Groundwater flows in an urbanised floodplain and implications for environmental management

A thesis submitted for the degree of Doctor of Philosophy

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Declaration

I confirm that this is my own work and the use of all material from other sources has been properly and fully acknowledged.

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Gooddy, D.C., Macdonald, D.M.J., Lapworth, D.J., Bennett, S.A., & Griffiths, K.J. (2014). Nitrogen sources, transport and processing in peri-urban floodplains. *Science of the Total Environment,* 494, 28-38.

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Ascott, M.J., Marchant, B.P., Macdonald, D.M.J., McKenzie, A.A., Bloomfield, J.P. (2017). Improved understanding of spatio-temporal controls on regional scale groundwater flooding using hydrograph analysis and impulse response functions. *Hydrological Processes*, 31(25), 4586-4599.

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Abstract

With population growth, the large lowland floodplains of our major rivers have become increasingly urbanised. Environmental issues have arisen with the juxtaposition of these urban developments and the aquatic and terrestrial ecosystems associated with the rivers and their floodplains. The floodplain sediments are often highly permeable, and hydraulically well-connected to the water courses, and therefore the interaction of the urban environment and groundwater is very important.

The overall aim of this PhD is improved understanding of the hydrological regime of urbanised floodplains, in particular groundwater hydrology, leading to better environmental management. The PhD uses as a case study, the floodplain of the River Thames in the city of Oxford. Through surveys, data collection via an extensive monitoring network, and the development of conceptual and numerical models, the floodplain has been characterised and the hydrological processes better understood. Focussed studies, working with key stakeholders, have been undertaken relating to fluxes of pollutants into and through the subsurface, and to the role of groundwater in urban flooding.

The research undertaken has resulted in a better understanding of:

- the impact of river management structures on water and nitrate exchange between rivers and floodplain aquifers;
- the influence of legacy waste dumps on water quality in floodplain aquifers, and quantification of the fluxes of associated pollutants to rivers via the subsurface;
- the conditions and mechanisms that control the occurrence of groundwater flooding in urbanised floodplains; and
- the role groundwater and shallow geology play in controlling the duration of flooding in urbanised floodplains, through the development and application of a model system for simulating flooding that links flood inundation and groundwater flow models.

Through these focussed studies a range of generic recommendations are made for environmental managers, as well as recommendations for future work.

Chapter 1: Introduction

1 Introduction

1.1 Riverine floodplains

1.1.1 Riverine floodplain services

Riverine floodplains are areas of low-lying ground adjacent to rivers, formed mainly of unconsolidated river sediment. These floodplains are highly complex natural systems of high biodiversity and societal value, but are often severely degraded and in urgent need of protection and rehabilitation (Erős et al., 2019). Modification and degradation of floodplains is ongoing due to urbanization, navigation, increasing levels of agriculture and the development of major hydropower projects, making large riverine floodplains one of the most threatened ecosystems on Earth (Arthington et al., 2010; Sommerwerk et al., 2010).

Historically floodplains have been attractive locations for urban development for reasons such as their coincidence with sources of water and food and their proximity to transportation routes (Montz, 2000). However, in recent decades there has been a rapid increase in the rate of urbanisation of floodplains (Monk et al., 2019). As a proxy for urban floodplain population, Jongman et al. (2012) estimated the global population in 2010 exposed to a 1 in 100 year return period fluvial flood as 805 million, approximately twice the equivalent population in 1970. EEA (2018) states that 15% of Europe's population is located on floodplains; in Austria, the Netherlands, Slovakia and Slovenia this figure rises to more than 25 %. Kummu et al. (2011) estimated that more than 50% of the world's population lives within three kms of freshwaters. Grizzetti et al. (2017), through an assessment of multiple pressures on European rivers, identified the area of floodplains that have been urbanised as a key predictor of ecological degradation.

As riparian zones, floodplains are defined as ecotones between terrestrial and aquatic realms (Gregory et al., 1991) extending from water bodies and including terrestrial vegetation associated with shallow groundwater (Naiman et al., 2000; Nilsson & Berggren, 2000). In lowland regions, floodplains associated with major rivers can occupy large areas; globally riverine floodplains cover more than 2 x 10^6 km^2 (Ramsar & IUCN, 1999). Floodplains are dynamic systems shaped by repeated erosion and deposition of sediment, flood water inundation, and complex groundwater-surface water exchange processes (Junk et al., 1989; Thorp et al., 2006). This dynamic nature makes floodplains highly biologically productive and diverse ecosystems.

Tockner and Stanford (2002) estimated the worldwide value of the services provided by floodplains as US$3.9 trillion annually, more than 25% of the value of all terrestrial ecosystem services although they cover only 1.4% of the land surface (Mitsch & Gosselink, 2000). In these services, Tockner and Stanford (2002) included flood regulation (37%), water supply (39%) and waste treatment (9%). However, anthropogenic development has had a devastating effect on the ecosystem services provided through floodplains and floodplain ecology, both in the developed and, increasingly, the developing world (Friberg et al., 2017). Expansion in agriculture is a primary reason for the alteration of floodplains. This includes removal of natural vegetation, mobilisation of sediment, drainage of land, and input of pollutants (Poff et al., 1997; Blann et al., 2009; Krause et al., 2011a). In Europe and North America, up to 90% of floodplains are already cultivated (Tockner & Stanford, 2002). Entwistle et al. (2019) report an increase in intensive agriculture in England from 38% of floodplain zones in 1990, to 64% in 2015. In tandem, they report that floodplain wetland areas in the form of fen, marsh, swamp and bog have been all but lost. Substantial efforts are ongoing to address the degradation of floodplain ecosystems and to improve the ecosystem services of floodplains, such as flood mitigation, by returning rivers to a more natural state (Palmer et al., 2014; Guida et al., 2016; Brown et al., 2018). Roni and Beechie (2013) estimated a global expenditure of approximately US$ 3 billion annually on interventions including the remeandering of rivers, riparian revegetation, the removal of flood embankments and weirs to return the connectivity of river habitats, and riverine wetland creation (Gilvear et al., 2013).

In addition to habitat alteration, Tockner et al. (2010) identified the main threats to floodplains and their ecosystems as pollution (discussed in Section 1.1.4), measures for flow and flood control (Poff et al., 1997) and species invasion. Changes to natural flow regimes within river–floodplain ecosystems have come about through the regulation of rivers for the purposes of water supply, hydropower generation, flood management and the development of transport routes. Nilsson and Berggren (2000), at the time, estimated that two-thirds of the fresh water flowing to oceans globally was obstructed by approximately 40,000 large dams and 800,000 smaller ones. CIA (2002) stated that more than 600,000 kms of inland waterways have been altered for navigation worldwide. Nearly all large rivers and their floodplains in central Europe and the USA are affected by dykes, with an estimated 40,000 km of dykes in the USA alone (Johnston Associates, 1989). In central Europe, the floodplain landscapes of the Rhine, the Lower Danube and the Middle Elbe Rivers, for example, have lost more than three-quarters of their natural inundation area as a result of dykes (Leyer, 2005). Within England and Wales, records from the Environment Agency, accessed in 2014, showed that on the 68,755 km of the river network there were 17,569 locks, weirs and control gates (Figure 1).

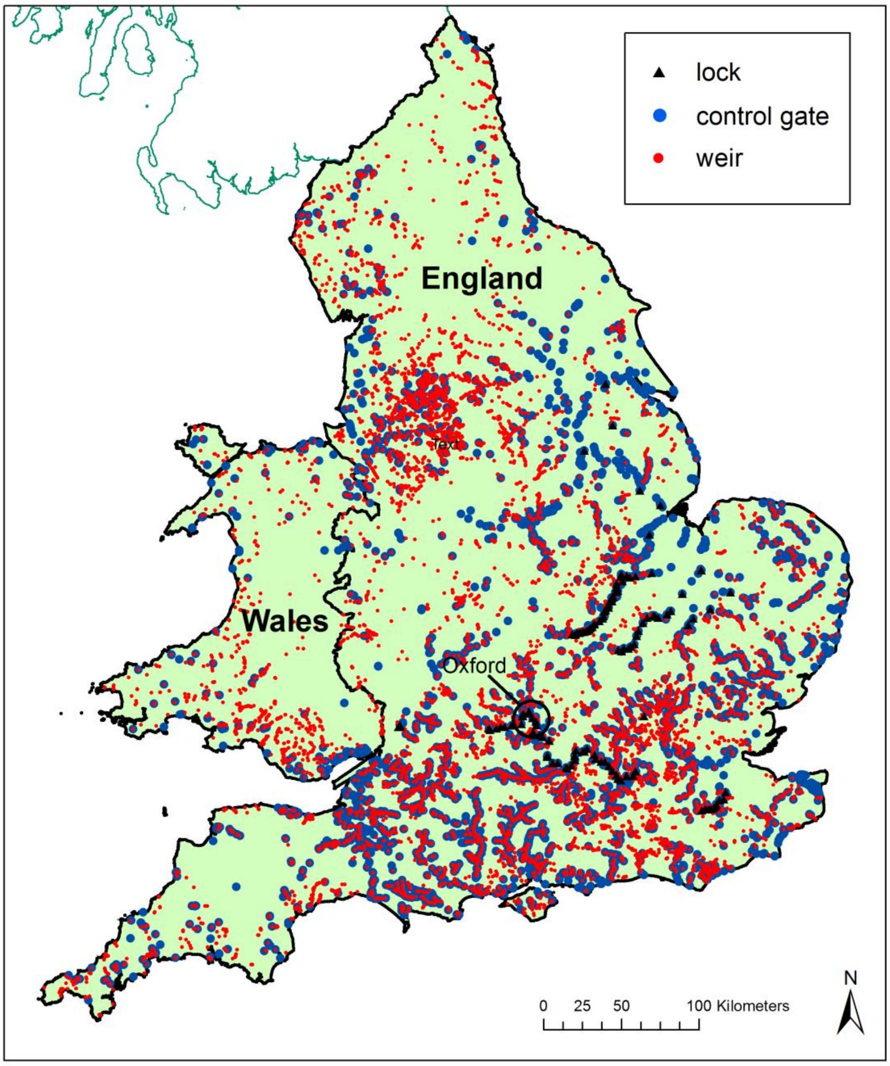

Figure 1 River management structures in England and Wales provided by the Environment Agency (EA), March 2014. These sites appeared on lists of: flood risk management assets, extracted from the EA Asset Information Management System; and locks, held as part of the EA navigation function. Contains Environment Agency data licensed under the Open Government Licence v3.0, and Ordnance Survey data © Crown copyright and database right (2018).

Flow has a major control on physical habitat conditions in streams, which in turn is a primary determinant of biotic composition. Aquatic species have evolved life strategies in direct response to natural flow regimes and the maintenance of natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species (Bunn & Arthington, 2002). The impact of the control on river flows is a reduction in flood peaks, flooding frequency and duration, and a change in the nature of dry periods (Kingsford, 2000; McMahon & Finlayson, 2003). These flow changes reduce lateral connectivity between the floodplain and the river (Diaz-Redondo et al., 2018), disrupt sediment transport (Wohl et al., 2015) and reduce channel-forming flows (Ward & Stanford, 1995).

Inundation–duration curves used to assess the hydroecological integrity of floodplain ecosystems indicate an almost linear relationship between water level and inundated area in natural floodplains (Benke et al., 2000; Van der Nat et al., 2002). In regulated rivers, floodplain inundation tends to be short and the increase in inundation area is abrupt. Along channelled rivers, floodplains only receive surface flooding from the river during major flood events. Therefore, large-scale floodplain development and water-resource development often lead to major decreases in the active floodplain area.

According to Tockner and Stanford (2002), species invasion is the second most important cause of the overall decline in aquatic biodiversity. Less susceptibility to processes that are highly restrictive to natural species is a primary pathway to the dominance of invasive species (MacDougall & Turkington, 2005); this can relate to groundwater conditions. For example, in examining the key drivers of compositional shifts in riparian plant species, Stromberg et al. (2007) identified that the anthropogenic alteration of stream-flow regime in the Gila and Lower Colorado basins in Arizona, USA, favoured introduced species; these species, in contrast to native species, have deep roots and have a narrow germination window, which mean they cope better than the native species with the lower groundwater levels and shifts in flood timing that have been a consequence of river flow regulation. Leyer (2005) demonstrated that reduced water level fluctuation caused by the construction of dams and dykes on the River Elbe, Germany, led to substantial changes in the spatial distribution of floodplain plant species due to native-species preference for highly fluctuating, over stable, water tables. Native species need high groundwater levels at the beginning of their growing period, as they have a strong requirement for soil moisture for growth. In the floodplain study area for this thesis, increases in invasive wetland species in a highly biodiverse and protected floodplain meadow have been linked with long-term changes in groundwater levels and flood frequency (Punalekar et al., 2016).

1.1.2 Groundwater flows in floodplain aquifers

Groundwater-surface water interaction

Floodplains are complex systems where rivers erode and deposit sediments ranging in size from clays to gravels (Bridge, 2009). Significant groundwater flows occur where there are appreciable thicknesses of coarser-grained and more permeable sediments. These sediments can be in hydraulic connection with underlying permeable bedrock, in which case they may form part of a regional groundwater flow system, or they may be isolated by low permeability bedrock, in which case, in riverine floodplains, flows are driven by the interaction with surface water bodies (Woessner, 2000). The latter is the case within the study area used in this thesis.

During low precipitation periods, groundwater tends to discharge to rivers and streams. In periods of high precipitation when flows increase, higher river levels, especially in the lower river reaches, can cause the river to change from influent to effluent condition, infiltrating its banks and recharging the aquifer (Sophocleous, 2002). The interaction of river and floodplain aquifer is influenced by the hydraulic conductivity of river bed material. Naganna et al. (2017) review factors that influence river bed hydraulic conductivity. The factors are many and include: sediment particle size, substratum heterogeneity, longitudinal variations in impervious surfaces such as bedrock and sills, bed material depth, channel geometry, variations in hydraulic radius, and roughness due to natural and anthropogenic modifications. As a result, the range of hydraulic conductivity varies over several orders of magnitude, reflecting variability that can be caused by settling, clogging of fine mineral or organic particles (colmation), and compaction due to the weight of eroded materials. Given the range of controlling factors, river bed hydraulic conductivity is highly spatially variable (Calver, 2001; Irvine et al., 2012) and can result in zones of focussed interaction between river and aquifer (Heeren et al., 2013).

The process of colmation occurs after extended periods of low flow velocity. It can also be induced by algal mats in eutrophic streams and by cohesive depositions in rivers receiving sewage effluent (Brunke, 1999). The removal of this layer, or decolmation, can result from the erosion that occurs during high flow conditions (Wong et al., 2015). Doppler et al. (2007) provide field evidence of the temporal variability of the river bed hydraulic conductivity, identifying that, depending on geometry and hydraulic characteristics of the riverbanks, it can be a function of the river stage, as at higher river levels more permeable zones of the river bed can contribute to infiltration of river water.

The alternating phases of colmation and decolmation are natural processes of sedimentation and erosion, however, the balance may be altered anthropogenically towards enhanced siltation (Petts, 1988; Doppler et al., 2007). The aggregation of fine-grained material associated with lower river velocity upstream of engineered river management structures and scouring of the river bed downstream of these structures, in combination with head gradients that increase and decrease the propensity for colmation, mean river bed hydraulic conductivity under the influence of river structures can be highly variable (Hatch et al., 2010).

As discussed in Section 1.1.1, river flows are often controlled by river management structures. Studies have shown how structures can cause groundwater levels in the associated aquifer to be raised and river reaches to switch from gaining water from the adjacent aquifer to losing water, when structures are introduced (Krause et al., 2007; Hill & Duval, 2009; Matula et al., 2014; Lee et al., 2015). There is some evidence that the influence of river management structures on groundwater levels can be linked to nutrient attenuation in the associated floodplain aquifers, but a limited number of studies have been undertaken on this topic (see section 1.2.1).

River-aquifer-floodplain interaction during high flows
This PhD has focussed primarily on high river flows and flooding on floodplains. Periods of high river flow result in raised river levels and, in some circumstances, out-of-bank flows onto the adjacent floodplain. Field studies have shown that rises in the river stageassociated with high river flow often cause a hydraulic gradient from river to aquifer resulting in temporary storage of river water in the riparian zone. For example, through a series of studies on a stretch of floodplain on the River Severn in the UK (Bates et al., 2000; Burt et al., 2002a; Claxton et al., 2003; Jung et al., 2004), observations and modelling were used to examine groundwater flows in a floodplain aquifer during high river flow events, involving both in-bank and overbank flooding. In both cases, river stage rise was shown to induce rapid responses of the water table over many tens of meters across the floodplain. Groundwater ridges were created, directed towards the base of the hillslope adjoining the floodplain, which switched off hillslope inputs to the riparian zone. In overbank flood events, the rise in groundwater level that occurred beneath the floodplain was a response to the river stage rise rather than a response to flood water recharge (Claxton et al., 2003).

Vidcon (2012) reports a similar study in a floodplain in a second-order stream draining a mostly agricultural watershed in Indiana, USA. During storms, larger water table fluctuations (approximately 100 cm) occurred near the stream, compared with near the toe of the hillslope at the edge of the floodplain (10 - 25 cm). A quick rise in the water table near the stream occurred for all storms studied. Water table fluctuations, groundwater flow velocities and electrical conductivity data indicated that riparian zone water table responses to precipitation were primarily regulated by pressure wave processes. Regardless of the storm, high water tables persisted for at least 2 days after the cessation of precipitation. Malzone et al. (2016) showed, through the analysis of a series of storm events, that the interaction between river and aquifer was dependent on antecedent conditions, with the time of year dictating the duration and magnitude of the exchange of water within the hyporheic zone.

Cloutier et al. (2014) and Buffin-Bélanger et al. (2016) in studies in a high hydraulic conductivity, unconfined, gravelly river floodplain aquifer in Eastern Canada, identified for a series of in-bank river floods, well-defined groundwater flood waves that propagated quickly through the entire floodplain (250 m). The largest flood event recorded in the studies affected local groundwater flow orientation by generating an inversion of the hydraulic gradient for 16 hours.

The diffusivity of the floodplain aquifer is a key controlling factor in the groundwater flood wave response (Pinder et al., 1969; Reynolds, 1987). The propagation of the flood wave is therefore very fast in confined aquifers with high hydraulic conductivity. García-Gil et al. (2015) used the equation for flood wave propagation derived by Pinder et al. (1969) to examine the influencing factors in groundwater flooding in the city of Zaragoza in north-eastern Spain, which is located on the floodplain of the Ebro River. These factors were: river level rise rate; absolute–relative height of the maximum stage; aquifer parameters; river–aquifer exchange rates; pre-event state of the aquifer; and distance to the aquifer boundary. Synthetic models derived by García-Gil et al. (2015) were used to assess risk to subsurface infrastructure from groundwater flooding and to improve their design.

The importance of groundwater recharge to floodplain aquifers from overbanked fluvial flood waters has been highlighted by Doble et al. (2012). Through the application of a fully coupled, surface-groundwater flow model they identified that the infiltration volume increased with the floodplain flood level and duration, and was limited by low values of: hydraulic conductivity of the river bed and surface layer of the floodplain; aquifer transmissivity; and unsaturated aquifer storage. Land development that has occurred in recent human history, such as the stripping of natural vegetation to allow agricultural activities, has contributed to higher loads of fine sediments within rivers globally (Walling & Fang, 2003; Macklin et al., 2010). The deposition of this material downstream has produced floodplain sediments that commonly have a low hydraulic conductivity surface layer. Ramberg et al. (2006) show in a study from Okavango Basin, Botswana, that with the absence of this low hydraulic conductivity layer, as well as deep groundwater levels, very substantial groundwater recharge can occur. This was also the case for a delta region to the south west of Barcelona, Spain, where overabstraction has caused groundwater levels to fall well below the bed of the Llobregat River (Vázquez-Suñé et al., 2007). As a result of a lack of a low hydraulic conductivity surface layer, a large proportion of fluvial flood waters recharge the unconsolidated aquifer, representing 40% of the total aquifer water inputs.

Accounting for groundwater recharge from overbank flooding is required to reduce uncertainty and error in river-loss terms and groundwater sustainable-yield calculations (Wang et al., 2015). However, continental- and global-scale models of surface water–groundwater interactions rarely include an explicit process to account for overbank flood recharge. Doble et al. (2014) calculated the proportion of overbank flood recharge to be at least 4% of the total change in groundwater storage in a modelled catchment, and at least 15 % of the riparian recharge. Accounting for overbank flood recharge is an important, but often overlooked, requirement for closing water balances in both the surface water and groundwater domains.

Groundwater and flooding

Groundwater flooding is the emergence of groundwater at the ground surface away from perennial river channels but can also involve the rising of groundwater into man-made ground and subsurface assets, including the basements of building and other infrastructure such as sewers (Macdonald et al., 2008; Booth et al., 2016). The impact of groundwater flooding can be severe under conditions where the 'normal' ranges of groundwater level and groundwater flow are exceeded. This is a form of flooding on which a limited amount of research has been undertaken, as its relevance has only been widely acknowledged in the last two decades, subsequent to widespread groundwater flooding in the

UK and France in 2000/01 (Finch et al., 2004). The only national study that could be found on the scale of impact from groundwater flooding is that of McKenzie and Ward (2015), which estimated that in England between 122,000 and 290,000 properties are located in areas of high groundwater flood risk. In an assessment of future groundwater flood risk in the UK, Sayers et al. (2015) estimated increases of 71% and 90% in the significant chance of flooding for residential and non-residential properties, respectively, by the 2080s.

Groundwater flooding takes a number of forms (Macdonald et al., 2008). *Clearwater flooding* is long-lasting, often regionally extensive groundwater flooding caused by the water table in an unconfined bedrock aquifer rising above the land surface as a response to extreme rainfall (Hughes et al., 2011; Naughton et al., 2017). It is this form of flooding that caused the significant damage to properties on the Chalk outcrop of southern England and northern France in recent years, often as the result of anomalous spring flows (Pinault et al., 2005; Ascott et al., 2017). It is of relevance to urbanised lowland floodplains where the alluvial floodplain sediments are in good hydraulic connection with underlying bedrock aquifers.

Flooding from *urban groundwater rebound* occurs where there has been a reduction in abstraction from large aquifers underlying major urban centres due to a decrease in industrial groundwater-reliant activities and a move away from groundwater as a source for domestic water supply (Soren, 1976; Lerner & Barrett, 1996; Jones, 2007). This allows lowered groundwater levels to recover causing the risk of flooding to subsurface infrastructure, such as tunnels and the basements of buildings, as well as changes in geotechnical and geochemical properties that can result in settlement and corrosion of deeply founded structures.

A form of groundwater flooding particularly relevant to urbanised floodplains is *permeable superficial deposit (PSD)* flooding. This flooding is associated with shallow unconsolidated sedimentary aquifers in hydraulic connection with rivers. These aquifers are susceptible as the storage capacity is often limited, direct rainfall recharge can be relatively high, and the sediments are very permeable. Few examples of the impact of this type of flooding have been reported in the international literature (see Section 1.2.1 for further detail).

In many cases, including in the study area on which this thesis is based, groundwater flooding is associated with floodplain aquifers isolated by underlying very low hydraulic conductivity bedrock. However, there are examples of locations where the lateral inflow of groundwater to floodplain aquifers from adjacent superficial and bedrock aquifers contributes to groundwater flooding, resulting in longer-lasting flood events. Gotkowitz et al. (2014) report on groundwater flooding that occurred in a floodplain terrace of the Wisconsin River, fed laterally and from beneath by a regional sandstone aquifer. This flow, in response to a period of intense rainfall and snow melt, caused flooding that lasted for a period of six months. Ó Dochartaigh et al. (2018) describe the groundwater dynamics of an upland floodplain aquifer in Scotland which although hydraulically well-connected to the river, with rapid groundwater level rise and recession over hours, is also strongly coupled with highly permeable extensive superficial deposits on the adjacent hillslopes. As a result of the lateral groundwater flow from these deposits, high heads can be maintained in the floodplain aquifer for weeks, sometimes with artesian conditions, with important implications for drainage and infrastructure.

Urban groundwater flooding also occurs in cities where there is *excess importation of water* to meet urban needs for both public water supply and industrial activities (Foster, 2001). Where large volumes of water are imported and a high proportion is lost from the distribution network and from wastewater collection systems, the resulting aquifer recharge can cause long-term groundwater level rise (e.g.: urban centres of the Middle East, George (1992); Riyadh, Rushton & Al-Othman (1994); urban centres in north-eastern Ukraine, Jakovljev et al. (2002)). This situation often occurs in cities in the developing world where the water supply and wastewater collection networks are not sufficiently well constructed or maintained. This is a particular issue in large urbanised floodplains where the river drainage network is insufficient to maintain groundwater at low levels.

The imbalance of aquifer recharge and discharge in urban areas, causing localised waterlogging, is a potential consequence of policies that promote *sustainable drainage systems* (SuDS; Dearden & Price, 2012). SuDS have a range of environmental aims which include: the alleviation of pluvial flooding caused by the sealing of the ground surface in urban areas; and the inability of the engineered piped drainage networks to deal with periods of intense rainfall. Pluvial flood alleviation is addressed by enhancing subsurface infiltration through a diffuse network of trenches and soakaways. However, the technique is often applied in settings that are not suitable, where the floodplains have low permeability and shallow water tables, causing localised waterlogging (Potter, 2006; Zheng et al., 2015).

The subsurface plays a key role in the growth of the built environment. It is the location for foundations, basements and subsurface infrastructure, such as sewerage networks and telecommunications. These constructions, which either create barriers to or conduits for groundwater, affect groundwater flows and, in turn, levels (Vázquez-Suñe et al., 2004; Pujades et al., 2012). Changing groundwater levels due to new construction can have implications for existing nearby property and infrastructure, potentially causing subsurface groundwater flooding (Paris et al., 2010). Other impacts associated with the rise of heads include: reduction of the bearing capacity of shallow foundations; expansion of heavily compacted fills under the foundation of structures; settlement of poorly compacted fills; increase in loads on basement walls of buildings; and increase in the need for drainage in temporary excavations (Marinos & Kavvadas, 1997). This may also cause the mobilisation of legacy pollutants located at shallow depths (Miller & Hutchins, 2017).

In fluvial flood events, much of the water that flows out of river channels onto the associated floodplain will find its way back to the river as its level recedes. However, natural levees and anthropogenic changes to topography in urban areas, may mean that shallow water is retained on the floodplain (Lewin & Ashworth, 2014), extending the period of flooding (McMillan & Brasington, 2007). In this context, Moftakhari et al. (2018) define *nuisance flooding* as shallow depths of flood water of 3-10 cm that do not pose significant threat to public safety or cause major property damage but do impact transport and public health. This impact may be caused by direct contact with flood waters but may also be indirect, for example by blocking the road network (Hammond et al., 2015). Moftakhari et al. (2017) showed that the cumulative exposure to frequent, relatively small flood events could cause a greater economic impact than exposure to less frequent extreme events. The recession of trapped flood waters will be slowed by low permeable surface geology and saturated ground associated with high groundwater levels. This is an aspect of flood inundation that has had little attention from the research community, however, with the potential for a greater frequency of flooding under future climate (Arnell & Gosling, 2016), nuisance flooding may become more prevalent. The economic drivers may be strong enough to require this form of flooding to be considered routinely in the modelling of flood hazards.

1.1.3 Modelling flooding in permeable floodplains

Although groundwater and surface water are often hydraulically well-connected, they are commonly considered as two separate systems and analysed independently. This is partly due to the difficulties in modelling their interactions but also because groundwater is considered to move over longer timescales than that of surface water (Liang et al., 2007). Evidence from Section 1.1.2 questions this stance in relation to flooding in permeable floodplains. Flood events in these settings can involve a range of interacting flooding mechanisms. Fluvial, groundwater and pluvial flooding, along with the performance of urban drainage systems, can all play a role in determining the nature and impact of urban flood events (Jha et al., 2012).

Complex, distributed models with detailed physics-based process representations, such as Mike-SHE, have been used to simulate variations in groundwater levels in two dimensions across floodplains, and their interactions with channels and the land surface (Bernard-Jannin et al., 2016; Clilverd et al., 2016; House et al., 2016; Thompson et al., 2017). These models have been used primarily to examine exchanges at the interface between surface water and groundwater in the context of floodplain ecosystems. An alternative approach to the modelling of surface water-groundwater interaction is via an exchange flux that appears in both the groundwater and surface water flow equations as general sink/source terms (Kollet & Maxwell, 2006). In this approach, the exchange rate is often expressed in terms of the conductance concept, which assumes an interface connecting the two domains (Anderson et al., 2015). Recent studies have included additional processes into the conductance concept to account for the influence of microtopography on surface saturation (VanderKwaak & Loague, 2001; Panday & Huyakom, 2004). The application of the conductance concept to natural systems can be problematic where a distinct interface between the surface and subsurface is absent (Cardenas & Zlontik, 2003).

While modelling studies commonly integrate a surface and subsurface component as well as surface water-groundwater exchanges, overbank flood events are rarely included in the analysis (Bernard-Jannin et al., 2016; Teng et al., 2017). However, the importance of including overbank flood water infiltration has been illustrated in floodplain modelling studies (Doble et al., 2012). For example, both Hester et al. (2014) and Claxton et al. (2003), show its importance in determining the overall residence time of water within floodplain environments.

However, even where groundwater is recognised as relevant to flooding, the modelling codes that incorporate groundwater-surface water interactions are not generally used by the flood forecasting community (Teng et al., 2017; Jain et al., 2018), given their highly parameterised nature and relatively long run-times. This limits their use in probabilistic ensemble forecasting frameworks. Bernard-Jannin et al. (2016) undertook a comprehensive study of surface water-groundwater exchanges during overbank flood events by coupling 2D modelling of surface flow with the Saint-Venant equation and 3D modelling of unsaturated groundwater flow with Richard's equation. One finding from the modelling of this unconfined system was that significant differences in exchanges occurred in response to floods of different magnitude. However, the distributed model was computationally expensive and could only be applied at the reach scale for a short time.

Conversely, groundwater fluxes are often neglected, or poorly represented, in reduced-complexity models, using kinematic or diffusive wave approximations, which are widely used to simulate inundated floodplains (see Teng et al., 2017 for a review), such as FloodMap (Yu & Lane, 2006), JFLOW (Bradbrook, 2006), and LISFLOOD-FP (Bates & De Roo, 2000). For example, in modelling groundwater flooding caused by the water table rising to the land surface in response to rainfall recharge, Morris et al. (2018) calculated groundwater discharge rates along a chalk valley using a simple Darcian calculation. These estimated flows drove their JFLOW flood inundation model but were calibrated during the modelling process by comparing simulated floodplain flows against targeted spot flow measurements at observed points of groundwater emergence and at a point near the perennial head of the river downstream, which were taken during the flood event. Such an approach does not allow for predictive simulation.

Flood mechanisms in urbanised river floodplains also involve flows via subsurface drainage (Hammond et al., 2015). Drainage networks are means to remove flood waters but can also act as pathways for flood waters. Few modelling studies that have addressed this issue have included groundwater (Elga et al., 2015) due to the complexity of linking models that incorporate flows in rivers, aquifers and drainage networks, as well as interactions between these components. There is a growing number of studies that integrate models that simulate the interaction of drainage and groundwater during pluvial flooding or to examine urban aquifer water balance (e.g. Domingo et al., 2010; Kidmose et al., 2015; Locatelli et al., 2017). Sommer et al. (2009) describe a model system developed to obtain a holistic understanding of flood risk in the city of Dresden on the floodplain of the River Elbe. The system incorporates models of river, groundwater and sewerage systems using model-inking software (MpCCI; (http://www.mpcci.de). The aquifer and the sewerage system are linked via Darcy flow equations. The coupling between the flood and sewerage system works on a smaller time step than with groundwater. The linking software interpolates between the individual model grids. It is not clear how river and aquifer are linked but flow here is conceptualised as being a minor proportion of the flow within the floodplain. Sommer et al. (2009) conclude that in the setting they have investigated: the inclusion of groundwater does not affect surface flood levels; groundwater infiltration is not significant in the overloading of the sewerage in comparison to storm runoff; problems can persist with the sewerage network, in part due to groundwater, if the river into which it is discharging does not recess quickly; and modelling helps to identify areas where high groundwater levels pose a risk to buildings.

1.1.4 Pollution

Floodplains are the locations of a range of receptors that are sensitive to pollution, in particular: aquatic ecosystems in rivers, lakes and ponds; wetland terrestrial ecosystems; and rivers and aquifers used for water supply. Rivers and floodplains are particularly exposed to high anthropogenic stress as they are located in the low-lying areas of the landscape in which the whole range of catchment modifications and impacts accumulate (Tockner et al., 2010). Urbanisation on floodplains also inevitably introduces contamination, particularly at the peri-urban interface. This is a zone of interaction, where urban and rural activities are juxtaposed, landscape features are subject to rapid modification, and often support large industrial developments (McGranahan et al., 2004; Douglas, 2006).

There are many potential sources of pollutants in urban environments, for example leaking sewers and other wastewater collection systems, accidental discharges from commerce and industry, and the maintenance of parks and other green spaces (Foster et al., 1999; Fetter et al., 2017). Peri-urban floodplains may also support some agricultural activity. In addition, many legacy landfills and waste dumps are located within urban and peri-urban floodplains (Laner et al., 2009; Brand et al., 2018).

Rivers themselves may be a source of pollution where flow into floodplain aquifers occurs in losing reaches and through overbank flooding during high flows (e.g. Stewart et al., 1998; Cabrera et al., 1999; Ciszewski & Grygar, 2016). Rivers can transport substantial amounts of pollutant associated with activities upstream, in particular excess nutrients used in agriculture and discharges from sewage treatment works (Mainstone & Parr, 2002; Withers & Lord, 2002; Jarvie et al., 2006; Bouraoui & Grizzetti, 2011).

Floodplain aquifers can be important as sources of drinking water (Ascott et al., 2016). These are often river bank infiltration (RBF) schemes (Hoehn, 2002) that take advantage of the improvement to water quality resulting from the attenuation that takes place in the riparian zone. RBF schemes can provide a high proportion of public water supply, for example 50% in Slovakia, 45% in Hungary and 16% in Germany (Hiscock and Grischek, 2002). Although these schemes can be highly effective at attenuating river pollution when in-bank (Ramli et al., 2017), Ascott et al. (2016) provide evidence that inundation of schemes from polluted overbanked flood waters can result in significant deterioration of public groundwater supplies for periods of weeks.

Outside of RBF schemes, groundwater sources within urban areas are less common in developed countries, given ready access to piped water systems and concerns about urban groundwater quality (Howard & Gelo, 2002). However, in developing countries, boreholes are a growing component of domestic water supply infrastructure, including community groundwater supplies for low income populations in peri-urban areas outside of the reach of public water supply networks (Foster et al., 2011). These communities are often located on marginal land on floodplains (Güneralp et al., 2015) and vulnerable to pollution associated with poor borehole construction and proximity to on-site sanitation (Back et al., 2018).

Nitrogen pollution in floodplain aquifers

Specific studies included in this thesis (Chapter 2 and 3) focus on nitrogen as a pollutant in floodplain environments. In recent decades anthropogenic inputs of nitrogen, for food production by intensive agriculture, and urbanisation, have caused increases in macronutrient fluxes, and have led to widespread nitrogen pollution of aquatic systems (Foster et al., 1982; Burt et al., 2011; Whitehead & Crossman, 2012; Lapworth et al., 2013). This is a global issue with implications for food production and security, water quality and land management/planning (Galloway et al., 2004; Krause et al., 2008).

Characterising redox zones constitutes an important framework for understanding the behaviour of nutrients in floodplain aquifers. Organic carbon is very important for the evolution of different redox regimes in aqueous environments, and the amount and reactivity of the organic carbon in the aquifers is therefore an important parameter when evaluating the state and trends of groundwater quality (Gooddy & Hinsby, 2008). In floodplain environments there is a ready supply of organic matter from river water inundation and groundwater-surface water exchange (Lapworth et al., 2009). This provides a seasonal impetus for microbial action and the formation of transitory redox zones.

Nitrate, the predominant oxidised form of nitrogen, is readily transported in water. High concentrations of nitrate are typically associated with diffuse agricultural pollution from fertilisers (Oakes et al., 1981; Addiscott et al., 1991), although oxidation of anthropogenic ammonium sources also causes high nitrate concentrations (Gooddy et al., 2002). Anaerobic carbon-rich sediments, characteristic of floodplains, have the potential to support large populations of denitrifying bacteria. The denitrification potential generally increases with organic matter content towards the soil surface (Burt et al., 1999) with consequent potential for denitrification increasing as the water table rises. Shallow water tables also help to create anaerobic conditions, as the aerobic unsaturated zone of the sediments is small (Burt et al., 2002b; Kellogg et al., 2005).

Studies have shown a relationship between denitrification potential and factors such as soil texture, discrete flow zones, channel shapes and fluctuating water levels. Dahm et al. (1998) found that hydrological and biogeochemical dynamics were linked to the sediment characteristics of the floodplain and stream bed interface and also to the degree of channel constraint, the availability of specific chemical forms of electron donors and electron acceptors and temporal changes in discharge. Pinay et al. (2000) found a significant relationship between denitrification rates in the floodplain soils and their texture; highest rates were measured in fine textured soils with high silt and clay content. Pfeiffer et al. (2006) found evidence of nitrate attenuation at depth which depended on topographic features and temporal variability. This flow system provided sources of dissolved organic carbon to deeper groundwater flow paths, leading to the consumption of dissolved oxygen and generating redox conditions suitable for denitrification and subsequently iron reduction. McCarty et al. (2007) also showed that the denitrification potential of a riparian wetland was both stratified and was limited by the presence of discrete seepages or upwelling zones rather than a uniform distribution. Kolbe et al. (2019) provide evidence of pronounced biogeochemical reactivity with depth at a series of locations in six aquifer settings across the USA and suggest that previous estimates of denitrification had underestimated the capacity of deep aquifers to remove nitrate, while overestimating nitrate removal in shallow flow paths. They argue that the increased oxygen and nitrate reduction identified related to relatively little organic carbon in agricultural soils and the excess nitrate input that has depleted solid phase electron donors near the surface. Burt et al. (2002b) showed how fluctuating water levels can control the degree of denitrification. Stating that denitrification potential generally increases towards the soil surface, they propose water table elevation can control the degree to which nitrate reduction is optimised.

Nitrogen in the form of ammonium is present naturally in groundwater as a result of anaerobic degradation of organic matter but also occurs in groundwater impacted by anthropogenic sources, for example as a result of domestic and agricultural waste water disposal practices as well as landfill leachates (Gooddy et al., 1998; Lawrence et al., 2000; Christensen et al., 2001; Heaton et al., 2005). A large number of landfill sites are located on floodplains, as is highlighted for England and Wales by combining data obtained from the Environment Agency on licensed landfill sites, with floodplain extent; 3797 of the 21030 sites, 18%, are located on floodplains (Figure 2). Siting of landfills on floodplains is not current practice and therefore it is assumed these sites are historical and, as the examples in the case study area used within this thesis (Chapter 3) would suggest, without adequate arrangements for the containment of leachate. Ammonium transport in the subsurface may be retarded by sorption, as it exhibits a high affinity for ion exchange and may be incorporated into clay mineral lattices (Ceazan et al., 1989; Buss et al., 2004), and ammonium can be attenuated through microbially induced transformations (DeSimone & Howes, 1998).

McClain et al. (2003) showed that rates and reactions of biogeochemical processes vary in space and time and these variations are often enhanced at terrestrial-aquatic interfaces. They defined biogeochemical 'hot spots' as patches that show disproportionately high reaction rates relative to the surrounding matrix, whereas 'hot moments' were short periods of time that exhibit disproportionately high reaction rates relative to longer intervening time periods. Hot spots occur where hydrological flow paths converge with substrates or other flow paths containing complementary or missing reactants. Hot moments occur when episodic hydrological flow paths reactivate and/or mobilize accumulated reactants.

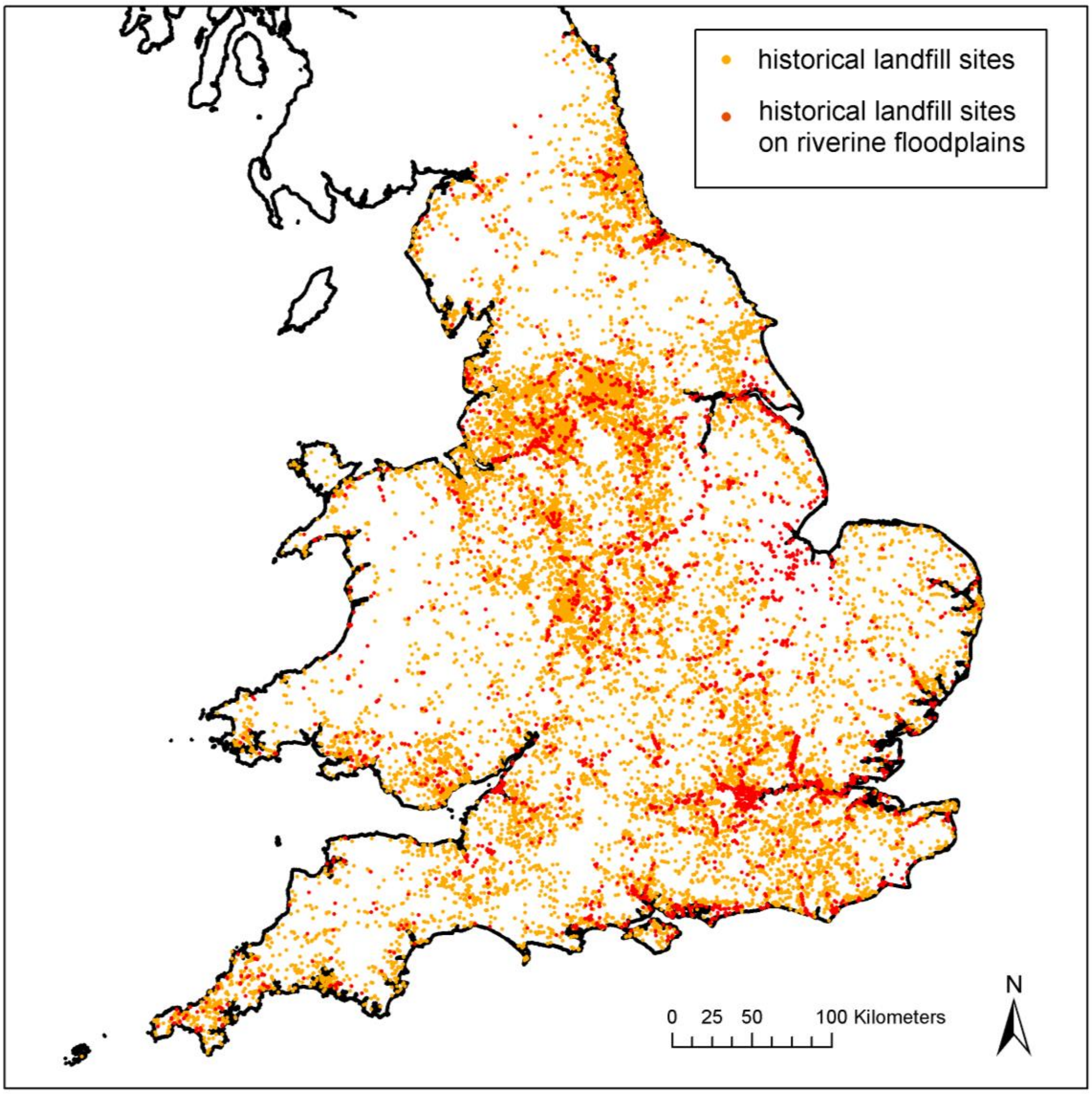

Figure 2 Historical landfill sites in England and Wales, licensed by the Environment Agency 2014, including those located on riverine floodplains. Floodplains as defined by the British Geological Survey Geological Indicators of Flooding dataset, Class 1 and 2 Fluvial Zones (Booth & Linley, 2010). Contains Environment Agency data licensed under the Open Government Licence v3.0, and Ordnance Survey data © Crown copyright and database right (2018).

Krause et al. (2017) highlight ecohydrological interfaces, the dynamic transition zones that often develop within ecotones or boundaries between adjacent ecosystems, as hot spots of ecological, biogeochemical, and hydrological processes. The hyporheic zone, the interface between river and groundwater (Burt et al., 2013), is a hot spot for nutrient processing (McClain et al., 2003; Hester & Gooseff, 2010; Krause et al., 2011b; Antiguedad et al., 2017). Exchanges of water, nutrients, and organic matter here occur in response to variations in discharge and bed topography and porosity (Boulton et al., 1998). Upwelling groundwater can supply stream organisms with nutrients while downwelling stream water can provide dissolved oxygen and organic matter to microbes and invertebrates. The hyporheic zone hosts a wide range of hydrologically controlled processes that are potentially coupled in complex ways. Understanding these processes and the connections between them is critical since these processes are not only important locally but integrate to impact increasingly larger-scale biogeochemical functioning of the river corridor up to the river network scale (Cardenas, 2015; Abbott et al., 2016; Pinay et al., 2017).

1.1.5 Environmental management drivers relevant to floodplain environments

The UK has a wide portfolio of environmental regulations, most of which reflect the requirements of European Union Directives, and many of which impinge on activities in urban and peri-urban floodplains (de Sosa et al., 2018). Although difficult to quantify, a link is identified between the amount of natural floodplains and achieving the key objectives of European policies (EEA, 2018). Floodplain management or protection is encouraged but only indirectly required under European environmental policies, but floodplain health is important for achieving multiple European policy objectives. The following policies are particularly relevant to floodplains. The relevance of research undertaken within the PhD to the environmental management requirements associated with these policies is highlighted.

EU Water Framework Directive (2000/60/EC)

The Directive establishes a framework for the protection of inland surface waters (rivers and lakes), transitional waters (estuaries), coastal waters and groundwater (Chave, 2001). It sets out to ensure that all aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands meet 'good status' by 2027. Achieving the goals of the Directive is facilitated through a series of River Basin Management Plans (RBMP) in which those water bodies not at good status have been identified, the causes investigated and, where not disproportionately costly, measures put in place to address these. Floodplain aquifers, at least in the UK, where extensive and sufficiently permeable, can form groundwater bodies on their own, for example, the Upper Thames Gravels in the Thames River Basin. Floodplains can also form a groundwater body with the underlying bedrock aquifer, if together they are sufficiently large and in good hydraulic connection, for example the River Colne Valley Groundwater Body in the Thames River Basin, which combines Lower Thames Gravels with the Chalk aquifer (Environment Agency, 2016). Where floodplains do not form groundwater bodies on their own, activities related to the floodplain deposits are likely to have a bearing on the status of associated river bodies (Dahl et al, 2007). In the context of floodplains, the Water Framework Directive provides protection for component water courses and groundwaters but also the terrestrial ecosystems that they support. The *Groundwater Directive* (2006/118/EC) complements the Water Framework Directive by establishing a regime which sets groundwater quality standards and introduces measures to prevent or limit inputs of pollutants into groundwater.

Pieces of legislation designed to protect groundwater against pollution and deterioration are part of a larger regulatory framework that can be traced back to the 1990s. The *Nitrates Directive* (91/676/EEC) aims to reduce and prevent water pollution caused by nitrates from agricultural sources. The *Plant Protection Products Directive* (91/414/EEC) and the *Biocides Directive* (98/8/EC) ensures that commercial plant protection and biocidal products, such as pesticides, herbicides, or fungicides, have no harmful effect on human health or on groundwater. These three Directives are relevant in peri-urban floodplains where agricultural activities can take place, and in urban and peri-urban floodplains if pollutants occur as a result of interaction with rivers that are sourced from heavily agricultural catchments (Pärn et al., 2012).

