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COUPLED HYDROLOGICAL/HYDRAULIC MODELLING OF RIVER RESTORATION IMPACTS AND FLOODPLAIN HYDRODYNAMICS

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ABSTRACT

Channelization and embankment of rivers has led to major ecological degradation of aquatic habitats worldwide. River restoration can be used to restore favourable hydrological conditions for target species or processes. However, the effects of river restoration on hydraulic and hydrological processes are complex and are often difficult to determine because of the long-term monitoring required before and after restoration works. Our study is based on rarely available, detailed pre-restoration and post-restoration hydrological data collected from a wet grassland meadow in Norfolk, UK, and provides important insights into the hydrological effects of river restoration. Groundwater hydrology and climate were monitored from 2007 to 2010. Based on our data, we developed coupled hydrological/hydraulic models of pre-embankment and post-embankment conditions using the MIKE-SHE/MIKE 11 system. Simulated groundwater levels compared well with observed groundwater. Removal of the river embankments resulted in widespread floodplain inundation at high river flows ($>1.7\,m^3\,s^{-1}$) and frequent localized flooding at the river edge during smaller events ($>0.6\,m^3\,s^{-1}$). Subsequently, groundwater levels were higher and subsurface storage was greater. The restoration had a moderate effect on flood peak attenuation and improved free drainage to the river. Our results suggest that embankment removal can increase river–floodplain hydrological connectivity to form a more natural wetland ecotone, driven by frequent localized flood disturbance. This has important implications for the planning and management of river restoration projects that aim to enhance floodwater storage, floodplain species composition and biogeochemical cycling of nutrients. © 2016 The Authors. *River Research and Applications* Published by John Wiley & Sons Ltd.

KEY WORDS: river restoration; embankment removal; hydrological model; MIKE SHE; MIKE 11; floodplain; river–floodplain connectivity; flood peak attenuation

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INTRODUCTION

Natural riparian and floodplain ecosystems form highly dynamic ecotones (i.e. transitional zones) between terrestrial and aquatic environments (Naiman and Décamps, 1997; Stanford, 2002). They support a range of diverse microhabitats and species, which are maintained by an active balance because of regular floods that continuously reshape the river channels and their banks, and deliver water, sediment and nutrients onto the floodplain (Junk *et al.,* 1989; Tockner and Stanford, 2002). Rivers and their connected riparian zones are widely recognized for the ecosystem services they provide, which are of both ecological and commercial value, such as the provision of habitat, flood water storage, nutrient attenuation and the maintenance of biodiversity (Forshay and Stanley, 2005; Hill, 1996; Naiman *et al.,* 2010; Ward *et al.,* 2002). These services are, however, dependent on strong hydrological links via overbank and subsurface flow that have, in many cases, been disrupted by anthropogenic modifications to rivers and floodplains (Kondolf *et al.,* 2006; Ward *et al.,* 1999; Zedler and Kercher, 2005).

Indeed, an estimated 50% of wetlands have been lost worldwide; this is largely attributed to the drainage of floodplains and riparian areas for agricultural and urban development, to water abstraction and pollution (Russi *et al.,* 2013). In England and Wales, over 40% of the total river length is classified as severely modified (Environment Agency, 2010). Channelization and embanking of rivers are ubiquitous anthropogenic disturbances that have led to major ecological degradation of aquatic ecosystems (Erskine, 1992; Nilsson and Svedmark, 2002; Pedroli *et al.,* 2002; Petts and Calow, 1996).

Examples of modification to the river environment are river embankments, which limit overbank flows onto the floodplain in order to protect adjacent land from flooding. However, local river embankment can severely impact flood defence downstream. Embankments lead to increased channel volume and flow depth and reduced resistance to flow, which in turn results in higher flow velocities, decreased contact time of water with sediments (important for the

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nutrient filtering capacity of aquatic environments) and increased downstream transport of water (Darby and Simon, 1999; Gilvear, 1999). In light of the recent severe flooding in England (December 2015 was the wettest and mildest December and 2013/2014 was the wettest winter on record in the UK) and continental Europe (2013 was one of the wettest summers on record in central Europe), and the likelihood of more frequent and intense rainfall events associated with a warmer climate (Jenkins *et al.,* 2010; Murphy *et al.,* 2010), the importance of providing 'room for rivers' has become apparent (DEFRA, 2004; Hooijer *et al.,* 2004; Met. Office, 2015b; Met. Office, 2015a; Wilby *et al.,* 2008).

River restoration involving the removal of river embankments is an increasingly popular management technique being used to restore a more natural, dynamic, flood-pulsed hydrological regime and to reclaim historical floodplain areas for floodwater storage (Acreman *et al.,* 2003; Blackwell and Maltby, 2006; Pescott and Wentworth, 2011). Hydrology, in terms of water quantity (duration and frequency of floods) and quality (supply of nutrients), is an important driver of floodplain biodiversity and nutrient attenuation capacity (Silvertown *et al.,* 1999; Baker and Vervier, 2004; Forshay and Stanley, 2005; Dwire *et al.*, 2006). Hence, river restoration that aims to create favourable hydrological conditions for floodplain biota and the biogeochemical cycling of nutrients such as nitrogen is also central to the legislative plans of governing bodies that aim to achieve good ecological and chemical status of European waters under the Water Framework Directive (Directive 2000/60/EC).

The effects of river restoration on hydraulic and hydrological processes are complex and are often difficult to determine if there is insufficient monitoring conducted before and after the restoration works (Darby and Sear, 2008; Kondolf, 1995). Understanding the long-term impacts of river restoration is important for predicting changes in wetland function and subsequent response patterns of the floodplain biota. For these reasons, hydrological modelling is increasingly used to better understand the effects of river restoration activities under a variety of hydrological conditions (Hammersmark *et al.,* 2010; Thompson *et al.,* 2009). One such deterministic model is MIKE SHE, which is employed in the current study and is a physically based, fully distributed comprehensive modelling system driven by daily air temperature, precipitation, evapotranspiration and gridded fields of physical properties (e.g. topography, geology, soil properties and vegetation cover). A finite-difference approach is used to solve the differential equations that describe saturated flow, unsaturated flow and overland flow. Channel flow is simulated using the one-dimensional hydraulic modelling system, MIKE 11. Dynamic coupling of the MIKE 11 river model and the MIKE SHE hydrological model enables the simulation of river–aquifer exchange, inundation from the river onto the floodplain and the return of overland flow to the river (DHI, 2007a; Thompson *et al.,* 2004).

This study used coupled MIKE SHE/MIKE11 hydrological/hydraulic models of pre-embankment and post-embankment conditions to simulate the hydrological impacts of river embankment removal along a reach of a lowland river in eastern England. More than 3 years of river discharge and meteorological data, and observed groundwater elevations, were used to parameterize and calibrate/validate the models, respectively. A companion paper (Clilverd *et al.,* 2013) described the hydrological and biogeochemical regime of the original embanked river floodplain and the initial response to embankment removal. Here, we address the following two research questions: (i) What are the effects of embankment removal on key components of river–floodplain hydrology (water table elevation, frequency and extent of floodplain inundation and flood peak attenuation)? and (ii) How will embankment removal impact river–floodplain hydrology under a range of expected river flow conditions?

STUDY AREA

The study was conducted at Hunworth Meadow on the River Glaven, a small (17 km long), lowland (elevation at Hunworth Meadow ~21 m above ordnance datum (AOD)), calcareous river in North Norfolk, UK (Figure 1). The River Glaven has a catchment area of 115 km^2 and flows through agricultural land, deciduous and coniferous woodland and grazing meadows, with most of the former floodplain environments currently disconnected from the river by embankments. Previous glaciation of this area has resulted in the formation of glacial hill features throughout the catchment in an otherwise flat landscape (Moorlock *et al.,* 2002). Hunworth Meadow is approximately 400 m long, 40–80 m wide and has an area of approximately 3 ha. It is bounded to the north-east by an arable and woodland hillslope (Figure 1). An agricultural drainage ditch on the meadow runs parallel to the river close to the base of this hillslope but has become blocked in recent years towards the downstream end of the meadow, impairing the site's drainage and leading to near-permanent surface water within a ponded area adjacent to the ditch. Prior to embankment removal (see succeeding text), the meadow comprised a degraded *Holcus lanatus–Juncus effusus* rush pasture community typically associated with waterlogged soils (Rodwell, 1992; Clilverd *et al.,* 2016).

Mean annual rainfall for the East Anglia region (for the period 1985–2015) is approximately 623 mm and is characterized by higher rainfall during the autumn and winter months (Figure 2). On average, the annual potential evapotranspiration [evaluated using the Hargreaves–Samani

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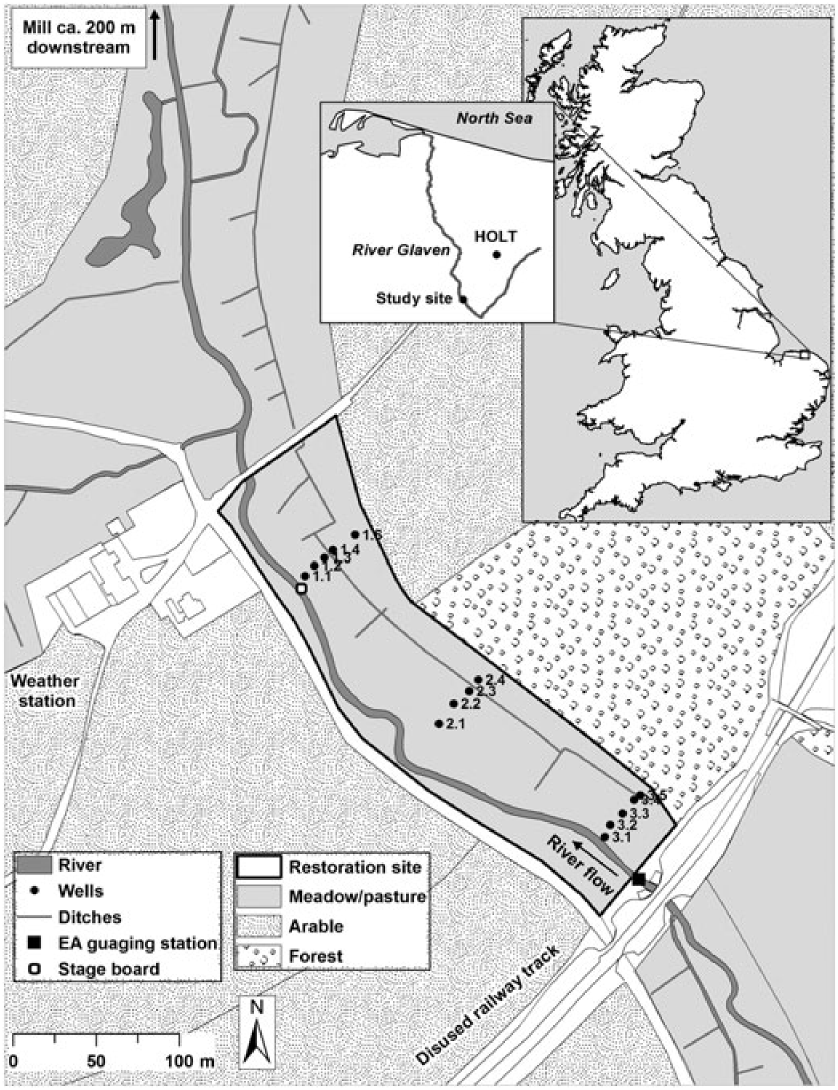

Figure 1. The River Glaven restoration site at Hunworth, north Norfolk. The woodland and arable border along the northeast of the meadow delineates the base of a hillslope. The River Glaven is shown inset, with the location of the study site at Hunworth

method (Hargreaves and Samani, 1985)] reaches 600 mm and exceeds precipitation in the summer. River discharge, measured at an Environment Agency gauging station (Station Number: 034052) immediately upstream of the study site, follows the typical annual hydrograph of a chalk stream (mean baseflow index = 0.81; Clilverd *et al.,* 2013), with increased discharge over the winter. Mean river discharge from 2001 to 2010 was $0.26\,m^3\,s^{-1}$. The largest discharge during this period was $3.1\,m^3\,s^{-1}$ (Figure 3).

