride from road salting. A large amount of industry is now, and historically has been, located adjacent to the river. With a narrow unsaturated zone, often permeable alluvial deposits and short groundwater flow paths, the river is at greater risk of contaminated groundwater discharge from these sites than from sites further away on the valley flanks.

Although the estimated contaminant flux from the groundwater to the river is not sufficient to cause a significant reduction in current surface-water quality, in the long term the groundwater, as a source of contaminants, does present a significant limiting factor in the future improvement of surface-water quality. Improvements in pipe-end discharge quality may be achieved relatively rapidly but improvements in groundwater quality will take significantly longer. The aquifer will continue to receive significant pollutant loading, via recharge in the urban environment, and long- term sources such as DNAPL pools are already known to be present within the aquifer.

The impact of the urban groundwater aquifer is dependent on conditions within the overlying river system. The Birmingham Aquifer discharges to a river that has risen in an urban catchment and the groundwater does not significantly alter the surface water quality. However, if Birmingham were located near the head-waters of a rural catchment, then the contribution of contaminant flux from the groundwater would be more significant.

Objective 4. To develop suitable monitoring methods to quantify contaminant flux to the river.

The use of riverbed piezometers was found particularly useful in this study, both for assessing the quality of the groundwater that is discharging to the river, and for investigating specific contaminant plumes. The riverbed coring device successfully provided samples of bed sediment to a maximum depth of 50 cm. The temperature probe proved to be a useful tool in characterising the riverbed and indicating areas of groundwater inflow. The pressure transducers that were constructed were economical and proved useful in measuring variations in water level in the narrow (20 mm I.D.) riverbank piezometers. The seepage meters, however, proved of limited use, mainly because of fast river flows and the ‘armoured’ nature of the riverbed. They need to be adapted to the conditions if they are to be used in the future. The techniques of riverbed investigation applied in this study may be more difficult to use on larger rivers, but because groundwater flows tend to be concentrated at the shoreline/channel sides they could still prove useful.

9.3 Policy Implications and the Water Framework Directive

The EU 2000 Water Framework Directive (WFD) requires integrated management of surface waters and groundwaters within a river basin, knowledge of the groundwater and surface water interactions, and the exchange of flows and contaminants that may occur. Data on the likely contribution of urban groundwater to the surface-water system are therefore vital to the understanding of each river basin and the necessary tools and data collection procedures must be developed to provide this knowledge. Research on the River Tame in Birmingham has given an insight into monitoring strategies and the problems associated with estimating contaminant flux to an urban river via the groundwater pathway.

Typically the quality of a river is determined by regular sampling of surface waters at several widely-spaced points using simple concentration values rather than total mass flux. This approach is unlikely to identify individual contaminant plumes discharging to the river due to dilution effects and masking by upstream inputs. The sampling programme along a 7.5 km section of the River Tame utilising piezometers in the river bed located discharge from plumes of both organic and inorganic contaminants which were not identified by the detailed surface-water sampling programme.

The WFD indicates that contaminant concentrations should be measured and regulated before dilution occurs at the discharge point to the receptor. To assess the total mass of contaminant flux from the plume to the river requires definition of the spatial extent of the plume, concentration values and discharge rates. This is time and data intensive and requires consideration of the temporal variability and the effect of any attenuation processes that may occur across the groundwater surface water interface.

Current UK water-quality guidelines are generally expressed as maximum concentration limits related to the possible toxic effects on humans and the wider ecology. However, to allow effective management of a river system, knowledge of the geochemical mass flux balance is required, though it is more problematic to obtain.

On a catchment scale, geochemical mass flux in the surface water can be readily obtained by linking regular water quality sampling with discharge measurements from the Environment Agency gauging stations, a practice which does not appear to happen at present. Useful estimates of geochemical mass flux in the surface water can also be obtained by combining the continuous discharge and conductivity measurements available at some gauging stations.

The groundwater contribution to the total surface water geochemical mass flux may be estimated from the surface water data or calculated using groundwater discharge estimates from computer modelling or direct measurement and groundwater quality data. A suitable monitoring network should be designed to sample the groundwater system that is discharging to the river. The Birmingham study has shown that a considerable difference in water quality exists between the deep abstraction wells and the shallow and riverbed piezometers, implying that the discharge to the river is dominated by the shallow groundwater. Ideally, *in situ* chemical flux measurement devices should be developed for emplacement in the riverbed as part of the network.

Data should be collected to establish whether the groundwater is improving or reducing the surface water quality. If groundwater is sustaining the river quality then restrictions on abstraction are prudent to safeguard the discharge levels.

The Baseflow Index (BFI) is often used as an initial screening tool to assess the importance of groundwater contributions to surface water flows (Young et al., 2002). However, care must be taken to assess the influence of industrial discharges especially near the head of a catchment where the contribution from these sources may be significant.

When dealing with stretches of river that may be at risk from the discharge of contaminated groundwater, regulatory compliance points must be set. These should be specified as maximum levels of both mass flux and concentration. The question is then to decide where the compliance point should be set, within a monitoring piezometer on the riverbank, within the riverbed (which may incorporate natural attenuation processes) or within the surface water column after dilution has occurred?

Further research is required to develop the necessary knowledge and monitoring techniques to allow effective implementation of the Water Framework Directive.

9.4 Further Work

Although suggestions for further work are centred upon the Birmingham Aquifer - River Tame system, many of the suggestions could be undertaken in other groundwater/surface water settings.

Further work on estimating the groundwater contribution from the Birmingham Aquifer to contaminant flux in the surface water could include the following considerations.