The *Urban Wastewater Treatment Directive* (91/271/EEC) aims to protect the environment from the adverse effects of discharges of urban wastewater and wastewaters from certain industrial sectors. The *Integrated Pollution Prevention and Control (IPPC) Directive* (96/61/EC) lays down measures designed to prevent or reduce air, water or soil pollution. The Directive applies to a significant number of mainly industrial activities with a high pollution potential. The *Landfill Directive* (99/31/EC) seeks to prevent or reduce the negative effects of landfill waste on the environment, including groundwater. However, this does not address the legacy contamination that is ubiquitous in the urban and peri-urban floodplains across Europe. These sites are addressed in the UK through legislation such as the Contaminated Land (England) Regulations 2006.

To help achieve good status of floodplain groundwater bodies and to contribute to the improvement of associated water bodies, contaminants from current and historic land use may need to be removed through improved land use practices or remediation, although in some situations natural attenuation of pollutants may be sufficient to reduce concentrations to acceptable levels (Ranalli & Macalady, 2010; Weber et al., 2011; Ciszewski & Grygar, 2016). In Chapters 2 and 3 of the thesis two aspects of water quality are addressed that are relevant to meeting the requirements of environmental regulations. In Chapter 2, an investigation is undertaken of the flux of nitrate into the floodplain aquifer in the case study area via the main river bed, and the floodplain during flood inundation. The natural attenuation of nitrate in the floodplain and in the river bed sediments is estimated. In Chapter 3, the leachate from an historical landfill on the floodplain is characterised, the degree of attenuation of ammonium in the leachate in the floodplain sediments examined, and the potential input to the case study river assessed.

EU Habitats and Birds Directives (1992/43/EEC & 2009/147/EC)

These Directives address nature conservation, protecting the highest value sites (Special Areas of Conservation) and species. In the UK, the EU legislation is enacted through the Wildlife and Countryside Act (1981) which also incorporates lower-protection Sites of Special Scientific Interest. This and other legislation underpin Biodiversity 2020, the strategy for England's wildlife and ecosystem services. Floodplain meadows, an important type of groundwater-dependent terrestrial ecosystem located within river floodplains, are a valuable resource, not only supporting some of the most diverse vegetation in the UK (up to 40 species per square metre) and associated fauna, but also performing key ecosystem services, such as flood storage and sediment retention (Wheeler et al., 2004). However, the UK now has less than 1500 ha of this unique habitat remaining (Jackson & Mcleod, 2002), most of which is protected under the EU Habitats Directive. Species composition is known to be tightly correlated to the hydrological regime (Silvertown et al., 2015), and related temperature and nutrient regime, both of which, in floodplain environments, are highly dependent on shallow groundwaters. In some cases it has been the long-term manipulation of the hydrological regime through water management systems that has established the ecosystems; changes to these hydrological regimes through lack of maintenance or through measures to address other aspects of the water environment, such as flooding, may have serious deleterious impacts on these vulnerable plant communities. Environmental regulation sets out to protect these ecosystems and provide a framework within which the competing societal needs can be optimised.

EU Floods Directive (2007/60/EC)

Environmental management regulation also aims to protect property and infrastructure. The Floods and Water Management Act (2010) in England, and equivalents in other regions of the UK, were a response to the increasing economic impact of flooding on urban areas within floodplains (Nones, 2015). The Act and associated regulations flow from the EU Floods Directive (2007/60/EC), established in 2007 (Klijn et al., 2008). Traditionally flood alleviation has focussed on flood defence to reduce overbank flooding from rivers, however, since major flooding in the late 1990s and early 2000s the whole-catchment approach has received more attention across Europe (Lane, 2017). Initiatives related to this approach, such as Making Space for Water (Defra, 2005) and Working with Natural Processes (Barlow et al., 2014) in the UK, Room for the River in the Netherlands (Warner & van Buuren, 2011), and pan-European Nature Based Solutions (EC, 2015) have shaped flood policy.

Groundwater and pluvial flooding have received more attention in the UK since the Pitt Review (Pitt, 2008) and subsequently from their inclusion in the EU Floods Directive. The Directive includes provisions for groundwater and pluvial flooding (Cobby et al., 2009): assessing risk; producing 'flood hazard maps'; and introducing measures to address any significant risk. In England, the responsibility for developing measures which address significant risk associated with the main river network falls with the Environment Agency (EA). For minor water courses, pluvial flooding and groundwater flooding, the responsibility resides with the Lead Local Flood Authority (LLFAs), with guidance provided by the EA. This has proved a challenge, as often the LLFAs do not have the capacity in terms of skillset or resources. Developing measures to address all forms of flooding, which are often interlinked, whether this is through improvements in flood defences, flood conveyance or upstream land management, is challenging but a basic understanding of the mechanisms and the risk are essential. Improving the understanding of groundwater flooding mechanisms is the focus of Chapter 4 of the thesis. A grey area in terms of responsibility is where flooding results from water from a range of sources, for example from the main river in combination with groundwater. Flood processes relevant to this situation are the topic of Chapter 5.

1.2 Research gaps addressed, and thesis aims and structure

1.2.1 Research gaps addressed

This PhD addresses research gaps related to the understanding of the hydrological regime of urbanised floodplains, in particular groundwater hydrology. It uses the case study of the lowland floodplain of the River Thames in the vicinity of the city of Oxford in the southern UK (see Section 1.3 for further detail).

Impact of river management structures on water and nitrate exchange in urbanised floodplain aquifers

A large number of studies have investigated nutrient pollution within alluvial aquifers associated with river floodplains (e.g. Haycock and Pinay, 1993; Correll et al., 1997; Clement et al., 2003; Forshay and Stanley, 2005; Krause et al., 2008). These investigations relate primarily to the quality of water in associated rivers, and measures that can be undertaken to bring nutrient concentrations below levels that are detrimental to the ecological status of the aquatic environment. Within this body of research, few studies have examined the influence of river management structures on processes that relate to nutrient cycling and, where undertaken, have addressed relatively simple hydrological settings.

In an assessment of the influence of dam operations on groundwater/surface water interaction, Hucks Sawyer et al. (2009) monitored groundwater levels, temperature, and specific conductivity along a transect perpendicular to the Colorado River (Texas, USA), downstream of a major dam. They report that the stage fluctuations, associated with the dam operation, that force river water into and out of the banks, fundamentally change the hydrological, thermal, and geochemical dynamics of riparian aquifers and their hyporheic zones. Matula et al. (2014) modelled the potential impact of the proposed construction of a weir on the River Labe in the Czech Republic to aid river transport. Their simulations indicated that as a result of the raised river level, the groundwater level in the adjacent aquifer would rise significantly and the increased flow of water, contaminated by industrial effluent, from river to aquifer, could potentially result in the pollution of the aquifer system. Lee et al. (2016) undertook a before-and-after assessment of the introduction of 16 weirs as part of a 'rivers restoration' project in Korea that aimed to: secure water resources; introduce comprehensive flood control measures; improve water quality; and restore river ecosystems. The study showed immediate increase in the groundwater levels associated with the weir installation but at the time of reporting had not seen any change in groundwater quality.

Specifically in relation to NO_3^-, Cisowska and Hutchins (2016) examined the effect of weirs on river NO_3^- concentrations by measuring river flow, water temperature and water quality data both before and after a weir was removed from the River Nidd in Yorkshire, UK. Modelling indicated that the removal of the weir reduced the annual fraction of the upstream NO_3^- load being retained along the river reach by ~2% over the two years monitored, however, the study incorporated only processes in the river bed sediment. Hill and Duval (2009) examined the hydrology and nitrogen biogeochemistry of a riparian zone before and after the construction of beaver dams along an agricultural stream in southern Ontario, Canada. The beaver dams increased surface flooding and raised the riparian water table by up to 1·0 m. Increased hydraulic gradients inland from the stream limited the entry of oxic nitrate-rich subsurface water from adjacent cropland. Permeable riparian sediments remained saturated during summer and autumn, whereas before dam construction a large area of the riparian zone was unsaturated in these seasons each year. In turn, beaver dam construction produced significant changes in riparian groundwater chemistry, including NO_3-N concentrations in autumn and spring, which were lower in the post-dam (0·03–0·07 mg L^{-1}) versus the pre-dam period (0·1–0·3 mg L^{-1}).

The studies described here show the impact of river structures on groundwater levels in adjacent aquifers and the potential implications for groundwater quality. With efforts to improve the ecological status of rivers there is growing interest in understanding the implications of the removal of dams and weirs on ecological status of rivers. Tullos et al. (2016), in a synthesis of common management concerns associated with dam removal, highlight the implications on groundwater supplies of removing structures, that cause associated groundwater levels and storage to decline. However, generally the focus of research in this context is on rivers, and not associated aquifers. The need to take a more holistic view of the environmental implications has been identified in reviews of research studies undertaken on this topic (Bellmore et al., 2017a, b; Ding et al., 2019).

In Chapter 2 of this thesis, a study is presented which aims to contribute to the research on the topic of the impact of river management structures on groundwater flows, and associated nutrient dynamics. It uses the case study floodplain where a high density of long-standing river management structures are located. It was not possible to assess conditions with and without structures, however, the research provides an overall understanding of the hydrology of the river network and interaction with groundwater within the study area, and uses this to quantify water and nutrient exchange, specifically focussing on NO_3^-, the predominant oxidised form of nitrogen. It uses a combination of water level observations, water chemistry sampling and a simple numerical model.

Influence of waste dumps on floodplain groundwaters and fluxes of pollutants to associated rivers

As described in Section 1.1.1, globally, floodplains are increasingly being encroached and developed due to anthropogenic pressures, including urbanisation and intensive agriculture (Tockner and Stanford 2002; Pinter 2005; Werritty 2006). In many parts of world there is a historical legacy of change in land use and associated historical pollution loading to floodplain groundwaters and surface waters (Burt et al., 2011; Stuart et al., 2011a). Urban and peri-urban floodplains are therefore complex, in terms of spatial heterogeneity in land use and topography, and temporal variability in recharge and redox processes (Burt et al., 2002; Burt and Pinay 2005; Macdonald et al. 2012a; MacDonald et al., 2014).

Due to fluctuating redox conditions, floodplains are considered hot-spots for nutrient attenuation (Devito et al., 1999; McClain et al., 2003; Harms and Grimm 2008). Groundwater levels are typically shallow and responsive to recharge events, and soil moisture conditions are also highly variable; together these have important implication for attenuation of oxidised N (Burt et al., 1999). Periods of inundation may stimulate denitrification due to the mobilisation of C pools at shallow depths; during periods of low groundwater level, denitrification may be restricted due to reduced C pools, low soil moisture and more oxidising conditions.

The important ecosystem services provided by floodplains and their general proximity to potentially damaging agricultural and urban nutrient sources means there is an imperative to understand associated nutrient processes, fluxes and attenuation mechanisms. Due to the complex hydrogeology of floodplain settings, estimating the residence times of groundwaters and legacy nitrogen pollutants is challenging. There is therefore uncertainty as to the timescale for changes in land management practices reducing baseflow nutrient fluxes to surface waters (Wang et al., 2011; Stuart et al., 2011b; Jackson et al., 2008).

Despite the environmental importance of NO_3^- and NH_4^+, there are few studies documenting their transport and reaction processes in aquifers (Heaton et al., 2005). Isotopic fractionation studies can provide an excellent tool for understanding N transport and speciation (Wassenaar, 1995; Böhlke et al., 2006). Isotopic fractionations have been reported for NH_4^+ sorption to clays (Karamanos and Rennie, 1978), with the remaining NH_4^+ in solution relatively depleted in ^{15}N. By contrast, nitrification results in a substantial increase in ^{15}N for the remaining NH_4^+ (Delwich and Steyn, 1970). Stable isotope ratios in NO_3^- have often been used to distinguish various sources of NO_3^- in groundwater, such as synthetic fertilisers and animal wastes (Gormley and Spalding, 1979; Flipse and Bonner, 1985). Denitrification causes an isotopic enrichment in the remaining NO_3^- (Mariotti et al., 1988). A disadvantage of the single isotope approach to denitrification studies is that processes such as ammonia volatilisation can also lead to enrichment of $\delta^{15}N$ in the residual NH_4^+ source material and in the NO_3^- produced during nitrification. However, by combining ^{15}N with an analysis of the $\delta^{18}O$ of the NO_3^-, a more reliable indicator of the denitrification process is achieved (Böttcher et al., 1990; Wassenaar et al., 1995; Kendall., 1998; Fukada et al., 2004). Through evaluating variations in concentration and isotopic composition, the opposing isotope fractionation effects make it possible to distinguish between sorption, nitrification and denitrification as major processes affecting N distribution in a field setting.

In Chapter 3, the temporal and spatial variations in N species is examined within the case study floodplain using geochemical indicators together with $\delta^{15}N$ of NH_4^+ and $\delta^{15}N$ and $\delta^{18}O$ of NO_3^-. The study focusses on an area of the floodplain down-gradient of a large waste dump. The peri-urban hydrological setting, legacy of pollution, and dependent terrestrial ecosystems within the wider floodplain mean the study is relevant across many industrialised and rapidly industrialising regions. The objectives are to evaluate the origin of N in groundwater and ascertain whether sorption, nitrification or denitrification are contributing to the attenuation of the nutrient load. In turn, through the understanding of the water movement and redox status across the floodplain, an assessment is made of the cumulative influence of nutrient fluxes from waste dumps on the adjoining river.

Conditions and mechanisms that control groundwater flooding in urbanised floodplains

As stated in Section 1.1.2, a form of groundwater flooding particularly relevant to urbanised floodplains is permeable superficial deposit flooding (PSD), which is associated with shallow unconsolidated sedimentary aquifers in hydraulic connection with rivers. Section 1.1.2 also details a number of studies that investigate river-aquifer-floodplain interaction during high flows (Bates et al., 2000; Burt et al., 2002a; Claxton et al.; Jung et al., 2004; Vidcon, 2012; Cloutier et al., 2014; Buffin-Bélanger et al., 2016), and explains the significance of floodplain aquifer diffusivity as a key controlling factor in groundwater flood wave propagation (Pinder et al., 1969; Reynolds, 1987). However, these studies focus on non-urbanised floodplain settings.

A number of examples of the impact on urbanised areas of PSD flooding have been reported. Kreibich and Thieken (2008) describe damage that occurred to buildings in the floodplain of the Rivers Elbe and Danube in Germany due to water pressure on basements, resulting from abnormally high groundwater levels associated with persistent high river levels. In the modelling studies referenced in Section 1.1.3, carried out in Dresden by Sommer et al. (2009) and Karpf et al. (2011), water flows in the urbanised floodplain were examined, identifying groundwater flow as one mechanisms resulting in flood impact, although surface water flooding were shown to dominate. García-Gil et al. (2015) references other studies from Germany in which PSD flooding accounted for ~20% of the costs of flood damage.

In a study of basement flooding along the Elbow River in Calgary, Canada, Abboud et al. (2018) identified groundwater ingress as the initial cause in 88% of the houses surveyed, while the remainder were flooded exclusively by groundwater. Of the 19 surveyed homes located outside of the 100-year overland flood zone, 47% were flooded by groundwater, indicating that groundwater flooding reached beyond overland water-flooded areas. Modelling linked groundwater flooding to the raised river stage in this PSD environment. Flood resilience strategies were proposed including groundwater level monitoring for early flood warning and design criteria for new homes.

These were the only examples within the international peer-reviewed literature that could be found that clearly address PSD groundwater flooding. However, given that urbanised floodplains are common, it is likely that PSD groundwater flooding is the cause of flood impact in many locations, with a substantial socio-economic cost. Without an understanding of the flood process, measures to reduce flood risk cannot be properly designed or instigated. Further studies are therefore required to highlight the mechanism to raise awareness, and also to examine how raised groundwater levels interact with property and infrastructure on urbanised floodplains.

The aim of the research presented in Chapter 4 of this thesis is to improve the understanding of groundwater flooding in urbanised permeable floodplains, to better understand the controls and mechanisms, to characterise the locations where this form of flooding is likely to occur and how it relates to other forms of flooding. A conceptual model is developed using a combination of floodplain characterisation, water level observations and local knowledge of flood impact, and the relevance to flood mitigation measures is discussed.

Role of groundwater and shallow geology in the persistence of flooding in urbanised floodplains

The author's personal experience of flooding within the case study city, underpinned by the data available from the comprehensive water level monitoring network there (see Section 1.3), has highlighted that flood waters can linger on the urban floodplain for many days after the peak in river levels. The greatest apparent impacts of the extended period of flooding are disruption to the transport network, a compromised sewerage network due to groundwater exfiltration and, in a few cases, persistent inundation of property. As was discussed in Section 1.1.2, natural levees and anthropogenic changes to topography in urban areas can cause a shallow depth of flood water to be retained on the floodplain after fluvial flood events (McMillan and Brasington, 2007; Lewin & Ashworth, 2014). This shallow flooding comes within the definition of 'nuisance flooding' (Moftakhari et al., 2018), not posing a significant threat to public safety or causing major property damage, but impacting on transport (Hammond et al., 2015) and public health.

In urban areas, engineered drainage may help remove flood waters (Hibbs & Sharp, 2012). Where drainage is absent, it is expected that low permeability surface geology and saturated ground associated with shallow groundwater levels will slow the recession of water trapped on the floodplain. Few surface water-groundwater interaction studies have examined the dynamics between inundated floodplains and shallow groundwater. As a consequence, the physics of the hydrological processes in areas with shallow groundwater is not well understood, and typically it is not included in river or groundwater models (Doble et al., 2012). As highlighted in Section 1.1.2, using a model of a conceptualised floodplain-aquifer system, Doble et al. (2012) showed that the surface layer of the floodplain is a key factor in controlling the period of flood inundation. They also showed that irregularities in the floodplain elevation have a large impact on the infiltration volume due to local ponding, but given the structure of the model, did not represent flow across the floodplain in 2D. However, they acknowledged that further work was required to investigate the spatio-temporal variability of vertical floodplain-aquifer fluxes in more complex settings, for example, with irregular floodplain topography in two dimensions and adjacent to non-linear channels.

As already discussed, complex, distributed models with detailed physics-based process representations that have been used within floodplain environments to simulate the interaction between groundwater and surface waters (Bernard-Jannin et al., 2016; Clilverd et al., 2016; House et al., 2016; Thompson et al., 2017) are not generally used by the flood forecasting community (Teng et al., 2017; Jain et al., 2018). Their highly parameterised nature and relatively long run-times, limit their use in probabilistic ensemble forecasting frameworks. Conversely, potentially relevant vertical subsurface fluxes are often neglected, or poorly represented, in reduced-complexity models, that are widely used to simulate inundated floodplains (see Teng et al., 2017).

Recognising the need for flood inundation models that incorporate groundwater and infiltration processes, Chapter 5 reports on the development of a model system that integrates the popular cellular, or 'storage cell', flood inundation model, LISFLOOD-FP, with a finite-difference groundwater model, allowing vertical exchange of water through a surface layer. The coupled model is used to estimate vertical fluxes across the case study floodplain and to test the influence that low permeability surface layers and shallow groundwater have on flood duration, and as a result explore an aspect of flood hazards with potential significant socio-economic impact that has until now been largely ignored.

1.2.2 Thesis aims and structure

To summarise Section 1.2.1, the aims of the research presented in this thesis are to:

1) improve the understanding of the impact of river management structures on water and nitrate exchange in urbanised floodplain aquifers;
2) better understand the influence of waste dumps on water quality in floodplain aquifers and to quantify the fluxes of associated pollutants to rivers via the subsurface;
3) determine conditions and mechanisms that control the occurrence of groundwater flooding in urbanised floodplains;
4) develop a flood model system that incorporates groundwater processes, to be used to understand the role groundwater and shallow geology play in controlling the persistence of flooding in urbanised floodplains.

The structure of the thesis is as follows:

Chapter 1

The context for the PhD research has already been set out in Sections 1.1 and 1.2.1 of this thesis. In the remainder of this chapter, the study area is described. It is recognised that in Chapters 2 to 5, aspects of the study area are described in the published and prepared papers and that as a result there is unavoidable repetition within the thesis.

Chapters 2 to 4

Chapters 2 to 4 are formed by three papers that have already been published in peer-reviewed journals. Each paper addresses individually, in the same order, the first three aims set out above.

Chapter 5

This chapter addresses the fourth aim above. It is written in the format of a journal paper to a standard that means it can be submitted to an international peer-reviewed journal in its current form.

Chapter 6

In this chapter the insights from the papers in Chapters 2 to 5 are brought together and the implications for environmental management in urban and peri-urban floodplains are discussed, making reference to issues raised in Sections 1.1 and 1.2.1. Remaining knowledge gaps are highlighted and recommendations are made for further research, building on the work presented in this thesis. Recommendations are also made as to how the results in this thesis could be translated into policy and practice for environmental managers.

Note that the PhD was written over the period of a number of years and also that Chapters 2 to 4 were not published in the order they are presented. Relevant developments in the conceptual understanding that happened subsequent to the writing of individual chapters are discussed in Chapter 6.

Individual reference lists are provided in Chapters 2 to 5. These references, as well as those used in Chapter 1, are pulled together into a complete reference list at the end of the thesis.

1.3 Study area

The city of Oxford is located in the south of the United Kingdom, in the upper reaches of the River Thames catchment (Figure 3). The main River Thames flows to the west of the historic centre of the city, which is located on high ground above the current floodplain (Figure 4). The river forms a series of channels as it passes Oxford. The River Cherwell joins the River Thames within the Oxford valley, downstream of the city centre. The mean annual flow of the River Thames upstream of Oxford is 18.48 m^3/s (Marsh & Hannaford, 2008). The baseflow index for the river at this location is 0.67, reflecting the influence of influent groundwater sourced from the limestone aquifers located in the headwaters, and the extensive floodplain gravel aquifers (Figure 3). The Thames in Oxford has a large number of river management structures associated with it, for historical navigation and hydropower purposes. Six locks and associated bypass flow channels are located on the main river. The study area for the PhD was defined as the River Thames floodplain in the Oxford valley between the upstream lock at Eynsham and downstream lock at Sandford (Figure 4).

The geology of the River Thames floodplain in Oxford is characterised by alluvial sediments (Figure 2 in Chapter 2), which are underlain by Oxford Clay bedrock throughout most of the study area (Newell, 2008). This is typical of much of the River Thames Catchment, where the extensive areas of alluvial deposits (alluvium and river terrace deposits) are associated with the wide, low-gradient bedrock mudstones, rather than the more steeply incised carbonate and sandstone bedrock (Figure 3; Bricker & Bloomfield, 2014).

The alluvium that underlies the floodplain typically comprises of silty clay. This overlies glacio-fluvial sands and gravels deposited by Late Devensian rivers (Newell, 2008). High ground surrounding the floodplain is formed of higher terrace gravel deposited during the latest Anglian Glaciation, and thin alternating layers of Late Jurassic limestones, sandstones and mudstones. The floodplain has an area of 20.4 km^2 and ranges in width from 410 to 2170 m. The floodplain narrows substantially downstream as it leaves the Oxford valley. The floodplain gravels are typically 2 to 4 m thick within the Oxford valley but can be up to 8 m thick in places. The alluvium forms a relatively uniform blanket around 1 m thick across the floodplain although in some areas it thins to a few decimetres and exceptionally can be up to 4 m thick (Newell, 2008).

Nine Sites of Special Scientific Interest are located on the floodplain in the Oxford valley (Figure 4), covering a total area of including 3.62 km^2. Four of these sites, including Port Meadow, a focus for research within this PhD, form the Oxford Meadows Special Area of Conservation. This is designated on the basis of the lowland hay meadows within and, in the case of Port Meadow, as it is one of only two known sites in the UK where creeping marshwort *Apium repens* is found (Gowing & Youngs, 2005). Port Meadow is also designated as a Scheduled Monument by Historic England due to evidence of Bronze Age and Iron Age activity (Lambrick, 1992). Its primary use is for the grazing of cattle and horses.

Fourteen historic waste dumps, licensed by the Environment Agency, are located on the Oxford floodplain, with a total area of 1.05 km^2 (Figure 4). At these sites waste has been deposited directly on to the floodplain, so does not fill excavations, and there is no lining beneath. They contain primarily domestic but also some industrial waste. The largest of these is Burgess Field (White & Macdonald, 2015), with an area of 0.34 km^2, and a maximum height of 4 m above the natural floodplain. This waste dump is located up-gradient of Port Meadow and the River Thames. It was the destination for domestic waste from the city from 1937 to 1980. There is a legacy of anthropogenic contamination in this part of the floodplain, with documented waste sites dating back to the late 1800s. There are a number of gravel pits within the floodplain, primarily within the northern areas (Figure 4), but there is currently no operating gravel excavation, with the last activity ending in the early 2000s.

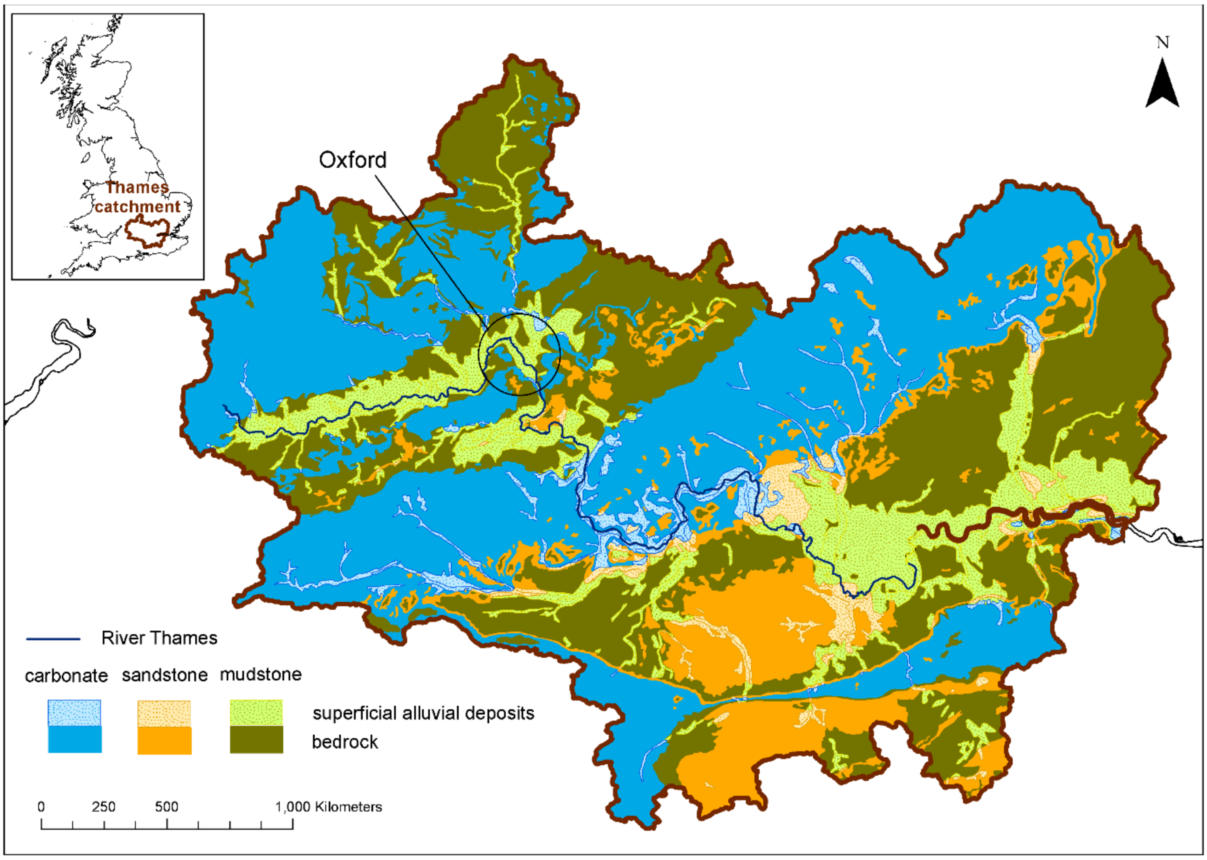

Figure 3 Bedrock geology within the River Thames catchment, categorised as carbonate, sandstone and mudstone, and overlying superficial alluvial deposits, coloured according to bedrock category. Contains Ordnance Survey data © Crown copyright and database right (2019).

The population of Oxford is approximately 171,000 with around 3400 domestic and commercial properties located on the River Thames floodplain. Approximately 14% of the floodplain is urbanised (Figure 4). The floodplain is also the location of key transport routes. The extensive river channel network limits the number of roads into the city centre. Flooding is a significant problem for the properties and transport routes on the floodplain, including main railway lines to London and Birmingham. The transport routes and urban development restrict the movement down-catchment of overbanked river waters during flood events, and reduce flood storage, raising fluvial flood levels (Figure 3 in Chapter 4). The 1 in 100 year flood envelope, as defined by the Environment Agency (Figure 4), covers approximately 85% of the floodplain. In the past two decades, flood frequency has increased with major events, in 1998, 2000, 2003, 2007, 2012 and 2014. The July 2007 flood was the largest of these events; identified as a 1 in 20 year flood, it inundated over 200 properties (Environment Agency, 2009). A significant number of properties affected by flooding this century have been as a result of rising groundwater, which was either the sole cause of flooding or the initial cause prior to inundation from fluvial waters. Flood waters lingering on the floodplain after the main river recession, extending the duration of flooding is also a major issue, especially for vehicular access to the city centre.

The Seacourt Stream, which becomes the Hinksey Stream in the south of the city (Figure 4), has few water management structures along its length. During flood events, this is the water course that sees the greatest rise in water levels and the most extensive flooding. Since the mid-2000s investigations have taken place in Oxford, led by the Environment Agency, to assess if a major scheme to address flooding is economic. The Oxford Flood Risk Management Study (Ball et al., 2009) concluded that the economic justification for a scheme could not be made within the rules set out by Government, however, following the 2014 floods the issue was revisited and as a result the Oxford Flood Alleviation Scheme was designed and funding for its construction found (Environment Agency, 2018). The scheme is due to begin construction in 2019, with the aim to improve conveyance of high flows through the Oxford valley, primarily along the course of the Seacourt Stream, and to better contain overbanked waters where flooding does occur.

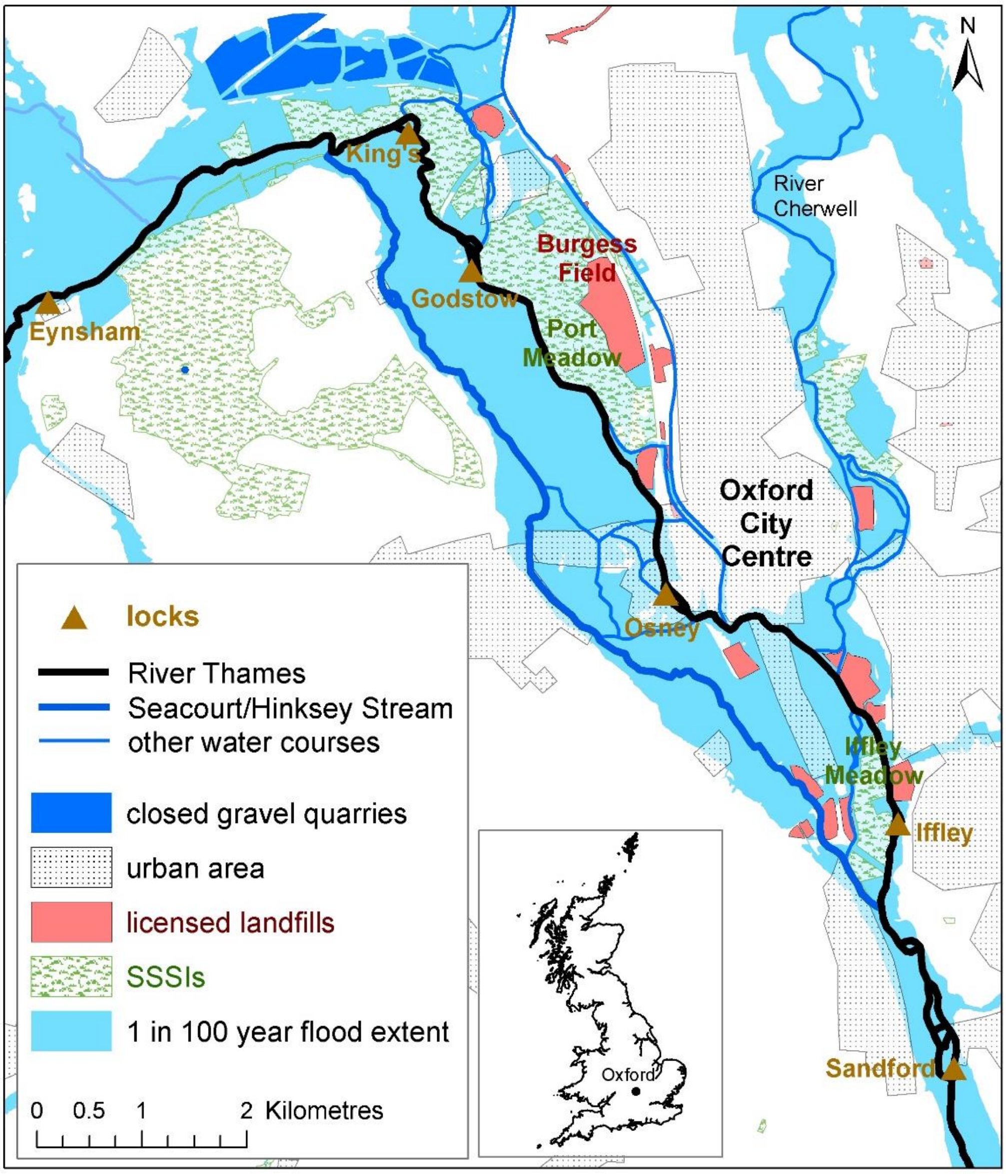

Figure 4 Features of the floodplain of the River Thames in the vicinity of Oxford, UK. Areas in white are higher ground above the 1 in 100 year flood extent. Contains Ordnance Survey data © Crown copyright and database right (2019)

Studies within the Oxford area relating to the status of floodplain ecosystems and flood risk have resulted in an extensive groundwater and surface water level network being developed (Figure 2 in Chapter 2). Groundwater levels in the aquifer are very shallow, typically fluctuating within 2 m of ground level. At some sites monitoring has been undertaken since the early 1980s (Institute of Hydrology, 1987). Manual monitoring of water levels was undertaken on a monthly basis between 2005 and 2009; the total number of monitoring points at the end of 2009 was 235. Digital water level recorders have been installed at some point within 110 of these monitoring points. At a number of sites there are combinations of two or more of the following: piezometers completed in the gravel aquifer; piezometers completed in the overlying alluvium; surface water stilling wells; and flood water level recorders. Data were made available from recorders installed in the upstream and downstream waters at four of the Environment Agency locks in the study area. During the July 2007 flood, a further six temporary manual flood water monitoring points were established, which provided additional data in urban areas for the rise and recession of flood waters.

References are made to a selection of the water level monitoring sites in Chapters 2 to 5. Each chapter has taken a different approach to naming the sites, as the papers within will normally be read in isolation. In Table 1 the monitoring sites are identified that are used in more than one chapter (n.b. none of the sites referenced in Chapter 4, apart from the locks, are included in the other chapters).

Table 1 **Naming of monitoring sites used in more than one of Chapters 2, 3 and 5. Each row relates to the same site.**

SITES		
Chapter 2	**Chapter 3**	**Chapter 5**
T1		S1
T1a		P2
T1b	PTM26D	
T1e	PTM29D	
L4		S2
	PTM11	P3
	PTM21	P1

As well as water levels, River Thames flows are routinely measured within the study area by the Environment Agency. Rainfall is measured on a 15-minute interval by the Environment Agency at Eynsham and Osney Locks. There is also a long climate record from the Radcliffe Meteorological Station in Oxford city centre (Burt & Shahgedanova, 1998).

Chapter 2: Water and nitrate exchange between a managed river and peri-urban floodplain aquifer: Quantification and management implications

The work presented in this chapter has been published in the following journal paper:

Macdonald, D.M.J., Dixon, A.J. & Gooddy, D.C. (2018). Water and nitrate exchange between a managed river and peri-urban floodplain aquifer: Quantification and management implications. *Ecological Engineering,* 123, 226-237.

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Water and nitrate exchange between a managed river and peri-urban floodplain aquifer: Quantification and management implications

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ABSTRACT

The management of rivers for navigation, hydropower and flood risk reduction involves the installation of in-channel structures. These influence river levels and can affect groundwater flow within hydraulically-connected riparian floodplain aquifers. A comprehensively monitored, peri-urban, lowland river floodplain in the southern United Kingdom was used to explore these dependencies and to examine the implications for the flux exchange of water and nitrate between the river and the floodplain alluvial aquifer. The study demonstrated that rivers maintained at high levels by management structures, result in raised groundwater levels in the adjacent aquifer and complex groundwater flow patterns. Engineered river management structures were shown to promote flow from river to aquifer through the river bed but the majority of the associated nitrate was removed in the hyporheic zone. High-nitrate groundwater recharge to the alluvial aquifer also occurred through overbank flood flows. Across the floodplain, substantial denitrification occurred due to anaerobic conditions resulting from carbon-rich sediments and the shallow water table, the latter linked to the river management structures. An upper limit on the total annual mass of nitrate removed from river water entering the floodplain aquifer was estimated for the study site (2.9×10^4 kg), which was three orders of magnitude lower than the estimate of annual in-channel nitrate flux (1.8×10^7 kg). However, this capacity of lowland floodplains to reduce groundwater nitrate concentrations has local benefits, for example for private and public water supplies sourced from alluvial aquifers. The insights from the study also have relevance for those considering schemes that include the installation, removal or redesign of river management structures, as the resultant change in groundwater levels may have consequences for floodplain meadows and the nutrient status of the aquatic system.

1. Introduction

Floodplains are locations of complex interactions between river water, groundwater and overland flow (Burt et al., 2002). The degree of interaction is dependent on a number of factors, including: the magnitude and direction of the head gradient between river and aquifer; the permeability of the alluvial sediments and the river bed material; and the capacity of the river channel to retain high flows (Sophocleous, 2002). Naganna et al. (2017) provide a comprehensive review of the controls on river bed permeability identifying the importance of the particle size and depth of the bed material, the river channel geometry and upstream sediment supply to the river. Colmation and bioclogging of macropores and associated lower bed permeabilities is more likely to occur in river reaches losing water to adjacent aquifers (Battin and Sengschmitt, 1999; Brunke, 1999; Krause et al., 2007; Younger et al., 1993). Given the range of controlling factors, river bed permeability will be highly spatially variable (Calver, 2001; Irvine et al., 2012). Bed scouring resulting from floods can induce temporal changes in streambed elevation and particle size composition, increasing hydraulic conductivity (Blasch et al., 2007; Doppler et al., 2007; Hatch et al., 2010). Where permeable near surface floodplain sediments occur, Doble et al. (2012) showed overbanking river water can result in substantial groundwater recharge.

The management of rivers for navigation, hydropower and flood risk reduction involves the installation of in-channel structures (Gregory, 2006). These structures are ubiquitous in many countries (Davies and Walker, 1986; Downs and Gregory, 2014). For example, within England and Wales, records from the Government environment regulator, the Environment Agency, accessed in 2014, showed that 17,569 locks, weirs and control gates were located on the 68,755 km of the river network. The operation of engineered river management structures disrupts the natural interaction of surface water and

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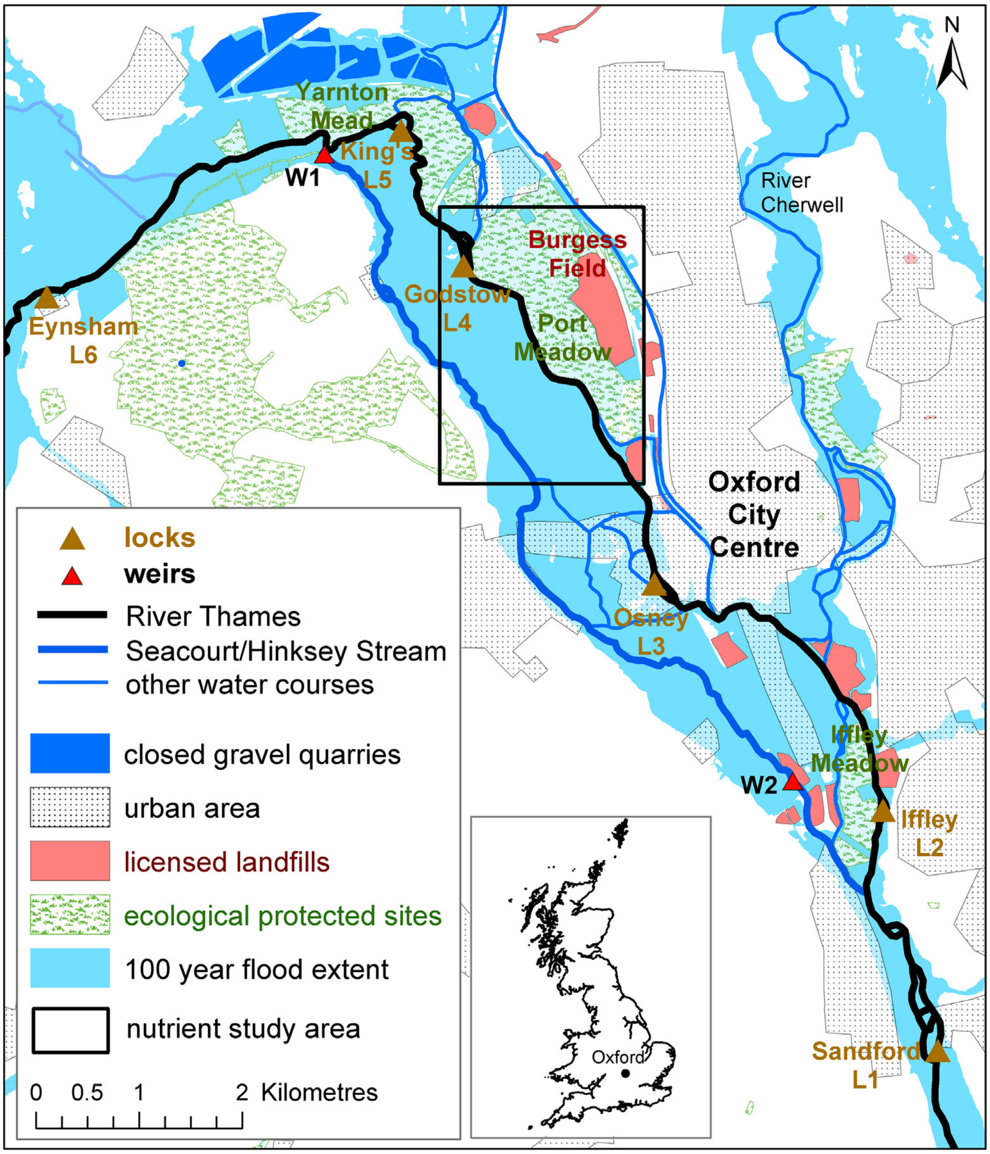

Fig. 1. River Thames floodplain in the vicinity of the city of Oxford, UK. Areas in white are higher ground above the 1-in-100 year flood extent. Contains Ordnance Survey data © Crown copyright and database right (2018).

groundwater. Studies have shown how structures can cause groundwater levels in the associated aquifer to be raised and river reaches to switch from gaining water from the adjacent aquifer to losing water when structures are introduced (Krause et al., 2007; Matula et al., 2014; Lee et al. 2015). The aggregation of fine-grained material associated with lower river velocity upstream of structures and scouring of the river bed downstream, in combination with head gradients that increase and decrease the propensity for colmation, mean bed permeability of rivers under the influence of river structures can be highly variable (Hatch et al., 2010; Naganna et al., 2017). Attempts to address poor river ecology have included the removal of weirs to return the connectivity of river habitats (Gilvear et al., 2013) with likely changes to potentially long-standing groundwater flow patterns and levels.

Groundwater levels are a factor in determining reduction-oxidation (redox) conditions within the subsurface that in turn are a major control on the processing of nutrients (Rivett et al., 2008). Nitrate (NO_3^-), the predominant oxidised form of nitrogen, is readily transported in water and is stable under a range of conditions. However, anaerobic carbon-rich sediments, characteristic of floodplains, have the potential to support large populations of denitrifying bacteria. Shallow water tables help to create these anaerobic conditions, as the aerobic unsaturated zone of the sediments is small (Burt et al., 2002; Kellogg et al., 2005). The rate of denitrification increases with organic matter (OM) content towards the soil surface (Burt et al., 1999) and there is a ready supply of OM to floodplains through inundation by sediment-laden river water. Pinay et al. (2000) found a significant relationship between denitrification rates in floodplain sediments and their texture; highest rates were measured in fine-textured soils with high silt and clay content. These finer-grained floodplain sediments are often found at the surface, as a result of historical clearance of natural vegetation and increased agriculture upstream (Macklin et al., 2010).

The hyporheic zone interface between river and groundwater (Burt et al., 2013) is also a hotspot for nutrient processing (Antiguedad et al., 2016; McClain et al., 2003). Exchanges of water, nutrients, and OM here occur in response to variations in discharge and bed topography and porosity (Boulton et al., 1998). Upwelling groundwater can supply stream organisms with nutrients while downwelling stream water can provide dissolved oxygen and OM to microbes and invertebrates in the hyporheic zone. The improvement to water quality resulting from the actions of the hyporheic zone are the basis of river bank infiltration schemes (Hoehn, 2002) and water sourced from such schemes can provide a large proportion of public groundwater supplies (Ascott et al., 2016). Although many authors acknowledge the part played by denitrification processes in the riparian zone in decreasing NO_3^- concentrations, the important role of dilution is also reported (Baillieux et al., 2014; Bernard-Jannin et al., 2016; Pinay et al., 1998).

A large number of studies have investigated nutrient pollution within alluvial aquifers associated with river floodplains (e.g. Haycock and Pinay, 1993; Correll et al., 1997; Clément et al., 2003; Forshay and Stanley, 2005; Krause et al., 2008). These investigations relate primarily to the quality of water in associated rivers, and measures that can be undertaken to bring nutrient concentrations below levels that are detrimental to the ecological status of the aquatic environment. Within this body of research few studies have examined the influence of river management structures on processes that relate to nutrient cycling (e.g. Hucks Sawyer et al., 2009; Cisowska and Hutchins, 2016) and where undertaken address relatively simple hydrological settings.

The aim of the study reported here was to examine flow and nutrient dynamics within the floodplain of a large lowland river system with a high density of long-standing river management structures. Hydrogeological and water level data from the floodplain were used to assess the influence of the engineered river management structures on groundwater flows and levels in the associated alluvial aquifers, and measurements of water chemistry and simple modelling were used to estimate the flux and removal of NO_3^- that resulted from the cycling of water through the floodplain aquifer. The significance of the nitrate loss was assessed in terms of river NO_3^- flux.

The floodplain studied was that of the River Thames in the vicinity of the city of Oxford in the southern United Kingdom. The many studies undertaken in the area over the period of recent decades have characterised the hydrogeology of the sediments and resulted in an extensive water level monitoring network (summarised in Macdonald et al., 2012).

2. Materials and methods

2.1. Study area

The River Thames flows along the western edge of Oxford (Fig. 1). The floodplain within the Oxford valley has an area of 20.4 km^2, varying in width from 410 to 2170 m. It is bordered by high ground formed from incised Quaternary river terraces and Jurassic bedrock. The floodplain is underlain by alluvial sediments; a layer of fine-grained silts and clays over very permeable sands and gravels, with a total thickness of two to six metres (Newell, 2008). Almost all of these sediments are bounded laterally and below by low permeability bedrock of Upper Jurassic Oxford Clay. The floodplain has down-valley gradient of 0.053% but locally contains shallow channels and raised interfluves, which can influence flood water distribution.