The River Glaven is slightly alkaline with an average pH of 7.3. It is moderately eutrophic, with nitrate concentrations averaging 6.2 mg NO_3^--N L^{-1} and phosphate concentrations of less than 0.05 mg P L^{-1} (Clilverd *et al.,* 2013). The River Glaven's catchment is characterized by chalk bedrock that is overlain by chalk-rich sandy till up to 40 m thick and glaciogenic sand and gravel deposits (Moorlock *et al.,* 2002). Floodplain soils consist of alluvial deposits estimated to be a maximum of 2 m thick and are predominantly sandy loam at the study site. A detailed description of the geology at the site is presented in Clilverd *et al.* (2013).

Chalk streams and rivers provide scarce and declining habitats. As such, they have received considerable conservation attention and are a priority habitat under the EU Habitats Directive (92/43/EEC). Like many rivers in Europe and indeed worldwide, the River Glaven has been modified by agricultural and flood management activities that have included river channelization, construction of artificial embankments and soil drainage. Nevertheless, the river flows through numerous habitat types that are of high conservation value (e.g. wet meadows, riparian woodlands,

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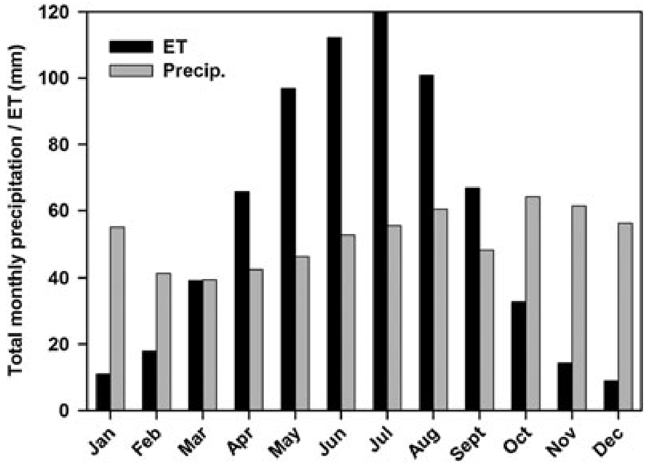

Figure 2. Mean total monthly precipitation and potential evapotranspiration (1985–2015) for East Anglia, UK. Climatology data are from UK Met Office regional climate summaries (Met. Office, 2016). Potential evapotranspiration was estimated using the Hargreaves–Samani method (Hargreaves and Samani, 1985)

shallow lakes and coastal marshes), which support several important and protected freshwater species such as brook lamprey (*Lampetra planeri*), white-clawed crayfish (*Austropotamobius pallipes*) and otter (*Lutra lutra*) (Sayer, 2014; Sayer and Lewin, 2002).

Chalk rivers such as the Glaven are low-energy systems poorly suited to autonomously reinstate their natural channel structure once it has been disturbed by engineering works. Therefore, river restoration through the reconfiguration of river embankments and the channel bed forms an integral part of returning the natural state and functioning of many chalk rivers. At Hunworth, the River Glaven was constrained by embankments that ranged from 0.4 to 1.1 m above the meadow surface, sufficient to prevent overbank flows onto the adjacent floodplain during the largest recorded discharges (Clilverd *et al.,* 2013). Restoration of the 400 m reach of river was undertaken between 18 and 27 March 2009 by the Environment Agency in collaboration with the River Glaven Conservation Group, the Wild Trout Trust and Natural England. The embankments were removed (with the exception of one section on the river bend midway along the meadow that was left to protect water vole burrows), and the spoil was removed from the site. This lowered the surface elevation of the riverbanks to the level of the adjacent meadow and reduced channel cross-sectional area by approximately 60% (Figure 4). It was anticipated that these changes would improve the connection between the river and its floodplain and in turn improve flood storage and establish a floodplain hydrological regime that will diversify wet meadow vegetation (e.g. Castellarin *et al.,* 2010; Hammersmark *et al.,* 2008; Viers *et al.,* 2012). The current study uses hydrological/hydraulic modelling to assess the impacts of the restoration on water table elevation, frequency and extent of floodplain inundation and flood peak attenuation.

METHODS

MIKE SHE model development

Coupled MIKE SHE/MIKE 11 hydrological/hydraulic models were developed for the pre-restoration (embanked) and post-restoration (no embankment) scenarios, which differed only in embankment and riverbed elevations resulting from the embankment removal. In both cases, the model

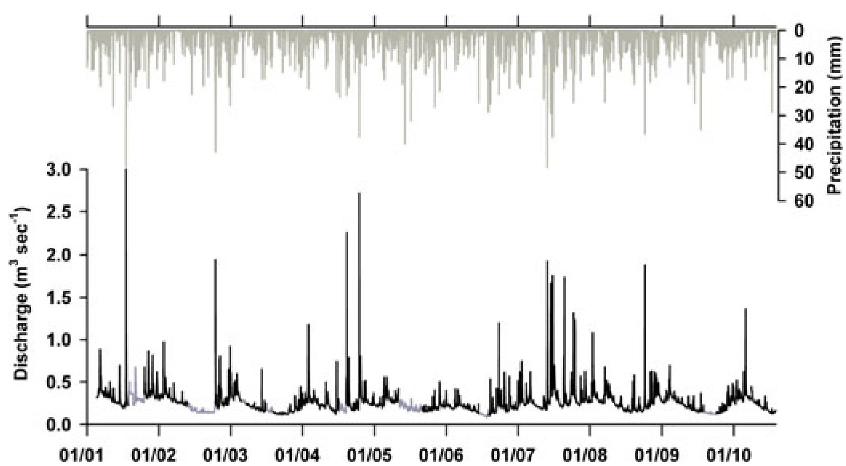

Figure 3. Time series of mean daily river discharge and total daily precipitation at Hunworth, Norfolk, from 2001 to 2010. Discharge data are from the Environment Agency (EA) gauging station (#034052) located at Hunworth. These data were supplemented with data from Bayfield EA gauging station (#034016) 5 km downstream (shown in grey) using the following regression ($y = 0.4087x + 0.0396$; $r^2 = 0.86$) during low-flow conditions when in-stream macrophyte growth affected the rating curve at Hunworth This figure is available in colour online at wileyonlinelibrary.com/journal/rra

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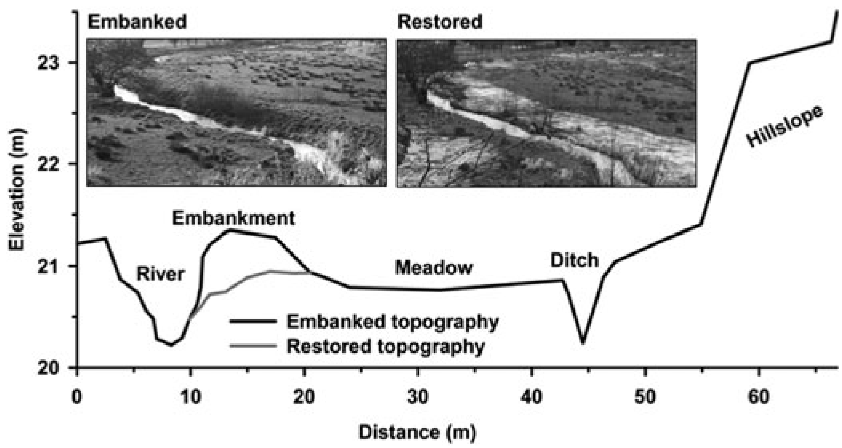

Figure 4. Cross sections and photographs (inset) of the embanked and restored River Glaven floodplain at well transect 3

domain included Hunworth Meadow and extended up to the summits of the adjacent hillsides on either side of the river. The upstream limit of the modelled area coincided with a disused railway embankment, whilst a smaller embankment carrying an agricultural track crossing the floodplain defined its downstream limit. The model domain was divided into 5038 grid cells of 5×5 m, although, as discussed later, initial calibration steps employed a 15×15 m grid (610 grid cells). The relatively fine discretization of the final model was needed to accurately characterize topographic variations across the floodplain including the blocked ditch and small-scale features such as shallow depressions and raised hummocks that can provide microhabitats of differing soil water content that are important for fostering high species diversity (Wheeler *et al.,* 2004). Two digital elevation models, one representing the embanked river, and the other the restored river, were derived from differential global positioning system (dGPS) surveys conducted before and after embankment removal (see Clilverd *et al.,* 2013 for detailed methods). Both digital elevation models (DEMs) were resampled to the MIKE SHE model grid.

The model included a relatively simple one-layer saturated zone (lower level 10 m above Ordnance Datum Newlyn) that represented the average geological conditions in the upper alluvial and glacial soils. These were considered separated from the chalk aquifer at the site by a layer of low-permeability boulder clay. Initial horizontal and vertical hydraulic conductivity values were guided by results from piezometer slug tests (mean $= 1.88 \times 10^{-6}$ m s^{-1}) conducted on the floodplain but were both subject to adjustment during model calibration. A combination of zero-flow and specified head subsurface boundary conditions were applied around the model domain (Figure 5) (e.g. Hammersmark *et al.,* 2008). A zero-flow boundary is the default condition and is realistic for watershed boundaries. The zero-flow boundaries are a simplification of the system but were justified for application along the summits of the hillsides on either side of the meadows following the assumption that the groundwater divide followed the topographic divide and provided a hydraulic boundary (e.g. Thompson, 2012). Similarly, the foundations of the railway embankment defined a physical boundary at the upstream end of the meadows that was assumed to restrict flow into the site. Some subsurface flow perpendicular to the river is, however, possible across the downstream boundary of the floodplain. To facilitate this exchange and provide a more realistic representation of actual conditions, a constant head boundary was specified at this location using mean groundwater elevation from a well transect at the downstream end of the meadow (see succeeding text). Specified-head and constant-head boundaries can supply an inexhaustible source of water no matter how much water is removed from a system model (e.g. Franke *et al.,* 1987). This is unlikely to cause a problem at the downstream boundary of the Hunworth model as the constant head value is based on mean groundwater elevation that fluctuated very little in this region of the floodplain. A manual sensitivity analysis of alternative boundary options (specified head, flux, zero-flow) was performed and demonstrated negligible effects on simulated groundwater elevations across the floodplain beyond the immediate location of the boundary conditions.

