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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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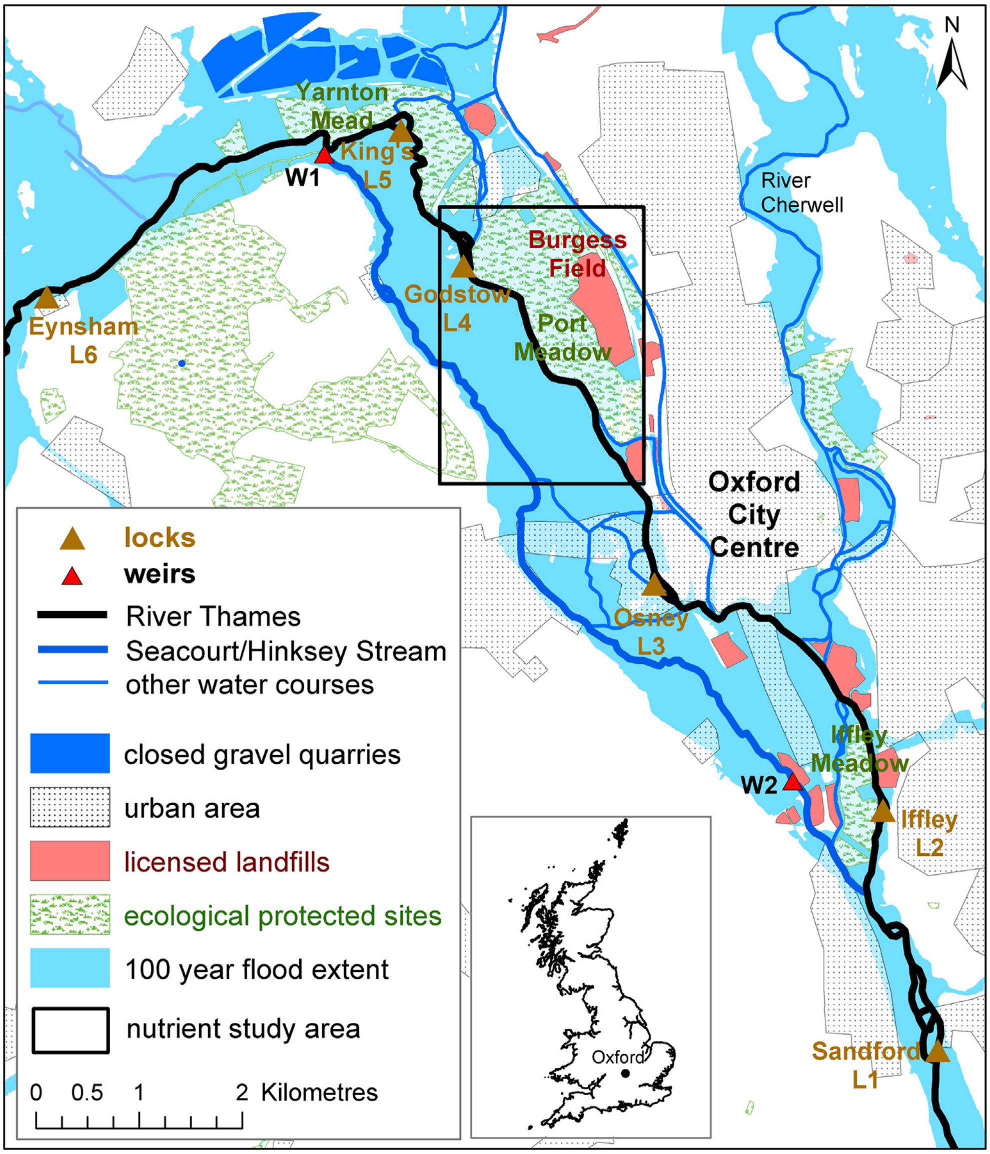

Fig. 1. River Thames floodplain in the vicinity of the city of Oxford, UK. Areas in white are higher ground above the 1-in-100 year flood extent. Contains Ordnance Survey data © Crown copyright and database right (2018).

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).