Although the local urban area mainly occupies the high ground surrounding the floodplain, approximately 14% of the floodplain is urbanised (Fig. 1). There are 20 historical licensed ‘landfills’ on the floodplain (Fig. 1), with a total surface area of 1.05 km^2. These landfills are mostly mounds of waste material sitting on the floodplain surface. A large gravel quarry, now closed, is located in the north of the floodplain (Fig. 1). Land designated as ecologically sensitive, primarily lowland floodplain meadows, occupies 3.62 km^2 (18%) of the floodplain (Fig. 1). Amongst other factors such as management practices, temperature and nutrient status, and soil pH, these types of meadows are highly sensitive to soil moisture and its temporal fluctuation that, in turn, is dependent on depth to groundwater (Wheeler et al., 2004; Punalekar et al., 2016).

The River Thames source is in the Jurassic limestone hills 60 kms to the west of Oxford. Within the Oxford valley the River Thames breaks up into a series of channels before reforming into a single channel as it flows out of the valley. The length of the River Thames in the study area is 16.1 km. The main secondary channel is the Seacourt Stream, which becomes the Hinksey Stream in the south of the valley (hereafter also referred to as the Seacourt Stream). The River Cherwell flows into the Thames to the south of the city centre. The long-term mean flow in the Thames, measured upstream of Oxford, is 18.48 m^3/s (Marsh et al., 2008). Since 2000 there have been five major flooding events that have affected the urbanised areas of the floodplain. Groundwater flooding is a significant component of the flooding in the city (Macdonald et al., 2012), mainly affecting subsurface infrastructure such as the inundation of house basements and the surcharging of sewers. Eighty-five per cent of the Oxford floodplain is inundated by the modelled 1-in-100 year return period flood (Environment Agency, 2009). The percentage of inundated floodplain resulting from the July 2007 flood, which was estimated to be between a 15- and 20-year return flood (Macdonald et al., 2012), was 63%.

The Thames has six locks and associated weirs within the Oxford valley (Fig. 1), with an average separation of 3.2 km; the locks furthest upstream (Eynsham) and downstream (Sandford) define the study area. All the locks, apart from Sandford, were most recently rebuilt in the first half of the 20th Century; Sandford was rebuilt in 1972. However, in all cases there have been weirs at these locations for centuries (Thacker, 1968). The difference between the mean water level at the tail of Eynsham Lock and the head of Sandford Lock is approximately 5 m. (NB, in Fig. 1 both the lock names and an alphanumeric identification are given, however in the remainder of the paper only the latter will be used when referring to the locks.)

The locks in Oxford are typical of those found on the non-tidal River Thames (354 km in length). Thirty-three locks are located over a 198 km reach of the river, with an average separation of 6.2 km. The purpose of the locks is to maintain the river upstream at navigable levels, higher than those that naturally occurred prior to their construction, enabling boats to move between the upstream and downstream levels.

2.2. Monitoring infrastructure and data

The water levels and flow regime in the floodplain aquifer system were investigated within the study area. Water chemistry from samples taken within a sub area of the floodplain (see Fig. 1) were used to examine nutrient dynamics.

There is a dense network of water level monitoring sites within the Oxford study area (Fig. 2). The study used data from 51 sites at which water levels were monitored over the previous three decades. Surface water monitoring sites were a combination of stilling wells with digital water level loggers, gaugeboards and locations, such as bridges, with known datums from which water levels were measured. Groundwater monitoring sites were drilled boreholes with diameters from 5 to 20 cm, completed in the gravel aquifer at least 1 m below the estimated minimum groundwater level. Measurements were made at the groundwater monitoring sites with a combination of digital water level loggers and manual water level meters. The monitoring network included eight paired surface water and groundwater level monitoring sites. At some locations these paired sites were combined with other groundwater sites to form water level monitoring transects. Water levels at monitoring sites not instrumented with loggers were measured manually as part of a series of floodplain-wide surveys. These surveys were undertaken on a monthly basis from May 2007 to March 2010.

River levels were also obtained from the Environment Agency. It monitors the upstream (head) and downstream (tail) river levels at five locks on the River Thames in the Oxford area (L2 to L6), as well as at four sites in the secondary streams (S1 to S4).

All monitoring sites had datums, the heights of which were surveyed relative to mean sea level. A map of groundwater level contours is presented in Section 3.1.3. This map is based on groundwater and surface water level measurements made over a two-day period in May 2007 when no rainfall occurred, converted to water levels relative to sea level, and hand-contoured. A raster dataset was created from these contours using the ‘Topo to Raster’ tool within ArcGIS (ESRI, 2015).

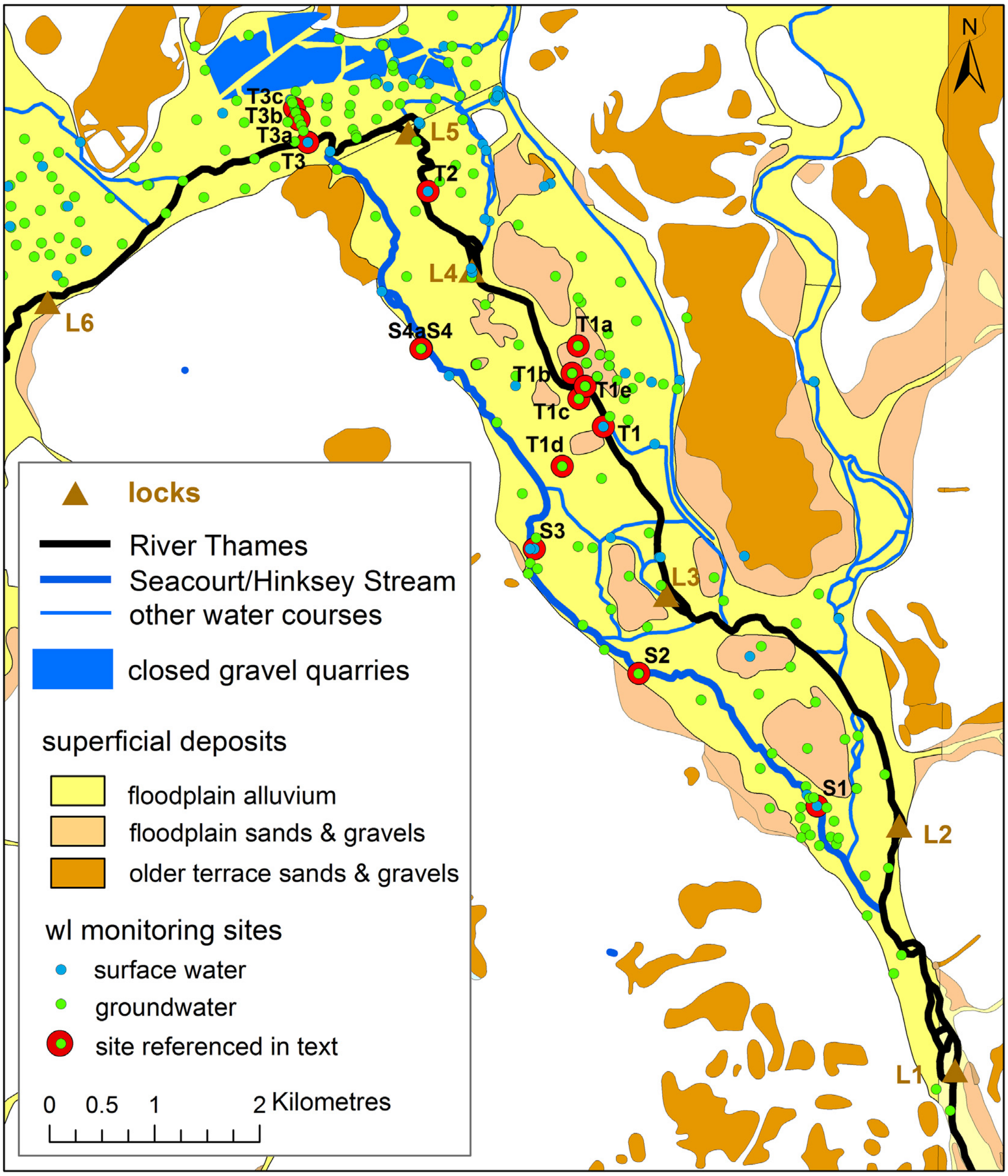

Fig. 2. Water level monitoring network and superficial geology. Sites referred to in the text are labelled. Note, some of the surface water monitoring sites are located on minor water courses not included in the figure. Contains Ordnance Survey data © Crown copyright and database right (2018).

This was combined with a digital elevation model, obtained using the Light Detection And Ranging (Lidar) surveying method by the Environment Agency (data licensed under the UK Open Government Licence v3.0), to produce a map of depth to groundwater.

Precipitation data for the area were obtained from the Radcliffe Meteorological Station in central Oxford (51° 45′40 N, 1° 15′50 W; Burt and Shahgedanova, 1998), via the archive of the Centre for Environmental Data Analysis (www.ceda.ac.uk). The mean annual precipitation and air temperature for 1986–2015, were 670 mm and 10.6 °C, respectively.

2.3. Groundwater nutrient concentrations

2.3.1. Water sampling

The focussed nutrient study was undertaken within a zone stretching across the width of the floodplain (Fig. 1) to better understand nutrient cycling in the subsurface, assess the importance of the hydrological regime in controlling this cycling, and quantify the flux of NO_3^- (note, all concentrations and fluxes are NO_3^-, rather than NO_3^--N). The study area included: to the east of the River Thames, an area of ecologically protected land used for communal grazing (Port Meadow) and a large waste dump (Burgess Field); and to the west, an area used primarily for pasture (Fig. 2). In this area, when the River Thames is out-of-bank, made ground immediately to the west forces river water to flood across Port Meadow to the east.

In addition to water level measurements, water samples were taken from within the area from 23 boreholes completed in the alluvial sediments (Gooddy et al., 2014), as well as from the River Thames (Fig. 2). A series of samples were obtained during 14 sampling rounds over the period May 2010 to March 2015. A minimum of three borehole volumes were purged from each site, and samples were collected when stable readings for pH, electrical conductivity (EC) and dissolved oxygen (DO) were obtained. Samples for chloride (Cl^-) and nitrogen species were filtered and collected in 30 ml plastic bottles. The samples were analysed for Cl^- and NO_3^- using ion chromatography, and ammonium (NH_4^-) by flow colorimetery. Samples for dissolved organic carbon (DOC) analysis were collected and filtered through 0.45 µm silver filters into 10 ml glass bottles and measured by the standard technique of acidification to pH < 3, then conversion to CO_2 by 680 °C combustion catalytic oxidation (Pt catalyst), followed by high sensitivity infra-red analysis of the gas. All analyses were carried out in the British Geological Survey's laboratories that are accredited by the UK Accreditation Service. Field data, including bicarbonate (HCO_3^-), pH, temperature, EC and DO, were all determined at site; a flow-through cell was used for unstable field parameters to ensure representative in-situ values were obtained.

2.3.2. Nitrate flux modelling

The assessment of nutrient cycling within this study focussed on the processing of groundwater NO_3^-. A conceptual model of groundwater-surface water interaction was developed in the study through an analysis of the water level data collected, in combination with the three-dimensional geological model of the floodplain aquifer (Newell, 2008). To examine dominant processes controlling groundwater NO_3^- concentrations, a simple single-cell mixing model was set up for the floodplain aquifer in the focussed nutrient study area and applied separately to the zones of the aquifer to the east and to the west of the River Thames. The model was oriented perpendicular to the river, approximately along a groundwater flow path (see Section 3.1.3). The model assumed a constant thickness aquifer, with the thickness averaged from the three-dimensional geological model; this was considered reasonable given the lack of variability in the thickness of the floodplain aquifer in this area (Newell, 2008). The model boundary conditions included lateral inflow (Q_L), rainfall recharge (R_P) and river flood water recharge (R_R). A water balance was maintained by making lateral discharge from the cell, the flow from the aquifer to the river (Q_A), equal to the total input over the period of a year (Fig. 3; Eq. (1)).

$$R_P + R_R + Q_L = Q_A \tag{1}$$

Lateral inflow was calculated using Darcy's Law. The hydraulic conductivity used was within the range specified for the Oxford floodplain gravels by Dixon (2004) and the hydraulic gradient was based on averaged observed groundwater levels for the modelled zone. Rainfall recharge was approximated using the output of a soil moisture balance model (Mansour and Hughes, 2004). Flood water recharge was estimated based on the volume of unsaturated material and the frequency of flooding, assuming all the unsaturated material was filled and recharged the aquifer, and the remainder of the flood water was rejected. The flood water recharge calculation used: i) an averaged depth to groundwater for winter months (i.e. December, January and February) over the 7 years prior to 2015, based on measurements from water level loggers in the zones of interest; ii) porosity of the alluvium measured within the floodplain (Hodgson, 2008; Gardner, 1991); and iii) a flood frequency based on the occurrence of major floods in the period of the study, defined by occurrences of river levels at T1 (Fig. 1) rising over 57.6 m above mean sea level (masl). The potential for the retention of flood water on the floodplain in topographical lows that allowed delayed recharge, was not accounted for. This meant the model may have slightly underestimated the flux of NO_3^- to the floodplain aquifer.

Nitrate concentrations associated with each of the inputs were based on the median of measurements obtained through the river and groundwater sampling campaign. However, the sensitivity of the model was tested using a wide range of flood water NO_3^- concentrations, recognising the non-linear relationship between river NO_3^- concentrations and river flows that have been measured in the River Thames (Neal et al., 2006). Nitrate inputs not included were those associated with: to the east, waterfowl, grazing horses and livestock, as these import a negligible amount of nitrogen; and to the west, pasture, as no fertiliser was added to this land. The model calculated NO_3^- concentrations assuming complete mixing within the aquifer cell; this was reasonable as the aquifer is thin, homogeneous and highly permeable and almost fully incised by the river (Newell, 2008). The model ran on a time step of one year, which was considered suitable given that the average residence time, estimated using Darcy's Law and the parameters given in Table 1, was greater than one year. The NO_3^- concentration at the end of timestep *i + 1* was calculated using Eq. (2)

$$C_{A_{i+1}} = \frac{C_R.\ R_{R_{i+1}} + C_L.\ Q_L + C_{A_i}.\ (A - R_P - R_{R_{i+1}} - Q_L)}{A} \tag{2}$$

where: C_R is the NO_3^- concentration in the river recharge water; $R_{R_{i+1}}$ is the river flood water recharge over timestep *i + 1*; C_L is the NO_3^- concentration in the lateral inflowing groundwater; C_A is the NO_3^- concentration in the aquifer cell; and A is the volume of the aquifer cell, equal to the width of the floodplain times the depth of alluvial sediments, multiplied by the porosity. It was assumed the NO_3^- concentration in the rainfall was negligible. A NO_3^- removal factor was used to match the average modelled NO_3^- concentration over one flood cycle with the average observed NO_3^- concentrations in all boreholes in the relevant zone for the period 2010 to 2015.

3. Results and discussion

3.1. Water levels and flows in the Oxford floodplain

3.1.1. Surface water levels

Water levels at the locks and intervening monitoring sites on the River Thames, and at sites on the Seacourt Stream were used to characterise the spatio-temporal variability of the river network within the Oxford area. The influence of the locks on river levels and the

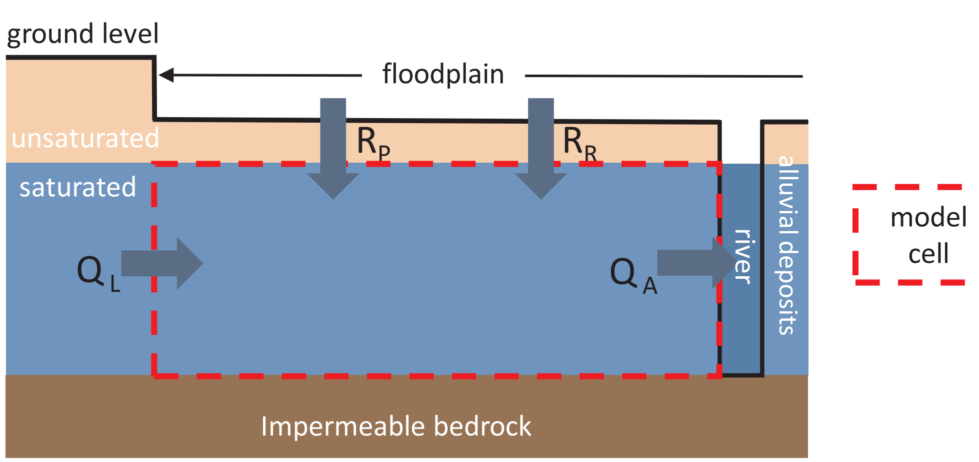

Fig. 3. Schematic of flows within the single-cell mixing model. See Section 2.3.2 for definition of terms.

Table 1
Parameters and input variables for the mixing-cell model of groundwater and NO_3^- in the aquifers on the east and west of the River Thames (Section 2.3) and necessary denitrification to match observed concentrations. Where the model is sensitive to parameters/input variables, values are shown in brackets that produce higher and lower aquifer NO_3^- concentrations and denitrification factors.

Parameters/input variables	unit	East of R. Thames	West of R. Thames
Aquifer thickness	m	2.5 (1.5/3.5)	2.5 (1.5/3.5)
Aquifer width	m	500	500
Aquifer porosity	–	0.2 (0.1/0.3)	0.2 (0.1/0.3)
Aquifer hydraulic conductivity	m/d	200	200
Groundwater level gradient	–	0.0004	0.00125
NO_3^- concentration of lateral inflow	mg/L	0	2.3 (4.6/1.5)
Annual rainfall recharge	m	0.1	0.1
Unsaturated zone depth prior to flood inundation	m	0.13 (0.20/0.08)	0.73 (1.00/0.50)
Alluvium porosity	–	0.4 (0.5/0.3)	0.4 (0.5/0.3)
Flood event frequency	years	2 (1.5/3)	2 (1.5/3)
Flood water NO_3^- concentration	mg/L	25 (35/15)	25 (35/15)
Resultant modelled aquifer NO_3^- concentration	mg/L	6.69 (28.16/0.94)	10.56 (35.25/3.04)
Average observed aquifer NO_3^- concentration	mg/L	0.04	2.62
Denitrification factor required to match modelled and observed	–	0.98 (1.00/0.79)	0.76 (0.95/0.07)

comparison with stream water levels is shown by a series of box plots produced from monitoring data for the period 1 April 2009 to 31 March 2013 (Fig. 4; Supplementary Information Table S1). The differences between median head and tail water levels at the locks range from 0.76 to 1.75 m. The impact of these steps in river level was that river gradients between locks on the River Thames (calculated using the median lock water levels) were small (average of 0.003%) compared with the gradients between monitoring sites on the Seacourt Stream (average of 0.034%). The different characteristics of lock heads and tails are illustrated (Fig. 4): tail water levels had an asymmetric distribution with a greater interquartile range and higher peak levels, similar to that of the more naturally flowing Seacourt Stream; and head water levels had an interquartile range that is an order of magnitude smaller than the tail water levels.

If it is assumed that lock tail levels are representative of ‘natural’ levels and, were locks not in place, the river levels between adjacent lock locations are proportional to distance along the river reach, then the differences between the current managed river levels and the natural river levels can be estimated. For example, the difference between managed and natural median river levels at T1, T2 and T3 would be 0.95, 0.69 and 0.61 m, respectively.

3.1.2. Groundwater-surface water interaction

Relative water levels within groups of monitoring sites were examined to assess the interaction of surface water bodies and aquifers. These relationships helped with the contouring of point measurements

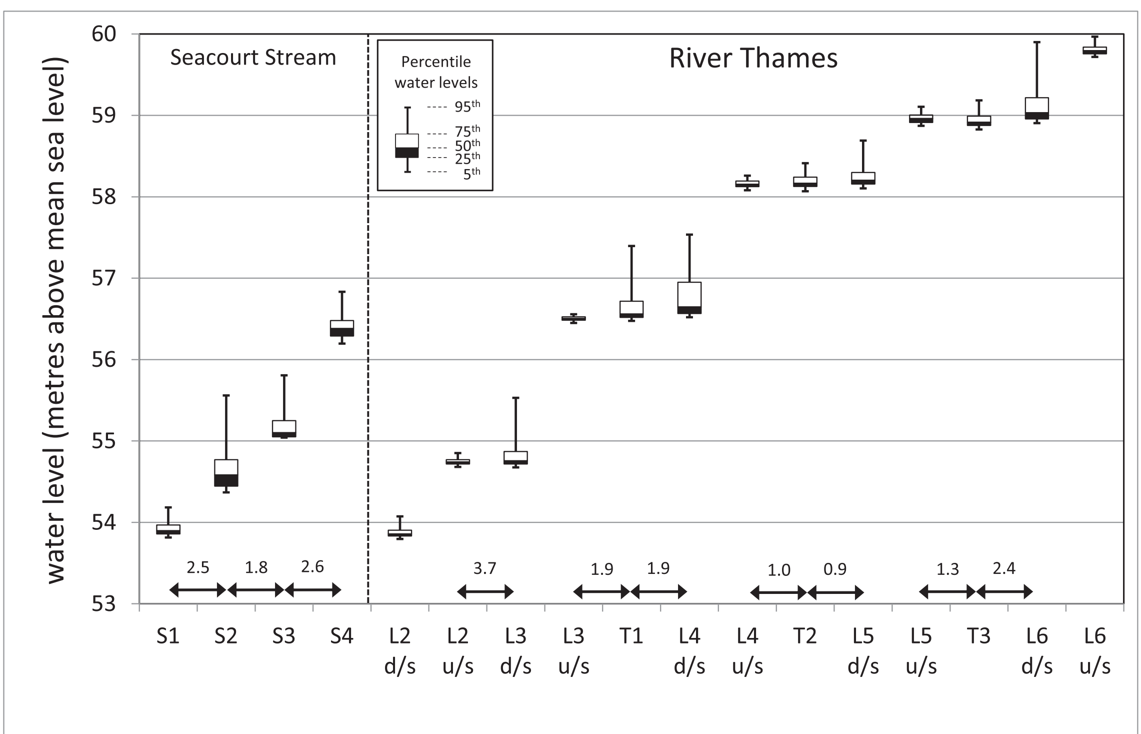

Fig. 4. Box plots of water levels at sites on the Seacourt Stream and the River Thames, from downstream (left) to upstream (right), for 1 April 2009 to 31 March 2013. For data used to produce the box plots see Supplementary Information Table S1. Lengths of reaches between monitoring sites (km) are shown above the arrows. See Fig. 2 for locations. In the case of the locks, levels are plotted for the downstream (d/s) and upstream (u/s) sides. Contains Environment Agency data licensed under the Open Government Licence v3.0.

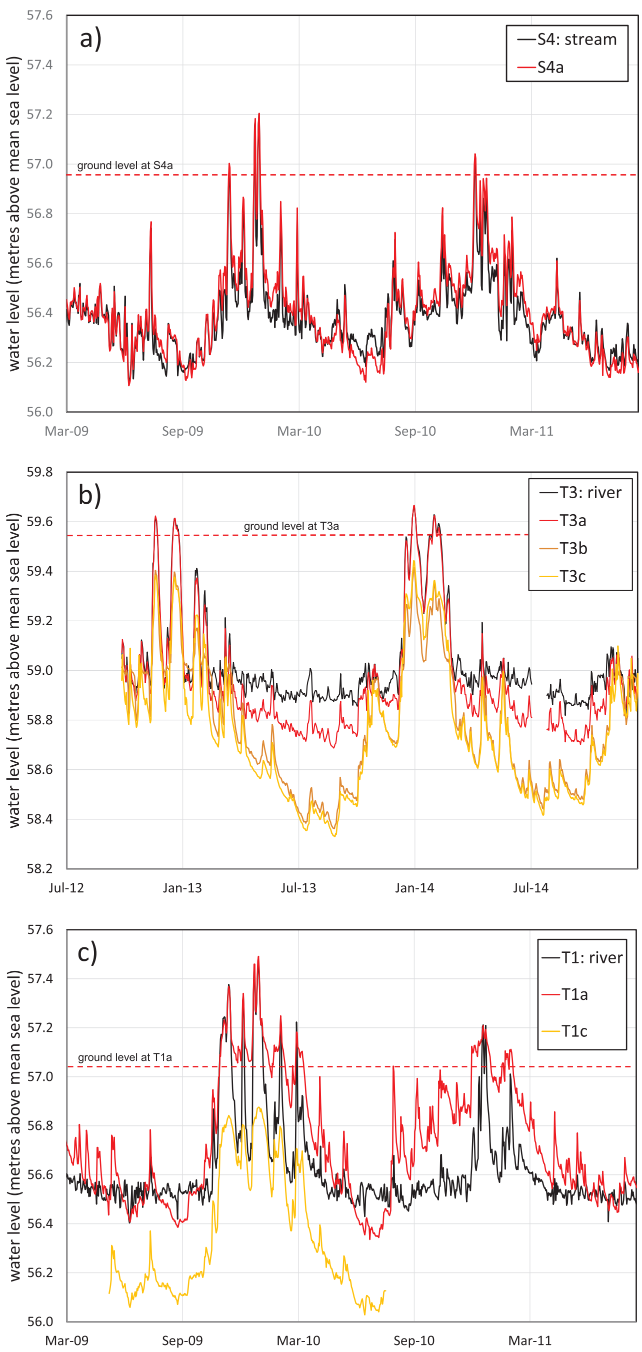

Fig. 5. Groundwater and surface water levels at three locations. Black lines are the river/stream water levels and other lines are groundwater levels (see Fig. 2 for locations): a) paired groundwater/surface water sites on the Seacourt Stream; b) a transect perpendicular to the River Thames upstream of L5; c) north-east to south-west transect through the River Thames at Port Meadow.

of water levels. The contours allowed groundwater flow directions to be determined and depths to groundwater from the ground surface to be mapped (see Section 3.1.3).

For the secondary streams, data show that surface water levels and adjacent groundwater levels had a very similar fluctuation pattern (e.g. Fig. 5a), and that the network of streams were gaining groundwater for the majority of the time. The interaction of aquifer and surface water body was different for the River Thames. In general, along reaches of the Thames upstream of the locks, for example upstream of L5 (Fig. 5b), steep gradients from the river towards the adjacent aquifer occurred for the majority of time. River bed elevation profiling, undertaken as part of the hydraulic modelling of the River Thames (Environment Agency, 2009), also suggests a saturated hydraulic connection between river and aquifer was maintained, i.e. the river did not become perched with an intervening unsaturated zone.

Along the reach of the River Thames between L3 and L4, adjacent to Port Meadow, the groundwater-river dynamics were different to those upstream of L5. To the west of the river, consistent steep gradients from river (T1) to aquifer (T1c) were again evident (Fig. 5c). To the east of the river, however, groundwater levels (T1a) were, for the majority of the time, above the river level. The groundwater gradient to the east of the river is likely to be due to a combination of two aspects: when flooding occurs it is always to the east, due to the raised bank to the west, resulting in relatively high groundwater recharge in this area; and the maintenance of a high river stage, due to the downstream lock, acts as a barrier to groundwater flowing westward towards the Seacourt Stream (see Section 3.1.3). The dynamics during periods of overbank flooding are illustrated in the winter months of 2009/10 in Fig. 5c. When the river overflowed its bank (compare the river level with the ground level at T1a, Fig. 5c), the groundwater level rose to match the river level. The groundwater levels took longer to recess than the river and, for a time, surface water on the floodplain was the result of groundwater flooding.

During the seasonal dry period, groundwater levels to the east of the river fell sufficiently to cause a temporary local reversal of the flow direction. Again, the depth of the river bed was such that the hydraulic connection between the river and aquifer was maintained during these periods.

3.1.3. *Groundwater levels and flow patterns*

In Fig. 6, maps of contoured groundwater levels relative to mean sea level (Fig. 6a) and depth to groundwater (Fig. 6b) are presented for May 2007. The groundwater flow lines and the relative depths of groundwater are typical for the alluvial aquifer in Oxford for all but the short periods when very high river levels occurred.

The flow lines show the complex patterns associated with the river management structures (Fig. 6a). Groundwater mounds coincide with raised river levels upstream of the locks. The steep water level gradients from river to aquifer in these reaches suggest that the river bed hydraulic conductivity is much lower than that of the floodplain aquifer. The low river bed hydraulic conductivity is expected as the water has a greater depth as a result of the locks, and therefore flows relatively slowly, allowing a greater proportion of fine sediment to be deposited. The positive river-to-groundwater gradient also enhances colmation.

The water entering the adjacent aquifer from the River Thames flows towards the lock bypass channels or nearby smaller streams, which form lines of groundwater discharge. The contours also show a groundwater mound created upstream of weir W2 on the Seacourt Stream in the southern part of the floodplain. It is notable in this area that a groundwater trough occurred between the River Thames and this mound, coincident with the narrow urbanised strip running north to south (compare Figs. 1 and 6a). It is postulated that the subsurface infrastructure associated with the urbanised area, such as the network of sewers and storm drains, provide a route for groundwater to discharge here, drawing down the groundwater level.

Another notable flow pattern is from north-east to south-west across the aquifer in the Port Meadow area, through the line of the River Thames, towards the Seacourt Stream. This flow pattern indicates that there is likely to be some recharge to the alluvial aquifer from higher terrace gravels to the north-east but also identifies discharge to the Seacourt Stream as having a strong influence on groundwater flows. The contours show that the level of the River Thames is higher than that of the Seacourt Stream by tens of centimetres.

As discussed in Section 3.1.2, the groundwater level gradient to the east of the River Thames was relatively low compared with that to the west, suggesting the maintenance of high levels of the River Thames in this area inhibits lateral groundwater flow. Data from the water level

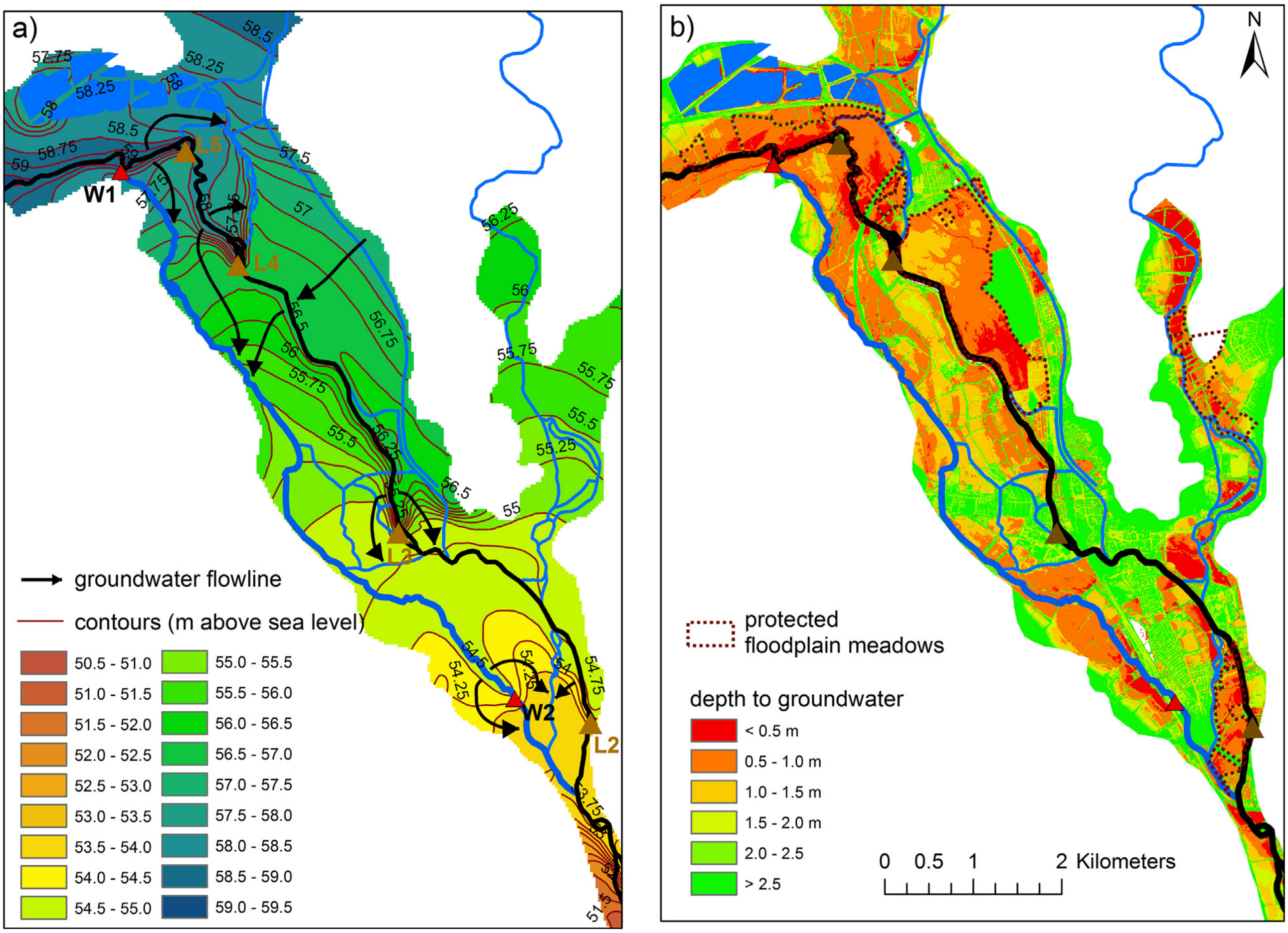

Fig. 6. a) contoured groundwater levels and groundwater flow lines within the alluvial floodplain aquifer; and b) depth to groundwater within the alluvial aquifer. Both maps are based on water levels measured in May 2007. Refer to Fig. 1 for elements not included in the legend. Contains Ordnance Survey data © Crown copyright and database right (2018).

loggers for the period 1 March 2010 to 28 February 2015 in boreholes T1a and T1d (Fig. 7), located on the western and eastern sides, respectively, of the River Thames in the Port Meadow area, highlight that the depth to groundwater in the floodplain aquifer on the eastern side was generally shallower (median: 1.01 mbgl in T1d; 0.24 mbgl in T1a). Geological logs from these boreholes show that for the majority of the time at T1a, the unsaturated zone remained within the fine-grained alluvium, whereas at T1d it extended to the underlying gravels.

The map of contoured depth to groundwater (Fig. 6b) highlights where the gravels of the modern floodplain rise up the valley sides, and the areas of man-made ground (e.g. Burgess Field licensed landfill), both resulting in relatively large depths to groundwater.

A comparison of maps in Fig. 6 shows that the raised levels of the River Thames created by the locks are associated with areas of aquifer where groundwater levels were relatively shallow, in the case of May 2007 often within 0.5 m of the ground surface (in red). Fig. 6b highlights the co-location of floodplain meadows and areas with relatively shallow groundwater. It may be that this co-location is, in part, related to waterlogging in the area upstream of river management structures, which historically made urban development more problematic, allowing floodplain meadows to survive. However, the co-location may also be because the soil moisture conditions associated with the shallow groundwaters created by the river management structures are suited to rare floodplain meadows plant communities. These plant communities have been shown to be highly sensitive to: soil moisture conditions, with centrimetric differences in groundwater level being linked with notably different plant assemblages (Silvertown et al., 2015); and the range of groundwater level fluctuations, influenced for example by dams and dykes (Leyer, 2005). Where river restoration schemes have been undertaken that have involved the raising of groundwater levels in the associated alluvial aquifers through the removal of deeply incised channels, studies of pre- and post-intervention have shown these are linked with substantial changes in floodplain vegetation composition (Loheide and Gorelick, 2007; Hammersmark et al., 2009).

3.2. Groundwater nitrate and associated parameters

Selected parameters measured at sites within the area, associated with nutrient cycling, are presented in Fig. 7 in the form of box plots. These box plots include all measurements from the October 2010 to February 2015 period. This dataset includes the same number of winter and summer sampling rounds (note, measurements below the detection limit are included in the box plots as half of the detection limit). The geographical grouping of sample sites is described in the caption of Fig. 7.

The data highlight the influence of the waste dump on the groundwater chemistry. High concentrations of DOC and NH_4^- in the LF group (medians 8.0 and 40 mg/L, respectively) are indicators that the waste dump is a significant pollution source. The NH_4^- concentrations in the LF group are possibly due to the reducing conditions in the alluvial sediments (median DO 0.47 mg/L). Anaerobic conditions and available OC also promote denitrification, which is consistent with the nitrous oxide (N_2O) concentrations measured by Gooddy et al. (2014), denitrification being the primary process that produces N_2O in groundwater (Jurado et al., 2017). Nitrate concentrations are above detection limit in only 38 of the 105 samples obtained from boreholes

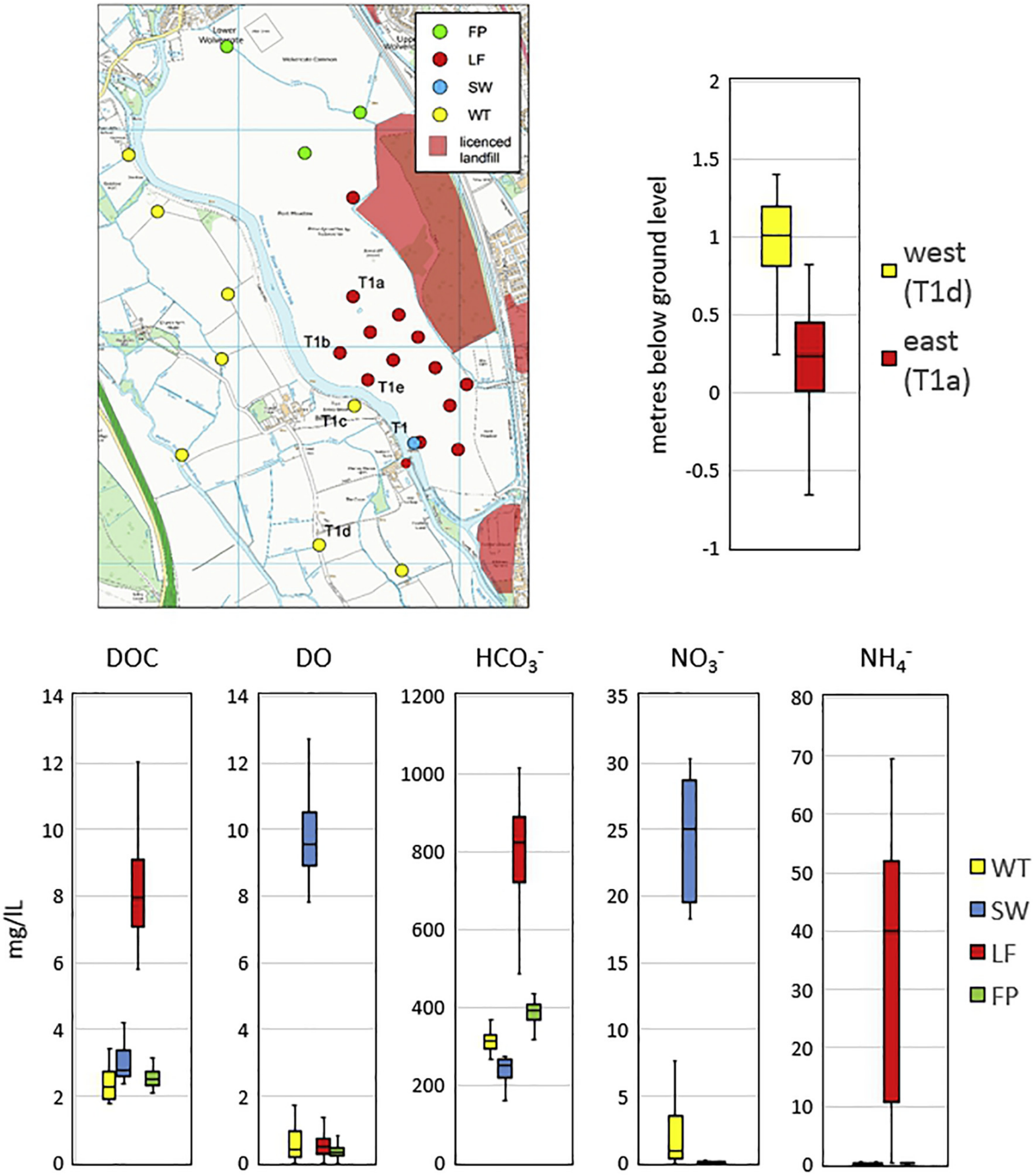

Fig. 7. Box plots of depths to groundwater and water chemistry for a series of monitoring sites (see: Fig. 1 for extent of this area within the overall study area; Fig. 4 for box plot legend; and Supplementary Information Table S1 for data used to produce the box plots). Sample sites are categorised as: groundwater west of the River Thames (WT); River Thames surface water (SW); groundwater east of the Thames in the floodplain under the influence of the Burgess Field landfill (LF); and groundwater east of the Thames in the other areas of the floodplain (FP). Labels are included for sites referred to in the text. Contains Ordnance Survey data © Crown copyright and database right (2018).

within the LF group. The median NO_3^- concentration in the LF group is low (0.04 mg/L), as is the standard deviation (0.26 mg/L), indicating there is limited spatial and temporal variability. Given the neutral pH levels in groundwater (Gooddy et al., 2014), high concentrations of HCO_3^- (median 823 mg/L) are also evidence of the oxidation of OC and denitrification (Vidon et al., 2010). Groundwaters from the FP group have comparatively low concentrations of NH_4^- (median 0.10 mg/L), low DO (median 0.32 mg/L), and high DOC and HCO_3^- concentrations (medians 2.5 and 393 mg/L, respectively). Very low groundwater NO_3^- concentrations (median 0.03 mg/L) suggest substantial denitrification in this zone, as in LF. The standard deviation in the NO_3^- concentration is low (0.05 mg/L), again indicating limited spatio-temporal variability.

The River Thames frequently flows out of bank onto the floodplain in the Port Meadow area to the east of the river; during the period of sampling there were two major floods. The NO_3^- concentration in the river water sampled as part of this study at T1 is high (median and interquartile range of 25 and 6.3 NO_3^- mg/L, respectively), in contrast to the groundwater. The low groundwater NO_3^- concentrations that occurred in the gravel aquifer here could be due to a range of factors: the limitation on recharge of high NO_3^- flood waters to the aquifer due to the shallow water table; dilution caused by low NO_3^- groundwater inflowing laterally to the area from the alluvial aquifer to the north; rainfall recharge; or NO_3^- removal by denitrification.

The mixing-cell model described in Section 2.3 was used to examine the likely contribution of each of these factors. Table 1 has values for the model parameters and input variables for the eastern zone of the River Thames (Port Meadow), as well as the resulting groundwater NO_3^- concentration, and the degree of denitrification required to match the observed average concentration here. Where the model is sensitive to parameters and input variables, low and high estimates of these are included to indicate the uncertainty in the denitrification factor calculated.

The model indicates that the groundwater NO_3^- concentrations are insensitive to both concentration of groundwater flowing laterally into this area of the aquifer, and the rainfall recharge. The model does show that the frequency of flooding, the NO_3^- concentration of river flood waters and the depth of groundwater prior to the flood are potentially major factors in determining the flux of NO_3^- into the floodplain aquifer. The model also shows that significant biogeochemical processing must be taking place for groundwater NO_3^- to remain as low as observed. The model requires a high NO_3^- removal factor for simulated NO_3^- concentrations to match the observed concentrations.

Nitrate concentrations in groundwater to the west of the river are higher than to the east, with a median of 0.90 mg/L, over an order of magnitude greater. The DO concentrations (median 0.40 mg/L) are similar to those to the east of the river. As presented in Section 3.1.3, water level contouring indicates groundwater flows north-east to south-west through the floodplain sediments and the line of the River Thames, towards the Seacourt Stream. A comparison of the EC of samples from a transect across the River Thames is used to estimate the proportion of the flow of groundwater in the western zone of the aquifer that is sourced from the river (assuming EC is conservative). The transect includes borehole T1e (44 m to the east of the River Thames), the river itself, and borehole T1c (42 m to the west of the river). Based on the median ECs (1773, 629 and 677 µS/cm, respectively) it is estimated that 95% of the lateral flow in the aquifer to the west of the Thames comes from the river. Although this river water had high NO_3^- concentration, the concentration in borehole T1c is low in comparison (median 2.3 mg/L), indicating that there is likely to be significant denitrification in the hyporheic zone of the river.

The mixing-cell model was also applied to the west of the Thames, using a lateral inflow of water with a NO_3^- concentration based on the average value from T1c. The model again identifies the frequency of flooding, the NO_3^- concentration of river flood waters and the depth of groundwater as important controls on the flux of NO_3^- into the floodplain aquifer but, in addition, that the lateral flow of water from the River Thames, with raised NO_3^- concentration, is also an important factor. Based on the model, the denitrification that occurs in this zone of the aquifer (Table 1), with its deeper water table, is less than to the east of the river although this has a greater degree of uncertainty. It is possible that the lower degree of denitrification compared with the eastern zone of the focussed study area is due in part to the deeper groundwater, that rises into the fine-grained alluvium for a shorter proportion of the year.

It is acknowledged that the mixing model is a simplified representation of the system that makes a number of assumptions about the flows within the floodplain system. However, it does indicate that it is very likely that overall there is substantial removal of NO_3^- within the floodplain sediments. This is important for the chemical quality of public and private groundwater supplies sourced from localised shallow aquifers. This indicates that in settings similar to those of the study area it may not be necessary to put in place measures to control river water quality if the primary purpose is to improve groundwater quality.

In this study it was not possible to compare pre- and post-construction conditions due to the historical nature of the engineered river management structures within the floodplain. However, evidence presented in Section 3.1 does show the influence of the locks on river and groundwater levels. There is a limited amount of peer-reviewed research that has been undertaken examining the influence of changes to structures in rivers on the nutrient status of groundwater in the associated aquifer through measurements before and after the intervention (Bellmore et al., 2017). However, in a study related to the impacts of the construction of small temporary dams, Hill and Duval (2009) showed that the rise in groundwater levels that occurred in the adjacent alluvial aquifer resulted in a significant reduction in groundwater NO_3^- concentrations; raised groundwater levels in the Oxford floodplain associated with the locks may have a similar influence.

3.3. Nitrate fluxes

This study highlights that in settings where there is no lateral regional-scale inflow of high-NO_3^- concentration groundwater, as in the case of Oxford where the floodplain aquifer is isolated by the underlying poorly permeable bedrock, river inflow can be the primary input of NO_3^- to groundwater. The influence of the river management structures on NO_3^- fluxes in the context of Oxford is complex. On river reaches influenced by the river management structures, where river levels were raised, river water recharged the alluvial aquifer due to the positive gradient from the river to the aquifer. The groundwater chemistry sampling shows that the hyporheic zone is highly efficient at reducing NO_3^- concentrations. To provide an approximation of the mass of NO_3^- that could be removed from recharging river water flowing through the river bed, we estimated a mass-balance for a section of the River Thames in the western zone of the focussed nutrient study area. The volume of water flowing through the river bed was estimated using Darcy's Law; parameter values were chosen to maximise the annual estimate of mass of NO_3^- removed. Parameters used were: depth of river 2 m (typical depth of the River Thames in the Oxford valley; Environment Agency, 2009); river-to-aquifer gradient 5%, based on median water levels over the period 2010 to 2015 at T1 and T1c; river bed permeability of 1 m/d (the maximum quoted for the River Thames by Younger et al., 1993); and a river NO_3^- concentration of 25 mg/L (the median concentration measured during the period of the focussed nutrient study; see Section 3.2). Assuming denitrification of 90% in river water passing through the river bed (as was seen to the west of the River Thames, based on median concentrations measured during the period of the focussed nutrient study), the mass of NO_3^- removed, calculated using these values, is 1.0×10^3 kg/a/km.