Soil properties were defined for a spatially uniform unsaturated zone that was represented using the two-layer water balance approach. This method is considered most appropriate in conditions that include high water tables and a rapid groundwater response to precipitation that characterize Hunworth Meadow (e.g. Thompson, 2012). The infiltration rate of the unsaturated zone (1.0×10^{-5} m sec^{-1}) was varied during calibration, with initial values guided by the piezometer slug test. Soil water contents at saturation and field capacity were additional calibration terms. However, initial values were based on measurements of the water release

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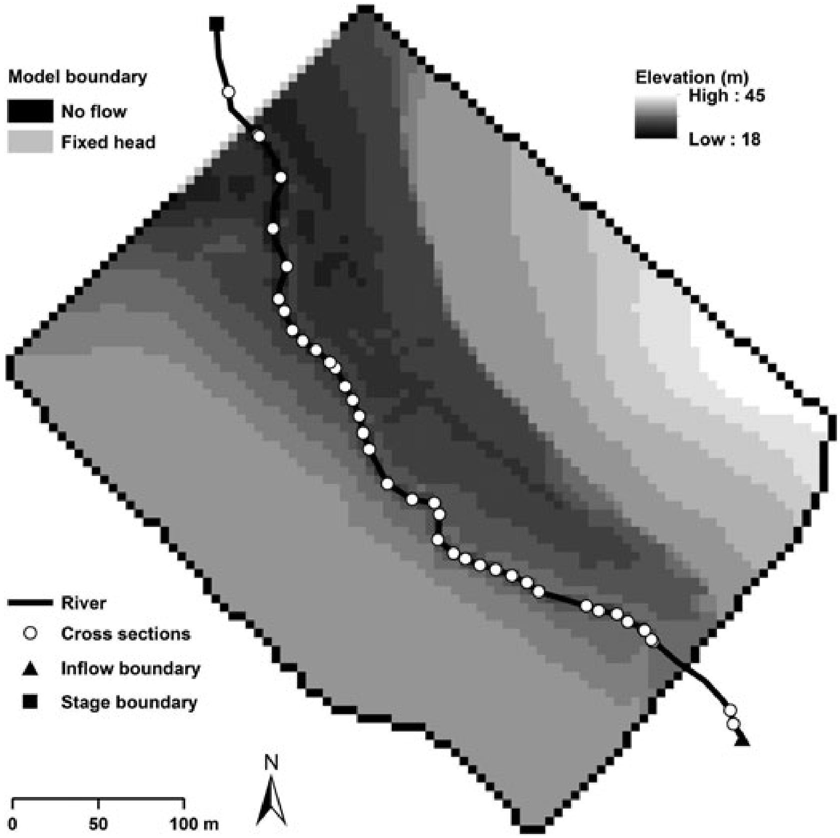

Figure 5. MIKE 11 river channel, cross sections and surface water boundary conditions of Hunworth Meadow superimposed upon the MIKE SHE model DEM (5-m grid resolution) and boundary

characteristic (pF-curve) using a manual 08.01 sandbox (Eijkelkamp, Giesbeek, The Netherlands), and averaged 0.7 (volumetric basis) and 0.2 (volumetric basis), respectively. Water content at wilting point was also varied during calibration, but was not measured. Therefore, a range of wilting point values for sandy loam soils were obtained from the literature (mean = 0.07) to guide the initial value (Zotarelli *et al.,* 2010). The final unsaturated zone parameter subject to calibration was the ET depth that determines the effective depth of evapotranspiration, that is, the thickness of the capillary zone. The maximum height of capillary rise for sandy loam soils at Hunworth Meadow was calculated as a function of soil pore size using Hazen's formula of capillary rise (Das, 2002) to be between 0.4–1.9 m. This is consistent with capillary rise values of >0.5 m in fine sands and silts reported by DHI (2007b), and with measurements in the range of 1.0–1.5 m for weakly compacted alluvial sandy loams and 1.5–2.0 m for alluvial loams (Chubarova, 1972).

The overland Manning's roughness coefficient was an additional calibration term initially set at a uniform value of 0.3 s m$^{-1/3}$, as guided by values for grassland in the literature (Thompson *et al.,* 2004; USDA, 1986). Subsequently, a value of 0.4 s m$^{-1/3}$ was applied to the woodland hillslope and to patches of rushes (*Juncus effusus*) in the vicinity of the ditch, based on values for light woodland underbrush and coarse grass given in USDA (1986).

Four different land use classes were defined within the Hunworth model: roads and buildings, arable land, riparian grassland and mixed deciduous/coniferous woodland. Constant rooting depths were applied to most land use classes, with the exception of the arable class, which was varied seasonally (range: 0–1.8 m). Root depth was set at 0.3 m on the meadow and 2.7 m for the mixed deciduous/coniferous woodland. Root depth values for the woodland and arable crop (classified as winter wheat) were taken from the literature (Canadell *et al.,* 1996; FAO, 2013; Thorup-Kristensen *et al.,* 2009), whereas the rooting depth for the meadow was based on investigations at the site and measurements of water table depth, which showed that a shallow region of topsoil was aerated during the growing season. Seasonal changes in leaf area index (LAI) were applied to the arable (range: 0–4), meadow (range: 1–4) and mixed woodland (range: 1–4) classes to account for increased LAI values during the growing season (Herbst *et al.,* 2008; Hough and

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Jones, 1997). Root depth and LAI was defined as 0 for the 'roads and buildings' land cover class.

In order to simulate the ponded conditions that were present at the downstream end of the meadow, an area of lower soil permeability was specified for the spatial extent of the pond to account for the accumulation of fine sediment in this region. A subsurface leakage coefficient of $1 \times 10^{-9}\,s^{-1}$ was used for the pond area, and detention storage and initial water depth were both set at 0.05 m. The MIKE SHE drainage option was used to represent relatively small-scale, fast runoff along the base of the hillslope and to route drainage into topographical lows along the agricultural ditch. A drainage level and a time constant were applied along the base of the hillslope and the ditch and were altered in the sensitivity analysis and model calibration. A drainage level of −1.6 m and a time constant of $6 \times 10^{-8}\,s^{-1}$ along the base of the hillslope, with a higher time constant of $2.6 \times 10^{-7}\,s^{-1}$ closer to the model boundary, provided the best overall fit (Table I).

Spatially uniform precipitation and potential evapotranspiration were specified, an approach justified by the small size of the model domain. Daily precipitation inputs were based on records from an automatic weather station (Skye MiniMet SDL 5400) installed 100 m from the meadow (Figure 1) supplemented, during periods of instrumental failure, with data from a nearby (<10 km) UK Met Office meteorological station (source ID: 24 219, Mannington Hall). Daily Penman–Monteith potential evapotranspiration (Monteith, 1965) was computed from air temperature, net radiation, relative humidity and wind speed observations from the onsite weather station. A detailed description of the evapotranspiration calculations is given in Clilverd *et al.* (2013).

The maximum allowed model time steps for the unsaturated flow (using the two-layer water balance method), saturated flow (finite difference) and evapotranspiration components were set at 24 h. A shorter time step of 0.25 h was specified for the overland flow (finite difference) component to ensure model stability. However, in flat areas with ponded water, such as on floodplains, the difference in water depth between grid cells is close to 0, which requires very small overland flow time steps. To allow the simulation to run with longer time steps and further reduce numerical instability, the calculated overland flows between cells were multiplied by a damping factor to reduce flow between cells when the flow gradient was close to 0. Rather than the default damping function in MIKE SHE, an alternative single parabolic function was specified, which approached zero more quickly and was consistent with the approach used in MIKE FLOOD (DHI, 2007b). This alternative damping function was applied below a specified gradient of 0.001. All model results were stored at 24-h intervals to coincide with the temporal frequency of observations.

MIKE 11 model development

Two MIKE 11 models were developed, one for the embanked river scenario and another for the restored scenario. Dynamic coupling of each MIKE 11 river model and the appropriate (embanked/restored) MIKE SHE model through the exchange of simulated water levels at MIKE 11 h-points (points where water level data are calculated along the river branch) and MIKE SHE river links enabled the simulation of river–aquifer exchange and inundation from the river onto the floodplain (DHI, 2007a; Thompson *et al.,* 2004). River–aquifer exchange was simulated using the aquifer-only formulation, where the river is assumed to be in full contact with the aquifer material. This was a suitable method given the similarity between river and groundwater chemistry along the riverbanks and the high baseflow index (0.81) and flow exceedance values for Q95 (51%), which indicated high groundwater contributions to discharge at the site (Clilverd *et al.,* 2013).

Table I. Final calibrated MIKE SHE and MIKE 11 parameter values

Parameter	Value
MIKE SHE	
Overland Manning's coefficient (sec $m^{-1/3}$)	0.30 (grass) 0.4 (light underbrush)
Water content at saturation (volumetric)	0.24
Water content at field capacity (volumetric)	0.10
Water content at wilting (volumetric)	0.05
Saturated hydraulic conductivity (m sec^{-1})	1×10^{-6}
Evapotranspiration surface depth (m)	1.10
Horizontal hydraulic conductivity (m sec^{-1})	9×10^{-7}
Vertical hydraulic conductivity (m sec^{-1})	1×10^{-7}
Drainage level (m)	−1.6
Drainage time constant (sec^{-1})	2.6×10^{-7}
MIKE 11	
River bed resistance (sec $m^{-1/3}$) (time varying)	0.058–0.15

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A 576-m section of the River Glaven beginning immediately upstream and ending just downstream of Hunworth Meadow was digitized in MIKE 11 using 1:10 000 Ordnance Survey digital data (Land-Line.Plus) (Figure 5). River cross sections for the two MIKE 11 models representing pre-restoration and post-restoration channel configurations were specified using the results from the dGPS surveys conducted before and after embankment removal. Cross sections were surveyed at approximately 10-m intervals.

A discharge boundary condition was specified at the upstream end of the MIKE 11 model using daily discharge records from the gauging station located immediately upstream of Hunworth Meadow. A constant water-level boundary condition of 18.6 m AOD was applied at the downstream end of the MIKE 11 model. This level was just above the bed of the river at the lowest cross section and prevented the river drying out whilst permitting discharge of water from the downstream end of the MIKE 11 model (e.g. Thompson *et al.,* 2004). An initial water depth of 0.2 m throughout the MIKE 11 model at the start of the simulation period was obtained from the records of a stage board installed in the river towards the downstream end of the reach (Figure 1).

A constant Manning's roughness coefficient for bed resistance of $0.08\,\mathrm{s\,m^{-1/3}}$ was initially applied to the model. However, this value resulted in river levels being too high in the winter and too low in the summer. Instead, a time varying Manning's roughness coefficient was specified throughout the MIKE 11 model based on the approach used by House *et al.* (2015), to account for seasonal differences in bed resistance associated with in-stream macrophyte growth. Seasonal macrophyte growth in the river was easily identified in the river discharge record (Clilverd *et al.,* 2013) as it impacted the rating curve and resulted in a slow increase in baseflow through the summer, despite low or no rainfall. This effect declined during the autumn because of macrophyte dieback or more abruptly during flood events because of de-vegetation of the river channel (e.g. Chambers *et al.,* 1991). Two general summer conditions were identified for varying Manning's roughness values among years: (1) high-flow summers where macrophyte growth was restricted and (2) low-flow summers where stable conditions resulted in substantial vegetation growth. A Manning's roughness coefficient of $0.058\,\mathrm{s\,m^{-1/3}}$ was applied in the winter, and maximum values of $0.08\,\mathrm{s\,m^{-1/3}}$ and $0.15\,\mathrm{s\,m^{-1/3}}$ were applied in June during high-flow and low-flow summers, respectively. These values are within the range of 0.045 to $0.353\,\mathrm{m^{1/3}\,s^{-1}}$ reported for a UK chalk stream by House *et al.* (2015). The growth period was defined as April to September, and Manning's roughness values during this period were interpolated between the winter and summer values, which was guided by macrophyte growth measurements in a UK chalk stream reported by Flynn *et al.* (2002).