1. Increasing the sample density of the riverbed piezometers to see if the estimated geochemical mass flux from the groundwater changes significantly with the discovery of further contaminant plumes. (The high concentration core of a plume may be very local and easily missed.) Flux estimates calculated from the riverbed data could be compared with those derived from further combined sampling and discharge measurements in order to determine the most effective method of groundwater flux estimation. Flux estimates for

other compounds of interest, e.g. pesticides and PAHs, should also be included. Work should be undertaken to calibrate the continuous conductivity measurements taken at Water Orton gauging station for comparison with surface-water quality.

A survey with relatively short sample spacing (50-100m) using a removable drive-point piezometer, pump and low volume flow cell could be undertaken to measure conductivity and pH in the riverbed to identify 'hot spots' for detailed sampling. Geostatistics could be performed to obtain the optimum sample spacing required to intercept the major plumes and obtain an improved estimate of the groundwater geochemical flux to the river.

Further attempts should be made to quantify the contribution from other sources such as the sewage treatment works to the surface water geochemical flux. Long-term monitoring for chlorinated solvents and heavy metals could be undertaken using semi-permeable membrane devices on the riverbed.

2. Tracer experiments could be performed within the riverbed to determine rates of groundwater discharge. A surface water tracer test of known mass should be combined with surface-water sampling to determine the dilution (groundwater inflow) and increase in mass flux for individual determinants along the reach. This method would go some way to removing the 'noise' introduced by the daily variations in sewage works discharges. The survey could incorporate not only stations on the river bank but also a mobile boat unit with fluorimeter, data-logger and GPS which could follow the tracer plume down river, repeatedly traversing and sampling it.

3. A better understanding is required of the relative contributions of groundwater to the river flow from the bedrock units (Coal Measures, Triassic sandstone and mudstone) and the overlying alluvial drift (gravel) which is present along the entire reach. This would be in the form of a mass balance to determine from where the other 54% of flow in the Tame is derived. This could be achieved by further measurements of discharge accretion along the reach, perhaps in conjunction with temporary stilling wells which could be used to remove the effect of the sewage treatment works. A study of the groundwater-supported tributaries of the Tame lying completely within the aquifer boundaries will increase the knowledge of the quantity and quality of aquifer baseflow.

4. Natural attenuation within the riverbed could be investigated by geochemical modelling and column experiments to examine the biological, physical, and chemical changes that occur across the groundwater/surface-water interface. Laboratory work could be linked with field work on known plumes to investigate biodegradation of chlorinated solvents and the transport of heavy metals.

   Finer resolution (<5 cm) is required in the vertical sample profile up to the surface of the river bed. Experiments on redox zonation and precipitation reactions could be undertaken in the riverbed by installing a column with an inert fill. A wider distribution of multilevel arrays would improve the data on the spatial variability/heterogeneity of the plume discharges and natural attenuation processes, including nitrate reduction, which has been observed in some locations. This could be undertaken in conjunction with an investigation of the amount of surface water passing through the hyporheic zone to allow an assessment of the importance of this zone in reducing surface water contamination. Estimates of groundwater and surface water residence times within the groundwater/surface water interface should also form part of this study.

5. Investigation should be made to further confirm that the most significant groundwater contribution to the surface water contaminant flux in Birmingham, comes from diffuse urban pollution rather than a few specific plumes which may be located and remediated. Some of the known plumes could be fully characterised in order to determine their full extent and to quantify their contaminant flux to the surface water. The fluoride plume detected in the current survey could be effectively delineated using a temporary drive-point piezometer and a specific ion probe. Instrumentation could be developed to measure directly, seepage and water quality in the Tame environment, both from the riverbed and through the river banks which may be very significant areas of discharge.

6. An ecological assessment of the riverbed could undertaken in relation to the specific plume studies and in other areas where water quality is known to vary between opposing banks.

7. The age and origin of the groundwater discharging across the riverbed could be investigated to determine whether deeper and cleaner groundwater is mixing with, and diluting, shallow contaminated groundwater. Predictions of the future variations in the quality of the discharging groundwater could be incorporated in the study. This would be accomplished by using deep multilevel piezometers adjacent to the river, detailed lateral multilevel sampling across the riverbed and through the use of isotope dating.

8. The existing regional groundwater models (Greswell, 1992 and Robinson, 2001) could be refined and used in conjunction with groundwater quality data to obtain better estimates of geochemical mass flux to the river. The model should include increased resolution along the river and calibration against surface water hydrographs. Abstraction well, land use, rainfall and pollutant-loading data could be combined in the model to develop a geochemical mass balance for the aquifer.

9. Computer modelling of the flood plain and river channel could be undertaken to examine the extent of the hyporheic zone, the surface water exchange with it, and the variation in residence times of both groundwater and surface water within this zone. The model could be used to simulate flood events to determine the contribution of groundwater to surface water flows, which has relevance to flood defence planning and surface-water quality during these events. The modelling should be three dimensional, incorporating the hyporheic zone, the effect of recharge and unsaturated flow processes. The effect of an increase in riverbed conductivity due to particle entrainment under high flows could also be examined.

10. Further information is required on the processes controlling the variability of groundwater contaminant flux to the surface water over time, on both the catchment scale and for individual plumes. Repeat sampling of riverbed piezometers and surface waters should be conducted to determine how variations in groundwater discharge quality are related to the level of baseflow and recharge. Continuous conductivity measurements from the Water Orton gauging station may be of use in this study.

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