The implication from the groundwater chemistry in the eastern zone of the focussed nutrient study, and the mixing-cell modelling, is that the majority of NO_3^- that infiltrates the aquifer as a result of river overbanking onto the floodplain is removed through denitrification. Using the maximum river-water aquifer recharge calculated in the mixing-cell model, an average annual mass of NO_3^- that could potentially be removed in this zone is estimated as 0.8×10^3 kg/a/km.

These calculations provide an upper estimate of the NO_3^- removed through interaction between river and alluvial aquifer for a setting such as Oxford in which the underlying impermeable bedrock limits lateral inflow of high NO_3^- groundwater. If we apply the per kilometre NO_3^- removal to the full length of the River Thames through the Oxford valley, then the annual mass is 2.9×10^4 kg. For comparison, the annual River Thames in-channel flux of NO_3^- was calculated using the mean annual flow entering the study area and the river NO_3^- concentration quoted in Table 1. The resultant mass of 1.8×10^7 kg is three orders of magnitude greater than the flux of river water NO_3^- into the floodplain sediments. Even though the evidence from the study indicates most of this NO_3^- would be removed from the aquatic system through denitrification, its removal would only have a small impact on the downstream NO_3^- flux.

4. Conclusions

High spatio-temporal density of water level monitoring and a series of water quality surveys provided detailed insight into the hydrology and water chemistry of a river system and its associated alluvial aquifer. Although the Oxford floodplain setting has its own specific characteristics, it was used here to highlight general issues relating to the influence of managed rivers on groundwater flows within floodplain aquifers, and the implications for the flux of NO_3^-.

Engineered river management structures are commonplace both along the length of the River Thames, and in England and Wales in general, with references indicating their widespread occurrence in other countries. The study shows the degree to which river management structures can control river levels. Raised river levels in reaches upstream of structures were shown to create zones of increased groundwater storage within adjacent alluvial aquifers. As a result other smaller water courses can become important locations for groundwater discharge. Such discharge patterns result in complex flows that cause water entering the aquifer to follow significantly longer flow paths to reach surface water discharge zones than might occur under a more natural hydrological regime.

The study highlights that in settings where there is no lateral regional-scale inflow of high NO_3^- concentration groundwater, a primary input of NO_3^- to groundwater can be from the inflow of river water to the alluvial aquifer when the river is in flood and via the river bed where there is a positive hydraulic gradient from river to aquifer.

In relation to NO_3^- fluxes, the influence of the river management structures can be complex, as illustrated in the Oxford study. Here, on river reaches influenced by the river management structures, the positive gradient from the river to the aquifer can increase the flux of NO_3^- into the floodplain aquifer, however, the hyporheic zone has been shown to be highly efficient at denitrifying the water.

Raised groundwater levels, associated with river management structures, also create conditions that help to control NO_3^- concentrations within the alluvial aquifer: a shallow water table limits the volume of high NO_3^- water that can infiltrate through the floodplain; a shallow unsaturated zone contained within the carbon-rich, fine-grained sediments promotes anaerobic conditions; and longer residence times associated with complex flowpaths mean more time for groundwater denitrification to occur. The simple mixed-cell modelling undertaken within this study indicates that a large proportion of the NO_3^- entering floodplain aquifers under similar conditions is likely to be removed by denitrification. However, as efficient as floodplain aquifers may be locally in removing the influx of NO_3^- where very shallow groundwater conditions occur, as this study has shown, the amount removed may be a small proportion of the in-channel flux where river NO_3^- concentrations are high due to upstream inputs, such as point source discharges and high NO_3^- groundwater baseflow associated with agricultural activities.

The study also identified that shallow groundwaters associated with river management structures were spatially correlated with the location of protected floodplain meadows. Other research is highlighted in which modifications to engineered river management structures, such as those related to river restoration schemes, that resulted in changes to groundwater levels, have had significant impacts on sensitive floodplain vegetation, as well as on the potential for the removal of nutrients from the aquatic system.

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Declarations of interest

None.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ecoleng.2018.09.005.

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SUPPLEMENTARY INFORMATION

Site	water level percentiles				
	5th	25th	50th	75th	95th
S1	53.81	53.86	53.90	53.97	54.21
S2	54.37	54.45	54.58	54.77	55.32
S3	55.04	55.05	55.10	55.25	55.78
S4	56.20	56.29	56.38	56.48	56.86
L2 d/s	53.80	53.83	53.86	53.90	54.12
L2 u/s	54.68	54.72	54.74	54.77	54.82
L3 d/s	54.68	54.72	54.76	54.87	55.56
L3 u/s	56.45	56.48	56.50	56.52	56.56
T1	56.48	56.52	56.56	56.72	57.29
L4 d/s	56.52	56.57	56.65	56.95	57.53
L4 u/s	58.08	58.12	58.15	58.19	58.26
T2	58.07	58.13	58.17	58.24	58.42
L5 d/s	58.10	58.16	58.20	58.30	58.73
L5 u/s	58.87	58.91	58.96	59.01	59.09
T3	58.83	58.88	58.92	58.99	59.17
L6 d/s	58.90	58.96	59.03	59.22	59.86
L6 u/s	59.72	59.76	59.80	59.84	59.99

Table S1 Percentile water levels for sites on the Seacourt Stream and River Thames (see Figure 2 for site locations), for 1 April 2009 to 31 March 2013. Note, d/s and u/s refer to downstream and upstream, respectively, water levels for locks.

Parameter	Site	No. measurements	percentiles				
			5th	25th	50th	75th	95th
Depth to groundwater	T1d	1826	0.20	0.82	1.01	1.20	1.32
	T1a	1826	-0.48	0.01	0.24	0.45	0.65
NO_3	WT	20	0.01	0.42	0.90	3.3	7.9
	SW	11	18	21	25	27	30
	LF	105	0.01	0.01	0.04	0.09	0.53
	FP	17	0.01	0.01	0.03	0.06	0.12
NPOC	WT	20	1.8	1.9	2.3	2.7	2.9
	SW	11	2.4	2.6	2.8	3.4	3.8
	LF	103	6.3	7.1	8.0	9.1	15
	FP	17	2.1	2.4	2.5	2.7	3.3
DO	WT	30	0.10	0.20	0.40	0.90	3.6
	SW	11	8.1	9.2	9.5	10	12
	LF	115	0.00	0.28	0.47	0.72	3.5
	FP	24	0.00	0.20	0.32	0.41	0.78
HCO_3	WT	43	266	294	315	328	351
	SW	14	178	222	251	265	271
	LF	121	583	724	823	890	969
	FP	23	298	371	393	404	433
NH_4	WT	26	0.00	0.04	0.08	0.19	0.32
	SW	20	0.02	0.05	0.12	0.25	0.98
	LF	93	0.32	11	40	52	59
	FP	17	0.04	0.08	0.10	0.12	0.63

Table S2 Percentiles for sites box plots of depth to groundwater and water chemistry for a series of monitoring sites shown in Figure 6 (units: depth to groundwater, m; chemical parameters, mg/l). Sample sites are categorised as: groundwater west of the River Thames (WT); River Thames surface water (SW); groundwater east of the Thames in the floodplain under the influence of the Burgess Field landfill (LF); and groundwater east of the Thames in the other areas of the floodplain (FP).

Chapter 3: Nitrogen sources, transport and processing in peri-urban floodplains

The work presented in this chapter has been published in the following journal paper:

Gooddy, D.C., Macdonald, D.M.J., Lapworth, D.J., Bennett, S.A. & Griffiths, K.J. (2014). Nitrogen sources, transport and processing in peri-urban floodplains. *Science of the Total Environment*, 494, 28-38.

Statement of contribution, as David Macdonald is not the lead author:

My overall percentage contribution to this paper was 35% and my specific contribution was to:

- devise the research question and design the methodology, with Gooddy and Lapworth;
- carry out the GIS analysis to identify how representative the study area is of the Thames catchment;
- co-design and supervise the installation of the borehole monitoring network;
- undertake elements of the groundwater sampling, as part of a team of five fieldworkers;
- coordinate the measurement of particle size distribution and calculate hydraulic conductivities;
- undertake the flux calculations;
- along with Gooddy and Lapworth, interpret the results; and
- write substantial sections of the methodology, results and discussion and produce 3 of the 6 figures.

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Nitrogen sources, transport and processing in peri-urban floodplains

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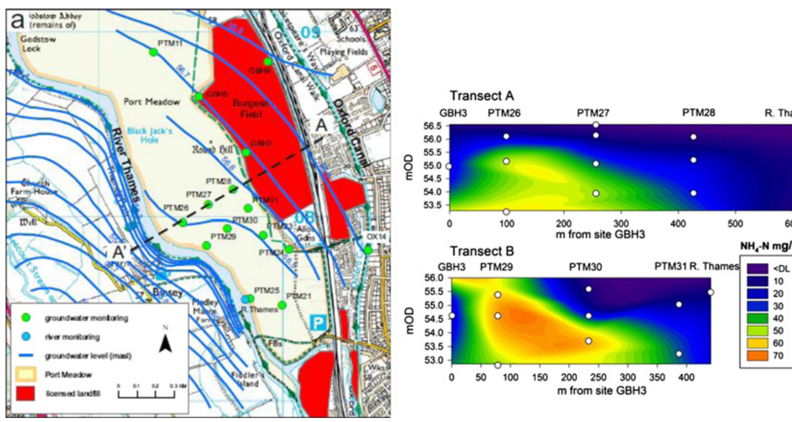
HIGHLIGHTS

- Peri-urban floodplains have been found to be populated with legacy landfills.
- Very high concentrations of nitrogen as ammonium are found.
- Isotopic methods identify legacy landfill as the ammonium source.
- Ammonium is not attenuated and is reaching the river.
- It is estimated that 27.5 tonnes of ammonium may be delivered to the river annually.

GRAPHICAL ABSTRACT



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ABSTRACT

Peri-urban floodplains are an important interface between developed land and the aquatic environment and may act as a source or sink for contaminants moving from urban areas towards surface water courses. With increasing pressure from urban development the functioning of floodplains is coming under greater scrutiny. A number of peri-urban sites have been found to be populated with legacy landfills which could potentially cause pollution of adjacent river bodies. Here, a peri-urban floodplain adjoining the city of Oxford, UK, with the River Thames has been investigated over a period of three years through repeated sampling of groundwaters from existing and specially constructed piezometers. A nearby landfill has been found to have imprinted a strong signal on the groundwater with particularly high concentrations of ammonium and generally low concentrations of nitrate and dissolved oxygen. An intensive study of nitrogen dynamics through the use of N-species chemistry, nitrogen isotopes and dissolved nitrous oxide reveals that there is little or no denitrification in the majority of the main landfill plume, and neither is the ammonium significantly retarded by sorption to the aquifer sediments. A simple model has determined the flux of total nitrogen and ammonium from the landfill, through the floodplain and into the river. Over an 8 km reach of the river, which has a number of other legacy landfills, it is estimated that 27.5 tonnes of ammonium may be delivered to the river annually. Although this is a relatively small contribution to the total river nitrogen, it may represent up to 15% of the ammonium loading at the study site and over the length of the reach could increase in-stream concentrations by nearly 40%. Catchment management plans that encompass floodplains in the peri-urban environment need to take into account the likely risk to groundwater and surface water quality that these environments pose.

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1. Introduction

In recent decades anthropogenic inputs of N, for food production by intensive agriculture and urbanisation, have caused increases in macronutrient fluxes, and have led to widespread N pollution of aquatic systems (Foster et al., 1982; Burt et al., 2011; Whitehead and Crossman, 2012; Lapworth et al., 2013). This is a global issue with implications for food production and security, water quality and land management/planning (Galloway, 1999; Galloway et al., 2004). In Europe and North America there have been a series of policies and accompanying regulations which have focused on reducing point and diffuse N and P pollution and associated problems of eutrophication and ecological degradation of freshwaters. For example, the European Water Framework Directive (WFD 2000) has the aim for fresh and marine waters to reach 'good ecological status' by 2015.

The zone of transition from rural to urban land-use is often referred to as the peri-urban area. Un-lined waste sites or landfills are a common source of N and C inputs to surface and groundwaters in peri-urban floodplains (Wakida and Lerner, 2005; Corniello et al., 2007). In Europe and North America there is a legacy of historic pollution from these sites (e.g. Heaton et al., 2005; Lorah et al., 2009). As such, landfill leachate plumes may contain high dissolved organic C (DOC) and NH_4^+ concentrations as well as ferrous iron, chloride and bicarbonate relative to natural floodplain conditions (Lorah et al., 2009). These commonly develop a series of distinct redox zones (Lyngkild and Christensen, 1992). In such systems, dilution, sorption and denitrification may be effective attenuation processes within the shallow groundwater system. The attenuation of N pollution from landfills, as well as other sources within peri-urban settings, requires fluctuations in redox conditions. These may be expected in floodplain settings due to rapid changes in water levels and episodes of surface water recharge and inundation.

Nitrogen in the form of ammonium (NH_4^+) is present naturally in groundwater as a result of anaerobic degradation of organic matter, and in the form of nitrate (NO_3^-) from the microbial oxidation of NH_4^+. However, NH_4^+ and NO_3^- also occur in groundwater from anthropogenic sources. High concentrations of NH_4^+ (10–1000 mg/L) have been found in groundwaters impacted by landfill leachates and as a result of domestic and agricultural waste water disposal practices (Gooddy et al., 1998; Lawrence et al., 2000; Christensen et al., 2001; Heaton et al., 2005). High concentrations of NO_3^- are typically association with diffuse agricultural pollution from fertilisers (Oakes et al., 1981; Addiscott et al., 1991) although oxidation of anthropogenic ammonium sources also causes high NO_3^- concentrations (Gooddy et al., 2002).

Ammonium transport in the subsurface may be retarded by sorption (Ceazan et al., 1989; Buss et al., 2004). Both NH_4^+ and NO_3^- can be attenuated through microbially-induced transformations (DeSimone and Howes, 1998). Ammonium oxidation commonly occurs in conjunction with oxygen reduction and is termed nitrification. This results in the production of nitrite (NO_2^-) followed by NO_3^-.

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O \quad (1)$$

$$NO_2^- + 0.5O_2 \rightarrow NO_3^- \quad (2)$$

In addition to degrading groundwater quality, NO_3^- and NH_4^+ can both be substantial sources of N in surface waters receiving groundwater (Jackson et al., 2007). Therefore it is desirable that this nitrogen is completely removed from the aquatic system. This requires bacterially mediated denitrification to convert the nitrate through intermediate stages to nitrous oxide and ultimately to nitrogen gas (Eq. (3)), which is a process requiring anaerobic conditions. This process is controlled by the availability of soluble carbon, redox status, pH as well as soil/groundwater residence times and hydrology (Haycock and Burt, 1993; Thomas et al., 1994; Burt et al., 1999; Gooddy et al., 2002).

$$NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2(g) \quad (3)$$

Therefore attenuation of N pollution from organic wastes requires alternating redox conditions, from oxidising to reducing, such as generated during water table fluctuations. Alternatively, through the anammox process, NH_4^+ can be oxidised anaerobically to nitrogen gas through the reduction of (NO_2^- derived from NO_3^- (see Eq. (3) above).

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \quad (4)$$

Despite the environmental importance of NO_3^- and NH_4^+, there are few studies documenting their transport and reaction processes in aquifers (Heaton et al., 2005). Isotopic fractionation studies can provide an excellent tool for understanding N transport and speciation (Wassenaar, 1995; Böhlke et al., 2006). Isotopic fractionations have been reported for NH_4^+ sorption to clays (Karamanos and Rennie, 1978), with the remaining NH_4^+ in solution relatively depleted in ^{15}N. By contrast, nitrification results in a substantial increase in ^{15}N for the remaining NH_4^+ (Delwiche and Steyn, 1970). Stable isotope ratios in NO_3^- have often been used to distinguish various sources of NO_3^- in groundwater, such as synthetic fertilisers and animal wastes (Gormly and Spalding, 1979; Flipse and Bonner, 1985). Denitrification causes an isotopic enrichment in the remaining nitrate (Mariotti et al., 1988). A disadvantage of the single isotope approach to denitrification studies is that process such as ammonia volatilisation can also lead to enrichment of ^{15}N in the residual NH_4^+ source material and in the NO_3^- produced during nitrification. However, by combining the $\delta^{18}O$ of the NO_3^-, a more reliable indicator of the denitrification process is achieved (Böttcher et al., 1990; Wassenaar, 1995; Kendall, 1998; Fukada et al., 2004). The opposing isotope fractionation effects make it possible to distinguish between sorption, nitrification and denitrification as major processes affecting N distribution in a field setting through evaluating variations in concentration and isotopic composition.

Riparian floodplains provide an important interface between terrestrial and aquatic systems (Harms and Grimm, 2008). Floodplain aquifers can be important sources of drinking water and sustain baseflow in surface waters, with important ecological implications (Sophocleous, 2002; Murray-Hudson et al., 2006). However, globally, floodplains are increasingly being encroached and developed due to anthropogenic pressures including urbanisation and intensive agriculture (Tockner and Stanford, 2002; Pinter, 2005; Werritty, 2006). In many parts of Europe, and elsewhere, there is a historical legacy of change in land use and associated historical pollution loading to floodplain groundwaters and surface waters (Burt et al., 2011; Stuart and Lapworth, 2011). Peri-urban floodplains are therefore complex, in terms of spatial heterogeneity in land use and topography, and temporal variability in recharge and redox processes (Burt et al., 2002; Burt and Pinay, 2005; Macdonald et al., 2012a; in press).

Due to changing redox conditions, floodplains and riparian settings are considered hot-spots for nutrient attenuation (Devito et al., 1999; McClain et al., 2003; Harms and Grimm, 2008). Groundwater levels within floodplain environments are typically shallow and responsive to recharge events. Vertical and spatial soil moisture conditions are also highly variable, and together these have important implication for attenuation of oxidised N (Burt et al., 1999). Periods of inundation may stimulate denitrification due to the mobilisation of C pools at shallow depths; during periods of low water tables denitrification may be restricted due to reduced C pools, low soil moisture and more oxidising conditions.

The important ecosystem services provided by floodplains and their general proximity to potentially damaging agricultural and urban nutrient sources means there is an imperative to understand nutrient processes, fluxes and attenuation mechanisms within. Due to the complex hydrogeology estimating the residence times of groundwaters and legacy nitrogen pollutants is challenging. There is therefore uncertainty as to the timescale for changes in management practices at the surface, reducing baseflow nutrient fluxes to surface waters (Wang et al., 2012; Stuart et al., 2011; Jackson et al., 2008).

This paper examines the temporal and spatial variations in N species within a typical peri-urban setting in the floodplain of the River Thames, UK. The peri-urban hydrological setting, legacy of pollution, and dependent terrestrial ecosystems means that the study is relevant across many industrialised and rapidly industrialising regions. The aim is to understand the water movement and redox status across the case study peri-urban floodplain and how this influences nutrient fluxes to the adjoining river. The objectives are to use geochemical indicators together with $\delta^{15}N$ of NH_4^+ and $\delta^{15}N$ and $\delta^{18}O$ of NO_3^- to: 1) evaluate the origin of the nitrogen in groundwater; 2) ascertain whether sorption, nitrification or denitrification was occurring to attenuate the nutrient loading; and 3) estimate the amount of nitrogen that is delivered to the river system.

2. Materials and methods

2.1. Study area

The study focuses on a section of the floodplain of the River Thames in the vicinity of the city of Oxford in the southern UK (Fig. 1). The study area, known as Port Meadow, is to the north-west of the historic city centre. Port Meadow is an area of communal grazing and is regularly flooded by the River Thames, which flows along its western border. Due to areas of persistent groundwater flooding, the meadow supports a large number of waterfowl. Port Meadow is part of the larger Oxford Meadows Special Area of Conservation (SAC). The study area (Fig. 2a) also includes: the Burgess Field Nature Reserve, a former waste dump located to the east of Port Meadow; various tracts of land to the east, sited on the current floodplain, which were formerly the location of industrial landfills; the urbanised higher ground to the east of the floodplain; and the agricultural areas to the west of Port Meadow and the River Thames. The study area is bounded to the north and the south by urban developments.

The River Thames flows in a southerly direction through the city of Oxford (Fig. 1). The mean annual flow of the Thames upstream of the study area is 18.48 m^3s^{-1} (Marsh and Hannaford, 2008). The baseflow index (the ratio of long-term baseflow to total flow) for the river at this location is 0.67, reflecting the influence of influent groundwater, sourced from the limestone aquifers located in the headwaters, and the extensive floodplain gravel aquifers. The River Thames floodplain is approximately 2 km wide in the Port Meadow area, with a low gradient of less than 0.1° (0.03%). Groundwater flow is predominantly north-east to south-west; Fig. 2a includes groundwater level contours for a low-flow period. Normally groundwater flows into the River Thames from the north-east along the full reach within the case study area. Water management structures on the Thames a few hundred metres downstream of Port Meadow are a control on the hydrology of the area, by maintaining the river at relatively high levels during low-flow periods (Macdonald et al, 2012a). In the furthest downstream reach of the River Thames in the study area this can cause a reversal in hydraulic gradient for a few weeks of the year, allowing eastward flow from the

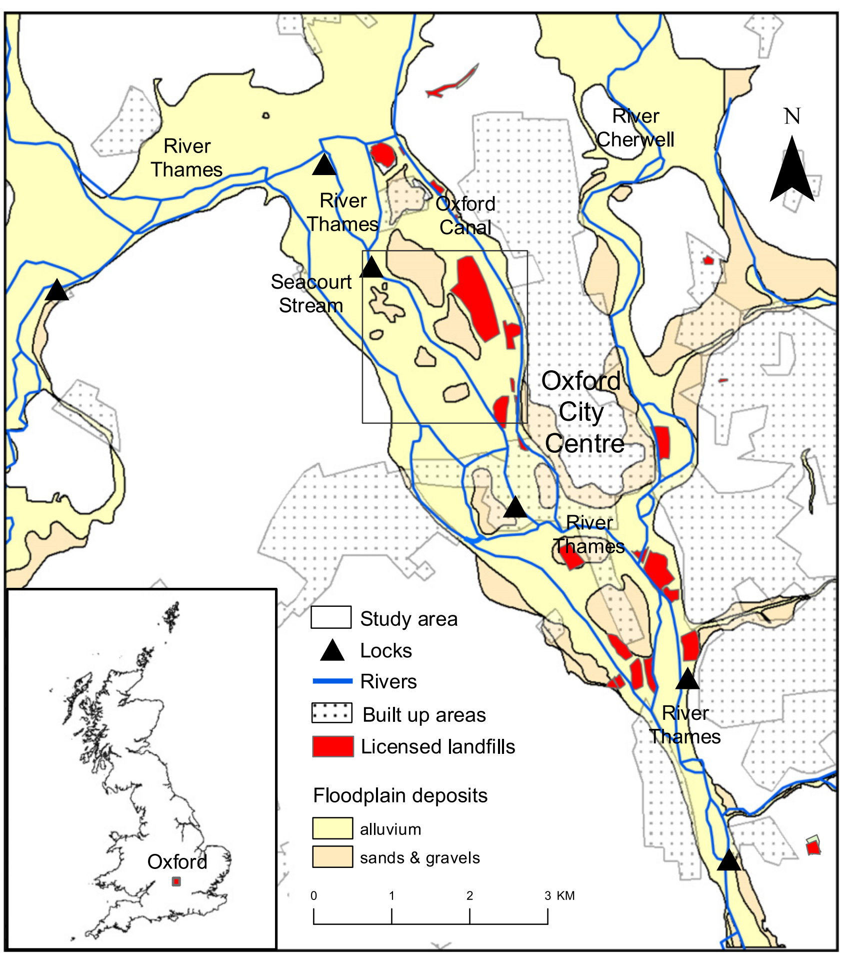

Fig. 1. The River Thames floodplain in the vicinity of the city of Oxford.

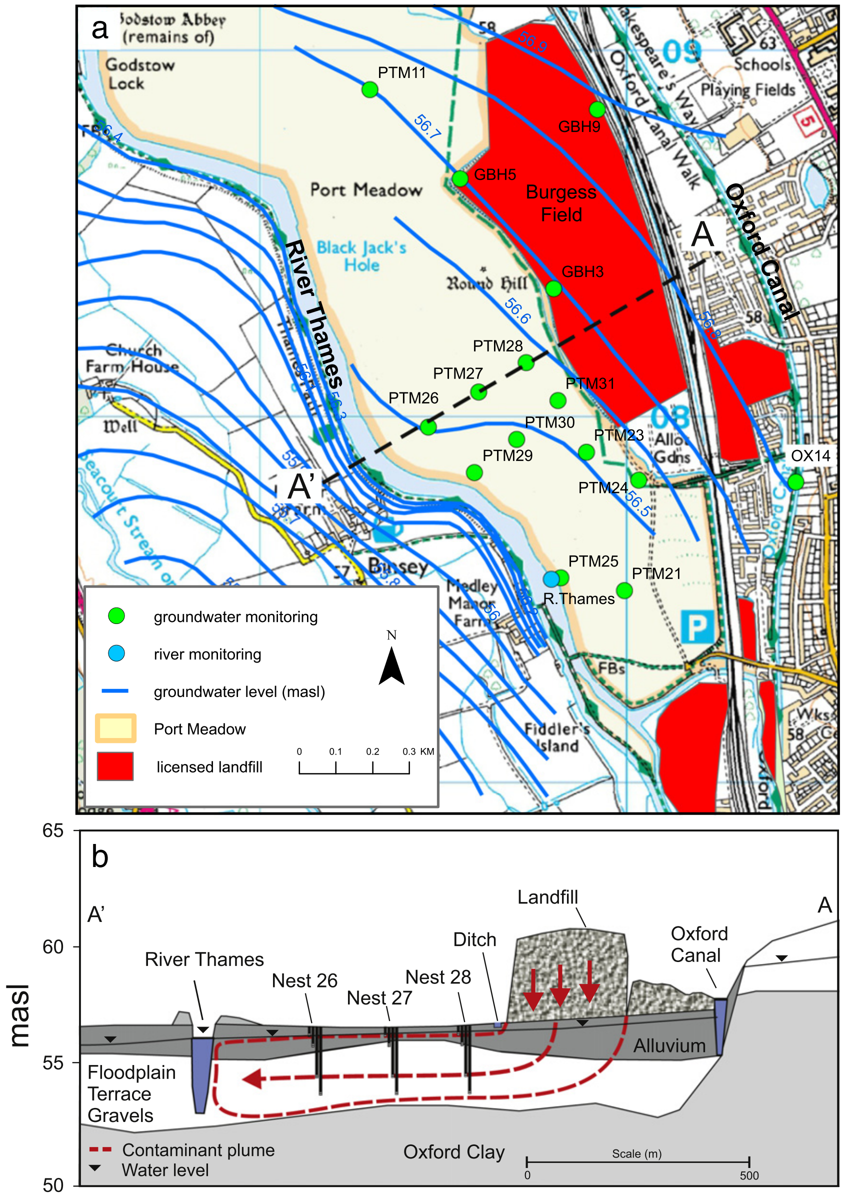

Fig. 2. a) Study area with monitoring network and groundwater level contours based on measurements made between 3 and 7 October 2011. Note, symbols for sites PTM26 to PTM31 indicate the centres of the piezometer nests. b) Simplified geological cross-section showing major topographic features and direction of landfill plume.

river into the floodplain deposits. The Seacourt Stream flows along the western edge of the floodplain. Flow in this stream comes in part from the River Thames to the north, controlled by a fixed head weir. The stream is also a major groundwater discharge line within the floodplain (Fig. 2a); groundwater flows from the direction of Thames. The Oxford Canal runs along the eastern edge of the floodplain. There are also a series of drainage ditches across the floodplain.

The floodplain deposits within the Oxford Valley are made up of a shallow surface layer of fine-grained alluvium, underlain by highly permeable sands and gravels (Fig. 2b). The alluvium thickness can be up to 4 m but is typically around 1.5 m (Macdonald et al., 2012a). The sand and gravel thickness is mostly within the range of 2 to 6 m. The floodplain sediments are underlain by mudstones of the Upper Jurassic Oxford Clay Formation with some limited connection between the current floodplain sediments and second terrace gravels forming higher ground to the east of Port Meadow.

Port Meadow and the surrounding area have a long legacy of anthropogenic contamination with documented waste sites dating back to the late 1800s (Macdonald et al., 2012b). Burgess Field was used for domestic waste from 1937 to 1980. The waste here was dumped onto the floodplain surface with no underlying or lateral impermeable barriers apparent. The dump covers an area of 0.34 km^2, and has an average height of ~4 m above the natural ground level. The categories of waste reported to have been dumped there are: inert materials (soil, brick, concrete, glass, clay, sand etc.); semi-inert materials (wood, paper, cardboard, plasterboard, plastic, etc.); biodegradable wastes (food, sewage sludge, household, garden etc.); difficult wastes (e.g. tyres, sludges); and special wastes (hazardous chemical wastes, asbestos). Since closing, the waste dump has been grassed and planted with trees and shrubs. To the south of Burgess Field (Fig. 2a) there are a number of other areas which received domestic, building and industrial wastes during the late 19th and early 20th centuries.

The pollutant load to the floodplain of the River Thames in Oxford is thought to be typical of a substantial proportion of the floodplain within the overall Thames catchment. To assess the proportion of the Thames floodplain that can be classed as peri-urban, within a GIS a 1 km buffer was delineated around the large urban areas defined within the Ordnance Survey Strategi dataset. The floodplain within the Thames catchment was approximated using the BGS Geological Indicators of Flooding dataset (Booth and Linley, 2010) and the interception between this and the peri-urban approximation was calculated (Fig. 3). The area of floodplain is estimated as 1619 km^2; the area of peri-urban floodplain is 720 km^2. This analysis indicates that significant areas within the Thames catchment can be classified as peri-urban and that insights from the Oxford case study will be have wider application.

2.2. Sampling

The sampling programme included 27 groundwater and 1 surface water sites across the study area (Fig. 2a). The sites include a range of land uses and were located up flow-gradient of the landfill and floodplain, down flow-gradient of the landfill and three sites located in the landfill. The existence of the unlined Burgess Field landfill was anticipated to exert an influence on the chemistry of the area so two new transects running perpendicular from the landfill towards the River Thames were installed (PTM26–28, PTM29–31). Each transect contains three nests of four piezometers, installed to target different depth horizons and sample the landfill plume (Fig. 2b). The piezometers within the nest were drilled separately, located within an area of approximately 4 m^2. The depth horizons approximately correspond to the soil, the alluvium, the top of the gravels and towards the base of the gravels. The piezometers within a nest are identified by letters A (shallowest) to D (deepest). The use of transects of nested piezometers with discrete screened intervals provides a good constraint on the vertical and lateral changes in hydrogeochemistry. Additional details of the sample sites included in this study are given in Macdonald et al. (2012b). To assess the impact of seasonal fluctuations in the groundwater level on redox conditions and nutrient hydrogeochemistry, sampling was conducted at approximately quarterly intervals between May 2010 and May 2012, with a further sampling round in August 2013; in total 12 rounds were undertaken. The piezometer headworks were completed in a 100 mm diameter chamber below ground level, for aesthetic, security and safety reasons, and capped with a metal plate below a grass sod so that it can be located with a metal detector. During periods of flooding a short length of 150 mm diameter pipe is driven into the ground around the chamber, acting as a cofferdam and allowing water to be pumped away from the headworks before sampling. The design and use of the piezometers is shown in Fig. 4.

A minimum of three borehole volumes were purged from each groundwater sampling site, and samples were not collected until stable readings for pH, specific electrical conductivity (SEC) and dissolved oxygen (DO) were obtained. Samples for chloride and nitrogen species were filtered at 0.45 μm and collected in 30 mL plastic bottles. Samples for ^{15}N analysis of NH_4 or NO_3 and ^{18}O-NO_3 were collected on August 4th 2013 and filtered into plastic 1 L bottles; the samples for NH_4 analysis being acidified in the field with HCl to pH 2–4. These samples were frozen and to be defrosted just before analysis. Samples for dissolved gases were collected in sealed steel ampoules at the same time as the samples for isotopic analysis as well as at selected sites on two other occasions in July 2011 and October 2011.

2.3. Chemical analyses

The samples were collected and analysed for Cl, NO_3 and NO_2 using ion chromatography (IC), and NH_4 by flow colorimetry. Field data including bicarbonate, pH, temperature, specific electrical conductance (SEC), and dissolved oxygen (DO) were all determined at site, and a flow through cell was used for unstable field parameters to obtain representative in-situ values. Dissolved nitrous oxide was measured at the British Geological Survey's Environmental Tracer Laboratories (Wallingford, UK) by Gas Chromatography with an electron capture detector (ECD) and a 3m Porapak™ Q column held isothermally at 40 °C. N_2O gas standards at 10 ppm and 100 ppm were used for calibration (Bedfont Scientific Limited, Rochester, England) and gave a linear response with the ECD. Analytical precision (1 SD) was typically <1% (Gooddy et al., 2002).

Isotope preparation and analysis was carried out at the NERC Isotope Geosciences Laboratory (Keyworth, UK). Nitrate was separated on anion resins and prepared as silver nitrate using the method of Silva et al. (2000). Ammonium was converted to NH_4SO_4 on acidified quartz filter papers using a static ammonia diffusion technique (Sebilo et al., 2004). The filters were combusted to produce N_2 for $^{15}N/^{14}N$ analysis. Ammonium and nitrate $^{15}N/^{14}N$ ratios were analysed by combustion in a Flash EA on-line to a Delta Plus XL mass spectrometer (ThermoFinnigan,

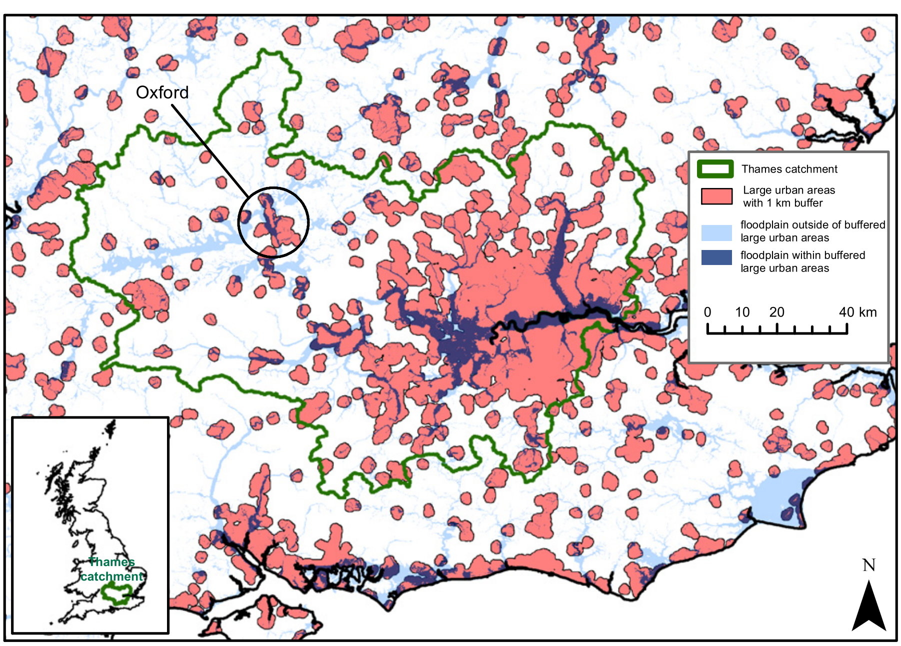

Fig. 3. Peri-urban floodplains in the catchment of the River Thames.

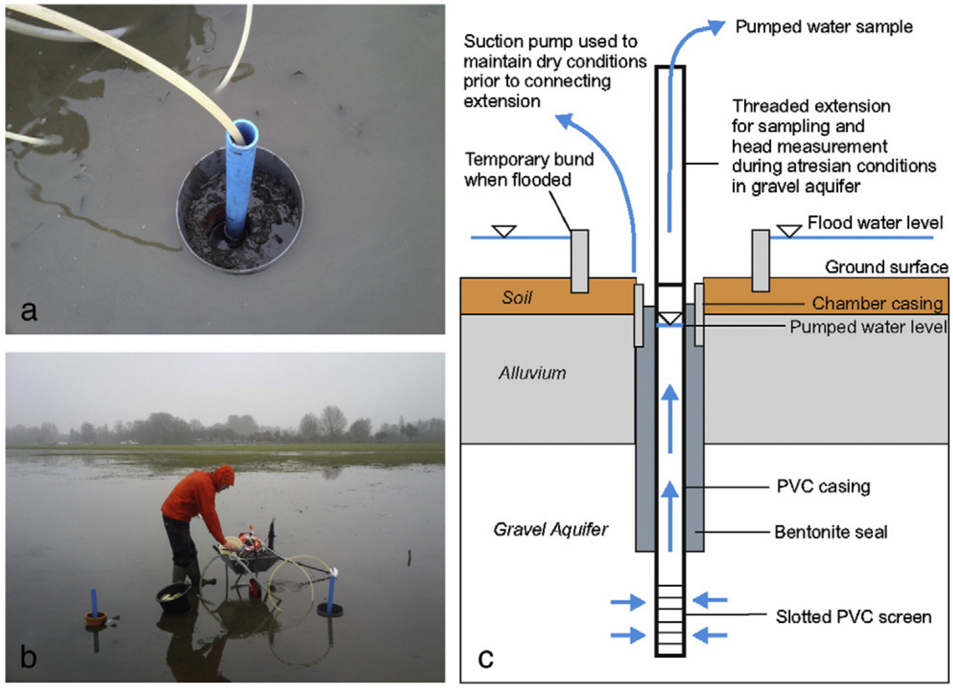

Fig. 4. Groundwater sampling during flood events: a) exposed well-head, b) sampling equipment, and c) piezometer design.

Bremen, Germany). Isotope ratios were calculated as $\delta^{15}N$ values versus air (atmospheric N_2) by comparison with standards calibrated against IAEA N-1 and N-2 assuming these had values of +0.4‰ and +20.3‰, respectively. Analytical precision (1 SD) was typically <0.8‰, from repeat analysis of a sample. $^{18}O/^{16}O$ ratios of nitrate were analysed by thermal conversion to CO gas at 1400 °C in a TC–EA on-line to a Delta Plus XL mass spectrometer (ThermoFinnigan, Bremen, Germany). Isotope ratios were calculated as $\delta^{18}O$ values versus VSMOW by comparison with IAEA-NO_3 assuming it had a value of +25.6‰. Analytical precision (1 SD) was typically <1.2‰.

2.4. Flux calculations

Darcy's Law was used to calculate the flux of a conservative tracer (in this case chloride) from the floodplain sediments into the River Thames along the reach parallel to the Burgess Field landfill. Parameters used to calculate the flux are given in Table 1. The hydraulic conductivities (K) of the alluvium and the sands and gravels were estimated based on particle-size analyses of material obtained during the drilling of piezometers in Port Meadow, using the method of Boonstra and de Ridder (1981); the method relates K to the specific surface area of the sediment. Although sufficiently permeable to allow groundwater recharge to occur, the alluvium was estimated as having K values three orders of magnitude smaller than the gravel K; the alluvium was therefore ignored in the calculation of lateral flux. Given the accuracy of the method, the average gravel K was rounded to the nearest 100 as was the range of potential K values (indicated in parentheses in Table 1).

Table 1
Parameters used to calculate flux from Burgess Field landfill to the River Thames through the floodplain sediments.

Parameters	Value
Gravel aquifer hydraulic conductivity (m/d)	400 (±200)
Groundwater river inflow cross-sectional area (m)	2.5
Groundwater level gradient (−)	0.0004
Landfill length, parallel to river (m)	1000
Groundwater inflow to river along affected reach (m^3/d)	400 (±200)
Mean daily river flow (m^3/d)	1.60×10^6

The river bed K was assumed to be equal to the gravel K as the correlation between river level and groundwater level in PTM25, which is adjacent to the river, was very high ($r^2 = 0.99$).

Geological logs of boreholes in the vicinity of the River Thames show an average thickness of alluvium of 0.8 m and depth to the base of the gravel aquifer of 4.4 m; a river survey undertaken for the Environment Agency of England has the bed elevation at its deepest as 2.8 m below the floodplain ground level. Taking these into account, as well as a likely vertical groundwater flow component to the river, a cross-sectional area for groundwater inflow to the river from the gravel aquifer of 2.5 m was used. Groundwater and river level monitoring allowed the calculation of the head gradient towards the river; the direction of flow is approximately perpendicular to the river for the majority of the period being considered (Macdonald et al., 2012b; Fig. 2a). The gradient outside of periods of short-lived high river stage ranges from 0.0008 to −0.0003; a median value of 0.0004 was used in the flux calculation. The total daily inflow of groundwater to the river along the Burgess Field reach was compared to the mean daily river flow, in Table 1.

Combining the inflows with the concentrations of dissolved species measured in boreholes between the landfill and the River Thames enabled the flux of species into the river from the floodplain aquifer to be compared with the downstream flux in the river (see the Results section).

3. Results

Table 2 shows the range of results observed for selected determinants measured over the past three years. There is relatively little seasonal variation for any given site with the greatest temporal changes being seen when sampling has followed a recent heavy rainfall event (Macdonald et al., 2012b). Consequently, greatest temporal variation is observed in the shallow piezometers. Data for electrical conductivity shows this particularly well, with the majority of measurements made in August 2013 (square brackets) within 5% of the mean conductivity sampled over the study period. Mean pH values are generally just below 7 with the highest values (>7) found in the shallower (<1 m) piezometers and the two boreholes to the far south (PTM21 and OX14)

Table 2
Borehole construction details and on-site field measurements data for selected boreholes in the Port Meadow network.

Site	Borehole depth[a] m	Geology at screen	SEC µS/cm	pH	DO mg/L
GBH3	5.09	See note[b]	2027 (1944–2110)	6.77 (6.68–6.86)	0.83 (0.46–1.20)
GBH5	5.60	See note[b]	1560 (1438–1681)	6.94 (6.68–7.19)	0.14 (<0.1–0.23)
GBH9	5.24	See note[b]	1463 (1403–1522)	6.86 (6.68–7.03)	0.38 (0.26–0.50)
OX14	2.72	Gravel	824 (743–977) [798]	7.43 (7.21–7.72) [7.34]	0.64 (0.19–2.04) [0.30]
PTM11	3.38	Gravel	772 (640–860) [755]	7.21 (6.93–7.99) [6.93]	0.40 (<0.1–0.79) [0.22]
PTM21	1.40	Gravel	1071 (914–1283) [1080]	7.19 (6.87–7.67) [7.67]	2.03 (0.66–4.81) [0.69]
PTM23	1.67	Gravel	1661 (957–1957) [1531]	6.99 (6.81–7.40) [6.89]	3.61 (0.45–10.6)
PTM24	0.99	Gravel	1075 (640–1250) [1180]	6.89 (6.65–7.18) [6.65]	1.24 (0.37–1.73)
PTM25	0.93	Gravel	1336 (691–1622) [1350]	6.92 (6.61–7.45) [6.61]	3.03 (0.45–10.6) [0.63]
PTM26B	0.83	Alluvium	1277 (447–1530)	6.91 (6.59–7.26)	0.64 (<0.1–1.24)
PTM26C	1.69	Gravel	1625 (1548–1846) [1566]	6.87 (6.64–7.16) [6.69]	0.43 (<0.1–0.95) [0.28]
PTM26D	2.90	Gravel	1812 (1619–19,850) [1705]	6.86 (6.64–7.17) [6.64]	0.56 (<0.1–1.33) [0.28]
PTM27B	0.76	Alluvium	1641 (1211–2060) [1821]	7.08 (6.75–7.44) [7.09]	0.62 (<0.1–1.14)
PTM27C	1.82	Gravel	1662 (1005–2090) [1878]	6.91 (6.57–7.48) [6.61]	0.52 (<0.1–0.85) [0.41]
PTM27D	2.94	Gravel	1963 (1854–2120) [1854]	6.87 (6.62–7.18) [6.65]	0.38 (<0.1–0.77) [0.25]
PTM28B	0.81	Alluvium	2263 (2010–2662)	6.91 (6.65–7.40)	1.28 (<0.1–3.6)
PTM28C	1.76	Gravel	1990 (1835–2160) [2010]	6.85 (6.64–7.12) [6.77]	0.89 (<0.1–1.71) [0.83]
PTM28D	3.64	Gravel	1895 (1755–2160) [1783]	6.83 (6.68–7.11) [6.68]	0.72 (<0.1–3.02) [0.39]
PTM29C	1.62	Gravel	1209 (1106–1303) [1190]	6.98 (6.73–7.25) [6.76]	0.38 (<0.1–0.63) [0.26]
PTM29D	3.74	Gravel	1733 (1075–1949) [1773]	6.90 (6.64–7.26) [6.64]	0.48 (<0.1–1.00) [0.21]
PTM30B	0.76	Alluvium	1168 (955–1420)	7.23 (7.04–7.36)	3.19 (0.81–7.92)
PTM30C	1.81	Gravel	1334 (688–1942) [1379]	7.00 (6.67–7.24) [6.67]	0.56 (<0.1–1.31) [0.47]
PTM30D	2.85	Gravel	1655 (911–1898) [1673]	6.93 (6.69–7.24) [6.69]	0.49 (<0.1–1.20) [0.46]
PTM31B	0.86	Alluvium	1969 (1577–2540)	7.19 (6.91–7.35)	1.36 (<0.1–2.93)
PTM31C	1.78	Gravel	1725 (1513–2050) [1590]	6.96 (6.72–7.25) [6.72]	0.56 (<0.1–1.35) [0.32]
PTM31D	3.87	Gravel	1745 (1140–1985) [1638]	6.99 (6.76–7.41) [6.76]	0.39 (<0.1–1.10) [0.39]
R. Thames	n/a	n/a	607 (450–714) [583]	8.19 (7.42–8.67) [7.42]	9.78 (7.80–12.7) [9.46]

Mean values that are presented with the range of concentrations found over the 12 sampling periods from May 2010 to August 2013 are shown in brackets. Measurements made in August 2013 are shown in square brackets.
[a] All screen sections are over the bottom 0.5 m of the borehole, apart from the GBH boreholes, drilled in the 1980s, for which no screen information is available.
[b] GBH boreholes are drilled into the gravel; it is expected that they are screened in the gravel and also the overlying alluvium and landfill material.

and to the far north (PTM11) of the landfill. Highest pH values are found in the River Thames. Dissolved oxygen mean values are typically <1 mg/L and often <0.1 mg/L in the deeper piezometers, in the shallower piezometers DO is up to 10 mg/L. The River Thames typically has DO concentrations at saturation for the atmospheric temperature in a given season.

Nitrate concentrations range from a high of ~30 mg/L in the river through below detection for the majority of the piezometers (Table 3).

Table 3
Nitrogen species and isotopic data for selected boreholes in the Port Meadow network.