The MIKE 11 models were set up to run at 1-min time steps. Once coupled to the MIKE SHE model, the specified MIKE SHE time step allowed storage of river flow and water levels at hourly intervals. Using the approach adopted by Thompson *et al.* (2004), flood codes were used to specify MIKE SHE model grid cells that could be directly inundated from the MIKE 11 model. Potentially flooded cells comprised the immediate riparian area, which included the grid cells through which the river ran, those coincident with embankments (if present) and the zone up to 10 m (two grid cells) onto the meadow. These MIKE SHE grid cells were flooded from the river if water levels simulated by MIKE 11 were higher than the corresponding MIKE SHE grid surface level. Once a grid cell was flooded, the overland flow component of MIKE SHE would simulate surface water movement onto adjacent model grid cells further away from the river. Infiltration and evapotranspiration from flooded cells would also be simulated in the same way as if flooding occurred from precipitation and surface runoff or the water table reaching the ground surface (Thompson *et al.,* 2004).

Model calibration and validation

A sensitivity analysis was performed as an initial step in the calibration process. Using the MIKE Zero automatic calibration procedure (Autocal), parameters were individually varied, and the most sensitive model parameters were then included in the model calibration (Table I). Model calibration and validation were principally undertaken through the comparison of observed and simulated groundwater levels. Observations were provided for 10 shallow (1–2 m depth) wells arranged in three transects across Hunworth Meadow (Figure 1). Groundwater table was recorded using a combination of manual well dipping and pressure transducers (Solinst 3.0 Levelogger corrected for barometric pressure changes using a single Solinst barologger). Further details of the well locations and instrumentation are given in Clilverd *et al.* (2013). Mean daily water levels were derived from the hourly observations from the pressure transducers for comparison with simulation results. In addition, simulated water levels in the River Glaven were compared with records from the stage board installed at the downstream end of the reach (Figure 1).

Calibration and initial model validation were undertaken for the embanked model using a split sample approach. The 13-month period 22 February 2007 to 14 March 2008 was used for calibration, and the following 12 months (15/03/2008 to 15/03/2009) for validation. The end of this period coincided with embankment removal, so that calibrated parameter values were specified within the model representing restored conditions with the subsequent 16 months (29/03/2009 to 25/07/2010) providing a second validation period. As described previously, a number of

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model parameters were varied during model calibration (Table I). Initial calibration was undertaken using an automatic calibration procedure that was based on the shuffled complex evolution method with the optimal parameter set being selected according to the lowest aggregate root mean square error (rmse), a measure of the average magnitude of error for the comparisons between observed and simulated groundwater and river water levels (DHI, 2007c; Duan *et al.,* 1992; Madsen, 2000, 2003). This approach was undertaken for the coarser 15×15 m model grid to reduce the computational time due to the number of individual model runs ($n = 480$) required for the automatic calibration routine to determine an optimal parameter set. Following auto-calibration, the model grid size was reduced to 5×5 m, and the calibration was checked and refined manually, with the model performance being assessed statistically using the RMSE, the correlation coefficient (R) and the Nash–Sutcliffe efficiency coefficient (Nash and Sutcliffe, 1970). These key statistics assess different aspects of the model performance (bias, correlation, goodness of fit) and have been widely used in similar studies including those where optimized parameter values from auto-calibration routines are refined manually (House *et al.,* 2015; Rochester, 2010; Thompson, 2012; Thompson *et al.,* 2013). The final values of the calibration terms defined at the end of this process are summarized in Table I. The same statistical measures were subsequently employed to assess model performance for both of the validation periods.

Impact assessment of embankment removal

The hydrological effects of removing the embankments along the River Glaven were investigated by running the two MIKE SHE/MIKE 11 models representing pre-restoration and post-restoration conditions for the same extended period with identical climatic and river flow conditions. This method avoids the differences in simulated hydrological conditions that are due to inter-annual climate variability within the pre-restoration and post-restoration periods used for model calibration and validation. For example, 2007 and 2008 (pre-restoration) were characteristically wetter than 2009 and 2010 following restoration (Table II). Simulating pre-restoration and post-restoration for the same period therefore enables the effects of embankment removal to be assessed directly. The simulation period for this assessment was the decade 2001–2010. As for the calibration and validation periods, the upstream boundary condition of the MIKE 11 model was specified as mean daily discharge at the Hunworth gauging station. In the absence of data from the local automatic weather station, daily precipitation and Penman–Monteith potential evapotranspiration were derived from records from the Mannington Hall meteorological station. These data represented a range of climate and river flow conditions, including extreme high river flow (i.e. 2001 and 2007) and low river flow (i.e. 2009) years (Figure 3), which enabled the simulation of a spectrum of probable flow conditions expected on the floodplain under both pre-restoration and post-restoration conditions.

Table II. Total precipitation and potential evapotranspiration (Penman–Monteith) and precipitation minus potential evapotranspiration for the hydrological years 2002–2010

Hydrological year	Precipitation (mm)	ET (mm)	Precipitation – ET (mm)
2002	598	475	86
2003	736	537	200
2004	829	520	309
2005	728	490	238
2006	697	522	175
2007[a]	972	475	496
2008[a]	738	460	277
2009[a]	669	497	172
2010[a]	766	511	255

ET, evapotranspiration.
[a]Calibration and validation period: climatology data from on-site weather station at Hunworth. Preceding years show data from the nearby (<10 km) UK Met Office meteorological station at Mannington Hall (source ID: 24219).

Bankfull capacity for the embanked river channel was estimated using a cubic regression between river stage and discharge ($r^2 = 0.999$; $y = -635.0860 + 162.9633 \times x + -11.7679 \times x^2 + 0.2606 \times x^3$). As bankfull discharges for the embanked scenario were not observed during the 10-year period of the discharge record, bankfull capacity was extrapolated beyond the available data and thus should be treated with caution. Bankfull capacity for the restored river was derived from MIKE 11 discharge-stage relationships and bankfull measurements from the river cross sections. In addition, bankfull capacity was evaluated using MIKE SHE results depicting the depth of overland water, which enabled the identification of two thresholds for overland flow: a high discharge threshold above which widespread inundation occurred and a lower threshold above which localized flooding (up to one grid cell—i.e. 5 m—from the river) occurred.

RESULTS

Model calibration and validation

For the majority of dip wells, there is very good agreement between the observed and simulated groundwater levels throughout the calibration and validation periods (Figure 6). The timing of simulated groundwater fluctuations fits well with the observed data. In particular, the rapid response of groundwater during high-magnitude rainfall and river flow events is captured well by the model. The observed and

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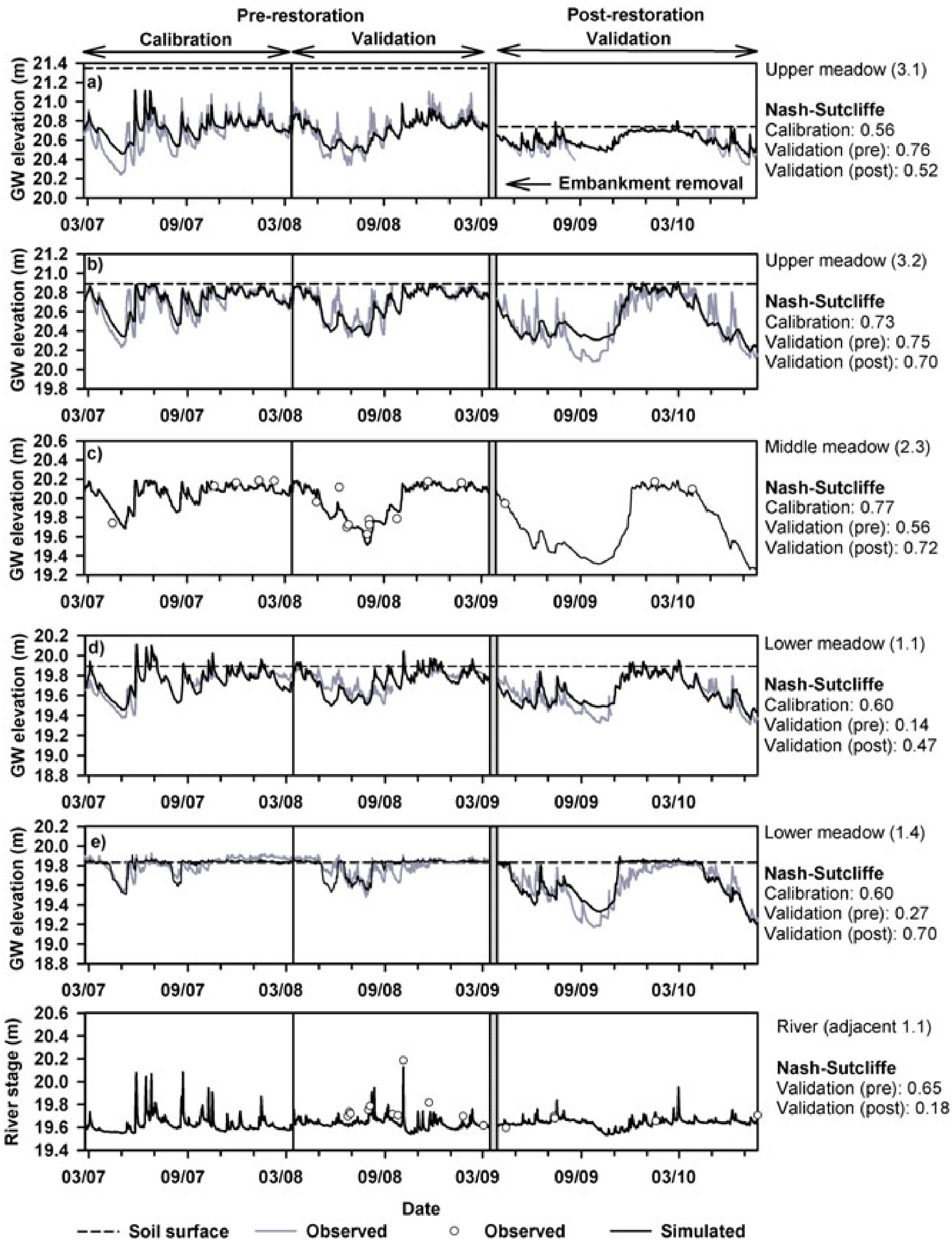

Figure 6. Comparison of observed and simulated groundwater depths for the calibration and validation periods for six representative wells across the floodplain. The embankment removal in March 2009 is highlighted by the vertical hashed bar This figure is available in colour online at wileyonlinelibrary.com/journal/rra

simulated rates of groundwater decline following periods of elevated water tables (typically March to May) also show generally good agreement each year. During some periods of low rainfall (e.g. August to mid-October 2009), simulated groundwater levels close to the river are higher than the observed levels, possibly because of overestimated in-stream macrophyte growth; however, this difference is <0.2 m.