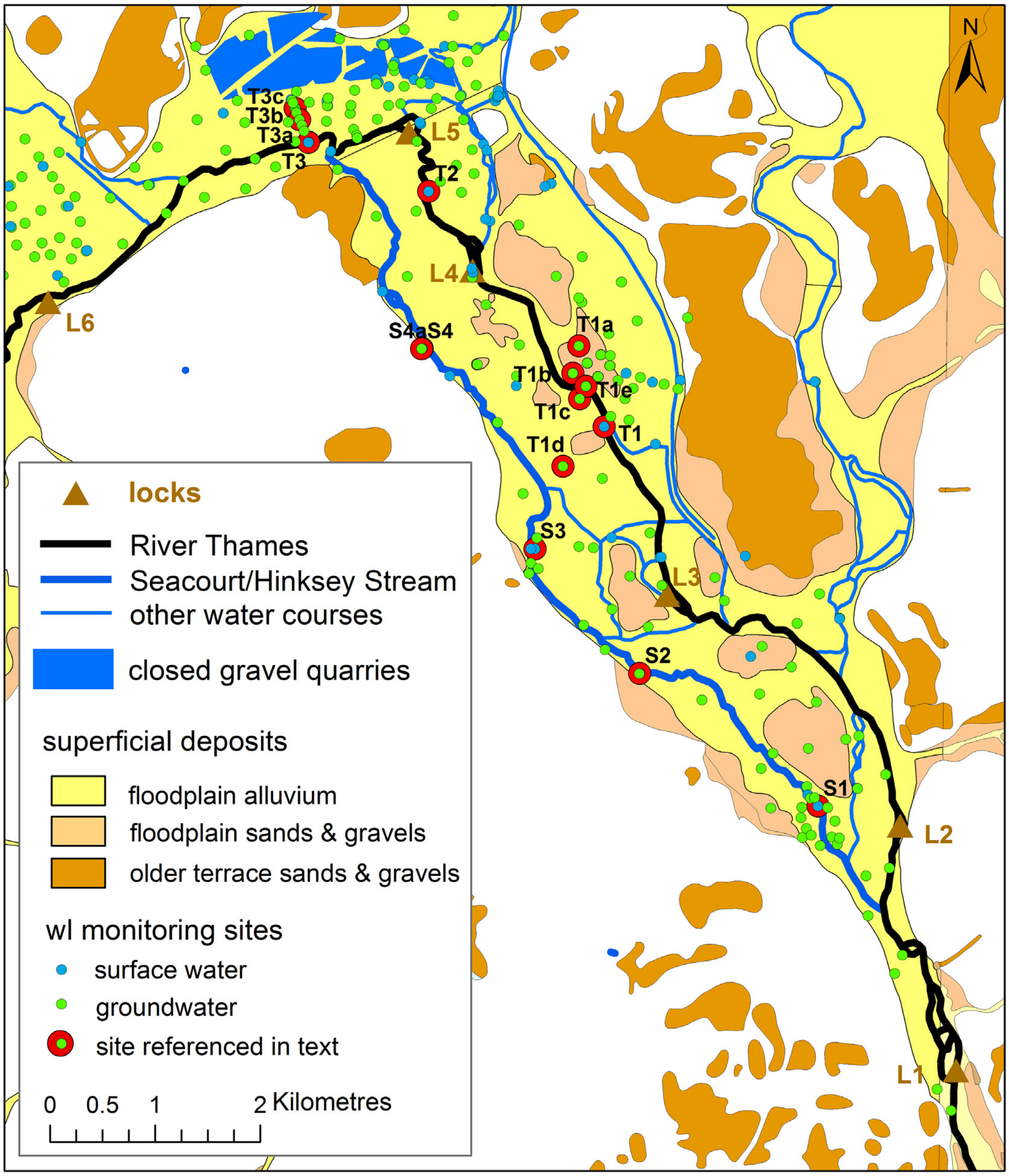

Fig. 2. Water level monitoring network and superficial geology. Sites referred to in the text are labelled. Note, some of the surface water monitoring sites are located on minor water courses not included in the figure. Contains Ordnance Survey data © Crown copyright and database right (2018).