Site	NO_3-N mg/L	$\delta^{15}N$-NO_3 ‰	$\delta^{18}O$-NO_3 ‰	N_2O µg/L	NO_2 µg/L	NH_4-N mg/L	$\delta^{15}N$-NH_4 ‰
GBH3	<0.01 (<0.01–5.6)			0.3	<10	22 (9.6–36) [14]	+12.5
GBH5	7.06 (2.55–16.2) [13.3]	+13.3	+6.2	27.9	<10	69 (9.6–78) [53]	+8.7
GBH9	16.5 (0.82–48) [0.82]	+8.7		19.9	<10 (<10–170)	42 (9.6–53) [39]	+7.0
OX14	0.12 (<0.01–0.48)			11.3 (<0.1–13.5)	<10 (<10–90)	3.3 (2.8–4.1) [3.0]	+8.9
PTM11	0.03 (<0.01–0.21)			<0.1 (<0.1–0.3)	<10	<0.2 (<0.2–2.2) [0.2]	+2.5
PTM21	0.03 (<0.0.1–0.35)			NA	<10	2.7 (1.58–6) [2.2]	+10.7
PTM23	0.12 (<0.01–1.63)			NA	<10 (<10–30)	11.5 (0.45–35.2) [25]	+8.7
PTM24	0.02 (<0.01–0.06)			NA	<10	3.3 (1.9–7.9) [5.0]	+10.5
PTM25	0.1 (<0.01–0.53)			0.1	<10 (<10–90)	0.43 (0.15–0.58) [0.4]	+19.2
PTM26B	0.09 (<0.01–0.38)			NA	<10	4.5 (1.2–24.2)	NA
PTM26C	0.06 (<0.01–0.32)			27 (<0.1–54)	<10	19 (13–22) [14]	+8.8
PTM26D	0.05 (<0.01–0.3)			1.75 (0.88–2.6)	<10 (<10–50)	27 (22–29) [18]	+8.3
PTM27B	0.05 (<0.01–2.8)			10.7 (0.5–17.3)	<10	7.7 (2.2–31)	NA
PTM27C	0.04 (<0.01–0.36)			1.8 (0.9–2.8)	<10 (<10–50)	33 (12–47) [28]	+8.3
PTM27D	0.03 (<0.01–0.09)			0.8 (<0.1–1.5)	<10	48 (2.5–50) [31]	+8.4
PTM28B	0.08 (<0.01–1.14)			NA	<10	17 (2.9–27)	NA
PTM28C	0.09 (<0.01–0.49) [0.49]	+3.7		1.1	<10	49 (43–99) [27]	+9.2
PTM28D	0.06 (<0.01–0.2)			3.5 (<0.1–39)	<10	56 (49–56) [25]	+6.9
PTM29C	0.04 (<0.01–0.17)			9.2 (2.5–15.9)	<10	21.3 (5–28) [28]	+8.0
PTM29D	0.03 (<0.01–0.14)			0.3 (<0.1–0.3)	<10	32 (5.8–37) [22]	+9.1
PTM30B	0.02 (<0.01–25.5)			1.12	<10 (<10–2800)	1.8 (0.45–15.9)	NA
PTM30C	0.05 (<0.01–0.15)			0.4	<10	14 (4–52) [41]	+7.8
PTM30D	0.05 (<0.01–0.51)			0.2 (0.1–0.3)	<10	58 (36–59) [36]	+9.1
PTM31B	0.07 (<0.01–0.69)			NA	<10	54 (1.2–60)	NA
PTM31C	0.08 (<0.01–0.27)			102 (13.7–191)	<10	63 (44–70) [44]	+8.7
PTM31D	0.06 (<0.01–0.14)			0.6 (0.2–1)	<10	48 (11–52) [39]	+9.1
R. Thames	21 (16.6–30) [18.4]	+9.92	+4.35	NA	70 (<10–140)	0.06 (<0.2–0.14)	NA

Mean values that are presented with the range of concentrations found over the 12 sampling periods from May 2010 to August 2013 are shown in brackets. Measurements made in August 2013 are shown in square brackets.

The main exceptions are three monitoring boreholes (GBH3, 5 and 9) completed within the Burgess Field waste site where NO_3-N concentrations range from <0.01 to 48 mg/L. Within the nested transect boreholes (PTM26–28 and PTM29–31), the shallow sites (B) have low (<2 mg/L) but uniformly higher mean NO_3-N concentrations compared to deeper sites. Reflected by these generally low nitrate concentrations is the relative paucity of data for nitrate isotopes, with just 2 samples yielding enough nitrate to enable measurement and one of these being the river. δ^{15}N-NO_3 values ranged from +10 to +13‰ and δ^{18}O-NO_3 ranged between +4 and +6‰. This was compounded by the high organic carbon concentrations within the plume. These high concentrations coincided with a low redox status and negligible dissolved oxygen which was observed year round for most sites.

Nitrite concentrations generally fall below the detection limit of 10 μg/L although there are exceptions from the river Thames, the site at OX14 and the highest concentration occurring from one of the landfill monitoring boreholes. Highest concentration of N_2O, at nearly 200 μg/L, also occurs in one of the boreholes adjacent to landfill. Other sites range from a near 'background' concentration of 0.2 μg/L to ~54 μg/L; the shallower transect boreholes are generally higher in concentration than the deeper transect boreholes (Table 3).

Ammonium concentrations show considerable variation from 0.05 to 99 mg/L NH_4-N with the highest concentration occurring in the landfill boreholes and the lowest in the River Thames (Fig. 5). Concentrations are generally higher the closer the borehole is to the waste site and the deeper the borehole, with concentrations decreasing closer to the river. Concentrations from the waste site boreholes show considerable variation, with the highest concentration detected in the same site that also had the greatest amounts of NO_3-N and N_2O. Similar to the NH_4-N data, the δ^{15}N-NH_4 values also show considerable variation from a minimum of 2.5‰ to a maximum of 19.2 of ‰ (mean of 9.1‰). Values in the main transect vary much less, with a mean of 8.5‰ and minima and maxima of 6.9‰ and 9.2‰. The highest value occurs to the south of the main plume, whereas the lowest value occurs to the north. Values in the waste dump boreholes vary from ~7‰ to 12.5‰.

Fig. 6 shows filled contour plots of mean N-NH_4 concentrations for the two transects within the floodplain along the flow gradient from the landfill site to the River Thames. There is a consistent trend of higher N-NH_4 concentrations with depth in the gravels, and the lowest concentrations are found in the two nests that are closest to the Thames. The highest concentrations are found in transect B, between 100 and 200 m from the edge of the landfill rather than in the landfill and at a depth greater than 55 mAOD.

The steps in estimating the percentage of the N and NH_4-N flux in the River Thames, due to that entering the river laterally from the floodplain sediments to the east, are set out in Table 3. The total N and NH_4-N groundwater influx is compared to the concentration within the River Thames by using a mean concentration in the river measured during the period of the study, and the long-term river flow upstream. Samples from the two borehole nests at the western end of the two transects (PTM26 and PTM29) provided an average deep gravel groundwater NH_4-N concentration (3.75 m borehole; 29.9 g/m^3) and total NH_4-N concentration (average of 1.75 and 3.75 m boreholes; 21.6 g/m^3). Estimated fluxes of NH_4-N in groundwater into the river along the Burgess Field reach are compared with the flux within the river; groundwater NH_4-N fluxes account for up to 15% (depending on the

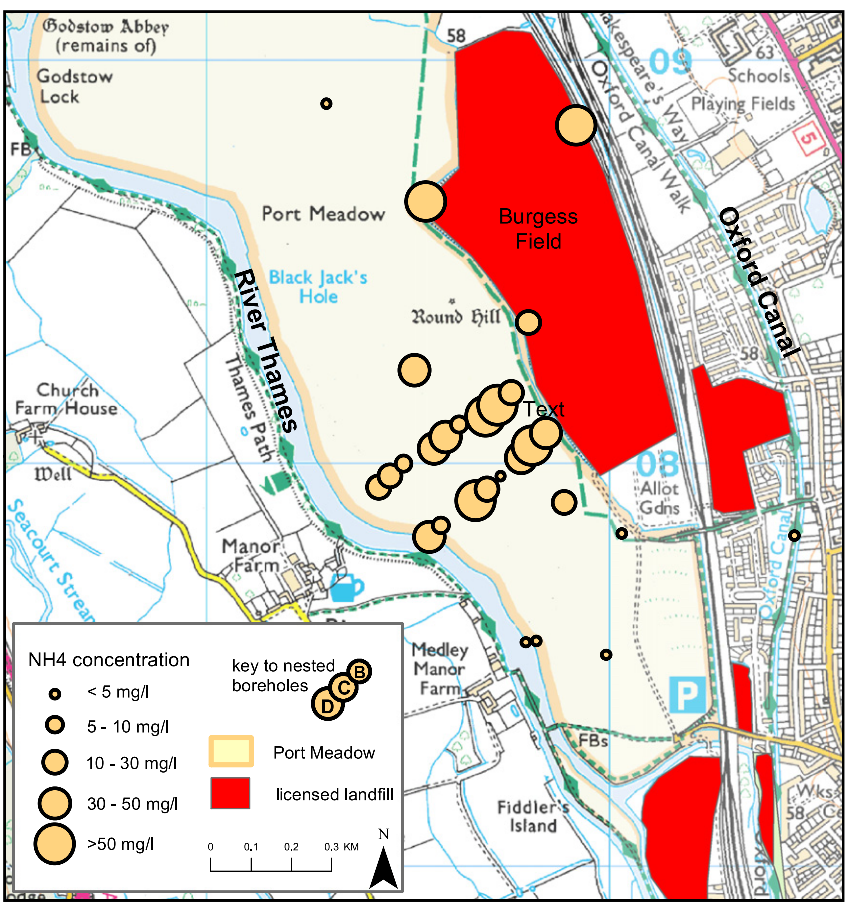

Fig. 5. Study area with average NH_4-N concentrations in groundwater from the piezometers. Graduated symbols indicate spatial variability in NH_4-N concentrations. Note, symbols for piezometers within the nests are offset to enable the concentrations within each to be visible; piezometer C is positioned in the centre of the nest, D to the south-west and B, where measurements are available, to the north-east.

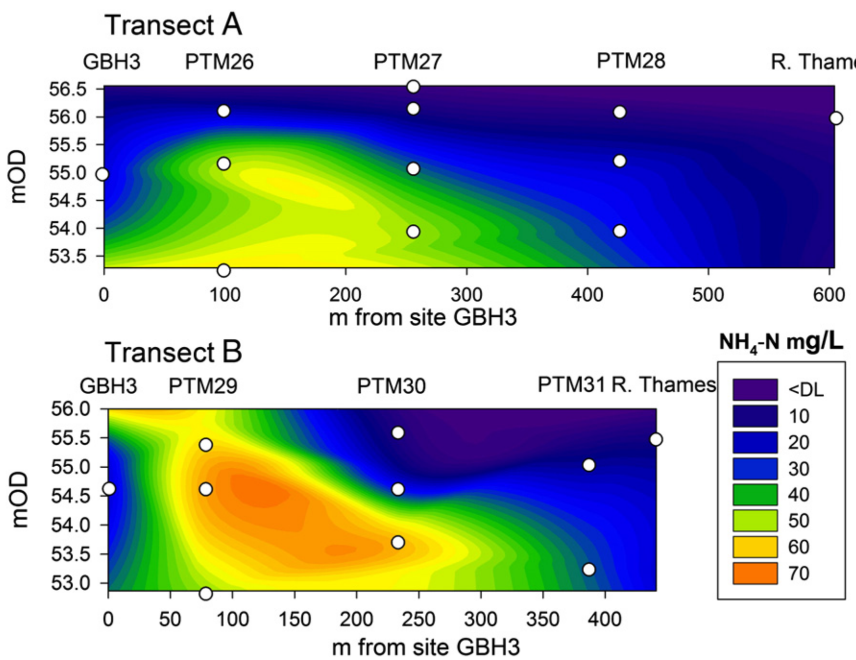

Fig. 6. Filled contour plot showing changes in average NH_4-N concentrations in groundwater across both transects in Port Meadow.

input concentration) of the NH_4-N in the river. Nitrate in the groundwater is negligible; total N in the river water is the sum of NO_3-N and NH_4-N. Table 3 also compares the estimated total N flux of groundwater into the river with the flux within the river which is approximately two orders of magnitude smaller than that for NH_4-N.

4. Discussion

The spatial variability in groundwater NH_4-N raises questions of source (i.e. does all the ammonium come from the landfill?) and attenuation mechanism within the floodplain sediments. A cross plot of NH_4-N and Cl for all available data during the study, with sites grouped by different hydrological zones within the peri-urban floodplain, is shown in Fig. 7. This shows the effect of dilution of the landfill contamination within the floodplain and the variability of NH_4-N and Cl within each zone. The spread of data in Fig. 7 can be understood in terms of ternary mixing between, i) groundwater leaching from the waste site with high NH_4-N concentrations, ii) local shallow recharge to the floodplain gravels and iii) the River Thames. Ammonium contamination in the shallow gravels is diluted relative to the deeper gravels, increasingly so down the flow gradient towards the River Thames. The samples with lower NH_4-N concentrations within the landfill sites are associated with low water level conditions. The shallow floodplain groundwaters have elevated Cl concentrations compared to other end members; this is perhaps evidence of evaporative enrichment of ponded surface water on the floodplain in the summer prior to recharge.

A cross plot of δ^{15}N-NH_4 against NH_4^+-N (Fig. 8) clearly shows how the majority of samples have δ^{15}N-NH_4 values that fall between ~7

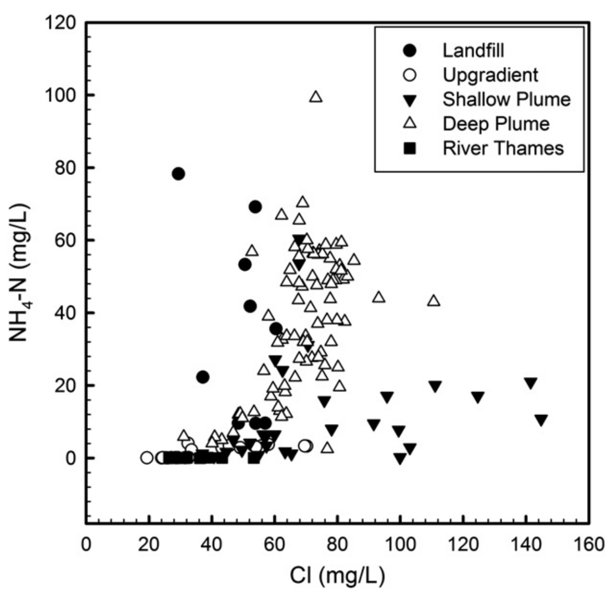

Fig. 7. Cross-plot of NH_4-N vs Cl in the Port Meadow area combined over the study period. Sites are grouped as: GBH3, 5 and 9 — Landfill; OX14 and PTM11 — Up gradient; PTM26–PTM31 A&B — Shallow Plume; PTM26–PTM31 C&D — Deep Plume.

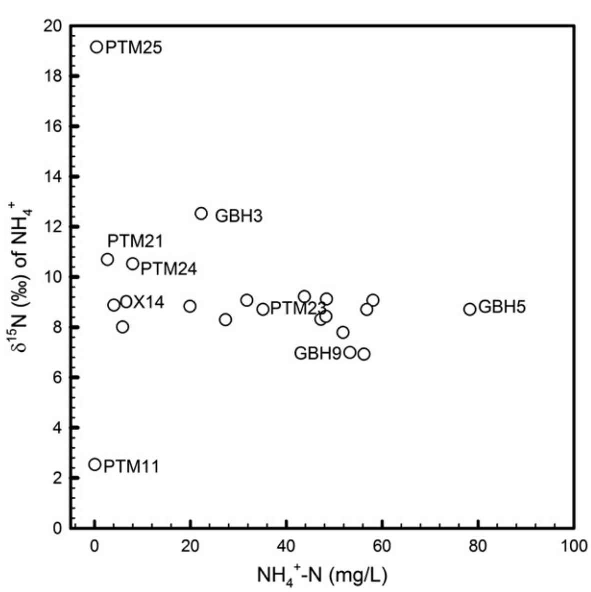

Fig. 8. δ^{15}N values and concentrations of NH_4^+-N in Port Meadow groundwaters. All unlabelled data points are from two main transects.

and 10‰ and are fairly independent of concentration. The two major outliers from this trend are PTM11 and PTM25. PTM11 is thought to be largely up-gradient of the main contaminant plume and it would appear this is borne out from the isotopic data. The isotopic ammonium delta value coupled with the much lower concentration of ammonium here is more similar to a soil delta value rather than any influence of the landfill (Heaton, 1986; Kendall, 1998). Similarly, PTM25 is in close proximity to the River Thames, and this very high value is probably more indicative of faecal waste (Heaton, 1986; Kendall, 1998) from two possible sources: treated waste water input to the River Thames upstream of Port Meadow which can move into the floodplain sediments due to the hydraulic gradient reversal that can occur in the area of PTM25 during summer months; and waterfowl that congregate close to this area where groundwater flooding within the floodplain persists longest.

The landfill boreholes show some of the most interesting N transformation chemistry. GBH5 contains high concentrations of both nitrate and ammonia as well as significant concentrations of N_2O. The presence of some measurable dissolved oxygen at times along with nitrate isotope values (δ^{15}N-NO_3 = 13.3‰ and δ^{18}O-NO_3 = 6.2‰) is highly indicative that some nitrification of ammonia has occurred followed by denitrification of the resultant nitrate, although the process has not moved to complete removal of nitrate. Similarly, lower concentrations of ammonium and some nitrate are also found at GBH9 and along with NO_2, albeit at low concentrations. This possibly suggests a lower rate of nitrification and no denitrification, since although NO_2 is an intermediate product in both process, the nitrate has to be formed before it can be removed. GBH3 has a much higher δ^{15}N-NH_4 value than the other two landfill monitoring boreholes, but it also has much lower NH_4^+-N concentrations. This might indicate some nitrification followed by complete conversion to N_2 via denitrification. The absence of any intermediate products might suggest that the reaction is complete so removing nitrogen from the system in a region where the water table is known to fluctuate.

OX14 and PTM23 have very similar δ^{15}N-NH_4 values to the transect boreholes. Site OX14 has nitrification/denitrification intermediates which could explain a much lower ammonium concentration. PTM23 however also has a high ammonium concentration and it would appear that this site is impacted by the plume. The shallow boreholes at PTM21 and PTM24 have some of the lowest ammonium concentrations and relative to the other transect boreholes have slightly elevated δ^{15}N-NH_4 values which may be indicative of some nitrification/denitrification reactions. All of these sites have relatively high DO concentrations at some times of the year as well as higher pH values (>7) which are more conducive to denitrifying bacteria (Thomas et al., 1994). Again, the absence of any intermediate products suggests that the reaction is complete so removing nitrogen from the system in a region where the water table is known to fluctuate.

One might speculate that the relatively consistent δ^{15}N-NH_4 values observed could be a result of nitrification, which would increase the delta value, and sorption which would decrease the delta value. However, given that the oxygen availability is generally low and that the sorption and nitrification processes would have to occur in fairly equal amounts (independent of concentration — see Fig. 8) this does seem a somewhat unlikely scenario. Therefore, since there is no relationship between ammonium concentration and δ^{15}N-NH_4, sorption on to clays can most probably be ruled out as an attenuation mechanism. Away from the main plume, and on the direct edge of the landfill, there is strong evidence for both nitrification and denitrification. However, this appears limited and restricted to pockets of groundwater table fluctuation, and the majority of ammonium in the plume is not effectively retarded by sorption, with concentrations only tempered by dilution. Where the aquifer remains saturated all year round (particularly at piezometer depths C and D) the ammonium can be considered as conservative. For the floodplain to be a buffer zone for nitrogen, as would generally be the case in more rural environments (Burt et al., 1999), then it is clear that the site conditions prevent this from occurring since in the landfill plume, there is insufficient dissolved oxygen to facilitate nitrification at depths greater than 1 m.

The area of landfills/waste dumps on the River Thames floodplain in the Oxford area is significant (Fig. 1); 1.05 km^2 of the 15.87 km^2 of floodplain. Making the assumption that the influx of NH_4-N to the river via the gravel aquifer from the Burgess Field waste dump is representative of the remainder on the floodplain, behaving in a similarly conservative manner, it is possible to estimate the total influx due to all the landfills/waste dumps. This takes into account the volume of the landfill and the dilution where the landfill is not located by the river bank. The calculated flux along an approximately 8 km reach of the River Thames is estimated as 75 kg ammonium (NH_4) per day or over 27.5 tonnes per year. Assuming no additional in-stream processing, this influx would lead to a river concentration of close to 0.1 mg/L at the end of the reach compared with 0.06 mg/L at Port Meadow — or roughly 40% of total ammonia in the river coming from legacy waste dumps.

5. Conclusions

A combination of nitrogen isotopes, nitrogen speciation and dissolved nitrous oxide has been used to understand the sources of nitrogen in a complex and heterogeneous peri-urban floodplain. Based on other nitrogen isotope studies, the dominant source of nitrogen is in the form of ammonium and this has been shown to have originated from a former domestic landfill which continues to act as a source of elevated nitrogen into the environment. Despite some evidence for enitrification in areas with water table fluctuations near to the landfill or on the fringes of the landfill plume, the prevalence of year round reducing conditions in the deeper (>1 m) floodplain has resulted in the transport of ammonium, with minimal biogeochemical processing or sorption to sediments, directly into a major river. Although the calculated flux is not a large contribution to the overall nitrogen in the river, it does represent the addition of a significant proportion (~10% but up to 15%, Table 4) of the total ammonium as the river goes past each landfill. Collectively this contribution to overall ammonium concentrations could be very high. This is a hitherto unconsidered source of river pollution from the peri-urban fringe. Given the large number of urban developments along the edge of floodplains and the associated historical and at times ad hoc waste dumps, this is likely to be a scenario reflected in many other parts of the developed world. Catchment management

Table 4
Determination of ammonium and total nitrogen flux from landfill site to the river at Port Meadow.

	Deep sample	Average of shallow and deep
NH_4-N groundwater concentration (g/m^3)	29.9	21.6
NH_4-N influx to river (g/d)	10000 (±5000)	8600 (±4300)
NH_4-N river concentration (g/m^3)	0.064	
NH_4-N flux within river (g/d)	1.02×10^5	
N river concentration — NH_4-N + NO_3-N (g/m^3)	5.45	
N flux within river (g/d)	8.7×10^6	
NH_4-N groundwater influx as percentage of river flux (—)	11.7 (±4.9)	8.5 (±4.2)
N groundwater influx as percentage of river flux (—)	0.14 (±0.07)	0.10 (±0.05)

plans that encompass floodplains in the peri-urban environment need to take into account the likely risk to groundwater and surface water quality from these legacy landfills.

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Chapter 4: Groundwater flooding within an urbanised flood plain

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Journal of
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Groundwater flooding within an urbanised flood plain

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Key words
Flood mitigation; flood plain; groundwater; groundwater flooding; Oxford; River Thames; urban flooding.

Abstract

In Europe in recent years, there has been recognition of the need to better understand the risk from groundwater flooding. This recognition has been due both to the occurrence of major flooding events clearly attributable to groundwater and the inclusion of groundwater flooding in European and national legislation. The case study of the city of Oxford on the River Thames flood plain in UK is used to examine the mechanisms for groundwater flooding in urbanised flood plain settings. Reference is made to an extensive data set gathered during a major flood event in 2007. Groundwater flooding of a significant number of properties is shown to occur in areas isolated from fluvial flooding because of high ground created historically to protect property and the transport network from flood inundation. The options for mitigating this form of flooding are discussed; measures to increase the rate of conveyance of flood waters through Oxford, designed to reduce fluvial flood risk, have also been recognised as a means for reducing groundwater flood risk within the city.

Introduction

Groundwater flooding is the emergence of groundwater at the ground surface away from perennial river channels and can also include the rising of groundwater into man-made ground, including basements and other subsurface infrastructure (Macdonald *et al*., 2008). The impact of groundwater flooding can be severe under conditions where the 'normal' ranges of groundwater level and groundwater flow are exceeded. In Europe, the risk from groundwater flooding has received more attention since its inclusion in the EU Floods Directive (2007/60/EC). The Directive, which came into force in November 2007, includes provisions for assessing the risk from groundwater flooding (Cobby *et al*., 2009), producing groundwater 'flood hazard maps' and introducing measures to address any significant risk. The inclusion of groundwater within the Directive follows serious groundwater flooding events in the past decade. The impact of groundwater flooding has been most severe in areas of Chalk outcrop and in the flood plains of major rivers. Groundwater flooding in Chalk catchments occurs where antecedent conditions of high groundwater levels and high unsaturated zone moisture content combine with intense rainfall. Resulting groundwater level rises of up to tens of metres can cause significantly increased stream base flow and spring flow, and the reactivation of dormant springs in dry valleys remote from perennial stream channels. Flooding is often prolonged because of high groundwater levels maintained by extended periods of drainage from the unsaturated zone (Pinault *et al*., 2005). Examples include the flooding in 2000 and 2003 in southern England (Finch *et al*., 2004) and the Somme Valley (Negrel and Petelet-Giraud, 2005).

The focus of this paper is groundwater flooding in the flood plains of major rivers. Where the deposits associated with flood plains are permeable, these are generally saturated to levels close to the ground surface and hydraulically well connected to the associated rivers. Here, groundwater can contribute significantly to river flow during summer months and extended dry periods; in wetter periods, rivers can be effluent, recharging the flood plain sediments. During periods of increased flow, and before the banks are overtopped, the naturally high ground of river levees can contain river water while low-lying ground beyond can be flooded because of rising groundwater. In the case of the River Danube flood plain in 2008, damage occurred to buildings because of water pressure on basements, resulting from abnormally high groundwater levels associated with high river levels (Kreibich and Thieken, 2008). The onset and recession of groundwater flood events in this type of setting is typically much shorter than that in Chalk catchments.

Flood plains in the past have been attractive locations for urban development for reasons such as their coincidence

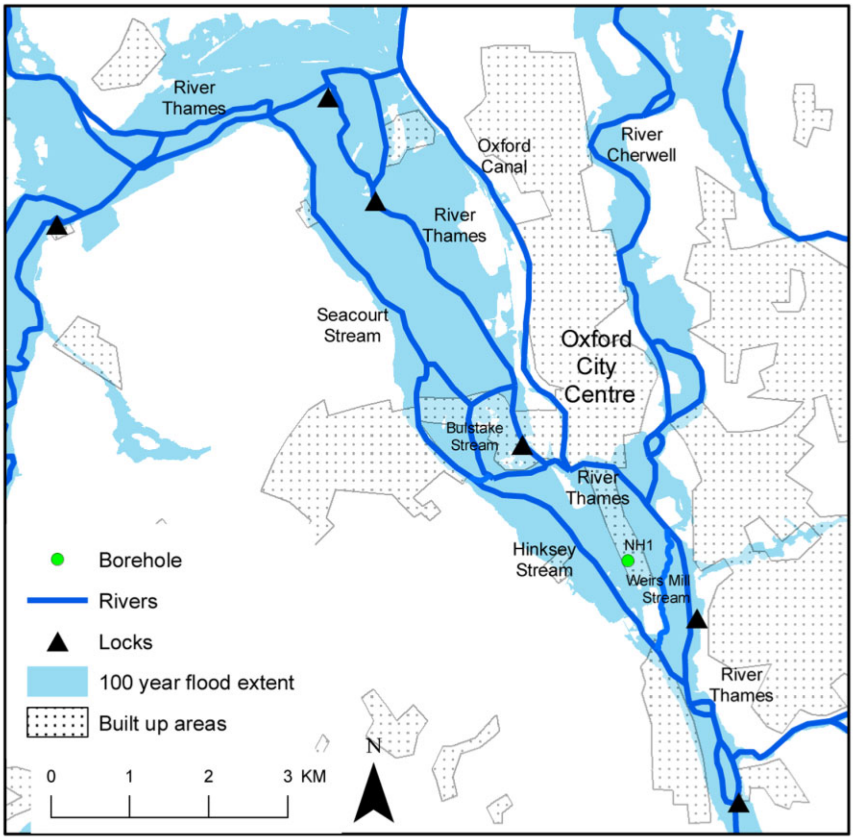

Figure 1 River Thames and its tributaries in the Oxford area. BGS © NERC 2011. Ordnance Survey topographic material reproduced with the permission of Ordnance Survey on behalf of The Controller of Her Majesty's Stationery Office, © Crown Copyright. Licence number 100017897/2011.

with sources of water and food, their potential for cultivation, and their proximity to transportation routes (Montz, 2000). Where settlements were originally located on high ground within or close to river valleys, these have inevitably spread onto the flood plain. Historically, this development has often continued even with the threat of frequent flooding. In urban areas on the flood plain, a cycle can develop in which defences to protect property and infrastructure from flooding increases the general risk from flooding, for example by removing flood storage, and hence raising flood levels, which in turn requires more substantial defences. In recent times, the folly of continuing this cycle has been recognised and a move has been made towards working with natural processes to manage the risk of flooding rather than trying to remove it (Fleming, 2002).

The topographical changes to flood plains resulting from urbanisation can have a significant influence on the location, timing and extent of groundwater flooding. This can make the assessment of risk highly complex. Identifying measures to reduce the risk of groundwater flooding as part of overall flood risk management schemes is challenging. A case study is reported here of the city of Oxford in the UK, which is located on the flood plain of the upper reaches of the River Thames. Groundwater flooding has been recognised in the city as a component of the overall flooding story (Macdonald *et al.*, 2007). The Oxford case study is used as a means to examine the risks from groundwater flooding and the options for mitigating these risks in an urbanised flood plain.

Background

The city of Oxford is situated within a relatively narrow valley of the upper reaches of the River Thames (Figure 1). The flood plain is on average 2 km wide; however, it narrows downstream to only 0.5 km. Although most of the city is located on older river terraces and bedrock above the current flood plain, from the late 19th century, pressure for housing near the city centre resulted in significant urban development on the flood plain; it is estimated that, currently, approximately 3400 primarily residential properties and a large number of commercial properties are located within the 100-year return period flood event envelope as defined by the Environment Agency. The urban areas of Oxford have

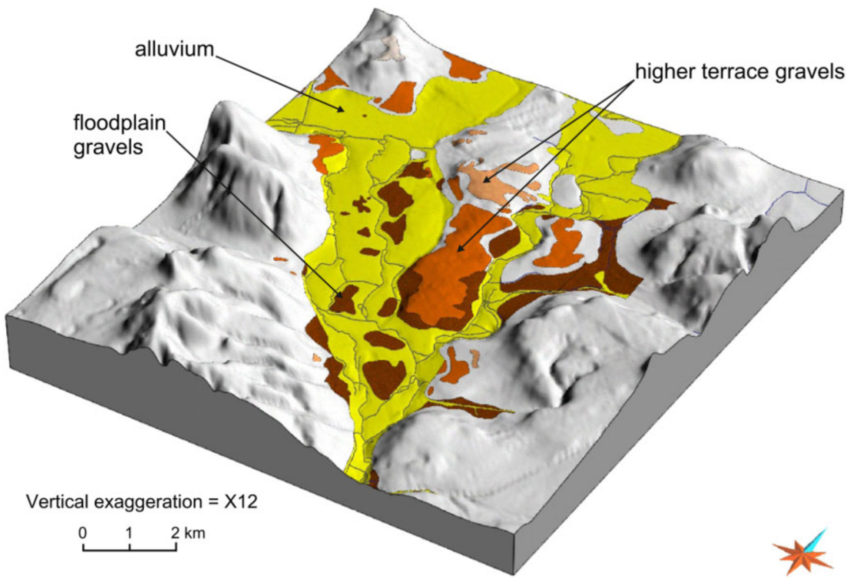

Figure 2 Superficial geology in the Oxford valley. BGS © NERC 2011. Some features of this map are based on digital spatial data licensed from the Centre for Ecology and Hydrology (Moore *et al*., 1994). This map includes NEXTMap Britain elevation data from Intermap Technologies.

historically suffered from serious flooding. There were relatively few major flood events in the second half of the 20th century; however, in recent years, there have been four notable floods over the period of a decade, in April 1998, December 2000, January 2003 and most recently in July 2007. The Oxford flood plain is underlain by permeable shallow sands and gravels (Figure 2), and a significant number of properties were affected by flooding from rising groundwater, which was either the sole cause or the initial cause prior to inundation from fluvial waters.

In February 2002, the Environment Agency of England and Wales commenced the Oxford Flood Risk Management Study (OFRMS) to identify options to reduce the flood risk in Oxford within the 100-year return period flood plain of the River Thames and its tributary, the River Cherwell (Ball *et al*., 2008). During the early stages of the OFRMS, it was recognised that groundwater flooding and the links to fluvial flooding were important considerations. Together with the British Geological Survey, a jointly funded research project was initiated in 2005 on the controls, location and timing of groundwater flooding in the city. The insights from this project have informed the choice of proposed flood mitigation measures.

In July 2007, serious flooding took place within the flood plain of the River Thames in Oxford. This was the result of a moist, subtropical air mass moving slowly north over central England, resulting in extreme rainfall on July 19 and 20. Some of the highest total rainfalls occurred in the headwaters of the River Thames, for example 140.1 mm in 24 h in Chastleton, Oxfordshire (Marsh and Hannaford, 2008). This rainfall event followed what was the wettest summer in England and Wales since 1912. Resulting flow in the River Thames to the south of Oxford at Sandford peaked at 224.8 m^3/s; this was the maximum flow at this location in 2007 and the fifth highest annual maximum flow since 1894. The peak river flow in Oxford occurred approximately 5 days after the rainfall event. The total rainfall in Oxford itself for July 19 and 20 was measured at 60.6 mm. The monitoring in place for the groundwater flooding element of the OFRMS was able to capture the impacts on water levels of the July 2007 event and provides the primary reference for this paper.

Methodology

The approach taken within the Oxford study was to develop a baseline understanding of the river and groundwater system, and use any flood events that occurred during the study period to explore the potential role of groundwater in flood events covering a range of return periods. The components of the study are described here.

Topographic data

In 2005, the Environment Agency undertook a topographic survey of the flood plain using LIDAR (Light Detection and Ranging), an optical remote sensing technology that measures ground elevation to centimetric accuracy (Cracknell and Hayes, 2007). Among other applications within this study, LIDAR data provided a means to assess the urban build-up on the flood plain and its impact on flood pathways, and the potential for groundwater flooding.

Three-dimensional (3D) modelling

A 3D geological model of the flood plain superficial deposits in the study area was built to aid the conceptualisation of the shallow groundwater system and to enable the potential storage capacity of the aquifer to be assessed. The model comprises three layers, including made ground, alluvium, and the underlying sands and gravels. The base of the model over the majority of the study area is the Oxford Clay, a Jurassic argillaceous sedimentary rock. ArcGIS, in conjunction with the 3D visualisation packages, geological surveying and investigation in 3 dimensions (Kessler *et al.*, 2009) and geological object computer aided design, were used to construct the model. Borehole logs were used to create a series of cross-sections with surfaces produced by interpolation between the sections. The layers within the geological model were attributed with estimates of specific yield made during previous studies within the Oxford flood plain (Institute of Hydrology, 1987; Dixon *et al.*, 1990; Gardner, 1991). The geological model was then combined with groundwater level surfaces contoured using measured groundwater and surface water levels.

Monitoring network

An extensive network of groundwater and surface water level monitoring points exists within the Oxford flood plain resulting from this and previous studies. At some sites, monitoring has been undertaken since the early 1980s (Institute of Hydrology, 1987). Manual monitoring of water levels was undertaken on a monthly basis between 2005 and 2009; the total number of monitoring points at the end of 2009 was 235. Digital water level recorders were installed in 51 of these monitoring points. At a number of sites, there are combinations of two or more of the following: piezometers completed in the gravel aquifer, piezometers completed in the overlying alluvium, surface water stilling wells and flood water recorders. Data were made available from recorders installed in the upstream and downstream waters at four of the Environment Agency locks in the study area. During the July 2007 flood, a further six temporary manual flood water monitoring points were established, which provided additional data in urban areas for the rise and recession of flood waters during the event. As well as water levels, river flows are routinely measured within the study area by the Environment Agency on the River Thames and its tributaries, the Rivers Evenlode and Cherwell. Rainfall is measured on a 15-min interval at two of the locks. There is also a long rainfall record from the Radcliffe Meteorological Station at Oxford.

Flood observation and impacts

During the period of the study, one major flood event (July 2007) and a number of minor floods occurred. Observation during floods is crucial in understanding flood mechanisms. Flood sources and pathways can vary over the period of a flood, and a monitoring network cannot feasibly capture all of this for a study area as large as the Oxford flood plain. A questionnaire sent out by the Environment Agency following the December 2000 flood event captured some information on the occurrence of groundwater flooding, but generally it is hard to obtain such information; in addition to the reluctance of owners to provide information on the impact of flooding on their property, it is often difficult to identify with confidence that the source of flooding is groundwater. Information was obtained on possible locations of groundwater flooding in 2003 and 2007 based on residents' observations.

Flood mapping

A national assessment of groundwater flood susceptibility undertaken by the British Geological Survey (McKenzie *et al.*, 2010) includes the River Thames flood plain in the Oxford area within its 'very high' category, that is, it has mapped the underlying superficial geology as permeable and the groundwater levels within 2 m of ground level. An attempt was made to improve on this broad-scale assessment in Oxford and to quantify risk rather than just susceptibility. Although separate river (1D and 2D) and groundwater models have been developed for Oxford, attempts to link these to enable simulation of the river–aquifer response during flood events is still in the development stage. Preliminary mapping of groundwater flood-prone urban areas was attempted based on the understanding of flood mechanisms gained during the study. LIDAR data were used to identify those low-lying urban areas that are isolated from fluvial flooding, at least for low return period events or in the early stages of higher return period events. For these areas, the minimum level at which groundwater flooding could potentially occur and the level at which fluvial waters would inundate a location were assessed. Based on the understanding of flood water pathways and levels during an event and the likely response of groundwater levels, the potential for groundwater flooding at the locations identified earlier was assessed. Where Environment Agency property threshold data were available, the groundwater flood levels identified were used to assess the likelihood of groundwater flooding affecting individual properties. Anecdotal information on the location of groundwater flooding in the three recent major floods was used to validate the outcome from this step. Where property threshold data were not available, the reported incidents of groundwater flooding were used to improve confidence. The output from an Environment

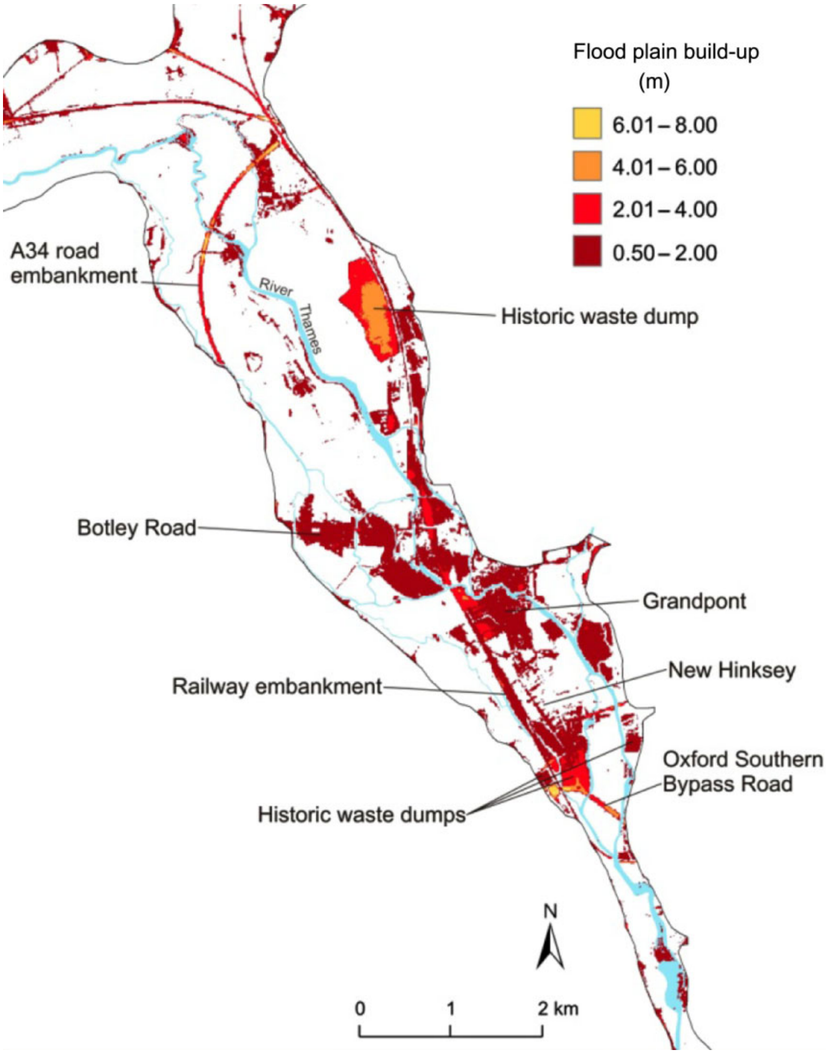

Figure 3 Thickness of made ground above the natural flood plain. BGS © NERC 2011.

Agency river flood model (Environment Agency, 2009), which gives the flood elevation for flood return periods of 2, 5, 10, 20, 50, 100, 200 and 500 years, was used to give a preliminary indication of the range of fluvial flood return periods for which groundwater flooding on its own might occur.

Results

Baseline conditions

Analysis of the topographic data for the flood plain, involving the picking of locations that are at the natural flood plain level and interpolating, allowed the thickness of the made ground above the natural flood plain to be mapped (Figure 3). This highlights the built-up residential and commercial areas within the city, the road and rail networks, and those sites previously used for dumping waste. The role of this high ground in constraining the movement of flood waters down the flood plain is important.

The geological logs used to create the 3D geological modelling of the superficial deposits underlying the Oxford flood plain all include some alluvium below the ground surface. The model produced has a continuous layer of alluvium of up to 4 m in thickness, but more typically 1.5 m, which covers a layer of sands and gravels beneath the flood plain, typically within a range of 2–6 m in thickness.

Water level data show that the flow of groundwater within the flood plain sediments is complex, due largely to the influence of locks and weirs on the River Thames, and associated bypass streams. These have created numerous zones of recharge from, and discharge to, the river network. Groundwater levels generally fluctuate within the upper few metres of the flood plain sediments with a greater range of fluctua-

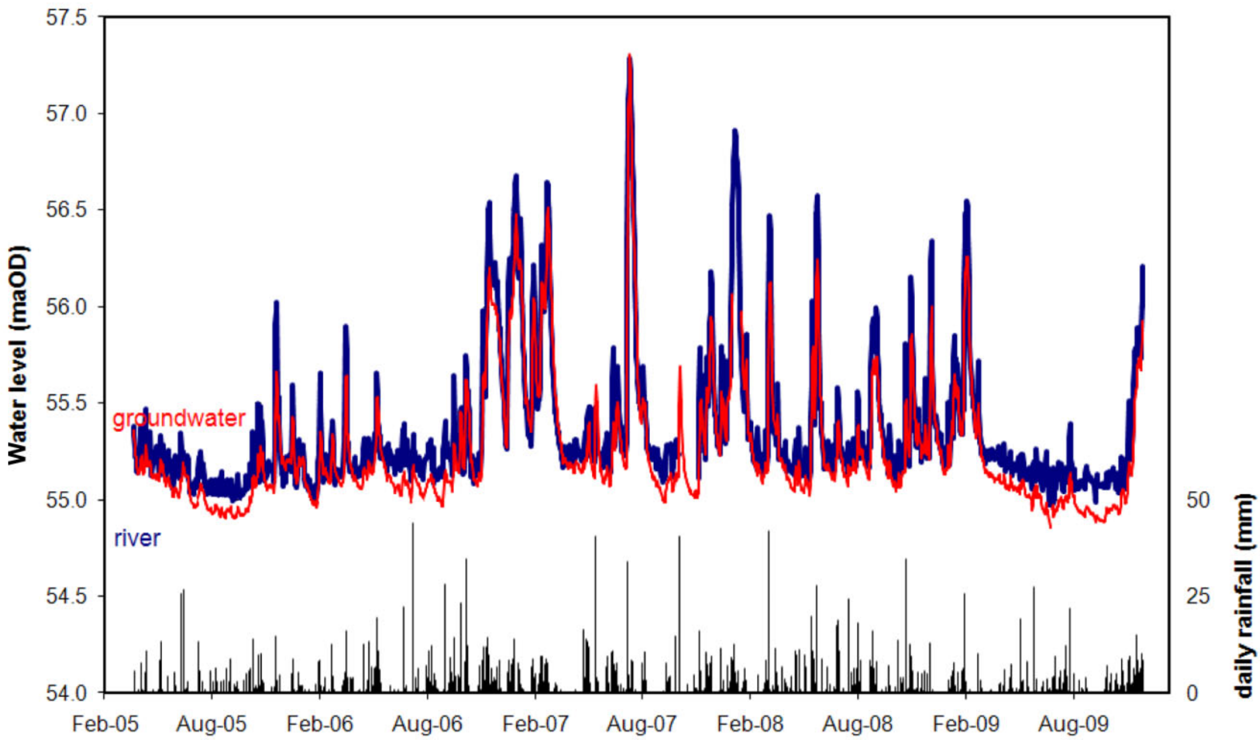

Figure 4 Typical river and groundwater hydrographs from the Oxford flood plain monitoring network. BGS © NERC 2011.

tion occurring at the flood plain margins. In the vicinity of rivers and streams, groundwater levels generally correlate well with surface water levels, indicating good hydraulic connection (Figure 4). Dixon (2004) reports hydraulic conductivity of the sands and gravels in the River Thames flood plain in Oxford as very high, ranging between 100 and 1000 m/d. This compares with an estimate of hydraulic conductivity of the alluvium of 0.3 m/d made by both Gardner (1991) and Hodgson (2008).

There is no evidence of buildings on the flood plain with basements constructed below the base of the alluvium, and building foundations are not thought to have a major influence on groundwater flow within the flood plain as these are generally less than a metre in depth for domestic properties, and therefore unlikely to affect lateral groundwater movement within the sands and gravels aquifer. The few large buildings on the flood plain were constructed on raised ground, and the foundations will not have penetrated the gravel aquifer sufficiently to significantly change groundwater flow patterns. There is only one location within the urbanised flood plain where a relatively small volume of gravel has been extracted and the void created has not been infilled.

Combining groundwater levels with the LIDAR data allows the depth to groundwater to be contoured; this is shown in Figure 5 for May 2007, a period when the groundwater levels were relatively low. This figure (in combination with Figure 3) highlights that the depth to groundwater is generally greater under the man-made ground and at the valley edges; groundwater is shallower in areas close to the River Thames upstream of locks, where raised heads cause enhanced aquifer recharge from the river, and also at the southern end of the valley where the narrowing of the flood plain causes restricted flow of groundwater down-valley.

The available storage within the unsaturated zone of the flood plain is very small compared with flood water volumes. Estimating this storage is highly problematic because of the variability of the lithology of the flood plain deposits, the challenge in estimating storage parameters and the influence of the capillary zone. However, using an approximation of less than 10% for specific yield of the flood plain deposits (Institute of Hydrology, 1987; Dixon *et al.*, 1990; Gardner, 1991), the volume of available subsurface storage within the flood plain (as defined by the outcrops of alluvium and flood plain gravels, Figure 2) for May 2007 is calculated to have been 2.6×10^6 m^3. For comparison, this is equivalent to just over 3 h of the peak flow in the River Thames downstream of Oxford during the July 2007 flood event. The available subsurface storage is significantly smaller during typical winter periods.