Groundwater levels on the floodplain are controlled by river stage and responses to rainfall. The model reproduces the close connection between groundwater and river water

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levels and captures the recession of groundwater levels in response to decreasing river levels (Figures 3 and 6). Seasonal changes in groundwater levels are reproduced well by the model. Levels at each of the well locations exhibit similar temporal patterns, with distinct seasonal fluctuation in groundwater levels in the range of 0.4–0.6 m. Across the floodplain, greater fluctuations in groundwater levels are simulated during the summer when drier conditions result in water levels that are typically lower in the soil profile, compared with the winter when surface soils are predominantly saturated (Figure 6). Consequently, greater variability in groundwater levels occurs between summers than between winters. The model clearly reproduces the lower groundwater levels observed during the dry summers of 2009 and 2010 following embankment removal, compared with the wet summers in 2007 and 2008 in which both observed and simulated groundwater levels are higher.

The ability of the model to represent observed conditions within the Hunworth Meadow is further demonstrated in Table III that summarizes the model performance statistics for each well for the calibration period and each of the validation periods (pre-restoration and post-restoration). The mean error for groundwater levels is typically less than ±0.05 m, and the correlation coefficient averages 0.85, 0.80 and 0.85 for the calibration and pre-validation and post-validation periods, respectively. Values of the Nash–Sutcliffe efficiency coefficient are between 0.5 and 0.8 for most of the wells, indicating fair to good model performance. In particular, excellent performance is indicated for wells 3.1 and 3.2. The first of these was the only well located on the embankment and as a result necessitated the re-installation of monitoring equipment after restoration (note the change in soil surface elevation in Figure 6a). Water levels simulated by the model also provide a good fit at Well 1.1, which was located next to the embankment at the downstream end of the meadow and at wells spanning the middle section of the meadow (Wells 2.1–2.3). The model performs less well at the edge of the ditch (i.e. at those wells that were in many cases within 1 m of this channel) and at the floodplain–hillslope margin. Model performance statistics indicate a poorer fit in this narrow section of the floodplain, with simulated groundwater levels being periodically slightly higher than observed at Well 2.4 and lower than observed at Wells 3.4 and 3.5 (Figure 6). Model performance in some of the lower meadow wells (e.g. Wells 1.1 and 1.3) is poor during the pre-restoration validation because of lower than observed groundwater levels during a period of low rainfall from April to May 2008.

Collectively, the comparisons between observed and simulated groundwater levels and the associated model performance statistics indicate a good ability of the model to reproduce groundwater levels across most of the meadow for periods both before and after the removal of river embankments. These results suggest that the model is an appropriate tool to assess the impacts of embankment removal upon hydrological conditions across the floodplain.

Impacts of embankment removal on overbank flows and floodplain inundation

The impact of embankment removal upon the potential for overbank flows is summarized in Figure 7. This shows the

Table III. Mean error (ME – m), correlation coefficient (R) and Nash–Sutcliffe model efficiency coefficient (NSE) for the calibration (22/02/07 to 14/03/2008) and validation (pre-restoration: 15/03/2008 to 15/03/2009; post-restoration: 29/03/2009 to 25/07/2010) periods

	Calibration (pre-restoration)			Validation (pre-restoration)			Validation (post-restoration)		
Well	ME	R	NSE	ME	R	NSE	ME	R	NSE
1.1	−0.02	0.79	0.60	0.02	0.71	0.14	−0.02	0.70	0.47
1.2[a]	−0.02	0.74	−0.23	−0.05	0.42	−1.71	−0.07	0.75	0.29
1.3	0.03	0.81	0.49	0.03	0.65	0.16	0.03	0.83	0.62
1.4	0.00	0.79	0.60	0.00	0.66	0.27	−0.03	0.86	0.70
1.6	0.04	0.74	0.12	0.00	0.65	0.17	0.01	0.68	0.37
2.1	−0.13	1.00	0.62	−0.05	0.91	0.75	−0.07	0.99	0.67
2.2	−0.10	0.99	0.67	−0.01	0.85	0.58	−0.02	0.99	−6.56
2.3	0.01	0.96	0.77	0.05	0.85	0.56	0.02	0.89	0.72
2.4	−0.05	0.62	−0.80	−0.04	0.80	0.31	0.05	0.91	0.54
3.1	−0.06	0.85	0.56	0.01	0.90	0.76	−0.04	0.84	0.52
3.2	−0.04	0.89	0.73	0.03	0.89	0.75	−0.03	0.84	0.70
3.3	−0.07	0.86	0.37	0.00	0.75	0.20	0.05	0.81	0.57
3.4	0.01	0.82	0.55	0.03	0.85	0.18	n/a	n/a	n/a
3.5	−0.04	0.89	0.45	0.00	0.88	0.69	0.11	0.81	0.26
River stage	n/a	n/a	n/a	0.08	0.97	0.65	−0.03	0.65	0.18

n/a, data not available for the period.
[a]Note that there were problems at times with the level logger at Well 1.2, so this well was discounted during model calibration and validation.

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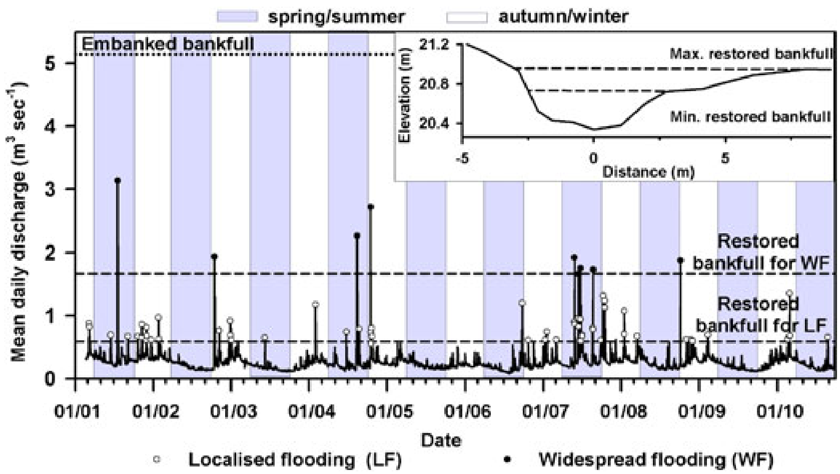

Figure 7. Mean daily river discharge from 2001 to 2010. The embanked and restored bankfull capacity is shown, above which widespread inundation of the floodplain would have occurred. Two bankfull thresholds, a minimum and maximum, are shown for the restored river, which correspond to the cross section inset; flows above these thresholds result in localized and widespread flooding, respectively. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

daily discharge at the Hunworth gauging station for the period 2001–2010 upon which are superimposed the estimated bankfull channel capacities under both embanked and restored conditions. Throughout the whole 10-year period, no overbank flows were simulated because the bankfull channel capacity ($5.1\,m^3\,s^{-1}$) was greater than the maximum observed flow of $3.1\,m^3\,s^{-1}$. In contrast, river flows frequently exceeded bankfull capacity in the restored model, where two thresholds for inundation on the floodplain were identified: the high-flow channel capacity ($1.67\,m^3\,s^{-1}$) above which widespread floodplain inundation occurred and the low-flow channel capacity ($0.6\,m^3\,s^{-1}$) that resulted in localized inundation at the river edge in an area corresponding to the former location of the embankments. Such flooding did not occur in pre-restoration conditions because of the steep sides of the embankments. Throughout the 10-year period, discharge exceeded the high-flow channel capacity for widespread flooding in the restored model on nine occasions, albeit only for short periods (1 day). Three large overbank events occurred over a month-long period from late May to June 2007, interspersed with eight smaller localized flooding events at the river edge. Localized flooding was much more frequent (61 occasions) and of longer duration (2–3 days) and is likely to result in a more dynamic and natural transitional zone between the river and the floodplain.

Surface flooding on the floodplain is simulated within the MIKE SHE models when groundwater levels intercept the ground surface (in which case precipitation cannot be infiltrated) or when the river overtops the channel banks. In the embanked model, groundwater was the only source of flooding on the floodplain, whereas under restored conditions, inundation also occurred because of overbank flows. Restoration of these overbank flows and the reconnection of the river and its floodplain therefore had a marked effect on simulated floodplain hydrology. This is clearly demonstrated in Figure 8, which shows the simulated extent and depth of surface water for the pre-restoration and post-restoration models for two high river flow events. The first (which occurred on 28/05/2007) is associated with a mean daily discharge of $1.9\,m^3\,s^{-1}$, just above the threshold channel capacity associated with widespread inundation under restored conditions, whilst the second (18/07/2001) is the largest event ($3.1\,m^3\,s^{-1}$) during the 10-year simulation period.

Results for the embanked, pre-restoration model show that river water was constrained within the river channel by the embankments, which were not flooded in both events shown in Figure 8 and indeed throughout the 10-year simulation period. During the smaller flood event (Figure 8a), flooding was limited to the margins of the floodplain ditch and the downstream ponded area and was driven by rising groundwater tables. During the larger river flow event (Figure 8c), there was limited groundwater flooding behind the embankments, with surface water depth ranging between 0.0 and 0.02 m across much of the meadow and up to 0.4 m in topographic depressions along the ditch and ponded area in the lower meadow. This was attributed to an extended period of low rainfall, high evapotranspiration and low water table depths that preceded the high flow event.

Under post-restoration conditions, overbank flows resulted in widespread inundation on the floodplain that would supplement groundwater-fed surface water. During the smaller flood event (Figure 8b), much of the floodplain was subject to shallow (<0.3 m depth) inundation. Embankment removal enabled some overbank flows at the top end

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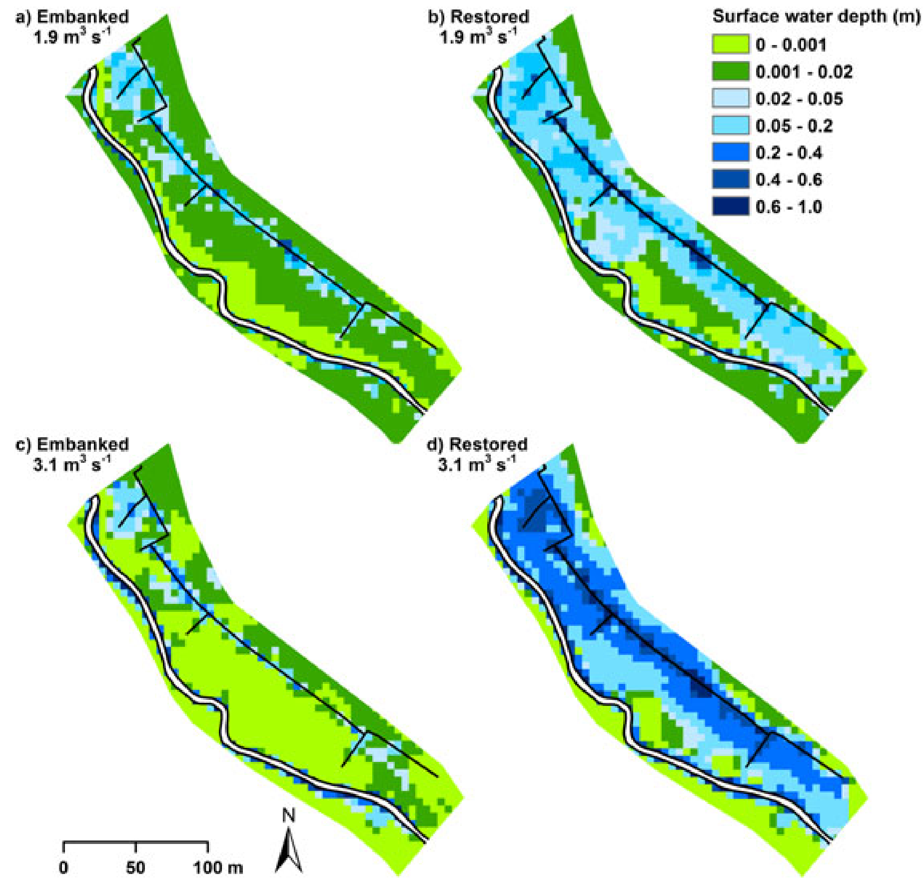