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)).

$$R_P + R_R + Q_L = Q_A \qquad (1)$$

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)

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

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

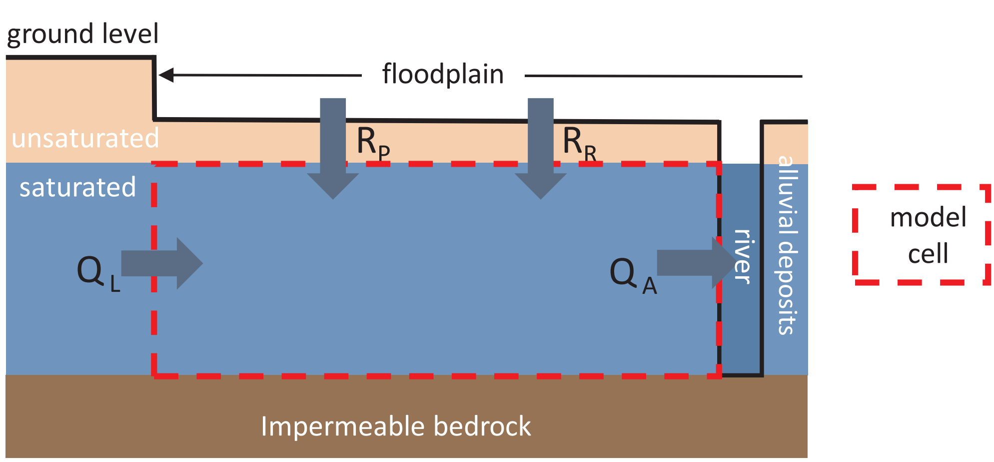

Fig. 3. Schematic of flows within the single-cell mixing model. See Section 2.3.2 for definition of terms.