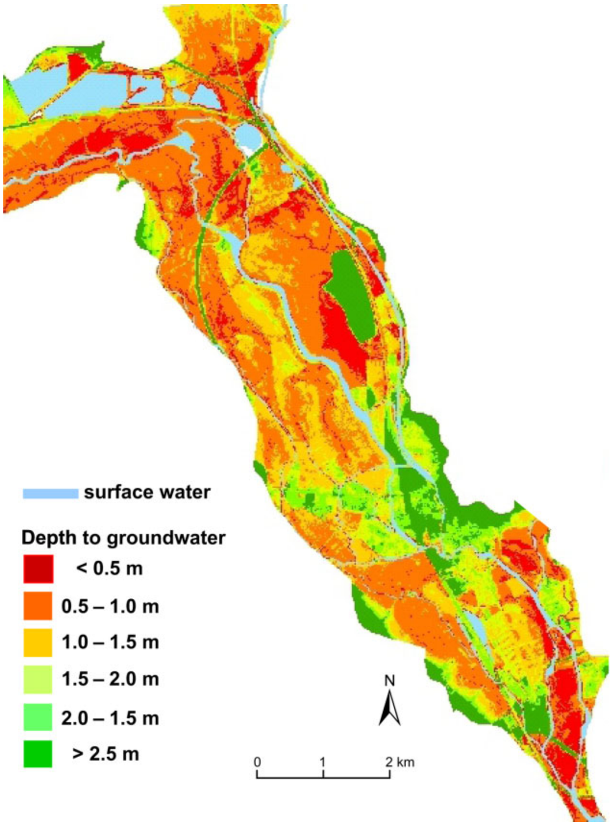

Figure 5 Depth to groundwater within the flood plain superficial deposits in the Oxford valley based on groundwater levels measured in May 2007. BGS © NERC 2011.

Flood conditions

Surface flood water pathways

Surface flood waters were observed following generally similar pathways in July 2007 to the floods in 2000 and 2003, which were also primarily River Thames floods (in comparison, the flood in 1998 was associated more with the River Cherwell catchment). Fluvial flood waters flowed south, parallel with the line of the River Thames. Major structures caused barriers to this flow, including the embankment of a dual carriageway (A34), the Botley Road and associated properties, and the Oxford southern bypass road and historic waste dumps in the south (Figure 3). In the south of Oxford, the main Birmingham to London railway line and the urban areas of Grandpont and New Hinksey (Figure 3), both of which run north to south, separated the flood waters in the west and the east of the valley. The result of all of these areas of high ground was the creation of a series of flood cells, which gradually filled as flood waters continued to enter the Oxford section of the Thames valley from upstream. The flood waters within these cells eventually overtopped, causing flooding of property in the Botley Road area and in the New Hinksey area. Approximately 160 properties were flooded internally in the flood of 2000, a similar number in 2003 and over 200 in 2007. The 2000 and 2003 floods were classified by the Environment Agency as 10- to 15-year return period events, and the 2007 flood event as a 15- to 20-year return period event. In New Hinksey, at a few locations, water flowed from the flood cell associated with the Hinksey Stream, east towards the River Thames which had flooded to a lower level. Figure 6 shows a map of approximate peak flood water elevations during the July 2007 flood, which identifies the flood cells.

Prior to flood waters overtopping structural barriers to flood, subsurface pipework was also seen to be a key pathway for flow. For example, water from flooded areas was flowing out of storm drains on the downstream side of topographic barriers. The ballast fill that surrounds underground pipes can also provide a high permeability pathway for groundwaters during flood events (J. Packman, pers. comm., Centre for Ecology and Hydrology).

Topographic controls on the location of groundwater flooding

River level, flood level and groundwater level data collected during the July 2007 event have helped understand the mechanism by which groundwater flooding occurs within the city of Oxford. Anecdotal information on the location and nature of flooding in 2000 and 2003 suggests the same mechanism controlled groundwater flooding in these events.

In the July 2007 event, responses in groundwater levels were seen both as a result of the rain falling directly on Oxford and the high river levels that occurred in the following days. In the 17 boreholes with automatic water level recorders completed in the shallow gravel aquifer (which are well distributed across the study area), increases in groundwater level seen in the day following the event ranged from 0.28 m to 1.23 m, with an average of 0.59 m. Groundwater levels did not rise above ground level at any of these sites at this time, although in all cases, groundwater levels were within the alluvium (*nota bene* the limited areas of standing water that occurred within the city immediately following the rainfall event were due to the drainage system being overwhelmed by the volume of rain water). Following the initial peak in groundwater levels, there was a period of recession of up to a few days. These groundwaters then responded to the rises in river levels caused by flood waters reaching Oxford from higher in the River Thames catchment. In the majority of the 17 groundwater monitoring sites, groundwater levels rose above ground level at their peak. Artesian conditions were also measured at 12 additional sites that were manually dipped approximately a day before the flood peak.

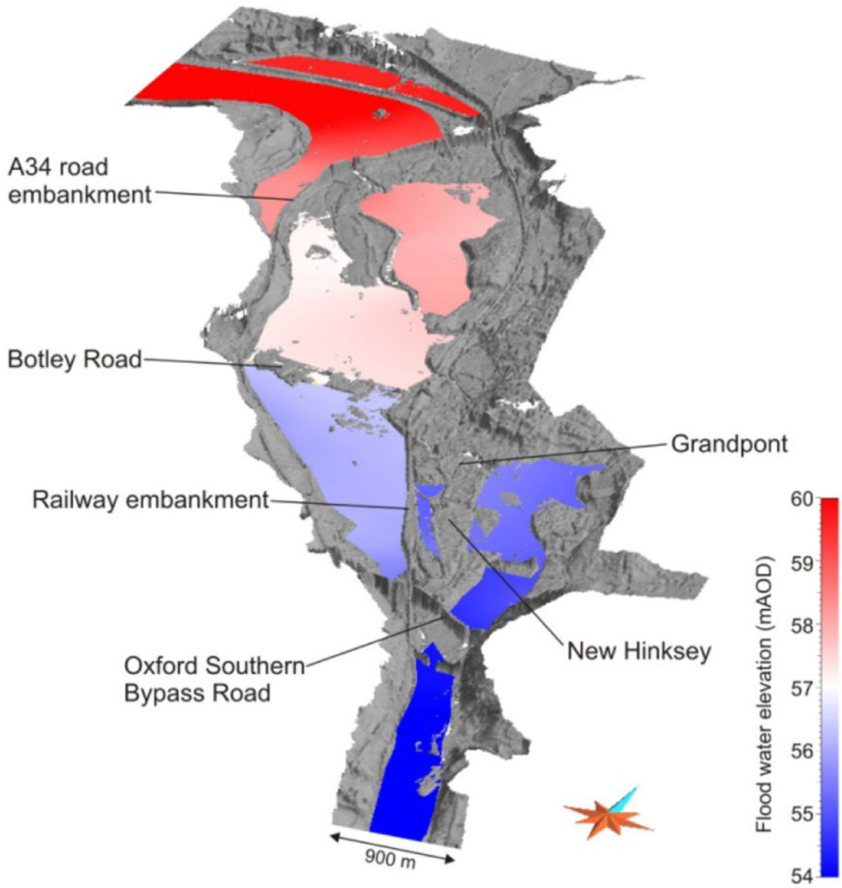

Figure 6 Estimated peak flood water elevation in the Oxford area during the July 2007 flood event. This map includes NEXTMap Britain elevation data from Intermap Technologies. BGS © NERC 2011.

Groundwater response to rainfall and fluvial flooding at sites in the south Oxford area during July 2007 is shown in Figure 7, along with river flood levels. These include (Figure 1) water levels associated with the Hinksey Stream, the River Thames upstream of Iffley lock and the Weirs Mill Stream, and the groundwater level in a borehole, NH1, completed in the sands and gravels in the New Hinksey area (Figure 6). Water levels prior to 19 July 2007 show that the Hinksey Stream and River Thames upstream of the Iffley Lock are at a similar elevation. The Weirs Mill Stream is at a much lower level, similar to the downstream elevation of Iffley Lock. A borehole located next to the Weirs Mill Stream provides evidence that the stream normally acts as a line of discharge from the gravel aquifer. This discharge influences the groundwater level in borehole NH1 (Figure 7). The response in these river and groundwater levels occurred within 4–8 h of the start of the rainfall event. The groundwater level in NH1 rose by 0.53 m, peaking within approximately 9 h of the start of the event. After this point, groundwater levels began to recess. The River Thames levels were partially controlled by the raising of the weir boards in anticipation of high flows to follow. The Hinksey Stream has limited management structures on it and continued to rise over the following days, overbanking and flooding into the surrounding area, primarily farm land. Eventually, flows in the River Thames were also too great and it too overbanked. After the initial rain-dependent peak, the groundwater level at NH1 recessed for approximately 2 days but then started to rise again. A double peak was seen in the flood waters associated with the Hinksey Stream, caused by the lag in flood waters moving from headwaters of a number of tributaries of the Thames. This double peak is reflected in the shape of the groundwater hydrograph for NH1, demonstrating that the fluvial flood waters have some control on the groundwater levels. The peaks allow a good estimate to be made of the lag between the fluvial floods and the groundwater at NH1; the lag between the first peaks was approximately 20 h and between the second peaks was approximately 22 h. Following the second peak, the groundwater levels again recessed, with the gravel aquifer drainage in the locality being controlled by the lower fluvial flood levels associated with downstream of Iffley Lock.

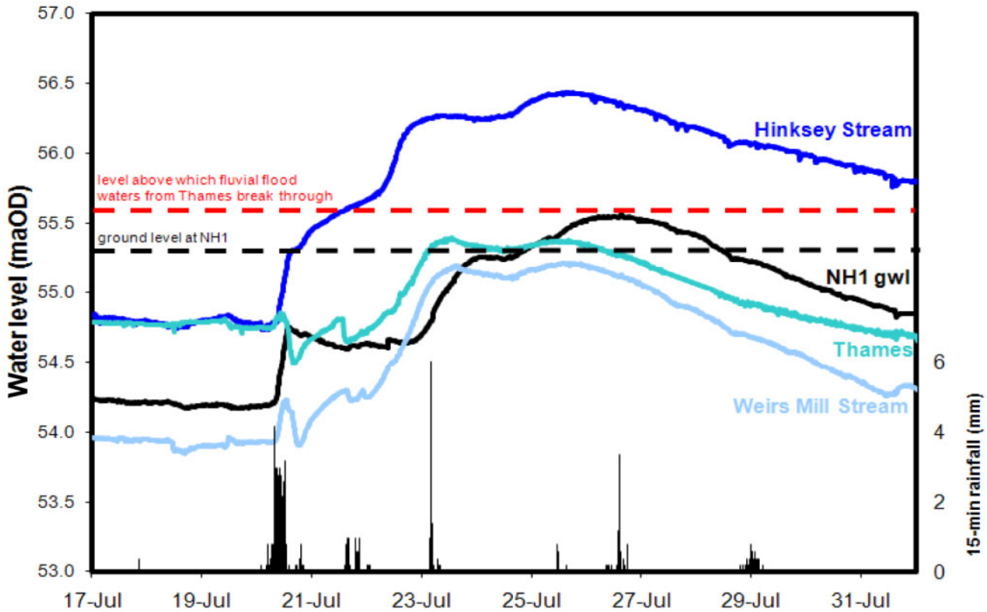

Figure 7 Groundwater levels in borehole NH1 in the New Hinksey area of Oxford during the July 2007 flood event, along with water levels nearby in the River Thames, the Weirs Mill Stream and the Hinksey Stream. Also shown, 15-min rainfall data, ground level at NH1 and the level of the threshold point above which the River Thames would flood the ground at NH1. BGS © NERC 2011. Rainfall and water level data for the River Thames were provided by the Environment Agency.

In the area of borehole NH1, there was significant flooding of gardens, a small number of low-lying properties flooded above the ground floor and many more flooded under ground-floor floorboards. This low-lying area has been protected from fluvial flooding in the past three major floods as it is surrounded by high ground created for the main railway line, roads and housing to raise them above the flood plain. The comparison of groundwater level in NH1 and flood water level in the vicinity, as well as observations of artesian flow, confirmed that the flooding was caused by the emergence of groundwater at surface. At peak flooding, the gravel water level was only 1 cm higher than the flood water level, indicating that the relatively low permeability alluvium, which is 1.1 m thick at this location, did not significantly inhibit the vertical movement of groundwater.

The depth of groundwater flooding at NH1 was, at its peak, 0.25 m. The peak flood waters had an elevation of 55.54 maOD; the threshold of the nearest flooded property to NH1 is 55.42 maOD. The level beyond which fluvial flooding from the River Thames to the east would breach the high ground and flow into the low-lying area in the vicinity of NH1 is approximately 0.2 m above the highest level to which the River Thames rose in this area.

The situation that occurred in the New Hinksey area was seen elsewhere in the urban flood plain areas of Oxford where built-up ground has isolated low-lying areas, protecting them from low return period fluvial flood events but also creating the conditions for groundwater flooding during these events. Often the groundwater flooding only impacts areas of relatively low importance, such as gardens or outhouses; however, there is a significant number of properties on low-lying ground that are vulnerable. Figure 8 illustrates the conditions during the July 2007 flood at a location in the Botley Road area of Oxford, where properties downgradient of the road were initially protected from fluvial flooding by high ground, but where flooding was reported which it was thought was due to rising groundwater. In the latter period of the event, the properties suffered fluvial flooding as surface flood waters overtopped the Botley Road. Figure 8 shows an upstream fluvial flood level and an interpolated groundwater level for the area of the properties based on monitoring in boreholes a few hundreds of metres from the location.

The greatest number of potentially vulnerable properties in Oxford, however, are those older properties in which the void created by raising the ground floor up from the flood plain when they were first built was converted to living space during the period of relatively infrequent high return period flood events in the second half of the 20th century. These rooms below the ground floor are relatively low-lying, at a level close to that of the natural flood plain. Tens of properties that have basement conversions have been identified in south Oxford, and there is evidence of some of these being flooded in the recent events. Some property owners have added waterproof membranes that have been successful in stopping groundwater ingress during flood events.

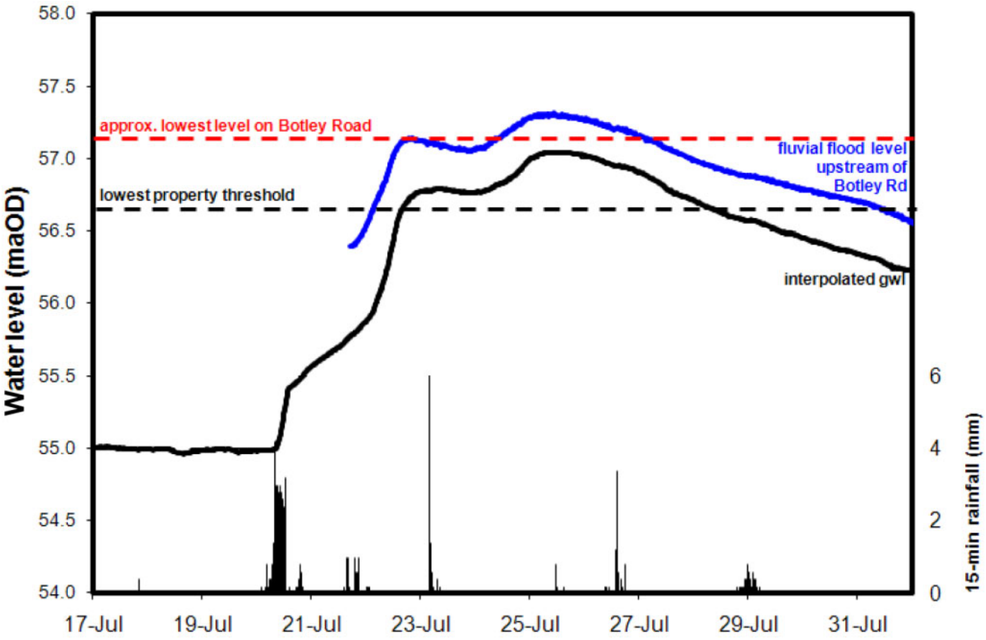

Figure 8 Fluvial flood water levels upstream of the Botley Road and interpolated groundwater levels at an urban location down-gradient of the Botley Road, Oxford during the July 2007 flood event. Also shown, 15-min rainfall data, the threshold of the lowest property in the urban area down-gradient of the Botley Road and the level above which the flood waters upstream of the Botley Road would overtop. BGS © NERC 2011. 15-min rainfall data were provided by the Environment Agency.

Recession of water levels after flooding events

A typical characteristic of groundwater floods is its relatively slow onset and recession in comparison with fluvial floods. For example, in the Chalk aquifer of northern Europe, flooding in some locations has been seen to last for months after the fluvial flood waters associated with the same events have recessed (Pinault *et al.*, 2005). In permeable flood plain deposits, the recession of groundwater levels will be significantly faster. The strong hydraulic connection that sees groundwater levels respond quickly to rises in river level also means that river channels are effective at draining aquifers once the fluvial flood event has passed. In July 2007, it took groundwater levels in the gravel aquifer between only 2 and 15 days to recede back to below ground level from first becoming artesian. However, it was observed that in some isolated low-lying areas, even though groundwater levels in the gravels recessed, flood waters sitting on top of the relatively low permeable alluvium sediments took a longer period to drain.

Mapping of groundwater flooding

The mapping of groundwater flooding was undertaken to help the Environment Agency gauge the scale of the vulnerability in Oxford from this form of flooding. The approach, described in the methodology section earlier, was a first-pass mapping exercise that made a number of significant assumptions. Any restriction on the vertical movement of groundwater from the gravel aquifer to above ground level due to the alluvium layer was not taken into account. The mapping exercise made the assumption that groundwater flooding could occur wherever the groundwater head was thought likely to be above ground level. In areas where there were few groundwater level data, an assumption was made, based on the strong hydraulic connection between river and aquifer, that groundwater levels could be estimated by interpolating between adjacent flood cells.

The mapping exercise showed that there are large areas of urban Oxford that could potentially be affected by groundwater flooding that for certain flood return periods would not be prone to fluvial flooding. However, the majority of these areas are gardens and not internal to property. There are estimated to be only tens of properties that may be affected by groundwater flooding at ground floor level where fluvial flooding had not already occurred. However, this assessment was limited by the availability of flood threshold data as there are over 200 properties in areas that could be vulnerable which have not had their ground-floor flood threshold level surveyed. The exercise identified a significant number of properties that are potentially vulnerable to groundwater flooding of rooms below ground floor level. Walking surveys identified over 80 properties with basements that may be in use as living areas or for storage.

With reference to the output of an Environment Agency river model, the flood events from which properties are most vulnerable to groundwater flooding alone were identified as those with return periods of 10–25 years. Lower return period events would appear to result in flood levels that are not sufficiently high to cause serious groundwater flooding; events with higher return periods would likely cause fluvial flooding to mask the initial groundwater flooding. For these events, groundwater would again be an issue if flood defence measures were introduced that held back fluvial flood waters but did not reduce the heads driving water into the gravel aquifer.

Discussion

The experience of flooding in Oxford highlights the complexities created by an urbanised area located on a major river flood plain. Topographical variations, resulting in many cases from construction to protect properties and the transport network from flooding, create complex flood pathways and areas of high vulnerability. Data collected during flood events in Oxford have shown that groundwater flood-prone areas can be created by the isolation of low-lying areas from fluvial flooding as a result of surrounding man-made ground. Groundwater levels rise in response to direct rainfall, and to elevated river levels and associated fluvial flood zones. Waters from the fluvial flood zones make their way to these isolated low-lying areas by passing through the permeable sediments underlying the flood plain. Although water level data indicate that the low permeability, near-surface alluvium inhibits the flow of water into and out of the underlying sands and gravels aquifer, there is a substantial amount of evidence that the alluvium is sufficiently permeable to allow groundwater flooding to occur. However, data on the alluvium is limited and is not currently sufficient to identify how vertical flow during periods of groundwater flooding is spatially distributed, and whether it is dominated by windows of higher permeability material. Where buildings have been constructed, the removal of alluvium also provides preferential pathways for upward groundwater movement as do man-made drainage channels.

Where artesian conditions do occur, these can result in flooded gardens, cause drainage problems (e.g. inundation of sewerage systems) and create dampness beneath floorboards. Groundwater can also flood properties in these low-lying areas where the ground floor or the inhabited basement is close to the level of the natural flood plain.

Quantifying the risk of groundwater flooding in these urban environments is difficult. The combination of flood mechanisms means that the degree of groundwater flooding will vary according to the nature of the overall flood event. The height to which groundwater levels reach during a flood event will depend on a number of flood event-specific factors: the amount of rainfall directly on the city as opposed to the upstream catchment, the rate at which river levels rise and the period for which fluvial flooding persists, as well as the antecedent soil and groundwater conditions. A methodology based on detailed topography, property thresholds, groundwater level data and the output from a river flood model has been used here to provide a preliminary assessment of groundwater flood risk that fits well with recent observed and reported flood events.

Options for mitigating groundwater flood risk are limited. Where basements are prone to flooding, these can be waterproofed or pumps can be installed, although the anecdotal evidence available from the 2007 flood event showed the latter to have variable effects. Impermeable barriers to the base of the shallow aquifer to stop groundwater flow into an area of housing is an option, although the change in groundwater flows patterns could have detrimental environmental impacts outside of flooding periods. The main approach being proposed by the Environment Agency for fluvial flood risk reduction in the Oxford area is to increase the conveyance of flood waters through the Thames flood plain. This would be achieved by the removal or widening of structures that restrict flow, maximising the use of existing channels, and the limited introduction of new channels. Baffles would be installed to maintain flows during low flow periods and to avoid the gravel aquifer being overdrained such that lowered groundwater levels have detrimental impacts on dependent ecosystems.

It is thought that this approach to fluvial flood risk reduction is an appropriate means to reduce groundwater flood risk in Oxford and in similar settings as it reduces groundwater recharge from fluvial flood zones. It is recognised that there remains the potential for groundwater levels to rise above the ground surface as a result of direct rainfall recharge. This will depend again on antecedent soil and groundwater levels. In the Oxford case, using the attributed 3D geological model and a water table surface contoured from monitored water level data collected prior to the July 2007 event, a very approximate value for the averaged capacity of the flood plain aquifer to accept recharge was calculated as equivalent to a rainfall event of the order of 150 mm. However, locally, including areas with groundwater flood-prone properties where the water table is relatively shallow, a substantially smaller rainfall event could result in groundwater levels rising above ground level due to direct rainfall recharge alone. In winter periods, the risk of this occurring is significantly higher due to the degree of saturation of the shallow flood plain sediments. This insight has implications for the application of sustainable urban drainage systems in similar settings, suggesting that techniques that delay recharge are worth considering.

Conclusions

1. In recent years in Europe, there has been recognition that there is a requirement to better understand the risk from flooding as the result of abnormally high groundwater levels. This has been due both to the occurrence of major flooding events clearly attributable to groundwater and the related inclusion of groundwater flooding in European and national legislation.
2. In flood plains underlain by highly permeable deposits, groundwater rise leading to groundwater flooding can be due to direct rainfall recharge as well as flow into the sediments from rivers with high water levels and areas inundated with fluvial flooding. However, the good hydraulic connection between river and aquifer means that the aquifer can drain quickly as fluvial flood waters recess. Groundwater flooding in these settings is relatively short-lived compared with other groundwater flood settings, for example Chalk catchments.
3. The Oxford case study has shown that man-made ground built up from the natural flood plain to reduce the risk of fluvial flooding of property and the transport network can create adjacent isolated low-lying areas that are prone to groundwater flooding. In Oxford, there are a limited number of properties that are flooded above ground level in these areas but potentially tens of properties with inhabited basements that could flood as a result of abnormally high groundwater levels. It is estimated that these properties are affected by groundwater flooding alone during relatively low return period flood events of 10–25 years; during higher return period events, groundwater flooding will precede fluvial flood inundation.
4. Appropriate options for mitigating groundwater flooding in these urban flood plain settings are limited particularly where it is the result of direct rainfall recharge. As is proposed through the Oxford Flood Risk Management Study, an effective means to reduce the risk of groundwater flooding at the city scale is to increase the rate of conveyance of flood waters, reducing the heads within the flood cells that can drive water into the underlying permeable sediments. The potential for groundwater flooding due to rainfall alone has implications for the implementation of sustainable urban drainage systems.
5. Generally, in urban flood plain settings, fluvial flood assessments will underestimate the extent of flooding if aquifer pathways are not taken into account. Further work is required to link river and groundwater models to simulate flood events, to enable better quantification of the risk of groundwater flooding and to assess the potential for using river level monitoring in early warning systems for groundwater flooding.

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Chapter 5: Persistence of flooding on permeable lowland floodplains: the role of shallow groundwater and surface infiltration

The work presented in this chapter is to a standard and in a format that enables it to be submitted to an international peer-reviewed journal.

Persistence of flooding on permeable lowland floodplains: the role of shallow groundwater and surface infiltration

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Abstract

Natural levees and modified topography can retain water on urbanised floodplains for long periods after the peak of fluvial flood events. The duration that flood waters remain on the floodplain can be reduced by drainage infrastructure but where this is absent, shallow groundwater and low hydraulic conductivity surface geology are key controls. Few flood inundation models sufficiently incorporate groundwater and infiltration processes. A model system was developed coupling a commonly used flood inundation model with a groundwater flow model which allows the vertical exchange of water through a surface layer. The model system was applied to an intensely monitored lowland floodplain in Oxford, UK. The model was used to examine water exchange during a large flood event and the duration of flooding was examined through a sensitivity analysis of the surface layer hydraulic conductivity. This modelling study shows that by taking into account low-depth flood waters, flooding can last up to weeks after the main fluvial flood event has passed, with the length of the period being very sensitive to the surface layer hydraulic conductivity. This persistent flooding can have a significant socio-economic impact, for example by restricting vehicle movement through road closures. There is an imperative for flood risk managers to collect evidence to assess the scale of the current and future impact of this type of flooding and to use modelling tools, such as developed in this study, to examine measures to mitigate where there is significant risk.

1. Introduction

The global socio-economic cost of flooding is substantial and increasing rapidly as a result of a higher frequency and severity of flood events, land subsidence and increasing density of vulnerable populations in flood-prone locations (Jongman et al., 2012; Ward et al., 2017; Willner et al., 2017). The characteristics of flooding that are most strongly correlated with impact are frequency, depth, flow and duration (Thieken et al., 2005; Kreibich et al., 2009). In fluvial flood events, much of the water that flows out of river channels onto the associated floodplain will find its way back to the river following the flood peak, as the river level recedes. However, natural levees and anthropogenic changes to topography, particularly in urban areas, may mean that shallow water is retained on the floodplain (Lewin & Ashworth, 2014), extending the period of flooding (McMillan and Brasington, 2007). In this context, Moftakhari et al. (2018) define 'nuisance flooding' as shallow depths of flood water of 3-10 cm, that do not pose significant threat to public safety or cause major property damage but do impact transport and public health. This impact may be caused by direct contact with flood waters but may also be indirect, for example by blocking the road network (Hammond et al., 2015). Moftakhari et al. (2017) showed that the cumulative exposure to frequent, relatively small flood events could cause a greater economic impact than exposure to less frequent extreme events.

In urban areas, engineered drainage may help remove flood waters (Hibbs & Sharp, 2012). Where drainage is absent, the recession of the flood waters will be influenced by the hydraulic conductivity of the underlying floodplain sediments. Land development that has occurred in recent human history, such as the stripping of natural vegetation to allow agriculture, has contributed to higher loads of fine sediments within rivers globally (Walling & Fang, 2003; Macklin et al., 2010). The deposition of this material downstream has produced floodplain sediments that commonly have a low hydraulic conductivity surface layer (alluvium), often contrasting with the substantially more permeable underlying deposits (Bricker & Bloomfield, 2014). Alluvium hydraulic conductivity can vary greatly depending on the grain size distribution, the degree of compaction and the amount of organic matter (Bridge, 2009).

In addition to the hydraulic conductivity of the shallow floodplain sediments, the recession of water trapped there by the topography may be limited by saturated ground associated with high groundwater levels. In highly permeable floodplains, within the relatively short timescale of a flood event, a significant rise in groundwater level can occur as a pressure response to raised river levels. This pressure response is determined by the diffusivity of the aquifer (Pinder et al., 1969) and hence is greater where confined conditions occur. Although river recession after flood events will cause groundwater levels to decline, these levels are always relatively shallow in floodplain environments.

Relatively few surface-groundwater interaction studies have examined the dynamics between inundated floodplains and shallow groundwater. As a consequence the physics of the hydrological processes in areas with shallow groundwater is not well understood, and typically it is not included in river or groundwater models (Doble et al., 2012). Intensive monitoring and modelling of short sections of rivers (Bates et al., 2000; Burt et al., 2002; Claxton et al., 2003) has shown that in such settings floodplain hydrology is predominately a two-dimensional (lateral) process and that down reach flow effects are only significant at the beginning and end of flood events. Where permeable sediments underlie the floodplain, the river stage is likely to be the principal driver of water table fluctuations (e.g. Jung et al., 2004; Lewandowski et al., 2009). Doble et al. (2012) showed that the hydraulic conductivity of the river bed and the surface layer of the floodplain is important by undertaking a sensitivity analysis on a 2D vertical slice model of a conceptualised floodplain-aquifer system perpendicular to a river. They also showed that irregularities in the floodplain elevation were found to have a large impact on the infiltration volume, but given the structure of the model, did not represent flow across the floodplain in 2D. Further work is required to investigate the spatio-temporal variability of vertical floodplain-aquifer fluxes in more complex settings, for example, with irregular floodplain topography in two dimensions and adjacent to non-linear channels.

Complex, distributed models with detailed physics-based process representations, such as Mike-SHE, have typically been used to simulate variations in groundwater levels in two dimensions across a floodplain, and their interaction with channels and the land surface (Bernard-Jannin et al., 2016; Clilverd et al., 2016; House et al., 2016; Thompson et al., 2017). However, such codes are not generally used by the flood forecasting community (Teng et al., 2017; Jain et al., 2018), given their highly parameterised nature and relatively long run-times, which limits their use in probabilistic ensemble forecasting frameworks. For example, Bernard-Jannin et al. (2016) undertook a comprehensive study of surface water-groundwater exchanges during overbank flood events by coupling 2D modelling of surface flow with the Saint-Venant equation, and 3D modelling of unsaturated groundwater flow with the Richard's equation. One finding from the modelling of this unconfined system was that significant differences in exchanges occurred in response to floods of different magnitude. However, the distributed model was computationally expensive and could only be applied at the reach scale for a short time. By aggregating 1 m LIDAR topographic data onto a 25 m grid they reduced model run-times but found that they incorrectly defined levees and introduced errors into the simulation of the flood extent. Other small structures, such as dykes and ditches, would also need to be represented at a higher resolution to improve the accuracy of the simulations of flood extent and floodplain-aquifer interaction.

Conversely, groundwater fluxes are often neglected, or poorly represented, in reduced-complexity models, using kinematic or diffusive wave approximations, which are widely-used to simulate inundated floodplains (see Teng et al., 2017 for a review), such as FloodMap (Yu & Lane, 2006), JFLOW (Bradbrook, 2006), and LISFLOOD-FP (Bates & De Roo, 2000). For example, in modelling groundwater flooding caused by the groundwater table rising to the land surface in response to rainfall recharge, Morris et al. (2018) calculated groundwater discharge rates along a chalk valley using a simple Darcian calculation. These estimated flows drove their JFLOW flood inundation model but were calibrated during the modelling process by comparing simulated floodplain flows against targeted spot flow measurements at observed points of groundwater emergence and at a point near the perennial head of the river downstream, which were taken during the flood event. Such an approach does not allow for predictive simulation, and for this reason, one objective of this study was to integrate a groundwater model with a cellular flood inundation model to be able to simulate floodplain aquifer interaction.

The overall aim of this research was to investigate the dynamics of interacting groundwater and flood water within a case study floodplain, to determine the influence that a low hydraulic conductivity surface layer and shallow groundwater have on flood duration. The study used the peri-urban floodplain of the River Thames within the city of Oxford, UK, an area within which the hydrogeology is well-characterised and there is a comprehensive hydrological monitoring network (Macdonald et al., 2018a). The study focussed on a major flood event on the River Thames, which occurred in July 2007. Observed river, floodplain and groundwater level data and aerial photography of the extent of flooding, provided a means to advance understanding of the response of the system during the flood event. Furthermore, these data were used to assess the performance of a new linked groundwater and flood inundation modelling code, developed by integrating the popular cellular, or 'storage cell', flood inundation model, LISFLOOD-FP, with a finite-difference groundwater model. The coupled model was then used to estimate vertical fluxes across the floodplain and to test the effect of different model parameterisations on flood persistence. The study was used to provide generic insights into the potential importance of the subsurface in increasing the impact of flooding.

2. Materials and methods

2.1 Study area

The urban and peri-urban floodplain of the River Thames at Oxford is a setting where the modification of the land surface and construction of infrastructure has produced complex flooding patterns and altered surface water-groundwater interactions (Macdonald et al., 2018a). The floodplain has an area of approximately 20.4 km^2, ranging in width from 410 to 2170 m (Figure 1). The valley of the Thames in the Oxford area is bounded by high ground to the west and east. Floodplain sediments in the incised river valley are almost entirely contained within impermeable clay bedrock (Newell, 2008).

The long-term mean flow in the River Thames, measured at the upstream end of the Oxford valley, is 18 m^{3}s^{-1} (Marsh & Hannaford, 2008). Within the valley the Thames splits into a series of channels which then rejoin at the downstream end. The confluence of the River Cherwell and the River Thames is within the Oxford valley. The water levels in the main River Thames channel are managed by a series of locks installed to increase river depths for navigation (Figure 1). These locks influence groundwater levels, creating complex flow patterns (Macdonald et al., 2018)

In Oxford, key transport routes and approximately 3400 properties are located on the Thames floodplain. These have had serious impacts from recent major floods in April 1998, December 2000, January 2003, July 2007, November 2012 and January 2014, involving both fluvial and groundwater flooding (Macdonald et al., 2012). Areas of flood water persist within parts of the city, days after river levels have receded.

This study focussed on Port Meadow (Figures 1 and 2), a flood-prone area of ground owned by the local authority. The meadow is covered primarily by short grass and is used for grazing of horses and cattle. It has environmental protection due to a rare plant species, and is designated as a historical monument due to archaeological features. Port Meadow was chosen due to its topography, as water is retained on the meadow for extended periods after flooding, and it has a good water level monitoring network.

The River Thames runs along the western boundary of Port Meadow (Figure 2a). A stream, which branches off the Thames towards the east, borders the south of the meadow. A series of shallow channels, of less than a metre in depth, cross the meadow in the northern part and also run along the eastern edges (Figure 2a). High ground immediately to the west of the River Thames stops fluvial flood waters flowing in that direction. To the east of Port Meadow, a railway line and a closed, licensed waste dump form high ground. Within Port Meadow in the south is a further historical waste dump (indicated in Figure 2b by high ground along the southern boundary of the model domain). Together these features act to contain the majority of flood waters. A levee on the eastern bank of the Thames acts to retain flood waters on the floodplain after river levels have returned to within-bank. The ground across the main area of the meadow slopes from north to south with an average gradient of 0.053%. A digital elevation model (DEM) was available for the study area derived from a 2 m LIDAR survey (Cracknell & Hayes, 2007).

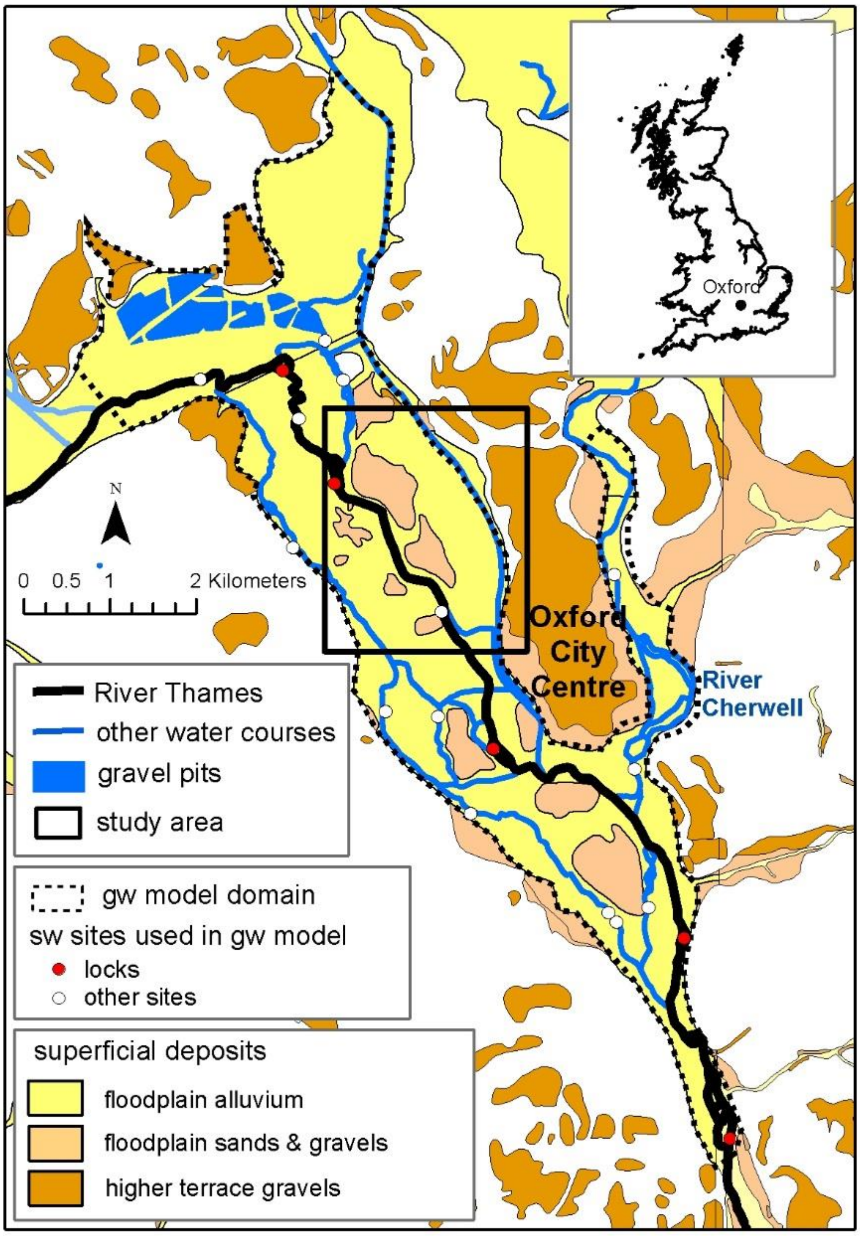


Figure 1 Superficial geology of the valley of the River Thames in the vicinity of the city of Oxford, UK. Areas in white are higher ground underlain by bedrock. The modelling study area is indicated, along with the full domain of the floodplain gravels groundwater model and surface water monitoring sites used in this model. Contains Ordnance Survey data © Crown copyright and database right (2018).

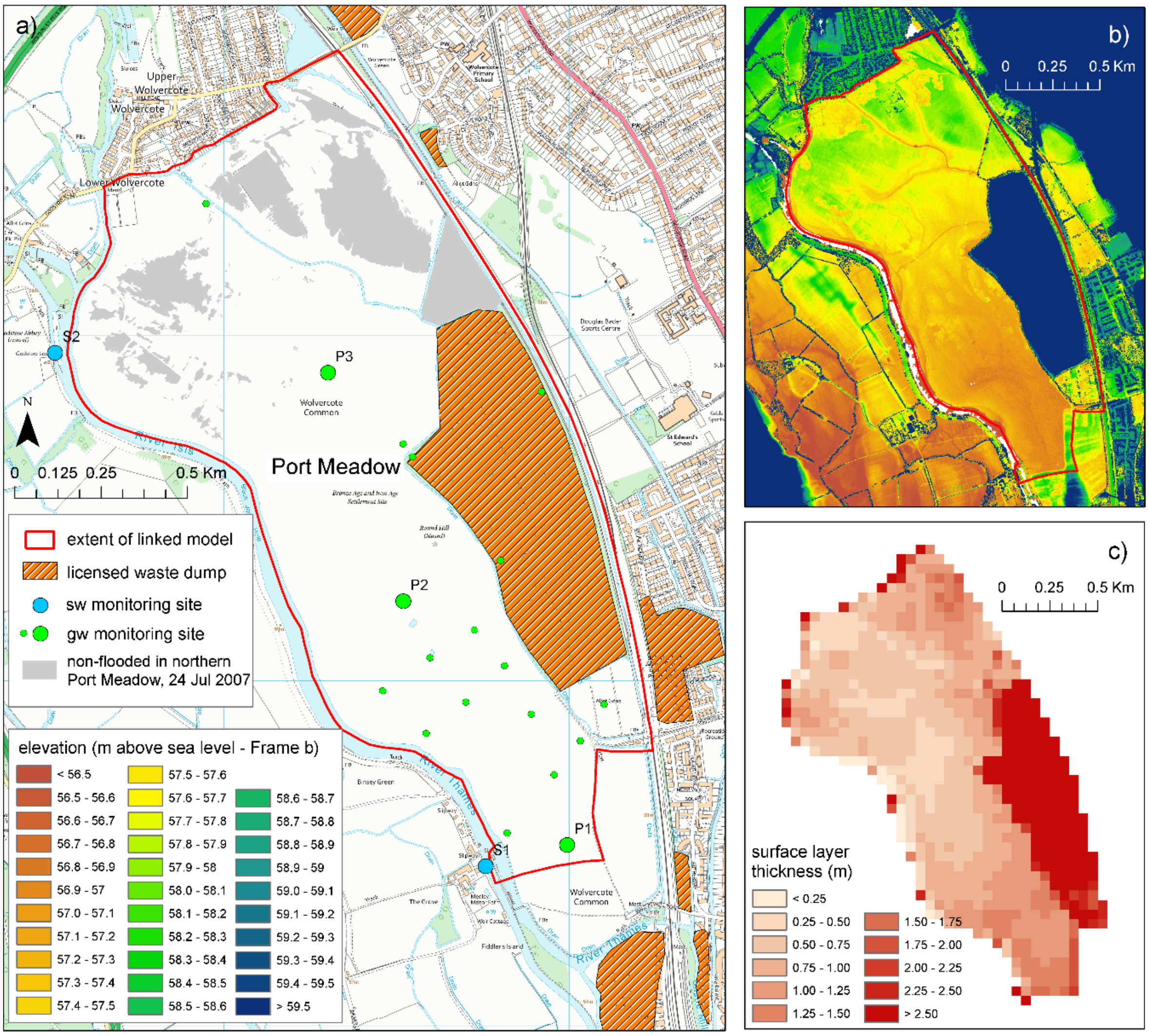


Figure 2 Port Meadow modelling study area showing a) basemap, linked model extent, relevant monitoring sites, waste dumps, and the areas of the northern part of Port Meadow that were not flooded, based on aerial photography taken by the British Geological Survey at 12:40 on 24 July 2007; b) elevation; and c) thickness of model surface layer (n.b. this layer includes made ground, hence the large thickness coincident with the waste dumps). Contains Ordnance Survey data © Crown copyright and database right (2018). Contains Environment Agency data licensed under the Open Government Licence v3.0.

The floodplain is underlain by alluvial sediments, a shallow layer of silty clay alluvium above sands and calcareous limestone gravels. In the three-dimensional (3D) geological model for the Oxford valley developed by Newell (2008), the thicknesses in the Port Meadow area range from 0.3 to 2.7 m for the alluvium (Figure 2c; n.b. this map also includes made ground) and 2.0 to 8.1 m for the sands and gravels. Hydraulic conductivities for the sands and gravels layer in the northern zone of the Oxford valley, including Port Meadow, were previously estimated from pumping and packer tests, and found to be in the range 100 to 1000 m d^{-1} (Dixon, 2004), whilst estimates based on grain size distribution measurements from sediments collected during the drilling of boreholes within Port Meadow gave a range of 200 to 600 m d^{-1} (Gooddy et al., 2014). In contrast, hydraulic conductivities for the alluvium estimated in studies undertaken in the Oxford valley are substantially lower, ranging from 0.03 to 0.3 m d^{-1} (Gardner, 1991; Gowing & Youngs, 2005; Hodgson, 2008).

Studies within the Oxford floodplain that have addressed the status of terrestrial ecosystems (Institute of Hydrology, 1987; Gowing & Youngs, 2005), flood risk management (Macdonald et al., 2012) and pollution from historical waste dumps (Gooddy et al., 2014), have resulted in an extensive water level monitoring network, both for surface and groundwaters (Figure 2a). Surface water levels are monitored in the River Thames at the downstream (S1) and upstream (S2) ends of the reach adjacent to Port Meadow, as well as an additional 24 sites across the Oxford floodplain (Figure 1). Nineteen monitoring boreholes (Figure 2a) located within Port Meadow, drilled over the period 2005 to 2010, were previously used to construct groundwater level contours for the area (Macdonald et al., 2018a). Of the 19 boreholes, 14 were equipped with digital water level recorders and, of these, seven, mostly located in the south of Port Meadow, were available for the full period used in the model calibration and the July 2007 flood event. The three boreholes that were used in this study for comparison with simulated groundwater levels produced from the linked model, P1, P2 and P3, are shown in Figure 2a; this subset was chosen to provide a more evenly distributed coverage of boreholes.

The boreholes were drilled at least 1 m into the sands and gravels beneath the alluvium and screened over the bottom 0.5 m. In addition, a hole augered to a few tens of centimetres into the alluvium, A1, was paired with gravel borehole P1. It was considered that when the water levels measured in A1 were above ground level these would be the same as the flood levels on the floodplain at this location. Within the study area, datums for the boreholes and augered sites and S1 were measured using a GPS system accurate to 3 cm. The ground levels at P1, P2 and P3 are 56.82, 57.05 and 57.59 m asl, respectively. Water level data were measured every 15 minutes at monitoring sites P1, P2, P3, A1, S1 and S2 using digital pressure loggers and compensated for atmospheric pressure using a barometric pressure logger located close to S2; both had an accuracy of 0.5 cm.

Data from the gravel monitoring sites show that for the majority of time the groundwater head fluctuates within the alluvium. Calculations of storage coefficient reported by Dixon (2004) for the Oxford valley indicate the gravel aquifer is semi-confined by the alluvium. Contours of groundwater levels across Port Meadow show an overall gradient, outside of periods of heavy rainfall, of 0.064% north-east to south-west, towards the River Thames (Macdonald et al., 2018a), implying that there is some recharge to the floodplain aquifer from higher terrace gravels to the east. To the west of the River Thames the water level gradient continues from north-east to south-west towards the Seacourt Stream, which gains groundwater in this area (Macdonald et al., 2018a).

2.2 The July 2007 flood event

Many parts of England and Wales experienced severe flooding during June and July 2007 due to extreme rainfall. Between May and July most of southern Britain registered more than twice the 1961-1990 average rainfall as a series of deep anticyclonic weather systems moved across the country (Marsh, 2008). Extreme rain fell on July 19 and 20 in the headwaters of the River Thames (e.g. 140.1 mm in 24 h in Chastleton, Oxfordshire; Marsh & Hannaford, 2007). Peak river flow in Oxford occurred approximately 5 days after the rainfall event (225 m^3s^{-1} at the lock furthest downstream within the Oxford reach of the Thames). Heavy rainfall in Oxford was 60.6 mm in total and limited to July 20. The start of the flood event in Oxford, for the purposes of this study, has been defined as 03:00 on 20 July 2007, the approximate time when the high intensity rainfall began. River levels monitored at S1 (Figure 3) and S2 began to rise within hours of the start of the rainfall at a rate of ~4 cm hr^{-1}. There were three peaks in the river level during the main event, which were due to differences in rainfall across sub-catchments of the River Thames and the timing of their contributions to flow at Oxford. The highest river level at S1, which occurred at 07:00 on 25 July 2007, was 1.22 m above the pre-event level. It took 40 days to return to the pre-event level, however there was a period of six days of rainfall in mid-August when 37.6 mm fell and river levels rose again by approximately 0.2 m.