Figure 8. Comparison of simulated surface water extent and depth for the embanked and restored scenarios during (a and b) a small overbank (post-restoration) event (28/05/07; flow = 1.9 $m^3 s^{-1}$) and (c and d) a larger overbank (post-restoration) event (18/07/01; flow = 3.1 $m^3 s^{-1}$). This figure is available in colour online at wileyonlinelibrary.com/journal/rra

of the floodplain, although a relatively high section of the riverbank and adjacent floodplain in the upper-middle part of the site was not flooded. Further downstream, the lower half of the floodplain was directly connected with the river, and the previously embanked area was inundated. During the largest flood event (Figure 8d), nearly the entire floodplain (with the exception of a few MIKE SHE riparian grid cells where the embankments were not removed) was directly connected to the river, and extensive and much deeper flooding (0.2–0.6 m) occurred.

Simulation results show that the ditch running parallel, but to the north of the river, played an important role in distributing floodwater. Surface water resulting from high water tables or overbanking of the river was channelled across and down the floodplain into the ditch, which then filled and contributed to flooding along the ditch marginal areas, ponding in topographic depressions and subsequent groundwater recharge leading to higher water table elevations (Figure 8). Surface water accumulated in the lower section of the meadow in the region of the pond. Prior to the restoration, the ponded area that was subject to groundwater flooding as well as being fed by the ditch, was saturated for much of the year. In this state, the embankment acted as a barrier for water that had accumulated in this part of the floodplain, preventing its return to the river. However, after the removal of the embankments, drainage of surface water from the floodplain to the river was restored. Water stored in this low-lying area of the meadow during flood events subsequently acted as a source of return flow to the river.

Impacts of embankment removal on groundwater

Throughout the 10-year simulation period, groundwater levels close to the river (i.e. within 30 m) were on average 0.01 m higher under restored conditions, whereas groundwater levels in low-lying areas of the meadow that were previously flooded were on average 0.01 m lower in the restored scenario. This is reflected in Figure 9, showing the differences in groundwater levels at the 14 wells simulated by the embanked and restored models. During periods of the highest river flows, groundwater levels were up to

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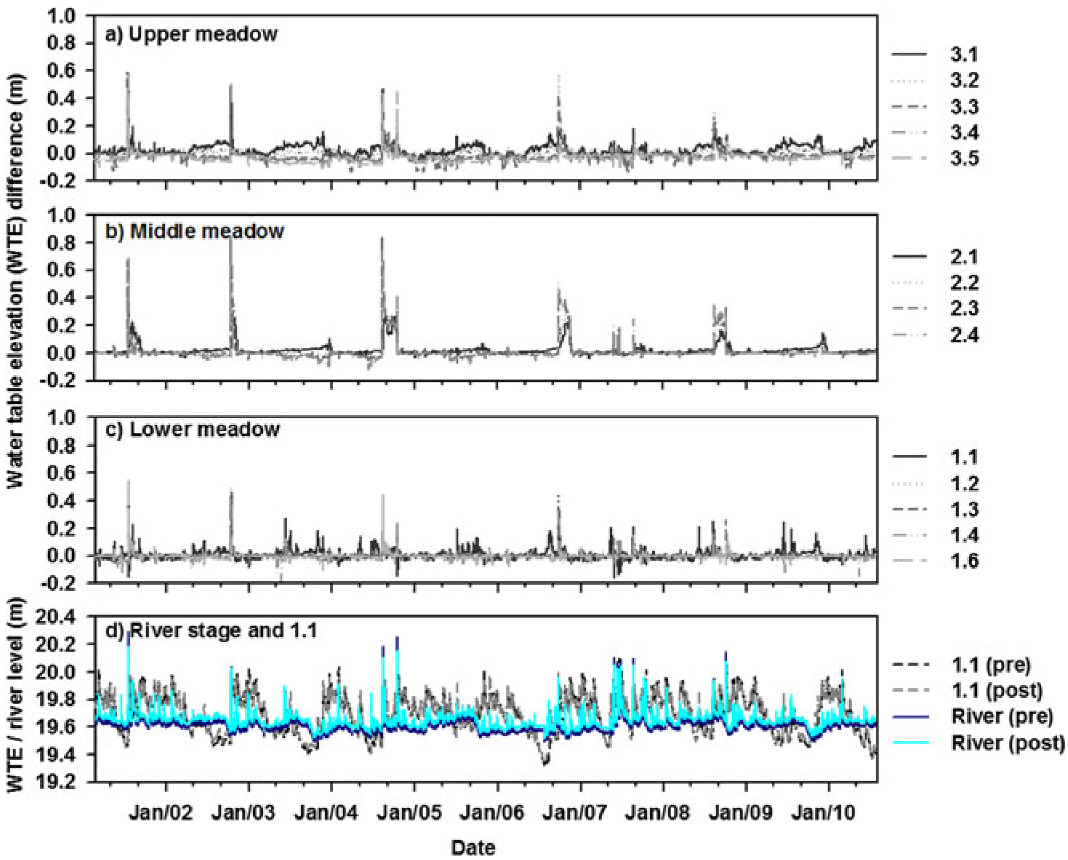

Figure 9. Time series of simulated water table elevation (WTE) differences (a–c) between the restored and embanked scenarios from 2001–2010 (a period that encompassed a range of wet and dry conditions). Positive differences indicate restored WTE > embanked WTE. Differences in WTE are shown at well locations across the floodplain at the (a) upper, (b) middle and (c) lower well transects (see well locations in Figure 1). Comparison of simulated WTE at Well 1.1 and river water level adjacent to Well 1.1 (d) for the embanked and restored scenarios from 2001–2010. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

0.8 m higher under restored conditions. The largest increases in water table elevation occurred along the river banks (e.g. Wells 3.1 and 1.1), in the region of the ditch (e.g. Wells 2.3 and 2.4) and on the relatively low-lying downstream end of the floodplain. The smallest effects were seen at Well 2.1 adjacent to the section of riverbank that was not restored, where increases in water table elevation during high river flow periods were typically less than 0.3 m (Figure 9b). This location corresponds to the relatively high part of the floodplain that, as discussed previously, was not flooded under post-restoration conditions (Figure 8b). Some short periods of slightly lower groundwater levels (up to −0.18 m) were simulated under restored conditions immediately after periods of groundwater and overbank flooding. These are most noticeable at Well 1.1 (Figure 9c) that was located close to the river and the ponded area in this part of the floodplain and at Well 1.4 that was located in a low-lying area next to the ditch. These changes are most likely due to the previously discussed improved drainage at the river–floodplain margin.

The greatest differences in water table elevation between the embanked and restored model results occurred in spring/summer during periods of low river flows. Simulated groundwater levels along the river (i.e. within 30 m) for the restored model were on average 0.03 m higher ($p < 0.05$) than those for the embanked model in the spring/summer. No significant differences were found in the autumn/winter ($p = 0.754$) (Figure 9). This can be attributed to increased surface flooding and floodplain storage during a number of inundation events that occurred in the summer months. Increased floodplain storage before the beginning of the spring/summer drawdown combined with periodic additions from summer flooding reduces the summer groundwater head recession within the meadow (Figure 9d). The higher simulated groundwater levels after embankment removal causes some differences in the hydraulic gradient between the river and floodplain during the summer. In comparison with the embanked model, results for the restored model show that summer groundwater levels at the river margin are closer to river water levels (Figure 9d), resulting in more

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frequent reversals in the hydraulic gradient and consequentially a more dynamic subsurface exchange.

During autumn and winter, simulated groundwater movements across the floodplain are complex. A groundwater divide is simulated at the upstream and midstream parts of the meadow with subsurface flow simulated from both the river and ditch to the central part of the floodplain (e.g. Figure 10 a). At this time of year, groundwater levels on the floodplain are close to or above river water levels. In the lower part of the meadows, the high water tables act as a source of water to the river, with some groundwater exchange back to the river being simulated (Figure 10). During dry summer conditions, simulated river levels are above groundwater levels in all wells (Figure 11). The hydraulic gradient from the floodplain to the river is reversed, and instead, simulated subsurface flows are predominantly directed from the river to the floodplain (Figure 10d). Short-term (1–2 days) groundwater ridging and increases in floodplain storage are simulated during periods of peak river flows (Figures 10b, d and 11). However, longer-term (2–3 months) reversal of the hydraulic gradient and the consequent loss of river water to groundwater storage are simulated during dry periods in the summer, possibly because of a dominant down-valley hydraulic gradient (Figures 10c and 11).