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

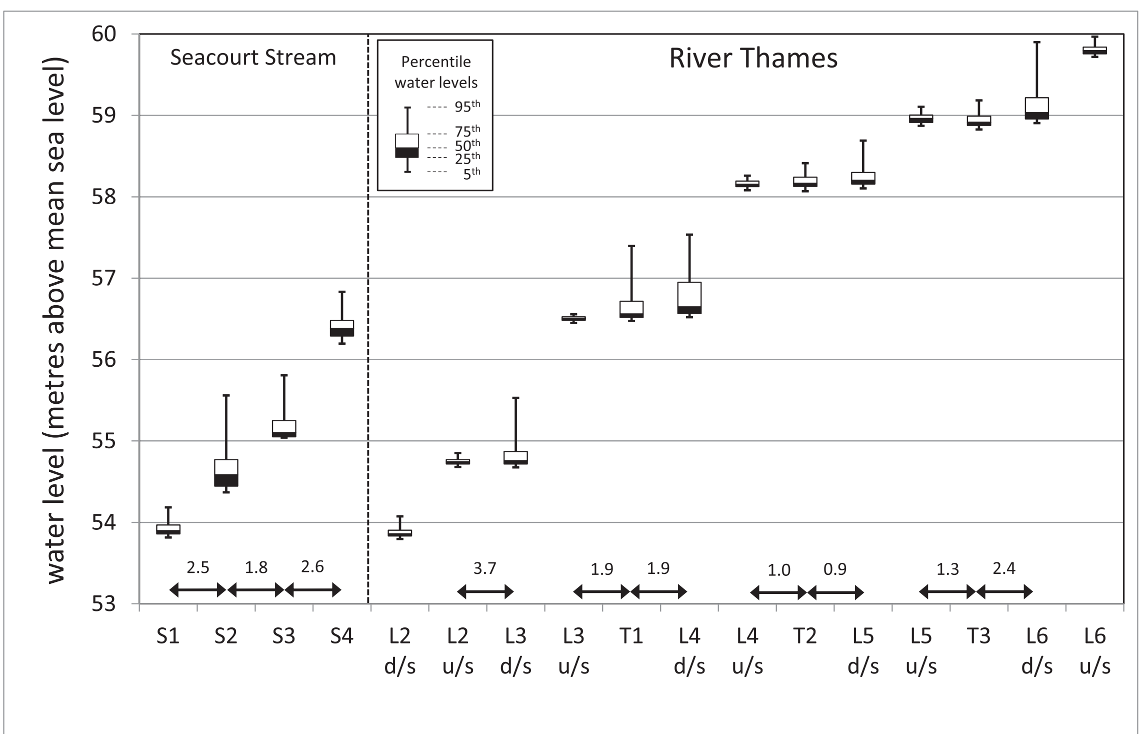

Fig. 4. Box plots of water levels at sites on the Seacourt Stream and the River Thames, from downstream (left) to upstream (right), for 1 April 2009 to 31 March 2013. For data used to produce the box plots see Supplementary Information Table S1. Lengths of reaches between monitoring sites (km) are shown 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.

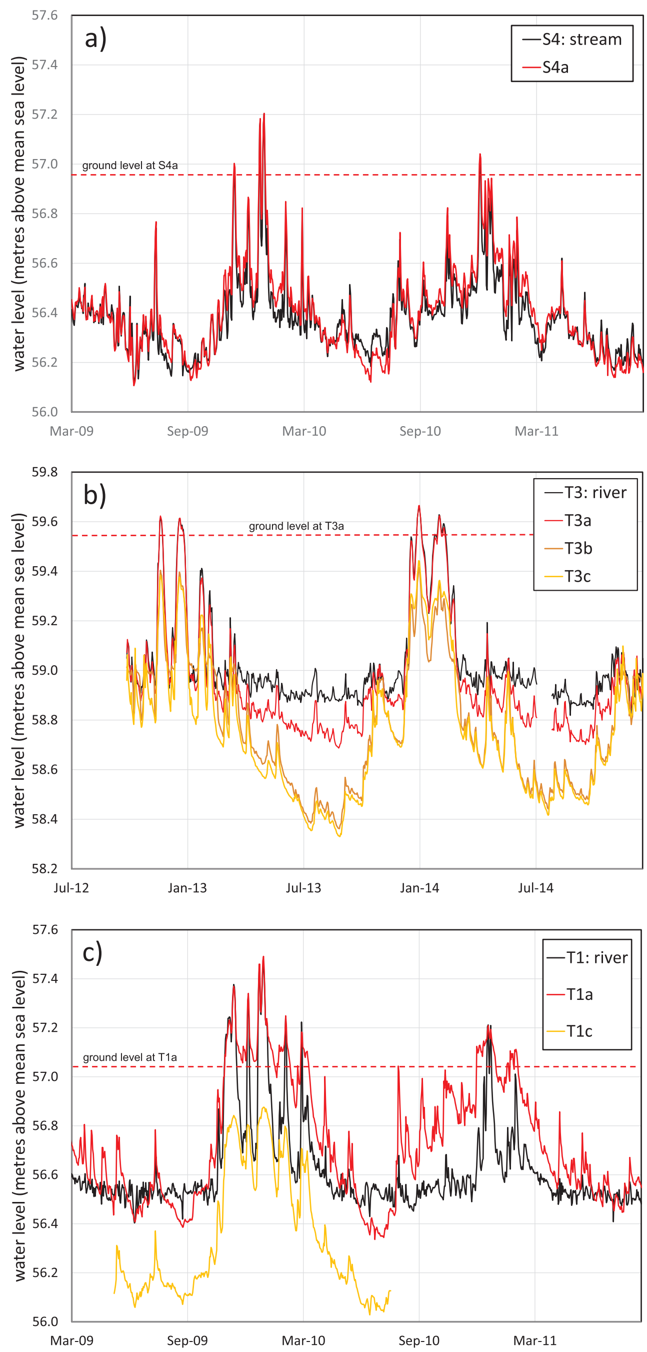

Fig. 5. Groundwater and surface water levels at three locations. Black lines are the river/stream water levels and other lines are groundwater levels (see Fig. 2 for locations): a) paired groundwater/surface water sites on the Seacourt Stream; b) a transect perpendicular to the River Thames upstream 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