Groundwater levels started to rise within a few hours of the start of the rainfall event (Figure 3). The groundwater levels at P1, P2 and P3 were 0.13, 0.36 and 0.76 m below ground level, respectively, prior to the event, and took 9, 6 and 38 hours to reach the ground surface. The drilling log for P3 shows the depth of the alluvium as 0.5 m, so the aquifer prior to the flood event was not confined (Figure 3); it took 8 hours from the start of the flood event to become confined. After the intense rainfall ended at

14:00 on 20 July 2007, groundwater levels quickly stabilised, or in the case of P3 decreased, but subsequently began to rise again following a similar pattern to that of the river. The groundwater levels at P1, P2 and P3 peaked at 0.85, 0.78 and 0.22 m, respectively, above ground level, 11, 5 and 7 hours after the time of the river level maximum at S1. The groundwater levels at P1, P2 and P3 declined to below ground on the 28 July, 12 August and 31 July 2007, respectively, 39, 64 and 11 days after the start of the event. Of note in the groundwater hydrographs were: the fast recession at P3 following rainfall, likely indicating a local groundwater discharge zone; and the diurnal fluctuations during the recession at P2 and P3, once groundwater fell below ground level and grass was able to transpire. The smaller diurnal fluctuations that can be seen in the surface water hydrographs are thought to be due to temperature effects on the barometric pressure loggers, as have been seen in other types of environmental sensors (Verhoef et al., 2006).

The monitoring data from A1 showed water appearing on the ground surface at the same time as the groundwater level at P1 rose above ground level (Figure 3a). The surface water level was above the groundwater level for almost all of the event, apart from an early period during the main event recession when the levels were coincident. The maximum depth of surface water at A1 during the event was 0.98 m. Surface water did not disappear at A1 until 22 September, 64 days after the start of the event and 25 days after the groundwater level at P1 had dropped below ground level.

In addition to the observational measurements, aerial photographs of the flooding of the northern half of Port Meadow were taken by the British Geological Survey at 12:40 on 24 July 2007. The areas that were not flooded at this time are shown by the grey polygons in Figure 2a.

2.3 Model codes and linkage

To simulate groundwater flow, floodplain inundation, and the interaction between groundwater and surface water we integrated a finite difference groundwater flow code with a 'storage cell' surface water model that solves for floodplain flows and levels.

2.3.1 Groundwater model

To simulate groundwater flow we use the ZOOMQ3D code (Jackson & Spink, 2004). This uses an implicit finite-difference scheme to solve the governing equation of groundwater flow on a three-dimensional Cartesian grid, which can be locally refined horizontally to increase resolution. The code simulates confined and unconfined, heterogeneous and anisotropic aquifers. River-aquifer interaction is simulated using a Darcian head-dependent flux based on the difference between groundwater head and river stage and is calculated as:

$$Q_z = \frac{Kr}{B} \, W \, L \, (h_a - h_r) \tag{1}$$

where Kr is the vertical hydraulic conductivity of the river bed (m day^{-1}), B is the thickness of the river bed (m), W is width of the river (m), L is length of the river reach (m), h_a is the groundwater head (m) and h_r is the river stage (m). Equation 1 is modified for 'free drainage' when the groundwater level falls below the base of the river bed. In this case the driving head is the difference between the river stage and the base of the river bed. The river stage can either be calculated from the simulated flow through the specification of a stage-discharge rating equation or rating table, or it can be specified over time.

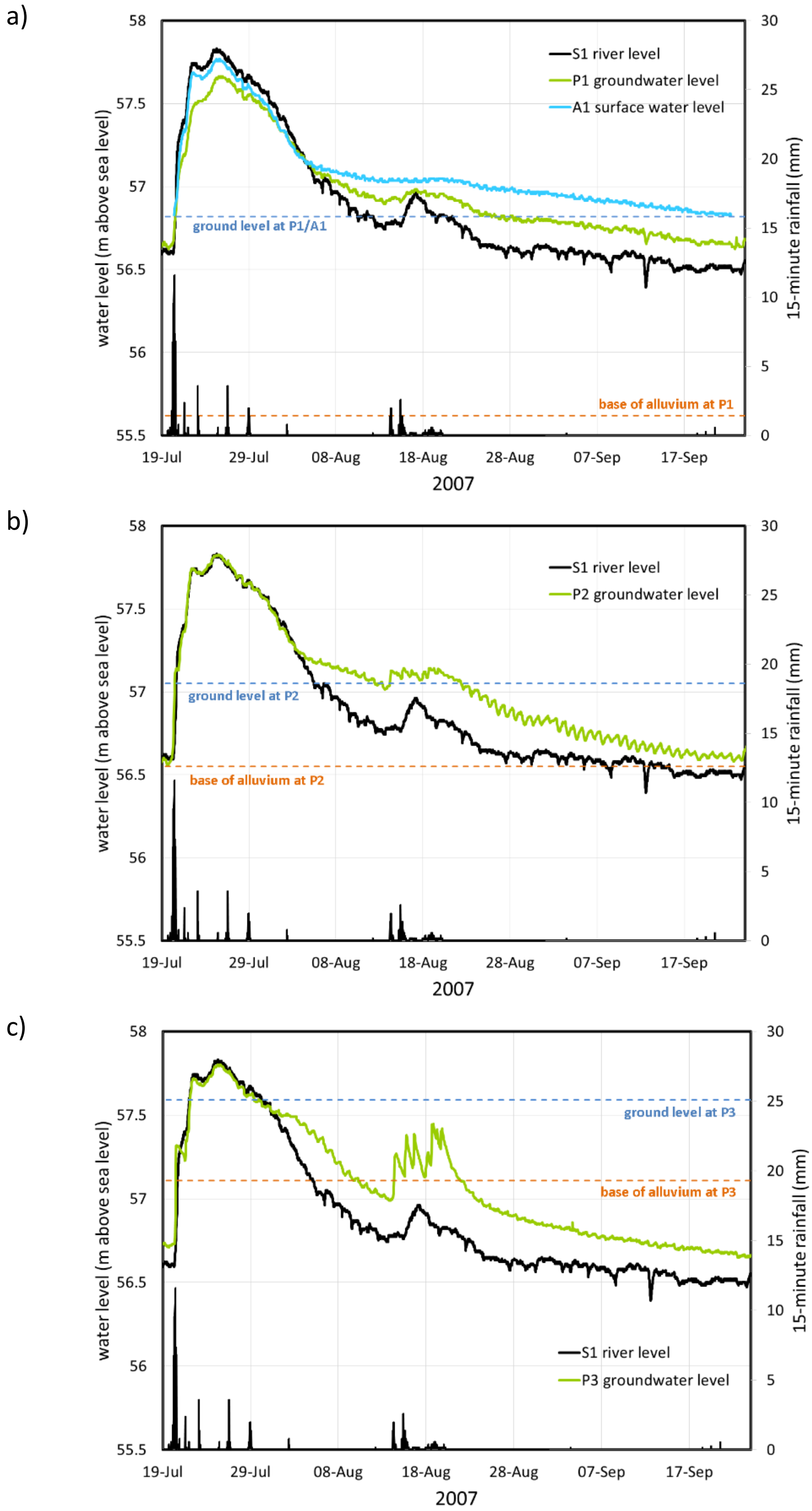


Figure 3 Observed groundwater and surface water levels for the Port Meadow area, as well as 15-minute rainfall data from Eynsham Lock in the north west of the Oxford floodplain, for the period 19 July to 26 September 2007. River level at S1 is plotted in each chart for comparison with the groundwater levels. Contains Environment Agency data licensed under the Open Government Licence v3.0.

2.3.2 Flood inundation model

To simulate flow over the floodplain, we use the method implemented in the cellular, or 'storage cell', model of Bates et al. (2010). This is one of the set of numerical solutions used in the widely-applied LISFLOOD-FP floodplain inundation modelling code (Bates & De Roo, 2000). Storage cell models have been developed as an alternative to models that solve the full shallow water equations in two dimensions, which can involve a significant computational overhead. This reduced complexity approach discretises the floodplain into cells and calculates the flux of water in each Cartesian direction analytically, potentially reducing the computational overhead to lower than that of equivalent numerical solutions of the full shallow water equations. They take advantage of the fact that flows over floodplains are typically slow and shallow, and gradients of the local free surface are very small. Consequently, the inertial terms of the governing equations of de Saint-Venant (Chow, 1988) can be neglected. However, Bates et al. (2010) incorporated a simple inertial term into the formulation of inter-cell fluxes to improve the stability of such schemes and their range of applicability. LISFLOOD-FP has been used to simulate flood inundation at high resolutions within complex urban settings (Neal et al., 2011; Sampson et al., 2012), across large areas (Schumann et al., 2013; Dottori et al., 2016), and over decadal to centennial time-scales (Coulthard et al., 2013; Barkwith et al., 2015; Liu & Coulthard, 2017). Considering that groundwater flood events can last from weeks to months (Habets et al., 2010; Hughes et al., 2011; Macdonald et al., 2012; Gotkowitz et al., 2014; Ascott et al., 2017), alongside the functionality, simplicity, and applicability of the storage-cell model of Bates et al. (2010), we selected it as the approach to adopt in this study; the numerical solution was coded into ZOOMQ3D in C++.

2.3.3 Flood water-groundwater interaction

Flood water-groundwater interaction is modelled using the conductance concept, in which there is an exchange flux of water through an interface connecting the two domains (Kollet & Maxwell, 2006; Anderson et al., 2015). This is simulated using a similar but modified version of Equation 1, again limiting the maximum flux to represent free drainage. Typically the width of floodplain model cells is less than 10 m, and as groundwater model cells are one to two orders of magnitude greater, multiple DEM cells will be associated with one groundwater solution point. Consequently, the total flow between the floodplain and aquifer is calculated by:

$$Q_z = \sum_{i=1}^{m} \sum_{j=1}^{n} \frac{K_z^{ij}}{B^{ij}} \, \Delta l^2 \left(h_a - h_s^{ij}\right) \tag{2}$$

where Δl is the width of the square DEM cells, m and n are the number of DEM cells in the x and y directions covered by the square or rectangular groundwater model grid cell, and h_s^{ij} is the surface water elevation at DEM cell i, j. The B and K_z terms are considered to be the thickness and hydraulic conductivity of the surface layer controlling the vertical exchange, data for which are specified at the same resolution as that of the DEM, hence their ij superscripts. Rather than incorporate this equation into the implicit finite-difference groundwater solution, to reduce computational cost, this flux is calculated explicitly at the start of a groundwater model time-step, t^0; flood infiltration or groundwater discharge is specified as an aquifer source or sink term, and the solution for the groundwater head at the end of the time-step, t^1, is calculated. The surface water level at time t^0 is adjusted, and then the surface water model is run to t^1, using its multiple, much shorter adaptive time-steps.

2.4 Model structure

2.4.1 Groundwater model structure

The ZOOMQ3D groundwater flow model of Oxford, the boundary of which is shown in Figure 1, was developed by Macdonald et al. (2018b). The extent of the model is limited to the floodplain alluvial sands and gravels and does not include the alluvium above, as its very low hydraulic conductivity contributes little to lateral groundwater flow, or higher terrace gravels, as these are not generally in direct hydraulic contact with the floodplain sediments. The floodplain sand and gravel aquifer was modelled as a single, homogeneous layer as there was insufficient spatial information that could be used to sub-divide it. The model mesh is regular horizontally and composed of 25 m square cells. The top and base of the sand and gravel layer were obtained from the 3D geological model developed by Newell (2008).

Rainfall recharge to the model was calculated on a daily basis using a soil moisture balance model (Mansour & Hughes, 2004). Boundary conditions at the edge of the model are no-flow, except at three locations where the flows are specified. This includes two sections across narrow widths of the floodplain, perpendicular to the course of the Rivers Thames and Cherwell where they enter the model domain, with modelled boundary groundwater inflows of 420 and 90 $m^3\ d^{-1}$, respectively. These are calculated using Darcy's Law with groundwater hydraulic gradients parallel to the river taken from hand-contoured groundwater levels from May 2007, aquifer thicknesses from the 3D geological model, and a hydraulic conductivity of 1000 $m\ d^{-1}$. The third location is a section adjacent to the higher terrace gravels underlying the city centre (see Figure 1; Macdonald et al., 2018b), with the total boundary flow equal to the modelled long-term average groundwater recharge over the higher terrace gravels in this area, distributed evenly over the relevant boundary nodes in space and time.

All rivers shown in Figure 1 are included in the model. The river node widths were averages for each river and stream within the model domain estimated from topographic maps, and the river bed thickness was a constant 1 m for all nodes. A single river bed hydraulic conductivity was assigned to each of two categories of water course, the main River Thames and other water courses; these were calibration parameters. These two categories were created as the River Thames is effluent over much of the model domain (Macdonald et al., 2018a), , due to the influence of locks, which is likely to result in lower river bed conductivities (Naganna et al., 2017), while the other water courses are mainly gaining. Observed river levels were imposed on model river nodes, using data from digital loggers at 22 monitoring stations within the Oxford floodplain (Figure 1); river levels between these locations were linearly interpolated.

2.4.2 Flood inundation model structure

The extent of the flood inundation model constructed for the Port Meadow area is shown in Figure 2. The features that form the boundaries are: to the west, the eastern edge of the channel of the River Thames; to the north, an urban area and road; to the east, a railway embankment; and to the south, the high ground associated with a historical waste dump. The ZOOMQ3D model domain within the linked model extends well beyond the boundary of the LISFLOOD-FP model, to areas of the floodplain that are also inundated with flood waters. The assumption is made that the exclusion of the interaction between fluvial flood waters and the floodplain aquifer in these areas does not influence fluxes within the LISFLOOD-FP model boundary.

The ground surface was defined using a 5 m DEM derived by averaging the 2 m DEM. This aggregation of the DEM was necessary to reduce run times to practicable lengths for the model sensitivity testing. Time-variant water levels were imposed on the River Thames channel cells in the west for the period 17 July to 30 September 2007, and updated every 15 minutes throughout the simulated period. These water levels were consistent with those specified at the corresponding river nodes of the groundwater model, and calculated using linear interpolation between measurements from monitoring stations S1 and S2 (Figure 2a). The River Thames was the only significant source and sink of water, to and from the flood inundation model domain.

The thickness of the floodplain alluvium was taken from the 3D geological model (Newell, 2008), interpolated onto a 5 m grid (Figure 2c), and used to define the surface layer. Flood water-groundwater exchange fluxes were calculated at the beginning of each 15 minute groundwater model time-step, and applied uniformly over the multiple, shorter adaptive time-steps of the flood inundation model. Spatial variability in the vertical hydraulic conductivity is anticipated (Hester et al., 2016), however it was not possible to map this within the study, and a single value was used across the model domain. It is anticipated that zones of relatively high surface layer hydraulic conductivity may exist and these could be significant for flood water recession; the sensitivity analysis on this parameter helped assess the significance.

2.5 Model application

The simulation of the 76-day period from 17 July 2007 using the linked model took 13 hours to run on a PC; the code does not yet take advantage of parallel computing. This limited possible approaches to exploring the model parameter space. Consequently, a two-stage approach was taken to calibrating the linked model. In stage one, the unlinked groundwater model was calibrated with a Monte Carlo process using monthly data over a period of eight years. In stage two, the parameters identified through the unlinked groundwater model calibration provided the starting point for further exploration of the linked model parameter space using the July 2007 flood event, and a sensitivity analysis focussing on the surface layer hydraulic conductivity.

2.5.1 Groundwater model calibration

The groundwater flow model of the Oxford floodplain sands and gravels was originally developed to provide a tool to investigate groundwater flow patterns and groundwater level responses to extreme river flows, in the context of groundwater flooding at a location 2 km downstream of the Port Meadow study area (Macdonald et al., 2018b). As a result, less attention had been paid to the fit of the model to the observations across Port Meadow. The calibration of the unlinked groundwater model was performed using an ensemble of 1000 simulations, varying the sand and gravel aquifer hydraulic conductivity and specific storage, and the river bed hydraulic conductivity. Each run simulated the period January 2004 to December 2012, and was evaluated against observations from January 2006 onwards to remove the effect of potentially biased initial conditions. This period was chosen to be long enough to cover a range of wet and dry years and to maximise the availability of groundwater level time-series data. The time-varying stage of the River Thames and other water courses were imposed on the model river nodes by interpolating monthly median values of observed levels at monitoring points along the channels. The model was run using mean monthly recharge rates and four time-steps per month. Parameter values were sampled uniformly from the ranges specified in Table 1, which also lists the justifications for these ranges; these include references to field studies within the Oxford floodplain (see Section 2.1) and the River Thames Basin, as well as relevant peer-reviewed literature. The calibration focussed on piezometers P1, P2 and P3 for which the root mean square errors (RMSE) between the simulated and observed monthly median groundwater levels were calculated. The groundwater model took approximately 15 minutes to run.

2.5.2 Model application to the July 2007 flood event

The linked groundwater and flood inundation model system was used to investigate river-floodplain-aquifer interaction and the implications for flood duration. The model runs simulated the period 17 July to 30 September 2007 using a 15 minute time-step for the groundwater model; the flood inundation model uses much shorter adaptive time-steps, as determined by the stability condition of its explicit scheme (see Bates et al., 2010).

The linked model parameter space was explored based on a combination of the results of the unlinked groundwater model calibration (Section 2.5.1), and the parameter ranges specified in Table 1 for the Manning's roughness coefficient and surface layer hydraulic conductivity. The calibrated groundwater model parameters were varied as follows: 50% higher and lower aquifer hydraulic conductivity; specific storage an order of magnitude higher and lower; and 50% higher and lower river bed hydraulic conductivity. The performance of the linked model was assessed against: the aerial photography of the flooding; groundwater levels recorded in piezometers P1-3; and the surface water level recorded at A1.

Two sets of model parameters were then used to examine the persistence of flooding on the floodplain, and the related dynamics of flood infiltration and groundwater discharge through the alluvium surface layer. In addition, simulations were performed in which the groundwater level was artificially held at a low level (10 cm above the base of the gravel aquifer) to assess flood persistence under conditions where shallow groundwater has no influence, as infiltration of flood water through the alluvium could occur at the free-drainage rate.

Table 1 Parameters of the linked model system and the range within which they were varied in the calibration process and sensitivity analysis

Model component	Model parameter	Range used in calibration	Units	Justification
Groundwater	Aquifer hydraulic conductivity	50 - 1500	m d^{-1}	Encompasses the range stated by Dixon (2004) for the Oxford floodplain gravels (see Section 2.1)
	Aquifer specific storage	10^{-5} - 10^{-2}	m^{-1}	
	Bed hydraulic conductivity - River Thames	0.05 – 20	m d^{-1}	Encompasses values reported for River Thames in Younger et al. (1993) and Calver (2001), and in the review by Naganna et al. (2017)
	Bed hydraulic conductivity - other water courses	0.5 – 200	m d^{-1}	An order of magnitude greater than River Thames as these water courses are generally gaining groundwater
Flood inundation	Manning's roughness coefficient	0.025 – 0.05	$m^{1/3}$ s^{-1}	Based on plausible range for grassland/pasture from Werner et al. (2005).
Surface layer	Surface layer vertical hydraulic conductivity	0.01 – 0.5	m d^{-1}	Encompasses the range of values measured in Oxford floodplain (see Section 2.1) and consistent with Doble et al. (2012)

3. Results

In the description of the observations and simulations of the July 2007 event, the period from the start of the event up to 5 August, at the end of the steep recession, is referred to as the main flood event; the subsequent period is referred to as the long recession (Figure 3).

3.1 Groundwater model calibration

The RMSE calculated between simulated and observed monthly median groundwater levels from boreholes P1, P2 and P3 for the 1000 Monte Carlo runs of the groundwater model are plotted in Figure 4. This shows that the only parameters within this model structure that have an identifiable control on model performance at the three piezometers are the hydraulic conductivities of the gravel aquifer and the bed of the River Thames. The average of the RMSEs for P1, P2 and P3 reduces as the aquifer hydraulic conductivity increases. Based on this, the value was set to 1000 m d^{-1} as this was the highest value within the range identified by Dixon (2004). The bed hydraulic conductivity of the River Thames was set to 10 m d^{-1} for the same reason, as this was the highest value identified for the River Thames in studies referenced by Calver (2001). Specific storage was not identifiable because the driving river level boundary conditions, which propagate rapidly through the very high-diffusivity aquifer, are the dominant control on the groundwater levels. A value of specific storage of 0.001 was initially chosen based on the results for south Oxford in Macdonald et al. (2018b). As the simulated groundwater levels at P1, P2 and P3 were not dependent on the hydraulic conductivity of the bed of the other rivers, this was set to the same value as that of the bed of the River Thames.

Using these parameters, the RMSE values over the 2006 to 2012 period for P1, P2 and P3 were 12, 15 and 18 cm respectively. Empirical cumulative distribution functions of simulated and observed mean monthly groundwater levels for the three piezometers are plotted in Figure 5. The simulated maxima over the period were 3 cm low at P1, and 16 and 19 cm high at P2 and P3, respectively, which was deemed acceptable considering that the unlinked groundwater model does not simulate fluxes between the aquifer and the floodplain. The lowest groundwater levels were simulated to be too high. We believe this is due to the inability of the single-layer groundwater model to represent groundwater flow west from Port Meadow under the River Thames to the Seacourt Stream, when levels are very low (Macdonald et al., 2018a). This could potentially occur if the thickness and hydraulic conductivity of the river bed sediments means the hydraulic connection between the river and aquifer is relatively poor at low river water levels, as has been reported in other studies (Doppler et al., 2007). However, groundwater levels observed in P1, P2 and P3 at the end of September 2007 are above 56.6 m asl, and as such are captured acceptably by the model.

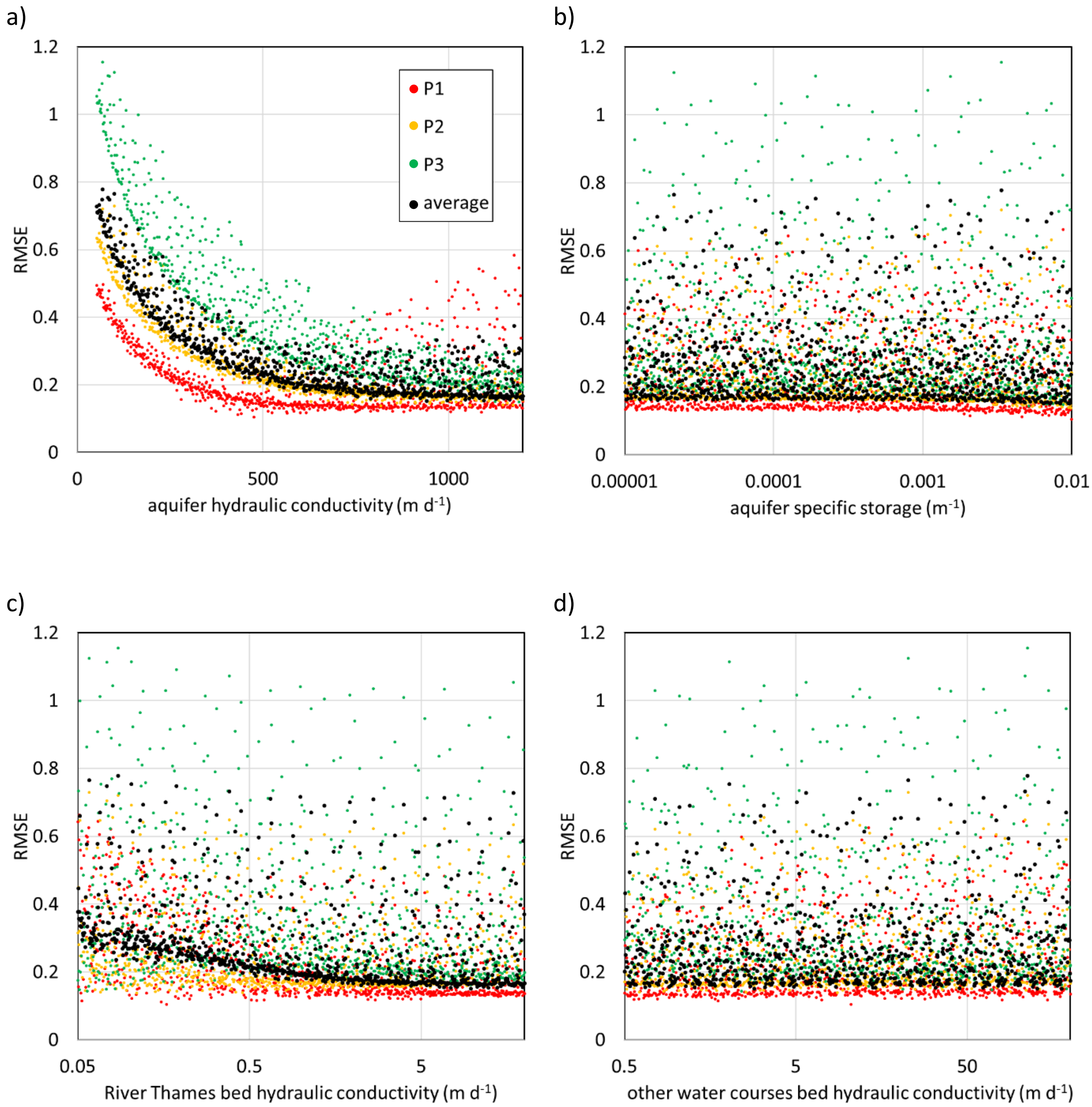


Figure 4 Individual and averaged root mean square error between simulated and observed monthly median groundwater levels from boreholes P1, P2 and P3 for the 1000 Monte Carlo runs of the ZOOMQ3D groundwater flow model for the Oxford gravel aquifer model over the period 2006 to 2012.

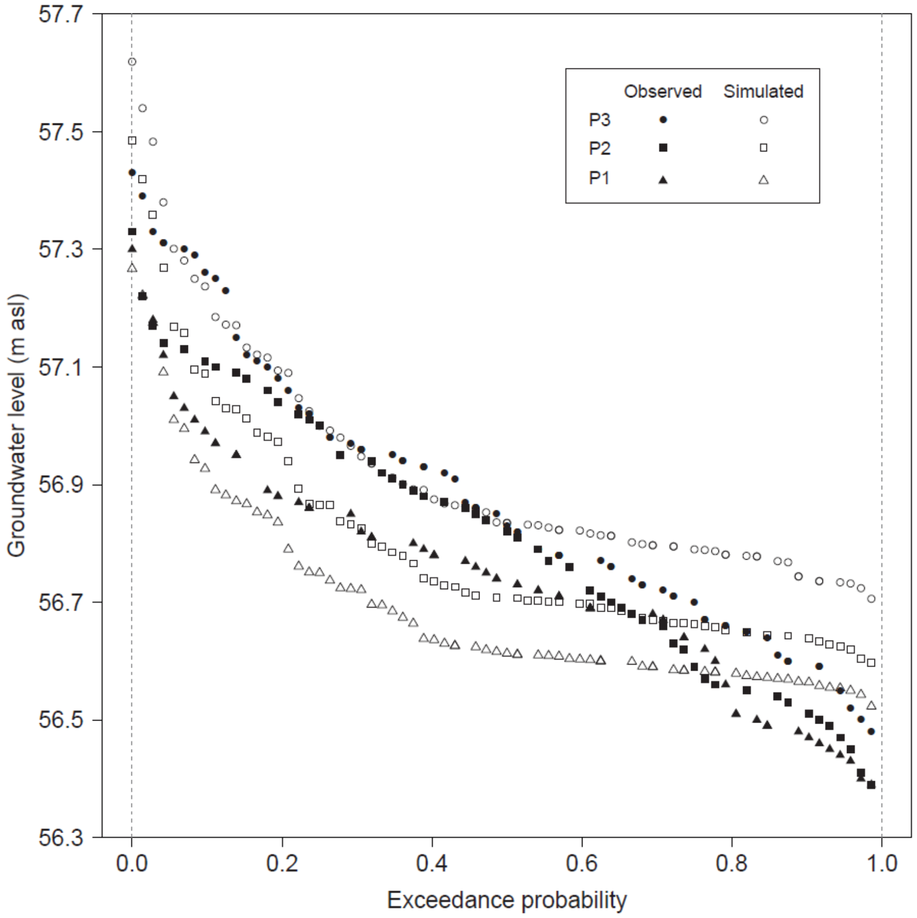


Figure 5 Cumulative distribution functions of median monthly groundwater levels over the period 2006 to 2012, from boreholes P1, P2 and P3: observed, and simulated using the ZOOMQ3D groundwater flow model for the Oxford gravel aquifer model with selected parameters identified through model calibration.

3.2 Simulation of the 2007 flood event

3.2.1 Comparison of observed and simulated water levels

Given the long run times of the linked model system, a Monte Carlo approach to assessing model performance was not possible. The linked model started with the groundwater model parameters identified from the model calibration in Section 3.1, and varied these parameters, along with Manning's roughness coefficient, *n*, and surface layer hydraulic conductivity, K_z, with the approach set out in Section 2.5.2. The model performance was assessed by varying the values of one parameter at a time and calculating the RMSE at P1, P2, P3 and A1. The results of this exploration of the parameter space are tabulated in Table S1 in Supplementary Information (SI).

Very little change was seen in RMSEs as a result of varying K_z (with *n* 0.025 $m^{1/3}$ s^{-1}). The RMSEs were in the ranges 7-8 cm, 7-10 cm, 15-19 cm and 5-6 cm for P1, P2, P3 and A1 respectively, with a K_z of 0.3 m d^{-1} giving the best average fit for the four monitoring sites, 8 cm (Figure 6a). A similar lack of variation in results occurred when the specific storage, *Ss*, was varied; here, two values of K_z were used, 0.03 and 0.3 m d^{-1}. RMSEs for P1, P2, P3 and A1 were in the ranges 6-7 cm, 8-10 cm, 15-19 cm and 5-6 cm, and 6-7 cm, 7-10 cm, 15-17 cm and 5 cm, for the K_z values of 0.03 and 0.3 m d^{-1}, respectively. The best average fit, 8 cm, was achieved with *Ss* of 0.001 m^{-1} and K_z of 0.3 m d^{-1} (Figure 6b).

The largest RMSE was obtained with the variation in the aquifer hydraulic conductivity, *K*. The model was run with values of *K* of 500, 1000 and 1350 m d^{-1}, with *Ss* 0.001 m^{-1}, K_z 0.3 m d^{-1} and n 0.025 $m^{1/3}$ s^{-}

[1]. RMSEs were in the ranges 7-16 cm, 7-23 cm, 13-39 cm and 5-8 cm for P1, P2, P3 and A1 respectively, however the RMSEs for the two higher values of K were very similar and at the lower end of the range of RMSEs for all sites (Figure 6c). The variations in river bed hydraulic conductivity and *n* made imperceptible differences to the RMSEs calculated for each monitoring site, and were not plotted, although are presented in Table S1.

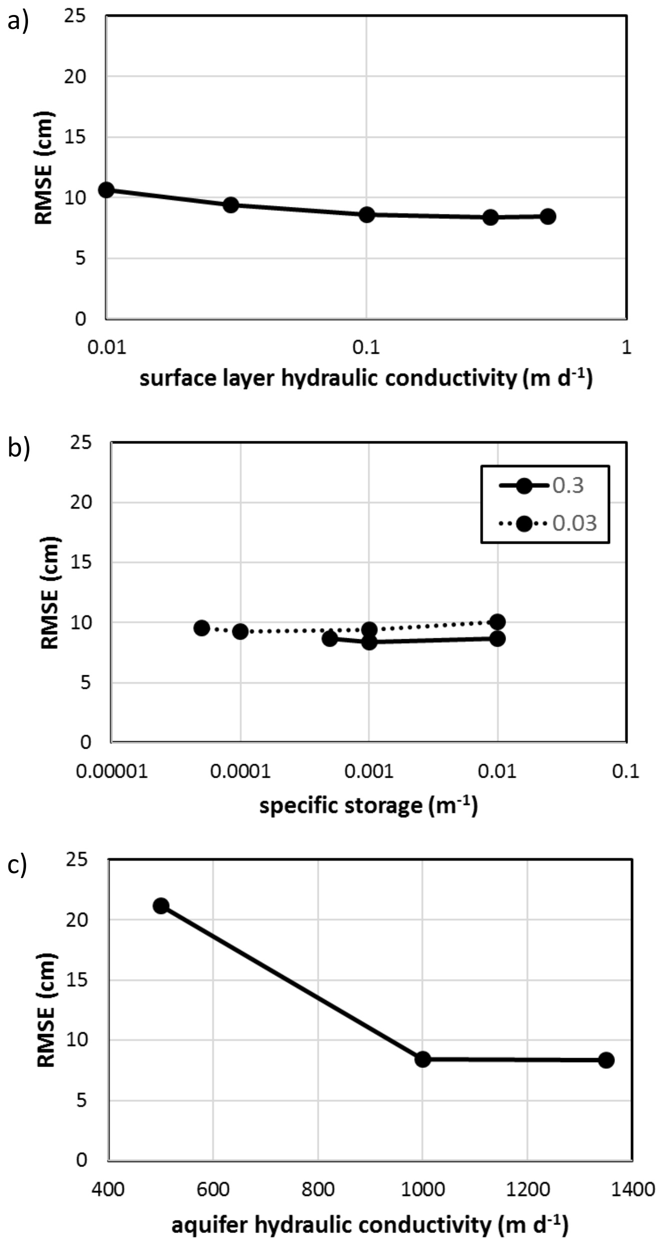


Figure 6 Average RMSEs for sites P1, P2, P3 and A1 for selected linked model parameter sets: a) K_z varied - K = 1000 m d^{-1}, Ss = 0.001 m^{-1}, Kr = 10 m d^{-1}, n = 0.025 m$^{1/3}$ s^{-1} ; b) Ss varied, with two values of K_z, 0.03 and 0.3 m d^{-1} - K = 1000 m d^{-1}, Kr = 10 m d^{-1}, n = 0.025 m$^{1/3}$ s^{-1} ; c) K varied - K_z = 0.3 m d^{-1}, Ss = 0.001 m^{-1}, Kr = 10 m d^{-1}, n = 0.025 m$^{1/3}$ s^{-1} .

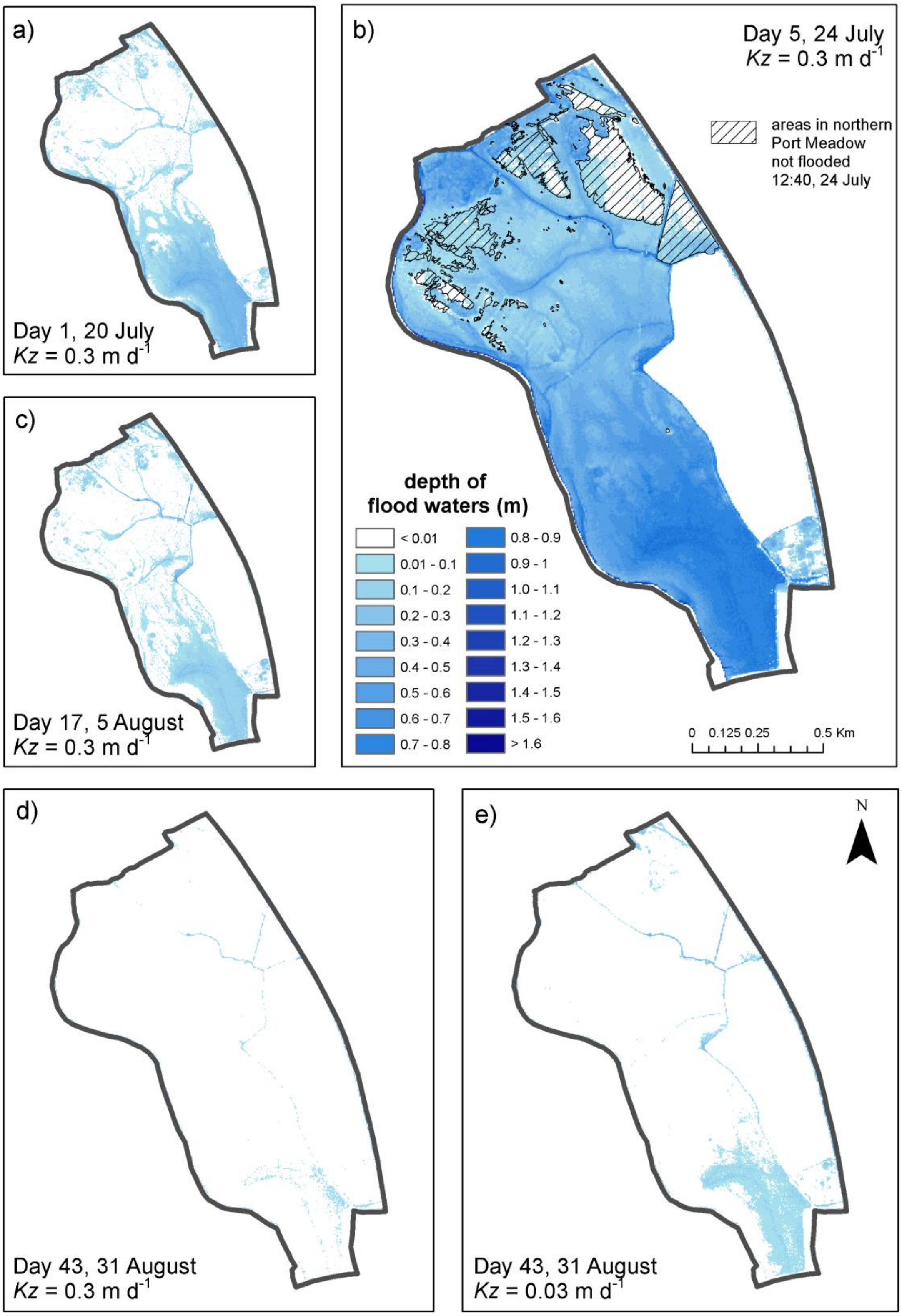


Figure 7 Simulated flood depth (m) on Port Meadow for specified days from the start of the flood event, from model runs using baseline parameters with: a) to d) K_z 0.3 m d^{-1}; and e) K_z 0.03 m d^{-1}. For b) includes areas in northern Port Meadow that were mapped as non-flooded from BGS aerial photographs.

3.2.2 Flood water extent

Based on the model parameter exploration presented in Section 3.2.1, the aquifer parameters identified in the groundwater model calibration in Section 3.1 were again used, along with K_z 0.3 m d^{-1} and n 0.025, to assess the model performance in simulating flood water extent. The output was compared with a model run using K_z 0.03 m d^{-1} to assess the influence of K_z on flood water persistence and vertical fluxes between aquifer and floodplain. Using the former of the two sets of parameters, the peak simulated flood water levels occurred at 14:30 on 25 July 2007, Day 6 of the flood event, when the observed depth at P1 was 1.05 m. The extent and depth of flooding across Port Meadow is shown in Figure 7a-d, for four times: 1, 5, 17 and 43 days after the start of the flood event. The model on Day 5 (Figure 7b) compares well to the aerial photography at this time, and reproduces the areas not inundated in the north, and the small mound in the centre of the meadow.

Figure 7a shows that at the end of 20 July, the first day of flooding, most of the southern end of the meadow is simulated to be inundated, with a maximum depth of water of 0.63 m. The simulated depth of water at P1 at this time is 0.37 m. By 5 August, Day 15 (Figure 7c), at the end of the steep recession from the main flood event, the southern end of the meadow is still flooded and there are other isolated areas of flooding. By 31 August, Day 43 (Figure 7d), only the drainage network and the southern end of the meadow are simulated to be under water. At this time, groundwater is below ground level at all but a few low-lying locations associated with drains (Figure 8). The effect of varying K_z on the results is only seen in the latter stages of the flood simulation. Decreasing the hydraulic conductivity of the alluvium ten-fold to 0.03 m d^{-1} increases the area of the floodplain inundated on Day 43 from 1.5%, or 5.78 ha, to 5.5%, or 21.86 ha (Figure 7e).

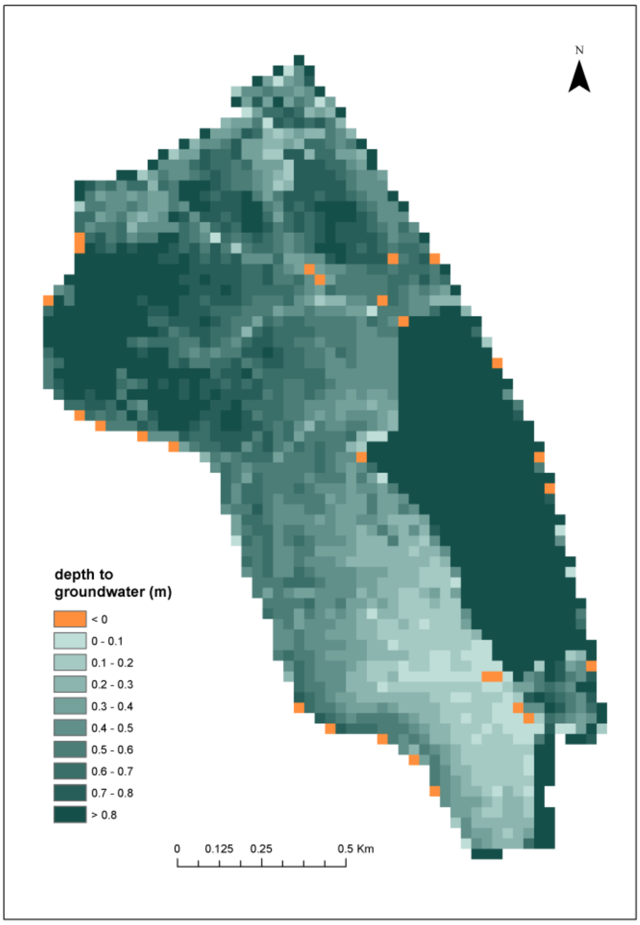


Figure 8 Depth to groundwater on 31 August 2007, Day 43 of the flood event, simulated by the baseline model using K_z = 0.03 m d^{-1}

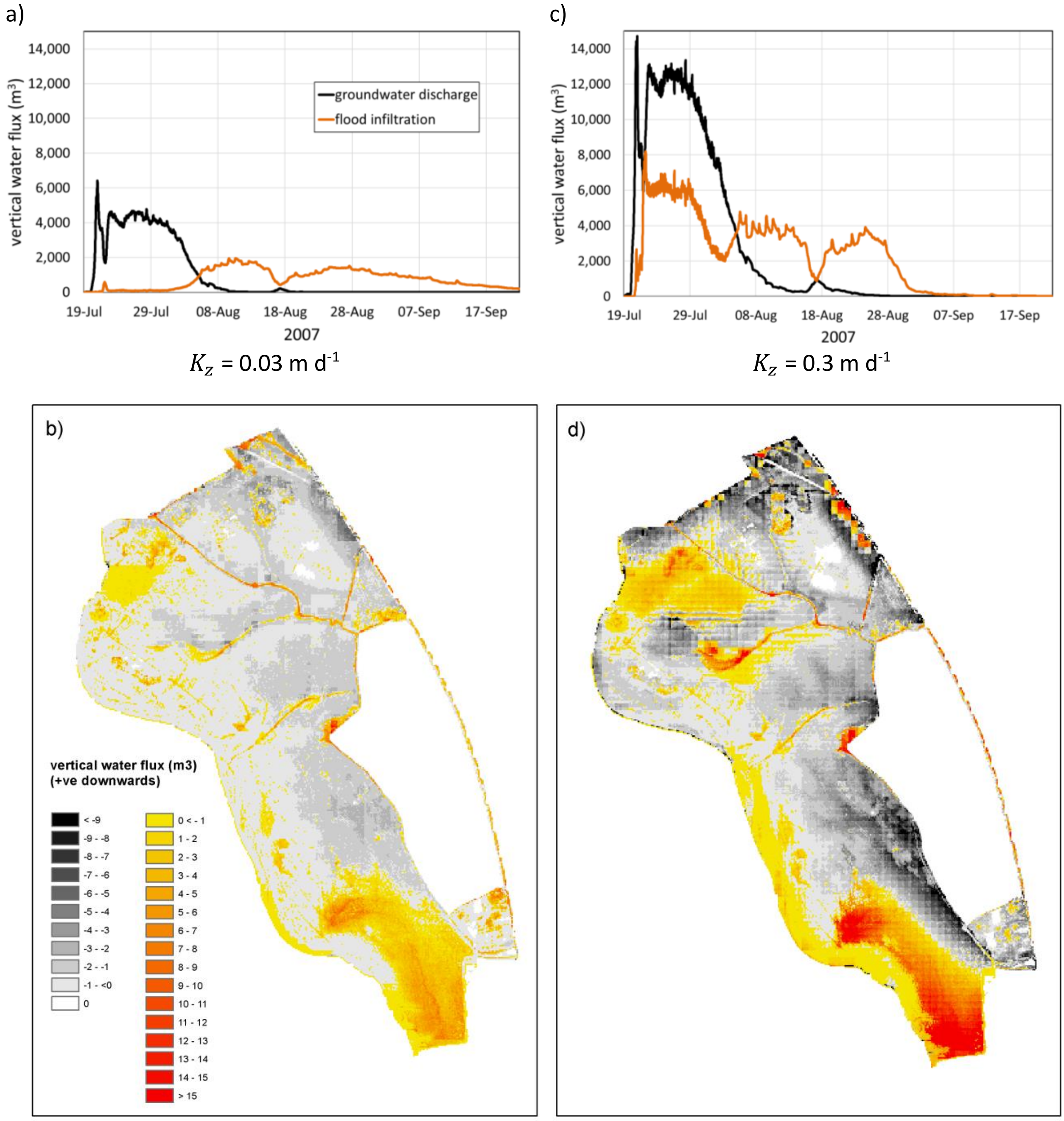


Figure 9 Simulated vertical fluxes from the baseline model runs using a), b) K_z = 0.03 m d^{-1}, and c), d) K_z = 0.3 m d^{-1}, as time-series of groundwater discharge and flood infiltration summed across the whole model domain, and as maps of the total net flux.

3.2.2 Groundwater-floodplain fluxes

The simulated total vertical flux of water between the aquifer and the floodplain is shown in Figure 9. In the simulations with values of K_z of 0.03 and 0.3 m d^{-1}, the total discharge from the aquifer to the floodplain over the period of the model run in both cases is greater than the total infiltration; 5.38 x 10^4 m^3 compared to 4.52 x 10^4 m^3 for K_z of 0.03 m d^{-1}, and 8.48 x 10^4 m^3 compared to 6.28 x 10^4 m^3 for K_z of 0.3 m d^{-1}. The majority of the discharge occurs within the main flood event in both runs. However, the pattern is different for infiltration: with the lower K_z it occurs almost totally within the long recession; while it occurs both during the main event and the long recession with the higher K_z. Maps of the total flux (Figure 9b, d) show that discharge occurs over most of the floodplain, although higher amounts occur in relatively low-lying areas away from the river channel. Infiltration is focused in the drains and depressions that may be associated with old stream channels, and also in the south of the floodplain where water that drains from the floodplain is trapped.

The linked model estimated flows onto the Port Meadow floodplain during the flood peak of over 200 m^3s^{-1}. This is similar to the peak flows in the River Thames in Oxford during the event, at the lock furthest downstream. In comparison, the total net groundwater flow through the river nodes of the groundwater model for the full 76-day period of the simulation, which is predominantly from the aquifer back into the river, was only 820 and 950 m^3, respectively for K_z of 0.03 and 0.3 m d^{-1}.

3.2.3 Response at monitoring sites

The sensitivity to variation in K_z of simulated groundwater and surface water levels and vertical fluxes at the three monitoring locations, P1/A1, P2 and P3, was investigated. The observed and simulated water levels are shown in Figure 10. The time-series of the vertical fluxes at these points are presented in Figure 11. First, results are described for each site in turn, and then for the simulations with the groundwater level held artificially low (the 'free-drainage' runs).