Impacts of embankment removal on floodplain storage and flood peak attenuation

The impacts of embankment removal upon both overland and subsurface water storage within Hunworth Meadow are summarized for the 10-year simulation period in Figure 12. The volume of simulated surface water stored on the floodplain is greater in the restored model (Figure 12a). Particularly large differences between the results of the two models are associated with periods when major overbank flood events are simulated under restored conditions. For example, the overland storage volume increases approximately sixfold during the highest flow event (18/07/2001) after simulated restoration (maximum storage increase of 2159 m^3 compared with 373 m^3 for the embanked model). As discussed previously, although surface water is stored on the floodplain in the embanked scenario during these periods, groundwater rather than river water overtopping the riverbanks is the source of flooding. Overbank flows substantially enhanced surface storage, which increased 600% from an average of 144 m^3 in the embanked model to an average of 841 m^3 in the restored model over the 14 peaks in overland storage shown in Figure 12a. Differences in the simulated volume of

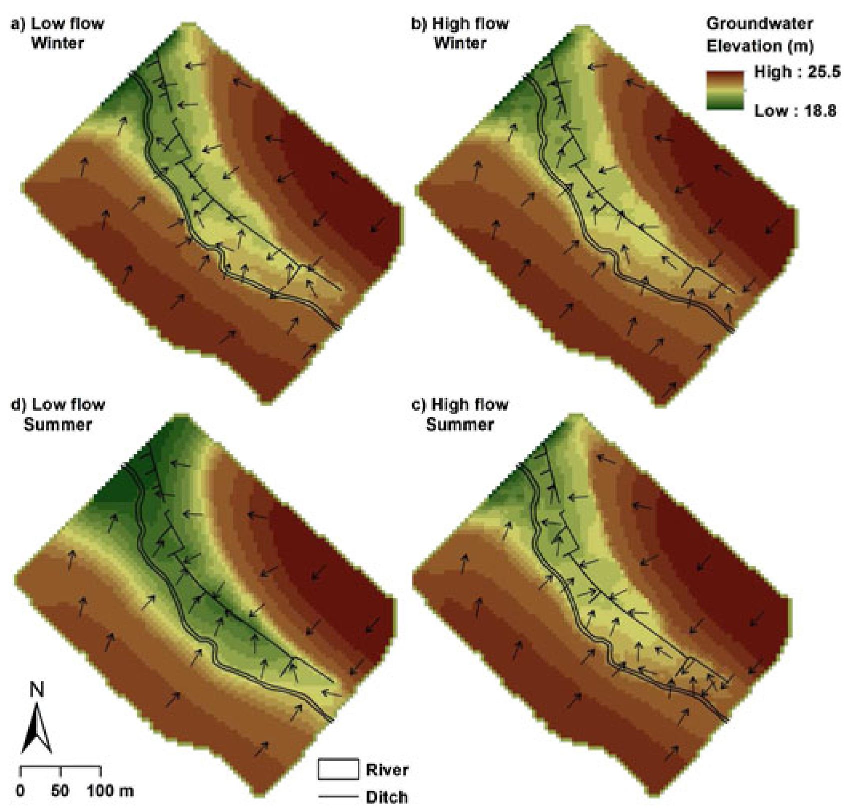

Figure 10. Simulated groundwater elevation and flow direction (arrows) during (a) low (01/01/07) and (b) high (15/10/04) river flow winter conditions and (c) low (01/09/09) and (d) high (28/05/07) river flow summer conditions simulated using the restored MIKE SHE scenario. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

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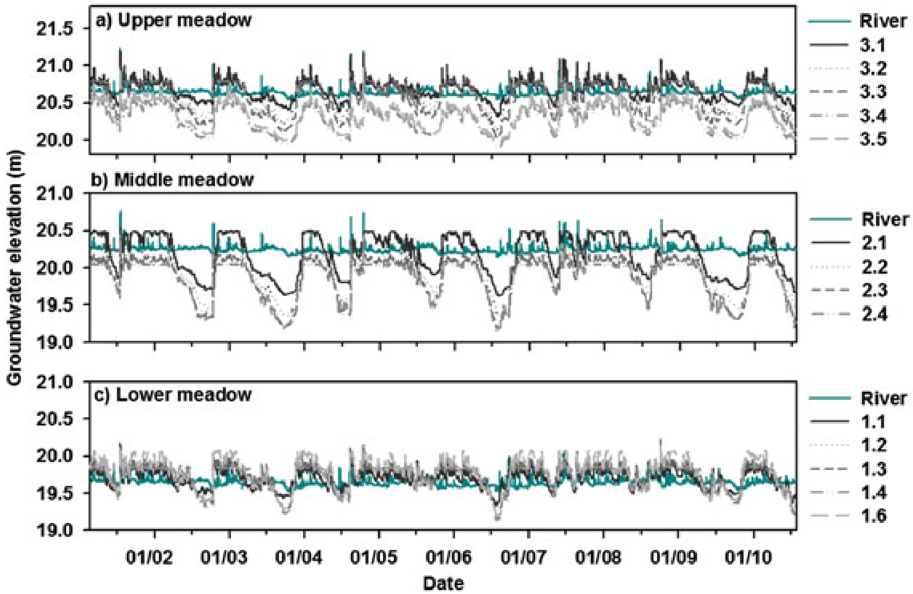

Figure 11. Time series of simulated post-restoration groundwater levels relative to the simulated river levels at each well transect from 2001 to 2010. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

subsurface storage between the embanked and restored models are much less pronounced (Figure 12b). During winter months, groundwater storage is very similar for both models as soils were typically at or near saturation and had limited available storage capacity. However, during the drier floodplain conditions that characterized summer months, subsurface storage is greater under restored conditions. The largest difference in subsurface storage occurred during a period of higher river flow at the end of the dry summer in 2004. At this time, storage change for the original embanked model was $-1099\,m^3$ compared with $-401\,m^3$ for the restored model (Figure 12b), equivalent to storage volumes in the floodplain of 38 022 and 38 675 m^3, respectively.

Although the annual actual evapotranspiration totals did not differ between embanked and restored models (Figure 13), the different components of total evapotranspiration were significantly different. Annual evapotranspiration from the unsaturated zone was on average 7% larger in the embanked model compared with the restored model ($p < 0.05$). This is the result of the higher water tables under restored conditions that limit the depth of the unsaturated zone and the duration of unsaturated conditions at the

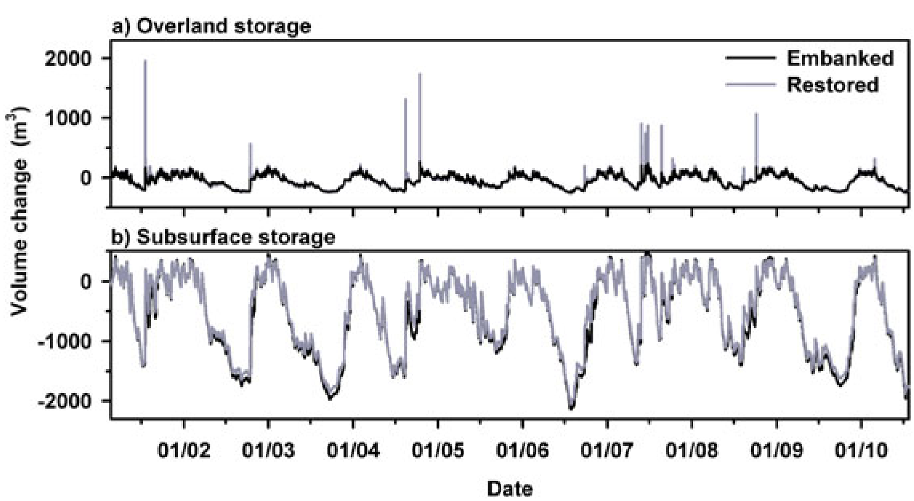

Figure 12. Times series of simulated change in (a) overland and (b) subsurface storage for the embanked and restored scenarios. Volume change is set at $0\,m^3$ at the beginning of the simulations (i.e. 20/02/2001) This figure is available in colour online at wileyonlinelibrary.com/journal/rra

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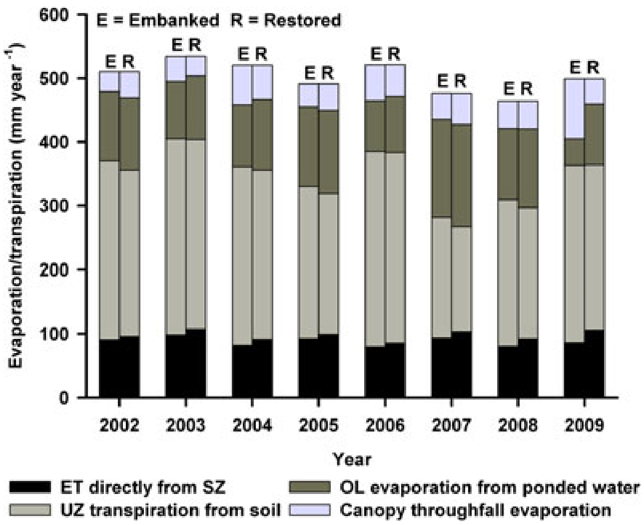

Figure 13. Annual evapotranspiration for the embanked (E) and restored (R) scenarios for the hydrological years 2002–2009. Total evapotranspiration is broken down into the contributing unsaturated, saturated, overland and canopy components. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

surface. Conversely, evapotranspiration from the saturated zone and evaporation from ponded overland water were on average 10% and 12% larger for the restored model ($p < 0.05$), respectively.

For the embanked model, river discharges were almost identical at the upstream and downstream ends of the modelled reach demonstrating that most flows are retained within the river channel (Figure 14). However, a slight reduction in outflows (of between 1% and 3%) is evident during the highest river flow events (flows $>1.2\,m^3\,s^{-1}$), likely associated with loss of flow to bank storage given the absence of simulated overbank flooding. For the restored model, differences between river inflows and outflows began at lower flows (around $1.0\,m^3\,s^{-1}$) compared with the embanked model. The largest overall reductions in river flow, however, occured during the largest overbank events ($>1.5\,m^3\,s^{-1}$) when inundation and recharge to the water table occurred across the floodplain. Embankment removal and restoration of overbank flows onto the floodplain had a moderate effect on flood peak attenuation. The peak discharge of the largest flood (18/07/2001) was reduced by 24% from $2.94\,m^3\,s^{-1}$ at the top of the restored reach to $2.31\,m^3\,s^{-1}$ at the downstream end (Figure 14). Following the highest river flows, outflow was marginally greater than inflow (maximum 2% and 3% in the embanked and restored scenarios, respectively), because of some return flow from the floodplain to the river. However, these differences were barely noticeable in Figure 14.

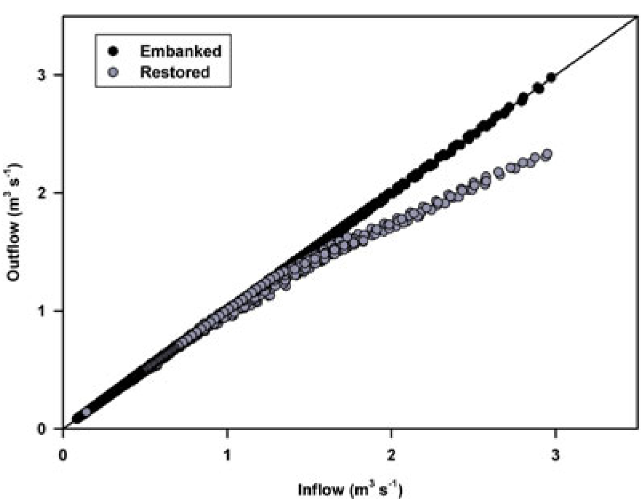

Figure 14. Comparison of simulated hourly river inflow versus outflow before and after river restoration. Values above the solid line indicate water loss (outflow > inflow), whereas values below the line indicate net retention (outflow < inflow) within the reach This figure is available in colour online at wileyonlinelibrary.com/journal/rra

DISCUSSION

River channelization and embankments constrain river flows within deeper, narrower cross sections to reduce overbank flows and thus restrict hydrological connectivity between rivers and their floodplains. In contrast, the bankfull discharge of more natural river channels is generally thought to be in the range of the 1- to 2-year recurrence interval flood event (Darby and Simon, 1999). Floodplain inundation is a major hydrological event that can attenuate downstream flood peaks through surface water storage and recharge of the floodplain aquifer and create a more heterogeneous riparian habitat through flood disturbance and deposition of nutrient-rich sediments (Amoros and Bornette, 2002; Naiman *et al.,* 2010; Shrestha *et al.,* 2014; Tockner *et al.,* 2000). Restoration of rivers to a more natural form is an increasingly accepted long-term solution for improving river health and functioning and is likely to increase in practice, encouraged through legislative requirements of the Water Framework (Directive 2000/60/EC) and Floods Directives (Directive 2007/60/EC) and the interests of local groups (Richter *et al.,* 2003; Perfect *et al.,* 2013.). Understanding how restoration affects river flow dynamics and connections with the floodplain is necessary to be able to predict and evaluate the success of restoration schemes and guide future practices. Hydrological/hydraulic modelling as undertaken in the current study offers enormous potential to improve understanding of river–floodplain interactions and the impacts of restoration projects. This study is one of few reported in the literature to present both pre-restoration and post-restoration hydrological data and to

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directly quantify the hydrological effects of river restoration using these data in combination with hydrological/hydraulic models.