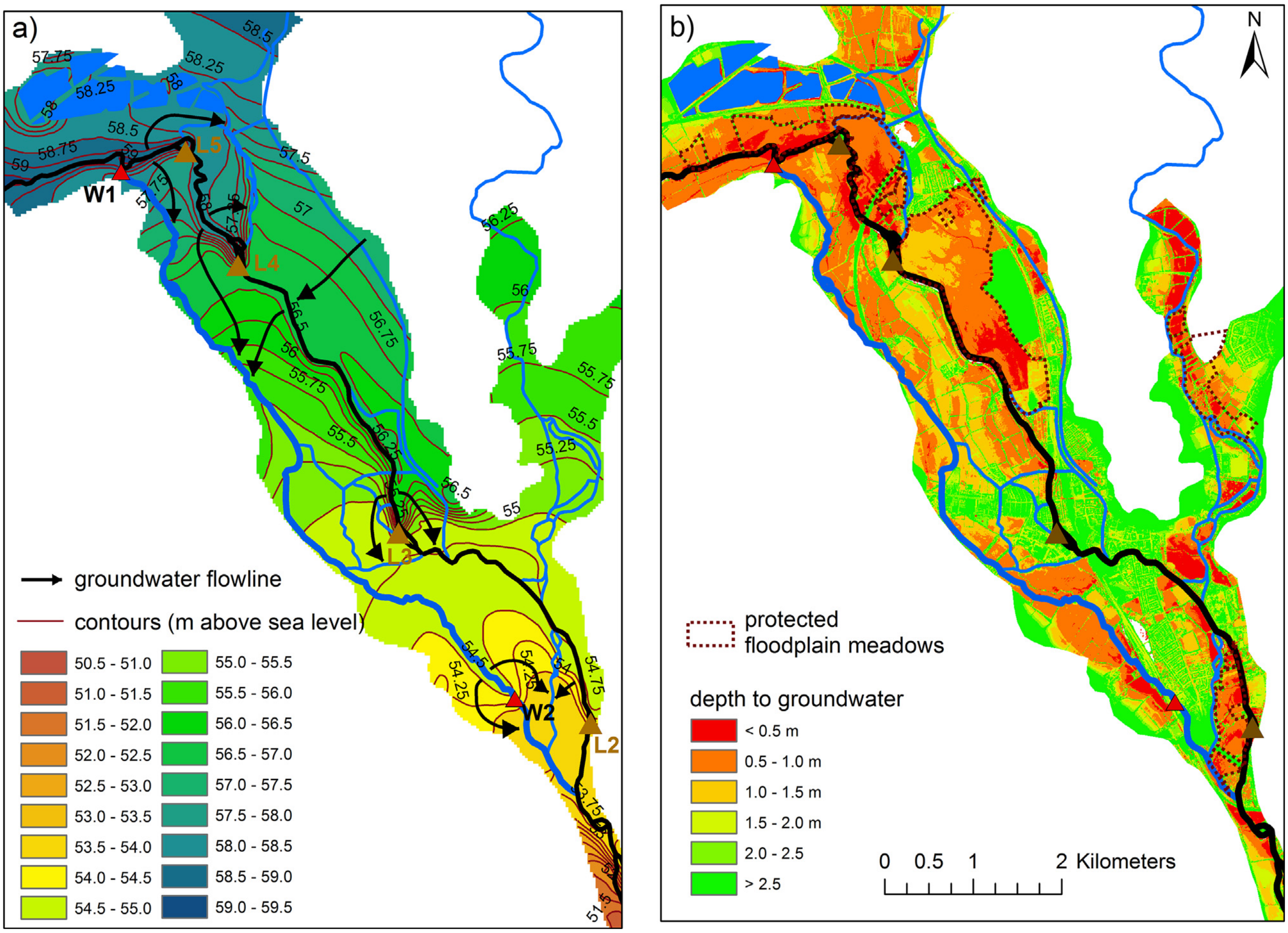

Fig. 6. a) contoured groundwater levels and groundwater flow lines within the alluvial floodplain aquifer; and b) depth to groundwater within the alluvial aquifer. Both maps are based on water levels measured in May 2007. Refer to Fig. 1 for elements not included in the legend. Contains Ordnance Survey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