At P1, as is the case for the observations, the simulated surface water levels were above the groundwater levels for the majority of the overall event, for every value of K_z. The simulated surface water levels were the same for every K_z until the end of the main event, as the dominant control during this period is overbank flow. The shape of the simulated surface water hydrograph matches that of the observed hydrograph well, although during the main flood event simulated levels are approximately 5 cm too high. The hydraulic conductivity of the surface layer has a substantial effect on the length of the long recession. With K_z = 0.01 m d^{-1} water remained on the floodplain at P1 until the end of the simulation. The difference in the duration of flooding for K_z values of 0.3 and 0.03 m d^{-1} was 18 days, although with the lower 3 cm threshold of nuisance flooding defined by Moftakhari et al. (2018) the difference reduced to 10 days.

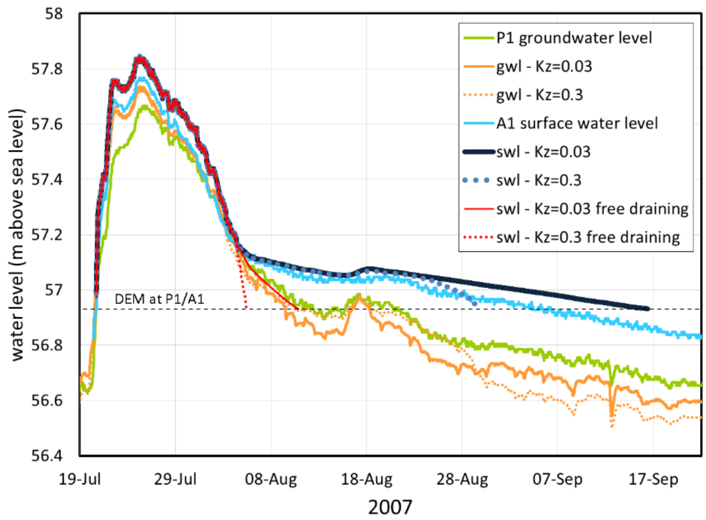
Differences between the groundwater level hydrographs during the main event, simulated using the range of K_z values, are small and of the order of 4 cm. The effect of the persistence of floodplain inundation on the groundwater hydrograph is apparent in the long recession. With more permeable alluvium (K_z = 0.3 m d^{-1}), the groundwater level is initially higher, due to increased infiltration, but then falls below the hydrograph for the lower hydraulic conductivity case (K_z = 0.03 m d^{-1}).

Figures 11a and b show the time-series of vertical fluxes at P1 from the simulations using K_z of 0.03 and 0.3 m d^{-1}, respectively. As with the observations, in both cases the simulated groundwater level is initially above the surface water level. This results in a short period of groundwater discharge to the floodplain. Subsequently, higher surface water levels drive infiltration until the floodplain dries. Infiltration increases with time as the groundwater level declines at a faster rate than the surface water level.

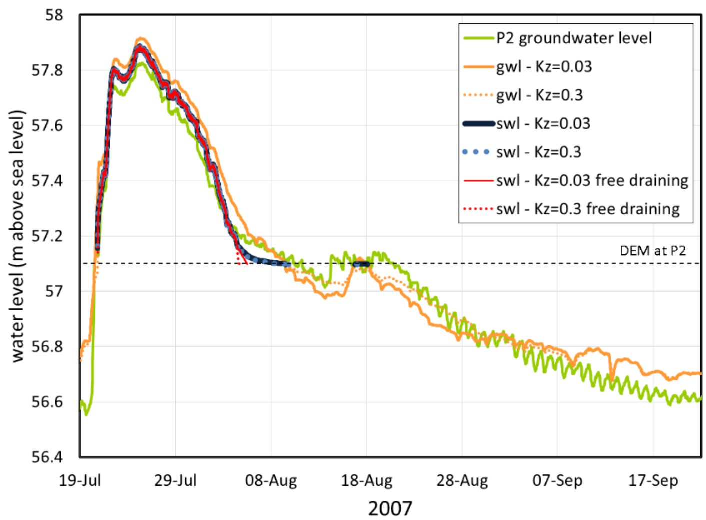
There are no surface water level observations at P2 but simulated groundwater levels compare well with the observations. With K_z set to 0.03 and 0.3 m d^{-1}, simulated peaks levels are only 9 and 6 cm higher than those observed, respectively. The gradient of the simulated recession is lower than the observed. However, the pattern of groundwater levels rising above and then falling below the ground surface in response to the mid-August rainfall was reproduced.

Groundwater levels are above surface water levels for the full period of each simulation at P2. The simulated groundwater level rises in response to the changing river stage. The rapid transfer of the pressure head through the high diffusivity aquifer resulted in a time lag between observed peak river level and simulated peak groundwater level at P2 of 1.5 and 1 hours, for K_z of 0.03 and 0.3 m d^{-1} respectively. The groundwater level rises higher than the predicted floodplain surface water level, thus driving groundwater discharge (Figures 11c and d). The driving head difference with K_z 0.3 m d^{-1} is less than 1 cm, compared to 8 cm with K_z 0.03 m d^{-1}, but results in discharge rates that are four times higher. The simulated surface water levels are the same for each simulation, as at P1, as these are controlled by overbank flows until the river returns to within its banks. There is a separate period of flooding on the floodplain at P2 associated with the rise of the groundwater level above the ground surface in mid-August (Figures 10b and 11d); during this period the model showed no fluvial flood waters reaching this location.

a)



b)



c)

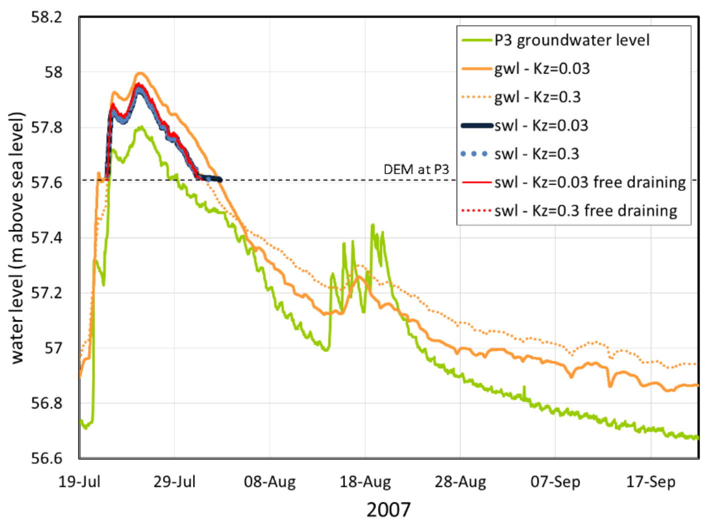


Figure 10 Observed and simulated water levels at a) P1, b) P2 and c) P3. Simulated water levels from baseline model and from free-draining version of baseline model, with K_z = 0.03 m d^{-1} and 0.3 m d^{-1}

Simulated groundwater levels at P3 do not compare as well as with observations as at other sites (Figure 10c), and the levels are different by approximately 20 cm. This difference is consistent over much of the hydrograph and it is possible, therefore, that at this site there was a mistake with the measurement of the datum. However, the relatively poor match of simulated and observed levels is consistent with the results of the calibrations runs of the groundwater model at this location. The recession of the simulated levels is shallower than that of the observations, and the groundwater level simulations do not replicate the flashy response to the rainfall in mid-August. As for the observations, simulated groundwater levels do not rise above ground surface in response to the mid-August rainfall.

Again, the simulated surface water levels are very similar for all the values of K_z applied, and in all cases are lower than the groundwater levels for the full period of the run. As at P2, all fluxes at this location are from the aquifer to the floodplain (Figures 11e and 11f) and with higher K_z values, the fluxes are substantially higher.

The results of simulations in which the groundwater level was held artificially just above the base of the aquifer, so that any infiltration occurred at the free-drainage rate, are also shown in Figure 10 for values of K_z of 0.3 and 0.03 m d^{-1}. At all three sites, and in both simulations, the surface water levels with and without fluctuating groundwater levels are the same. The influence of groundwater was seen once the river levels recess to within bank. The free draining model at P1 with K_z of 0.03 m d^{-1} has the surface water disappearing quickly after the main flood event, on 10 August 2007 (Figure 10a), 37 days before water disappears in the model with dynamic groundwater. For this site, with K_z of 0.3 m d^{-1}, the free-draining simulation results in the floodplain drying 5 days earlier than with K_z of 0.03 m d^{-1}.

At P2 and P3, the period that the simulated surface waters remain on the floodplain is not long when compared with P1. With the free-drainage model and K_z of 0.03 m d^{-1}, surface water disappears at P2 and P3 by 5 August and 31 July; that is 13 and 3 days, respectively, before that simulated by the model with dynamic groundwater. With K_z of 0.3 m d^{-1}, the difference in the timing of the disappearance of the surface water between the two models is only 1 day at P2, with no difference at P3.

4. Discussion

4.1 Limitations of the model system and transferability

In assessing the ability of the linked model system to simulate water levels in a flooded floodplain and its transferability to other settings, the model limitations need to be considered.

Groundwater recharge is calculated by the ZOODRM model (Mansour & Hughes, 2004) prior to running the ZOOMQ3D model. This approach introduces inconsistencies where soil water balances are used to calculate recharge in areas that are inundated with flood water. In addition, within the linked model system there are a number of time-variant processes that are not simulated during run time that may be particularly significant when investigating the persistence of flood waters on the floodplain. Rainfall input to areas already flooded is not simulated, nor is evaporation. In very shallow groundwater conditions, evapotranspiration directly from the saturated zone in areas where flood waters have abated may also be significant for the water balance, especially for summer flood events. In the application of the linked model system reported here, ignoring the rainfall input is not considered significant due to the low amounts during the period that flood waters occur on the floodplain. However, it is acknowledged that direct evaporation from flood waters could have resulted in a faster recession of the flood water level, potentially allowing a better fit between modelled and observed water levels with lower surface layer hydraulic conductivities.

The linked model system does not simulate flow within the unsaturated zone. The assumption is that in floodplain environments the total saturation of sediments occurs within a comparable timescale to flood inundation. In the case study floodplain this is particularly so due to the shallow range within which groundwater fluctuates, and the high diffusivities that result in rapid propagation of the flood wave from the river. In floodplain settings where the sediments are unconfined and less permeable, with a deeper unsaturated zone, the lack of explicit modelling of the unsaturated zone may be problematic, and further examination of the significance of the assumptions of the model in this context is required.

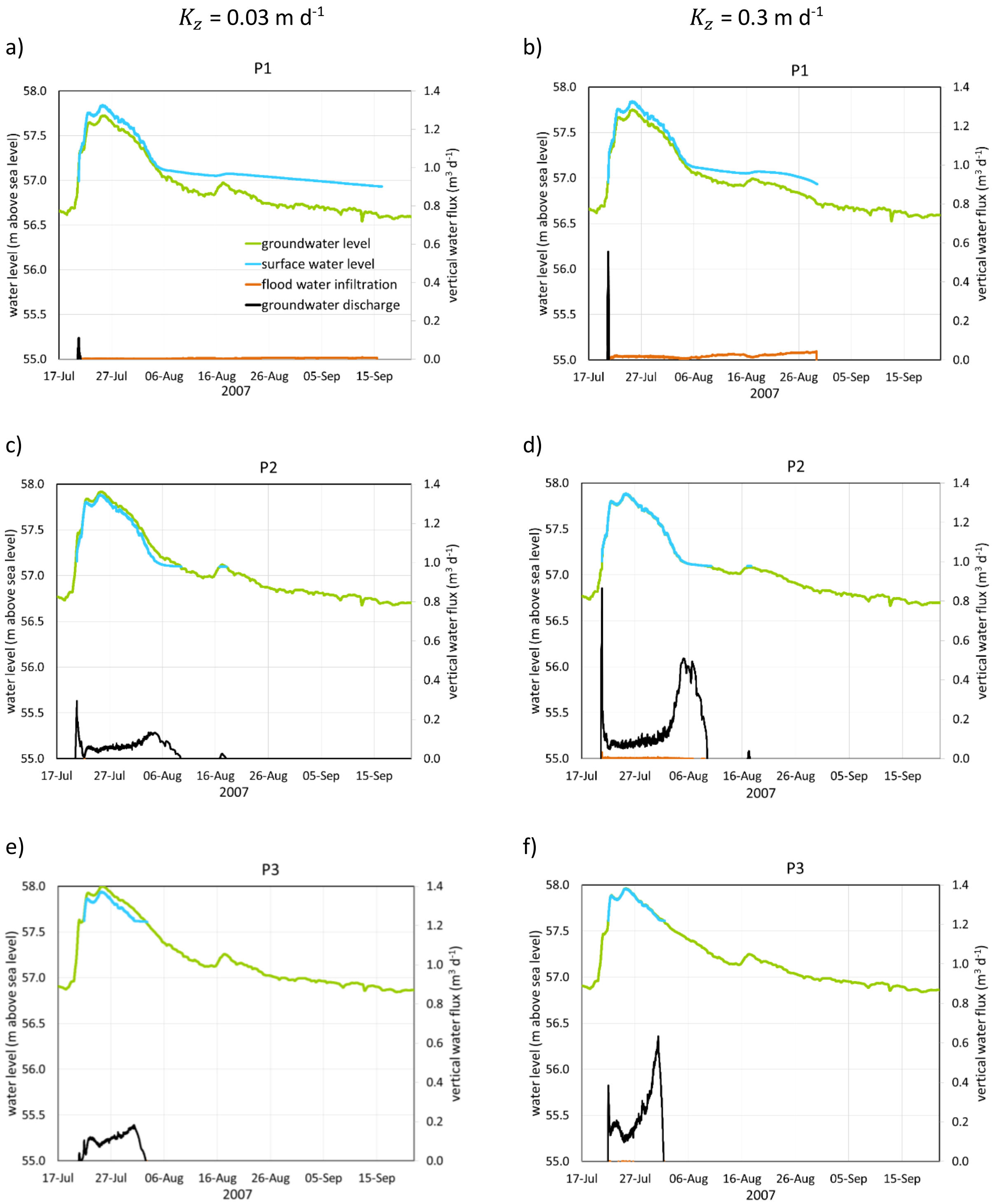


Figure 11 Simulations of water levels and vertical fluxes for P1 (a, b), P2 (c, d) and P3 (e, f) from the baseline model using K_z 0.03 m d^{-1} and 0.3 m d^{-1}. Vertical fluxes in m^3 d^{-1} over the 5 x 5 m cell containing the observation point.

Although time-varying river flows and levels can be simulated by ZOOMQ3D, in the current version of the code the translation of these onto the linked model DEM has to be prescribed *a priori* as a boundary condition. This has been possible in this study due to the high density of river level monitoring stations. However, for application to other areas where such data are not available, the code needs to be developed to transfer the river level simulated by ZOOMQ3D to LISFLOOD-FP during run-time.

The incorporation of groundwater-floodplain interaction in the finite-difference solution of the groundwater model using an explicit difference term (Wang and Anderson, 1982), where the flux depends on the groundwater and surface water levels at the start of the groundwater time-step, and the differing lengths of the ZOOMQ3D and LISFLOOD-FP time-steps, may introduce some oscillations into these flux time-series. This makes it possible for there to be circumstances when groundwater discharge and floodplain infiltration occur during alternate time steps, as can be seen in Figure 11d, though here the floodplain recharge fluxes are negligible.

The linked model system incorporates the assumptions of the component codes. In particular for LISFLOOD-FP there is the requirement that flows are slow and shallow, and gradients of the local free water level surface are small (Bates et al., 2010).

Crucially, a limitation in the linked model system is that it does not allow the effects of subsurface drainage infrastructure to be simulated. This would likely increase the rate at which retained water is removed from the floodplain. This infrastructure also provides pathways for the movement of flood waters during the onset of flooding. The area of the Oxford floodplain used in this case study is without subsurface drainage and has provided a good test case for the newly developed model system, however, further development of the model system is required to enable this subsurface route for water flow to be incorporated prior to the transfer of the model to urban settings. Other challenges in urban areas include: the mapping of surface layer thickness, for example due to the impact of the construction of building foundations; and the hydraulic conductivity attribution of the surface layer given the mix of spatially complex geology and anthropogenic changes to the land surface. There would also be benefit in using parallel processing to substantially reduce run times, especially in urban areas with complex topography where high resolution DEMs are needed.

However, as the case study described here has shown, the model can be successfully applied to peri-urban floodplains where there is no subsurface infrastructure but there has been some anthropogenic modification of the floodplain topography. In addition, the model system could be used to explore issues such as: groundwater recharge from flood inundation, a mechanism that has been identified in some settings as a key component of aquifer water balances; spatiotemporal variation in the ingress of polluted surface waters to floodplain aquifers; and the efficacy of some natural flood management measures.

4.2 Fit of model simulations to observations

The linked model system has enabled the dynamics of the July 2007 flood event to be reproduced and provided a good fit to the observed groundwater and surface water level time-series. RMSEs of approximately 7, 7, 15 and 5 cm were achieved at sites P1, P2, P3 and A1, respectively. Within the model parameter space explored, the range of the RMSEs at these sites was limited to a few cm, apart from when the lowest value of aquifer hydraulic conductivity was applied. The combination of the general confined nature of the gravel aquifer, and associated low storage coefficient, with its high hydraulic conductivity produces very high aquifer diffusivity. As a result, groundwater levels across the aquifer, both those observed and modelled, closely followed the rapid rise in levels in the River Thames during the July 2007 flood event. The overriding control that river levels have on groundwater levels was the reason for the limited range of variability in modelled water levels, leading to the equifinality within the model parameter space.

The spatial extent of the flooding simulated by the linked model matched well that derived from aerial photography. However, it is recognised that the confidence in the results produced by the model would have been greater if more observations of surface water level on the floodplain were available. It is also recognised that on flat floodplains, relatively small differences in level can be highly significant. This is particularly so when quantifying the duration of flooding at shallow depths of less than a few tens of centimetres. This highlights the need for accurately measured datums and water levels. Great efforts were made in this study to produce accurate measurements, but the potential for the accumulation of errors in the observed data and the DEM must be recognised when assessing the fits between simulated and observed water levels.

Floodplain sediments can be highly heterogeneous, with associated variation in aquifer parameters. Due to a lack of data at the necessary spatial resolution, in developing the Oxford floodplain model, significant simplications had to be made both for the aquifer parameters of the sands and gravels aquifer and the surface layer alluvium. However, given these uncertainties, the study still provides important insights into the functioning of floodplains during flood events.

4.3 Floodplain water fluxes

As the modelled groundwater levels were generally higher than the levels of the overbanked river water on the meadow, although only by a few cm (Figure 11), groundwater discharge was simulated across much of the floodplain during the main flood event (Figure 9). It is thought that this occurs because groundwater levels are controlled by distal river levels, and overbanked flood waters, once beyond the river levee, are lower than these over much of the floodplain. The simulated groundwater discharge was not substantial; the total net volume discharging over the main flood event was equivalent to average depths of water of 35 and 55 mm, for values of K_z of 0.03 and 0.3 m d^{-1}, respectively. However, the uncertainty in modelled groundwater and surface water levels needs to be taken into account in this context and could mean the reversal of the direction of vertical fluxes during the main flood event, although these fluxes are still likely to be small.

With K_z of 0.03 m d^{-1}, the majority of flood infiltration occurred during the long recession and, if averaged over the whole of the Port Meadow floodplain, was equivalent to a depth of water of 29 mm. This is approximately 25% of the long-term average annual recharge estimated using a soil moisture balance model (Macdonald et al., 2018a). The temporal distribution of flood infiltration from the simulations with K_z of 0.3 m d^{-1} was different to that with K_z of 0.03 m d^{-1}, with the more permeable alluvial layer allowing infiltration to occur during the main event. Net infiltration during the long recession with K_z of 0.3 m d^{-1} was equivalent to a depth across the floodplain of 41 mm. However, in comparison to the modelled groundwater discharge that occurred during the flood event, modelled flood infiltration was not widespread, occurring only in areas that are relatively low-lying. The larger water level gradients in these low-lying areas mean uncertainties in modelled water levels may not have been as significant for flux calculations as they were during the main flood event. Macdonald et al. (2018a) identified the infiltrating flood water as providing a significant input of nitrate to the floodplain aquifer; the results of the current study indicate that these high nitrate waters do not infiltrate uniformly across the floodplain but are spatially focussed.

The reconnection of rivers with their floodplains and the construction of levees to retain water upstream of vulnerable urban centres, known as offline storage, is an approach that is used to manage downstream flood risk by lowering flood peaks (Lane, 2017). The consequence of the high groundwater levels during the main flood event at Port Meadow was to reduce the storage height of the floodplain but only by a few centimetres. This compares to the equivalent average storage height of the floodplain at the peak of the July 2007 flood, which was 48 cm. Therefore in this case study, groundwater discharge to the floodplain does not compromise the storage of the floodplain significantly but under other circumstances, groundwater conditions may need to be taken into account in the design of offline storage schemes.

4.4 Flood duration

The study highlights the importance of modifications of the surface topography as a factor in flood duration. In areas to which water drains and is trapped by surrounding high ground, modelling has shown that the groundwater level and the hydraulic conductivity of the underlying sediments are key in controlling the duration that flood waters remain on the ground surface. The slow decline in water level lengthened the period of flooding by days or weeks. Although depths of flood water may be very shallow, other research (Moftakhari et al., 2018) has shown that in urban areas as little as 3 cm of water can cause significant nuisance. The modelling here showed that an order of magnitude difference in the hydraulic conductivity of the floodplain surface sediments resulted in a difference in the period of flooding to the 3 cm threshold of 10 days. In the theoretical case when shallow groundwater was artificially held at a low level, with K_z of 0.03 m d^{-1}, the period of flooding reduced by 32 days compared to the simulation in which groundwater levels were not fixed.

Given the heterogeneity of floodplain sediments, both vertically and laterally, challenges exist in identifying areas where flood waters are likely to persist due to underlying ground conditions. However, with the potential for a greater frequency of flooding, and groundwater levels that are maintained at shallower depths in lowland floodplains by raised levels in the associated rivers, nuisance flooding may become more prevalent. The economic drivers may be strong enough to require this form of flooding to be considered routinely in the modelling of flood hazards. If so, it will be necessary to include processes in standard flood routing models that allow the vertical fluxes at the floodplain to be incorporated.

5. Conclusions

Through this study a groundwater flow model has been linked to a widely-applied floodplain inundation model to investigate the dynamics of interacting groundwater and flood water. This model system was developed recognising the lack of flood inundation models that have a physics-based representation of groundwater, that are able to simulate over periods of weeks to months with sufficiently fast run times to undertake sensitivity analyses. While recognising the limitations of the linked model system and the uncertainties in the simulations, the model application to a lowland peri-urban floodplain was considered successful. The model set-up, and the hypotheses that this represents in terms of hydrological processes, worked well for the study area when the model was constrained using observed hydraulic conductivities and tested against observed flood water and groundwater levels.

The modelling has shown that the fluxes of water between aquifer and floodplain during a flood event can be highly spatially and temporally variable and that the effective infiltration of flood waters to the aquifer occurred in the long period of flood recession and was focussed in relatively low-lying areas. In locations where natural levees or modifications to the floodplain topography retain flood waters on the floodplain, the modelling highlighted that the duration of flooding is highly sensitive to the hydraulic conductivity of surface geology and the presence of shallow groundwater. This can result in flood waters remaining for weeks at depths that can have significant socio-economic impact.

This research highlights that there is an imperative for flood risk managers to collect evidence to assess the scale of the current and future impact of this type of nuisance flooding and to use modelling tools, such as developed in this study, to examine measures to mitigate where there is significant risk.

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SUPPLEMENTARY INFORMATION

Aquifer hydraulic conductivity (m d^{-1})	Aquifer specific storage (m^{-1})	River Thames bed hydraulic conductivity (m d^{-1})	Surface layer hydraulic conductivity (m d^{-1})	Manning's roughness coefficient ($m^{1/3}$ s^{-1})	P1 RMSE (cm)	P2 RMSE (cm)	P3 RMSE (cm)	A1 RMSE (cm)	Average RMSE for P1, P2, P3, A1 (cm)
1000	0.001	10	0.01	0.025	8	10	19	6	11
1000	0.001	10	0.03	0.025	7	9	17	5	9
1000	0.001	10	0.1	0.025	7	7	16	5	9
1000	0.001	10	0.3	0.025	7	7	15	5	8
1000	0.001	10	0.5	0.025	7	7	15	5	8
1000	0.00005	10	0.03	0.025	7	10	15	5	10
1000	0.0001	10	0.03	0.025	7	8	16	6	10
1000	0.001	10	0.03	0.025	7	9	17	5	9
1000	0.01	10	0.03	0.025	6	10	19	6	10
1000	0.0005	10	0.3	0.025	7	8	15	5	9
1000	0.001	10	0.3	0.025	7	7	15	5	8
1000	0.01	10	0.3	0.025	6	7	17	5	9
500	0.001	10	0.3	0.025	16	23	39	8	21
1000	0.001	10	0.3	0.025	7	7	15	5	8
1350	0.001	10	0.3	0.025	9	7	13	5	8
1000	0.001	5	0.3	0.025	7	7	17	5	9
1000	0.001	10	0.3	0.025	7	7	15	5	8
1000	0.001	20	0.3	0.025	7	7	15	5	8
1000	0.001	10	0.3	0.025	7	7	15	5	8
1000	0.001	10	0.3	0.05	7	7	15	5	8

Table S1 Linked model simulations for the July 2007 flood event: model parameters and RMSE between observed and simulated water levels at P1, P2, P3 and A1. Note, some model run results are repeated for ease of comparison.

Chapter 6: Conclusions

6 Conclusions

The broad aim of this thesis has been to improve the understanding of the interaction between surface waters and groundwaters in permeable lowland floodplains, with a particular focus on water and nutrient fluxes, and flooding. The conclusions from the research are presented here in relation to the four specific aims set out in Section 1.2.1. Although the research in each of Chapters 2 to 5 map on to individual aims, in assessing how the results of each chapter have addressed these aims, insights are drawn in from across the full range of research undertaken. Section 6.1 includes the implications for environmental management; areas for further work are presented in Section 6.2.

6.1 Addressing aims, and implications for environmental management

1) Improved understanding of the impact of river management structures on water and nitrate exchange in urbanised floodplain aquifers

Evidence was provided that highlighted that engineered river management structures are common on the river networks of England and Wales, and in other countries globally. Evidence from the study area showed that these structures can have a substantial effect on both river levels, and groundwater levels in adjacent permeable floodplain aquifers. Raised groundwater levels are associated with increased aquifer storage, which may be important where these are used as sources of groundwater supplies. The study also showed that in areas with dense networks of water courses, the combination of rivers with raised water levels and natural or constructed bypass channels that are generally receiving groundwater, can result in complex groundwater flow patterns. It was proposed that these complex flow patterns can result in water residing in the subsurface for longer than might have been the case with a more natural systems, with possible implications for groundwater quality.

An observation from the Oxford floodplain was that sites upstream of locks, where groundwater levels are particularly shallow, are often the locations of floodplain meadows. In Oxford these sites have national and European-level protection, having rich and diverse flora and fauna. The plant species are highly dependent on soil moisture conditions and any change to these conditions can have significant implications for species composition. Therefore any change to the design or operation of structures in similar settings could have implications for the status of these types of ecosystem.

The fluxes of water and nitrate were the focus of Chapter 2. The Oxford floodplain aquifer is isolated from regional aquifers as it is underlain almost completely by very low permeability bedrock. As a result, high nitrate groundwater does not flow laterally from outside of the floodplain. It is notable that in the River Thames catchment, extensive areas of alluvial gravels are generally underlain by mudstone bedrock as these form wide, shallow, low-gradient zones in which alluvial sediments most readily accumulate (Section 1.3, Figure 3). The hydrogeological setting of Oxford is therefore not atypical. In the study area the input of high nitrate concentration water to the floodplain aquifer is a combination of local sources, primarily urban, and water from the River Thames. A conceptual model was proposed focussing on the latter of these two inputs. Two mechanisms were identified for the recharge of the floodplain aquifer by this high nitrate water: raised river levels, resulting from river management structures, causing a flux of water from river to aquifer through the river bed; and overbanking of river water infiltrating through the floodplain surface during periods of flooding. Simple models used evidence from measurements of groundwater chemistry to show that there must be substantial nitrate removal occurring in the river bed sediments, and in the floodplain more diffusely, as groundwater nitrate concentrations measured were generally very low. Groundwater chemistry data at three depths were available at six sites with nested boreholes (see Chapter 3) but did not show any vertical variation in nitrate concentration although stratification in redox-reactivity might be expected (Kolbe et al., 2018). (N.B., in Chapter 2 reference is made to denitrification in the hyporheic zone. It is recognised that although denitrification is likely to be the dominant process, other processes might have been contributing to the lowering of nitrate concentration in groundwater, and therefore the more generic term 'nitrate removal' should have been used. In addition, as there is debate within the peer-reviewed literature as to the definition of the hyporheic zone, it is recognised that it would have been more appropriate to have referred to 'river bed sediments' as the location for much of the nitrate removal.)

The insights from research reported in Chapter 5 raised some issues with the calculations presented in Chapter 2. The calibration of the groundwater flow component of the model presented in Chapter 5 indicated that the hydraulic conductivity of the bed of the River Thames may be substantially higher than that used in calculations in Chapter 2; the river bed hydraulic conductivity used in the baseline model in Chapter 5 was 10 m d^{-1}, and in Chapter 2 was 1 m d^{-1}. This would have increased the estimate of the amount of nitrate attenuation occurring within the river bed sediments, calculated in Chapter 2 as 1×10^3 kg/a/km, by an order of magnitude. The modelling undertaken in Chapter 5 also improved the understanding of water fluxes during flood events. The modelling indicated that the simple approach taken to estimate groundwater recharge in Chapter 2 may not be valid, i.e. equating recharge with the average volume of the unsaturated zone in winter months. The modelling in Chapter 5 showed groundwater discharge, rather than recharge, occurring over most of the floodplain during the main period of the July 2007 flood event. In this modelling study, net groundwater recharge was shown to occur during the latter period of the flood and was of a similar order to that produced by the simple model in Chapter 2; the modelled estimate of recharge resulting from a single event, when averaged over the whole floodplain, was 52 mm in Chapter 2, compared with the equivalent in Chapter 5 of 29 and 41 mm, using surface layer hydraulic conductivities of 0.03 and 0.3 m d^{-1}, respectively. However, the recharge from the linked model system in Chapter 5 was focussed primarily in the south of Port Meadow. Groundwater from boreholes in this area of the floodplain (PTM27 and PTM30 in Table 3 in Chapter 3) show a greater range in nitrate concentrations than others in the study, indicating a higher influx of nitrate from fluvial flood waters. However, the median values of groundwater nitrate concentrations are still very low in these boreholes, showing almost complete nitrate removal is occurring.

These improvements to the conceptual model do not change the conclusions of this aspect of the thesis, that the flux of nitrate into the floodplain aquifer is potentially significant for groundwater quality, but that conditions within the floodplain result in almost complete nitrate removal. The anaerobic conditions created by the shallow unsaturated zone and carbon-rich, fine-grained sediments promote this groundwater nitrate removal. It is proposed that the river management structures contribute by creating raised groundwater levels.

Although it is acknowledged that there is considerable uncertainty in the results presented in Chapter 2, the research highlights the need for environmental managers to understand the hydrogeological setting, potential nitrate sources and capacity for attenuation within floodplains when considering whether high nitrate groundwater concentrations are likely where there are sensitive receptors. There are also implications for those who are considering changes to existing river management structures as part of river restoration schemes as the resulting changes to groundwater levels may have unintended environmental consequences. The research shows that there are benefits associated with the groundwater conditions produced by these structures in terms of the potential for enhanced attenuation of pollutants as well as creating optimal settings for some types of groundwater-dependent terrestrial ecosystems.

2) Better understanding of the influence of waste dumps on water quality in floodplain aquifers and quantification of the fluxes of associated pollutants to rivers via the subsurface

Data presented in Chapters 1 and 3 highlighted that there are substantial areas of floodplain in the UK that are within urban or peri-urban areas, and that many legacy waste dumps are located here. As these waste dumps may have no containment measures, this creates a serious risk to the environment that it was argued is not sufficiently recognised. The research presented in Chapter 3 helps to characterise the plume of pollution produced by the largest of the historical waste dumps in the River

Thames floodplain in Oxford. The study of nitrogen dynamics through the use of N-species chemistry, nitrogen isotopes and dissolved nitrous oxide revealed that the ammonium emanating from the waste dump is not significantly retarded by sorption to the aquifer sediments. It is hypothesised in Chapter 3 that this is because the floodplain sediments down-gradient of the historic waste dump have reached their capacity for adsorption. The lack of dissolved oxygen in the floodplain sediments which might otherwise have facilitated nitrification, is noted as a key factor in the conservation of the ammonium.

In Chapter 3, estimates are made of the potential flux of ammonium into the River Thames. This assumes that the impact of the waste dump studied was representative of others located on the floodplain in Oxford. The flux of ammonium estimated was substantial at 27.5 tonnes per annum, which, assuming no attenuation within the hyporheic zone or in-stream processing, would mean around 40% of the ammonium concentration measured in the river could be sourced from the waste dumps.

The understanding of the aquifer system that developed since the research in Chapter 3 was undertaken has some implications for the calculations made and the interpretation. The aquifer hydraulic conductivity estimated through particle size distribution calculations, reported in Chapter 3, is lower than the values resulting from the calibration of the groundwater flow model in Chapter 5. If the higher value is used, the groundwater flows, and therefore ammonium fluxes, would be much greater; the influx to the River Thames from waste dumps on the Oxford valley floodplain would be a factor of 2.5 greater than the 75 kg per day calculated. However, an assumption was made in Chapter 3 that the river bed hydraulic conductivity was the same as the aquifer hydraulic conductivity, based on the correlation between the river level and groundwater levels in a borehole very close to the river. Subsequent analysis presented in Chapters 2 and 5 showed that although possibly not as low as reported in other studies in the Thames catchment, the river bed hydraulic conductivity is likely to be at least an order of magnitude lower than the floodplain gravel aquifer. Data presented in Chapter 2 also showed that there is likely to be some flow west beneath the River Thames into an area of the floodplain aquifer with a deeper unsaturated zone where attenuation of ammonium could occur prior to discharge to the Seacourt Stream. All of this, together with the evidence of the attenuation capacity of the river bed sediments of the River Thames in Chapter 2, suggests the flux into the river may be lower than proposed in Chapter 3.

Recognising that there large uncertainties in the calculations made, it is argued that the implications of the research presented in Chapter 3 are still highly significant. Pollution from legacy waste dumps on our floodplains is likely to be substantial and this could be having a major impact on environmental receptors. The mitigation options are limited due to the setting and the cost of removal or containment. The requirement of environmental managers is, initially at least, risk assessment, including monitoring.

3) Conditions and mechanisms that control the occurrence of groundwater flooding in urbanised floodplains

The research presented in Chapter 4 was one of the first detailed studies undertaken internationally on groundwater flooding in a permeable floodplain setting. The study identified that although rapid groundwater level rise could occur in response to direct rainfall recharge, the primary driver for groundwater flooding in this type of setting is the stage in the associated river. Based on local knowledge, survey evidence and data from an extensive water level monitoring network, including temporary flood gauges, it was shown that in urbanised areas groundwater flooding was only apparent where property was protected from fluvial flood waters. These were located within residential areas surrounded by high ground mainly associated with transport routes. Research presented in Chapter 5 indicates that groundwater discharge would likely occur throughout the floodplain but is masked by fluvial flood waters.

Property can be impacted by groundwater flooding at ground floor level where it is built on low-lying ground in areas protected from fluvial flooding. However, the greatest impact on property has been on the basements of houses that have been converted into living areas. Where land for development was known historically to be particularly vulnerable to flooding, houses were built with the ground floor higher above the floodplain than elsewhere. This created a tall void beneath the ground floor that was perceived during the second half of the 20th Century, a period of relatively infrequent flooding and rising house prices, as a risk-free and cost-effective option for adding living space. In addition to property, Chapter 4 also reported that infrastructure was vulnerable to groundwater flooding. High groundwater levels resulted in: inflow to the sewer network compromising its operation; and inundation of telecommunication wiring. As highlighted in Chapter 5, persistent low level nuisance flooding, which is influenced by subsurface conditions, also caused issues with the transport network.

It was commonly thought that the subsurface would soak up overbanked water during periods of flooding. In Chapter 4, the capacity of the subsurface to accept water was calculated and shown to be insignificant in relation to the volumes of river water flowing through the floodplain during major events. This calculation used gravel porosity values but it may have been more appropriate to use the equivalent for the alluvium, although this would not have changed the conclusion.

The research reported in Chapter 4 was undertaken in collaboration with the Environment Agency, as part of the Oxford Flood Risk Management Study (OFRMS). General measures to reduce groundwater flood risk were discussed in Chapter 4. These were considered as part of the OFRMS, but at a city scale the only cost-effective option to address groundwater flooding was to reduce the period and height of the fluvial flood peak. This is likely to be the case in other similar urban settings. This is best done by increasing conveyance of river water through urbanised floodplain areas. This was also the optimum option identified by the OFRMS to reduce fluvial flood risk in Oxford. The scheme designed did not get implemented because the economic case was not strong enough to get Government funding. However, following further flooding in 2014, additional analysis was undertaken (I acted as an advisor to this project), and as a consequence the Oxford Flood Alleviation Scheme was designed, on the same principles, and has received funding.

The research undertaken as part of this thesis occurred at a time when flood policy was recognising the need to assess groundwater flood risk as part of a holistic approach to flood management. However, perhaps contradictory to this ethos, as discussed in Section 1.1.5, the UK response to the EU Floods Directive requirements and the concerns raised as part of the Pitt Review (Pitt, 2008), was to devolve responsibility on groundwater flooding, as well as pluvial flooding, to local authorities (in the flood context, referred to as Lead Local Flood Authorities; LLFAs). The understanding developed through research undertaken here is an aid to those responsible for assessing the significance of groundwater flooding. The insights into the potential speed of response of groundwater levels to rainfall recharge is also of relevance to the design of SuDS as measures to aid pluvial flood alleviation. The groundwater flooding research reported in this thesis has informed the development by the British Geological Survey of national maps of groundwater flood susceptibility at 1:50,000 scale (that I also have been involved in producing). These have been provided to LLFAs to help identify areas of high groundwater flood risk, a requirement of the EU Floods Directive.

4) A flood model system incorporating groundwater processes, used to understand the role groundwater and floodplain properties play in controlling the persistence of flooding in urbanised floodplains

Observations from the study area and reference to the international research literature has highlighted the impact of persistent, shallow flood waters that extend the length of flood events and have tangible economic consequences. The study presented in Chapter 5 explored the extent to which groundwater levels and the hydraulic conductivity of the surface alluvium, control the duration of this type of flooding in permeable floodplains. Flood inundation models that incorporate subsurface processes are not common. Those models that have been used to simulate floodplain-aquifer interactions, have generally been applied within research studies to understand physical processes. The models are commonly complex, physics-based codes that require long run-times. As part of the study, to address the lack of routinely used codes for flood simulation and prediction that represent groundwater, a model system was developed that links a groundwater flow model, ZOOMQ3D, to the widely-applied, computationally-efficient, cellular flood inundation model, LISFLOOD-FP. The models are linked via a low hydraulic conductivity layer that allows vertical fluxes to be quantified and passed between the two models during run time.

It is recognised that there are limitations to the linked model system; these are described in the Section 4 of Chapter 5. For example, there is an inconsistency in the model system, in that groundwater recharge is calculated in advance of the linked model run in locations that are inundated by flood waters. The lack of capture of direct rainfall input to, and evaporation from, flood waters on the floodplain may be significant under certain circumstances, as may ignoring unsaturated zone flow processes. In addition, the model does not incorporate the effects of subsurface infrastructure on the recession of groundwater and flood waters, which limits its application in urban environments. However, the case study application does allow the performance of the model system to be tested in a moderately modified floodplain setting.

The linked model system was applied to the Oxford floodplain, simulating the alluvial gravel aquifer, surface flows on the floodplain, and flows between the two via the shallow alluvium. The linked model system was used to simulate flooding processes in Port Meadow, a well-monitored area of the River Thames floodplain that regularly floods, and where flood waters are retained due to natural levees and anthropogenically-modified topography.

There were issues with simulating groundwater and flood water levels in the study that would be experienced with any application of the model. Floodplain aquifers are highly heterogeneous systems. It is challenging to both set-up a model structure that adequately captures all the key components, and for it to be parameterised. Observed water levels, including flood waters, are required to assess the performance of the model system, and monitoring networks in these settings are rare. Gradients are low and even minor errors in topography, datums and water level measurements can be problematic. Some of these issues were apparent in the modelling study presented in Chapter 5 but nevertheless the output of the model system produced a good fit to the observed groundwater and surface water level time-series, and allowed the research aims to be met.

The linked model system provided a good match to the flood extent when compared with photographs taken close to the peak of the event and to water level time-series at specific locations. The insights from the fluxes produced by the model were of particular interest, as already discussed in this Section. As shown in Chapter 5, where the alluvial aquifer is confined and where three dimensions are considered, water fluxes can be complex. The modelling of Port Meadow during the July 2007 flood event, simulated net groundwater discharge over most of the floodplain, as the groundwater heads in the high diffusivity aquifer closely followed the River Thames levels, which were generally higher than the flood water levels on the floodplain. Net recharge only occurred where flood waters accumulated due to topographical controls. This is a significant contribution to the international scientific literature on this topic, which generally reports large and uniform volumes of groundwater recharge occurring across floodplains as a result of fluvial flooding.

In the case of Port Meadow in the July 2007 event, the vertically discharging groundwater was a minor component of the floodplain water balance. As discussed in Chapter 1, the use of floodplains to store flood waters upstream of vulnerable urban centres is gaining interest as one of the options for flood risk management that does not rely on hard flood defences. This is achieved with embankments on floodplains and managed river inlets. Questions have been posed as to whether groundwater might compromise the performance of these systems. The insights from Chapter 5 are of relevance as Port Meadow acts in a similar way to an upstream flood store. In this case groundwater inflow to the floodplain is not a high proportion of the floodplain storage capacity and would not have compromised its performance if it had been used to store flood waters. However, the research does highlight that groundwater can interact with floodplain stores and it is important for flood managers to understand floodplain hydrogeology when designing them.

Chapter 5 confirms that shallow groundwater and superficial deposits could have a significant bearing on flood duration if the depth threshold of impactful flooding considered for flood risk assessment is lowered. As flooding becomes more frequent in the future, the economic consequences of this aspect of flooding may become substantial and consideration should be given to it by the relevant agencies when planning the gathering of evidence during floods. When undertaking flood risk modelling it should also be assessed whether the circumstances require the simulation of the interaction of flood waters with groundwater, as well as the nature of the superficial geology to be taken into consideration. The model developed in Chapter 5 provides an excellent tool in this context. In terms of mitigation, improved drainage for the removal of flood waters is the primary option. However, this needs to be designed recognising that drainage routes during recession can be flood water pathways during main flood events.

6.2 Further work

Recommendations for further work relate to the study area, to further understand aspects of the floodplain system, as well as more generic applications:

- The Port Meadow study involved a limited amount of field measurement of parameters used within the linked model system, in particular surface layer and gravel thickness, and aquifer, river bed and surface layer hydraulic conductivities. Better quantification of these parameters would help to constrain the ranges used within the calibration stage of the model development and the sensitivity analysis. These parameters need to be measured at a density that matches their likely spatial variability. This should include a combination of point measurements and geophysical surveying. Geophysical surveying, using a technique such as 3D electrical resistivity tomography, combined with a high-density intrusive methodology such as penetration with a steel rod to identify the alluvium base, and further shallow drilling (Chambers et al., 2014), would allow the thickness of both the alluvium and the gravels to be better mapped. Aquifer tests, both pumping tests and falling head tests, could be undertaken on newly drilled and existing boreholes within Port Meadow to estimate hydraulic conductivity and specific storage of the gravel aquifer. Permeameter tests at a range of depths within the alluvium would help to quantify its hydraulic conductivity (MacDonald et al., 2012).

  Tests within the river bed sediments could be undertaken to quantify its hydraulic conductivity, including methods using such permeameters, seepage meters, and freeze coring along with lab permeability measurement (Brunner et al., 2017). It is recognised that this involves significant challenges given the length, width and depth of the River Thames in the Port Meadow area, and the public access that makes it problematic if equipment needs to be left in situ. An additional approach to parameter quantification would be to use available analytical numerical methods on measured river and groundwater level time-series data (Barlow & Moench, 1998).

  Further and higher density high-frequency monitoring of flood water and groundwater levels would also improve the calibration of the model. The linked model system provides a tool for exploring processes within floodplains, in relation to pollutant processing, ecosystem status as well as flooding.

- It was suggested in the thesis that the complex groundwater flow pathways, resulting from managed river systems and their associated bypass channels, could result in longer residence times for groundwaters in floodplain aquifers, and that this could affect pollutant attenuation and possibly recovery from flooding. Research to explore this aspect of floodplain hydrology would be of value. This would involve focussed monitoring of zones where these complex pathways occur, including higher spatial density of nested piezometers, to allow groundwater heads, temperature and water chemistry to be measured. Additional spatial characterisation of the floodplain aquifer and associated water courses, using techniques described above, would also be necessary.

- The research in this thesis identified that interventions to improve ecological status of aquatic and terrestrial ecosystems, that change groundwater levels, could have potential unintended consequences, in particular for pollutant attenuation capacity of the subsurface, and soil moisture conditions to which floodplain vegetation can be so sensitive. Few studies have been undertaken which adequately compare pre- and post-intervention conditions to assess the benefits and disadvantages. This research is of value in the context of river restoration schemes, wetland development and natural flood management measures.

- At a national scale, the monitoring of floodplain aquifers needs to be improved. Our national groundwater level monitoring network has been developed primarily for groundwater resource assessment. Water supplied from floodplain aquifers is not insignificant, however, the network of monitoring boreholes in this setting is relatively small. It could be argued that the dense river monitoring network provides a good measure of the storage in floodplain aquifers, however, there is a need for a greater number of boreholes that are issue- and receptor-specific. These would be located to provide warning of flooding to vulnerable communities or monitor pollution plumes in relation to important ecological or water supply sources.

- The study identified that a large number of legacy waste dumps are located on floodplain aquifers, with possibly limited containment measures. It is suggested that further investigation is undertaken in the Oxford floodplain to examine the cumulative impact of waste dumps on groundwater and river quality. The flux of ammonium into the River Thames in the Port Meadow study area was estimated to be potentially substantial, however the degree of attenuation in the river bed sediments was not measured. In the context of a wider floodplain study, further work to examine processes in this zone would be of value as would investigation of in-river attenuation.

- This research would be valuable in the context of a national scale assessment of the impact of legacy waste dumps on our floodplain aquifers and associated aquatic and terrestrial ecosystems. An initial screening study could be relatively easily undertaken as collated datasets on licensed historical landfills are available from the environment regulators.

- The linked model system developed in this study could be applied to urbanised areas of the Oxford floodplain where, additionally, the impact of drainage and the built environment on the alluvium layer would need to be assessed. Model systems such as this should include interaction of surface and groundwater flows with subsurface infrastructure, to allow a holistic simulation of flood mechanisms. Although research is being undertaken to incorporate these processes in flood models, there is still much work to be done to produce models for operational use.

- The linked model system could also be applied to other groundwater flood settings, in particular on the outcrop of the Chalk outcrop. This is work that BGS plans to undertake.

- The issue of nuisance flooding has been raised in this thesis. This issue needs to be considered further and evidence gathered to assess if it is a significant aspect of flooding elsewhere in urban floodplain settings. An economic analysis in the context of urbanised UK floodplains needs to be undertaken. Models, such as the one developed as part of this study, may need to be applied more routinely within flood risk management assessments.

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