Observed groundwater levels on the floodplain at Hunworth Meadow before and after embankment removal were simulated well by the two coupled MIKE SHE/MIKE 11 models. The models successfully reproduced groundwater responses to high-magnitude flood events, although they overestimated groundwater levels at the base of the hillslope (e.g. Well 2.4). This may be because either the model grid resolution was unable to sufficiently represent the topography of the ditch and its immediate surroundings or the MIKE SHE drainage function did not adequately simulate drainage towards the topographic lows in the region of the ditch. Nonetheless, the coupled MIKE SHE/MIKE 11 models were able to adequately predict temporal changes in groundwater levels across the floodplain, capturing intra-annual variations in these levels associated with climate as well as changes in hydrological fluxes related to the restoration. Sensitivity analyses during model calibration revealed that the models were responsive to the overland Manning's coefficient. Greater resistance to flow on the floodplain (e.g. applying Manning's *n* values for woodland versus grassland) reduced overbank flow depth and, after flooding, increased flood retention on the meadow. This demonstrates the importance of vegetation type for the management of riparian lands for reducing flood risk downstream (e.g. Piegay, 1997; Tabacchi *et al.,* 2000).

The results from the two models developed using identical hydrometeorological conditions, but with different topographical characteristics to reflect pre-restoration and post-restoration conditions, indicate four main hydrological responses to embankment removal on the River Glaven: (1) an increase in the frequency at which bankfull discharges are exceeded and in turn overbank inundation of the floodplain that was not simulated under embanked conditions; (2) increased groundwater levels and subsurface storage within the floodplain; (3) increased overland storage on the floodplain surface, especially during winter; and (4) moderate declines in downstream flood peaks. These responses are consistent with those reported following embankment removal and 'pond and plug' meadow restorations (where floodplain alluvium is excavated to plug-incised channels) on, for example, the River Cherwell, Southeast England (Acreman *et al.,* 2003), the headwaters of the Feather River, Northern California (Loheide and Gorelick, 2007) and Bear Creek, Northern California (Hammersmark *et al.,* 2008).

A major aim of the river restoration at Hunworth Meadow was the re-establishment of hydrological linkages between the river channel and floodplain. Model results suggest that prior to restoration, the embankments restricted river flows to the channel, which limited river–floodplain hydrological exchange to slow lateral subsurface flow (Clilverd *et al.,* 2013). Removing the embankments has restored overbank water transfers onto the floodplain, modifying the floodplain's hydrological regime, to form a more natural and dynamic wetland ecotone driven by flood disturbance. Widespread inundation occurred across the floodplain during high river flows ($>1.7\,m^3\,s^{-1}$) and reached as far as the hillslope (~50 m from the river). Large overbank flows were of short duration (around a day) and were separated by large time intervals (2.9 year return period). Localized inundation of the immediate riparian area (within 5 m of the channel) was a much more frequent event (0.22-year return period). Increased river water incursions on to the floodplain is likely to improve continuity with groundwater and enhance the supply of river nutrients to soil microbes and plant roots, an important influence on species composition, richness, primary productivity and nutrient cycling (e.g. nitrification, denitrification and methanogenesis) within wetland environments (Amoros and Bornette, 2002; Clilverd *et al.,* 2008; Hedin *et al.,* 1998; Pinay *et al.,* 2002).

The groundwater regime is one of the most important factors determining the plant communities that are present on floodplains (Castelli *et al.,* 2000; Silvertown *et al.,* 1999). Hydrological models such as MIKE SHE therefore provide useful tools for evaluating the effects of river restoration on water table depths, which can in turn be used to predict shifts in vegetation communities and guide floodplain management (e.g. Thompson *et al.,* 2009). At Hunworth Meadow, groundwater levels responded differently across the floodplain to embankment removal. Substantial increases in groundwater levels (0.4–0.6 m) occurred at the river–floodplain margin, where connectivity with the river was greatest and frequent localized overbank flooding occurred. This resulted in increased surface soil saturation throughout the year, which is likely to promote colonization by wetland plant species that can tolerate waterlogging (e.g. Wheeler *et al.,* 2004). Restoration also improved drainage between flood events, which could reduce flooding stress and lessen the impact of large floods on plant communities during the growing season. Smaller increases in water table elevation occurred as distance from the river increased, with the exception of the ditch area that received floodwaters during large overbank events. As a result, the effects of restoration on floodplain biota are expected to vary spatially across the floodplain. Surface flooding and consequent surface water storage increased the volume of subsurface storage and reduced aquifer head recession over the summer. This was due to increased surface water inundation at the river–floodplain margin and ponding of floodwater in topographic depressions on the floodplain. The simulated increases in groundwater levels and subsurface storage in this study are consistent with modelled increases in groundwater levels simulated by Hammersmark *et al.* (2008) using a MIKE SHE model of floodplain restoration in Northern California.

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Prior to restoration, model results suggest that bank storage contributed to a slight (maximum 3%) decrease in downstream flood peaks. River water intrusion increased during periods of elevated river stage, which reversed the hydraulic gradient on the floodplain and directed some subsurface flow away from the river. However, removal of the embankments resulted in a substantially more marked response in flood peak attenuation. Most of the overbank water was stored temporarily on the floodplain surface and in the ditch. Most floodwater returned to the channel downstream with improved drainage being facilitated by embankment removal whereas prior to restoration embankments acted as a barrier for surface water exchange from the floodplain to the river. Whilst some overbank water was infiltrated, no noticeable changes in baseflow due to return flows occurred following inundation events.

Before embankment removal, the floodplain at Hunworth Meadow was a groundwater-dominated system. Rapid groundwater recharge occurred in response to precipitation and rising river levels, likely associated with pressure differences across the floodplain (e.g. MacDonald *et al.,* 2014). During high winter river flows, groundwater was typically close to the soil surface, which limited the capacity for subsurface storage. Increased storage was available in soils in the summer. Therefore, after restoration, the greatest attenuation of flood peaks occurred when floods followed a period of low rainfall (in particular during warm and dry summers). Although restoration increased surface water inundation and surface water storage, total evapotranspiration was unchanged. This was attributed to the rapid response of groundwater to river levels and subsequent groundwater flooding that resulted in saturated surface soils in both pre-restoration and post-restoration conditions. This response may vary in different hydrogeological settings, where evapotranspiration from inundated areas may act to reduce overland runoff and further attenuate flood peaks.

Expansive inundation and storage of floodwaters on Hunworth Meadow resulted in a maximum reduction in peak river flows of 6–24%, along the length of restored reach (~400 m). This is a similar contribution to flood peak attenuation reported by other modelling studies. For instance, reductions in peak flows of 10–15% were simulated along a 5-km reach of the River Cherwell, UK (Acreman *et al.,* 2003), and 13–25% reductions in river discharge were reported along 3.6 km of restored channel at Bear Creek, Northern California (Hammersmark *et al.,* 2008). Logically, providing increased room for floodwater storage on floodplains favours greater reductions in flood peaks, which is an appreciable benefit of river restoration.

Many recent reviews have identified the need for larger-scale restorations that include an environmental management plan for the catchment as a whole, particularly where problems persist throughout the catchment, for example, agricultural fertilizer runoff, habitat fragmentation and urbanization (Bernhardt and Palmer, 2011; Harper *et al.,* 1999; Wharton and Gilvear, 2007). Indeed, this project is part of a wider landscape approach to restoration being implemented along the River Glaven to reconnect and buffer an array of aquatic habitats of varying sizes (e.g. rivers, streams, ponds and ditches), with the aim of repairing autonomous river processes and associated ecosystem services (e.g. biodiversity and water quality) within the catchment (Sayer, 2014). The removal of embankments along other reaches of the river that is proposed as part of this project could therefore be expected to have a cumulative impact of flood peak recession.

River restoration, and the associated improvements to river–floodplain functioning (e.g. enhanced hydrological connectivity, groundwater retention and flood peak attenuation), may provide an important tool for buffering the hydrological regime of wetlands and other aquatic ecosystems against some of the extreme climate variability predicted over the next century (IPCC, 2014). In the UK, five of the six wettest years have occurred since 2000, and eight of the warmest years have all occurred since 2002 (Met. Office, 2015a). The wettest May to July on record since 1766 occurred in 2007 during the observational period of this restoration study (IPCC, 2014). Indeed, 2014 was the wettest winter and warmest year in the UK for over 100 years, suggesting a trend towards warmer and wetter weather (Met. Office, 2015a, 2015b). The majority of climate change scenarios for the UK predict that the frequency and magnitude of floods will increase because of increased winter precipitation (Thompson, 2012; Wilby *et al.,* 2008). Increases in air temperature will also likely alter evapotranspiration rates and groundwater recharge, which is likely to affect wetland species that are sensitive to changes in hydrological regime (e.g. Araya *et al.,* 2011; Gowing *et al.,* 1998). For example, a climate impacts study conducted by Thompson *et al.* (2009) using MIKE SHE/MIKE 11 and UK Climate Impacts Programme (UKCIP) projections for the 2050s simulated lower water table depths and reduced magnitude and duration of surface water inundation within the Elmley Marshes, Southeast England. It was suggested that these hydrological changes would lead to a loss of specialist wetland plants adapted to the current high water tables. Similarly, House *et al.* (2016) used a MIKE SHE/MIKE 11 model of a riparian wetland on a tributary of the River Thames to demonstrate spatially varying hydrological impacts because of climate change that would have implications for both wetland flora and fauna. Such results point to the potential for further analysis using the hydrological/hydraulic models of the Hunworth Meadow to assess the capacity of river restoration to proof wetlands from the hydrological impacts of climate change.

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CONCLUSIONS

This study employed coupled MIKE SHE-MIKE 11 hydrological-hydraulic models to investigate the hydrological impacts of river restoration involving embankment removal along a stretch of River Glaven, North Norfolk, UK. The removal of the river embankments provided the physical geomorphological conditions to allow regular overbank flows and provided space for water to spill out onto the adjacent floodplain. The restoration increased river–floodplain hydrological connectivity, creating a more disturbance-based riparian zone that extended laterally from the river towards the edge of the floodplain. Model results have the potential to be used in the prediction of ecological responses to changes in water table depth and duration of water surface resulting from floodplain restoration.

Our approach was used to quantify the impact of restoration and could be used at other similar sites when inter-annual climate variability and relatively short observational periods prevent direct pre-restoration and post-restoration comparisons. In addition, it could be used to understand the effects of past restorations when pre-restoration observational data are not available. This approach may be applied in the planning stage of restoration projects to determine the suitability of the site and whether desired hydrological conditions can be achieved. When combined with climate projections, models such as those developed in this study offer the potential to predict future conditions at a restoration site under a changing climate.

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