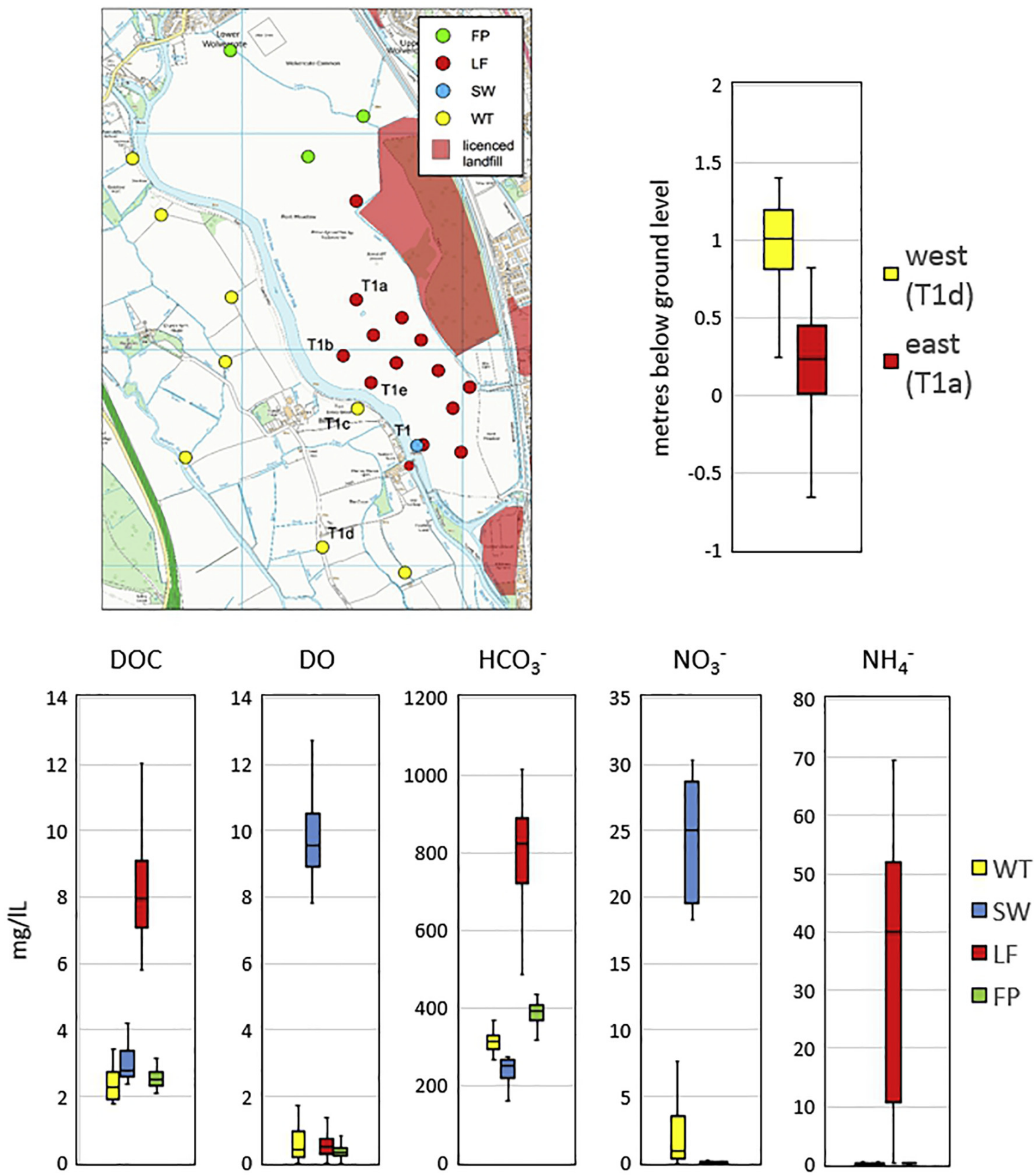

Fig. 7. Box plots of depths to groundwater and water chemistry for a series of monitoring sites (see: Fig. 1 for extent of this area within the overall study area; Fig. 4 for box plot legend; and Supplementary Information Table S1 for data used to produce the box plots). Sample sites are categorised 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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