



Projecting impacts of climate change on hydrological conditions and biotic responses in a chalk valley riparian wetland



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SUMMARY

Projected changes in climate are likely to substantially impact wetland hydrological conditions that will in turn have implications for wetland ecology. Assessing ecohydrological impacts of climate change requires models that can accurately simulate water levels at the fine-scale resolution to which species and communities respond. Hydrological conditions within the Lambourn Observatory at Boxford, Berkshire, UK were simulated using the physically based, distributed model MIKE SHE, calibrated to contemporary surface and groundwater levels. The site is a 10 ha lowland riparian wetland where complex geological conditions and channel management exert strong influences on the hydrological regime. Projected changes in precipitation, potential evapotranspiration, channel discharge and groundwater level were derived from the UK Climate Projections 2009 ensemble of climate models for the 2080s under different scenarios. Hydrological impacts of climate change differ through the wetland over short distances depending on the degree of groundwater/surface-water interaction. Discrete areas of groundwater upwelling are associated with an exaggerated response of water levels to climate change compared to non-upwelling areas. These are coincident with regions where a weathered chalk layer, which otherwise separates two main aquifers, is absent. Simulated water levels were linked to requirements of the MG8 plant community and Desmoulin's whorl snail (*Vertigo moulinsiana*) for which the site is designated. Impacts on each are shown to differ spatially and in line with hydrological impacts. Differences in water level requirements for this vegetation community and single species highlight the need for separate management strategies in distinct areas of the wetland.

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1. Introduction

Wetlands are highly vulnerable to climate change due to the primary importance of the hydrological regime in controlling their ecological characteristics (e.g. Baker et al., 2009). Unequivocal warming of the climate (IPCC, 2014) will alter precipitation and evapotranspiration rates, and result in changes to runoff and groundwater levels. The key roles of these processes in controlling wetland vegetation (Baldwin et al., 2001; Wheeler et al., 2009), animals (Ausden et al., 2001; McMenamin et al., 2008) and biogeochemical cycling (McClain et al., 2003; Lischeid et al., 2007) means that climate change is likely to have major impacts on the world's wetlands, their flora and fauna as well as delivery of the many ecosystem services which they provide.

Groundwater may contribute a significant proportion of the water balance in riparian wetlands (Bravo et al., 2002; Krause and Bronstert, 2005; House et al., 2015b), which can strongly influence the hydrological regime, nutrient status and species composition (Wheeler et al., 2009; House et al., 2015a). Groundwater/surface-water interactions are inherently complex, being time dependent (Hunt et al., 1999), spatially heterogeneous (Hunt et al., 1996; Lowry et al., 2007; House et al., 2015a), and sensitive to topographical, geological and climatic controls (Winter, 1999; Sophocleous, 2002). The effects of climate change on regional aquifers and catchment runoff may cause intricate and significantly detrimental impacts to wetlands underlain by permeable geology, such as the chalk lowlands of southeast UK (Herrera-Pantoja et al., 2012). The impacts of climate change upon such wetlands should ideally therefore be assessed on an individual basis in relation to their water supply mechanisms and position within the catchment (Acreman et al., 2007).

Hydrological changes due to climate change may be linked to water level requirements of different species and communities to

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infer ecological impacts (Wheeler et al., 2004; Acreman et al., 2009). For instance, water table level regime is a dominant control on wetland plant communities (Silvertown et al., 1999). In the UK, the preferred water levels and depths to groundwater for wetland plants and communities have been well documented (Elkington et al., 1991; Newbold and Mountford, 1997; Gowing et al., 2002; Wheeler et al., 2004, 2009). Modifications to a wetland's hydrological regime may also be linked to changes in animal species distribution. Focus has centred on the indirect impacts to wading birds through the habitat requirements of macroinvertebrates that serve as their prey and the penetrability of soils by their beaks. Waterlogged areas sustain a higher biomass of surface-active and aerial invertebrates (Plum, 2005; Eglington et al., 2010). For example, the distribution of the near threatened Desmoulin's whorl snail (*Vertigo moulinsiana*) (Killeen et al., 2012) has been directly linked to water levels (Tattersfield and McInnes, 2003). Indirectly, softer ground allows snipe (*Gallinago gallinago*) to forage for food more easily (Ausden et al., 2001; Smart et al., 2008), while drains and wet rills provide favoured feeding grounds for lapwing (*Vanellus vanellus*) and redshank (*Tringa totanus*). Alterations to a wetland's water balance, and in turn its water level regime, due to climate change could therefore lead to shifts in habitat availability (Johnson et al., 2005), and affect the capacity of a wetland to support populations of conservation importance (Sorenson et al., 1998; Herron et al., 2002; Thompson et al., 2009).

There are few hydrological modelling studies at a suitable resolution which link water table predictions directly to plant and animal requirements for individual wetlands (Thompson et al., 2009; Carroll et al., 2015). To our knowledge, none do so for individual wetlands with groundwater contributions. However, an ability to accurately predict the impacts of climate change is vital for wetland management where species conservation and ecosystem service provision relies on managing hydrological functions (Acreman et al., 2009). Models able to accurately represent wetland hydrology will enable the assessment of possible degradation to wetland ecosystems through climate change (Acreman and Jose, 2000). In turn, such models will permit assessment of the likely success of modifications to wetland management designed to mitigate the impacts of climate change. Models are required that can accurately simulate groundwater levels at the fine-scale resolution associated with water level requirements of different species and communities (Thompson et al., 2009). Changes in water table level of less than 0.1 m may have profound effects on species composition, and provide conditions which favour distinct species or communities over those currently dominant at a given site (Wheeler et al., 2004). Whilst, as shown in Table 1, hydrological modelling has been used to assess some ecological impacts of climate change, in many cases this has not been undertaken at a resolution sufficient to directly infer impacts for particular species and communities; instead surmising effects through changes in habitat availability (Johnson et al., 2005; Candela et al., 2009; Barron et al., 2012). Other studies have postulated impacts generalised over regional scales (Acreman et al., 2009; Herrera-Pantoja et al., 2012).

The aim of this study is to assess the impacts of climate change on a riparian wetland in the chalk lowlands of the UK. The objectives are to: (1) Project changes in hydrometeorological inputs to a distributed hydrological/hydraulic model of the wetland under scenarios of different climate sensitivities to incorporate the uncertainty associated with climate change, (2) use the hydrological model to investigate how climate change scenarios affect wetland hydrology, and (3) compare simulated water levels under each climate change scenario to the requirements of conservation species/communities for which the site is designated. In this way the study provides an assessment of the potential ecohydrological effects of climate change upon the wetland and resulting management implications of these changes.

2. Study area

The Centre for Ecology & Hydrology (CEH) River Lambourn Observatory located in Berkshire, UK (51.445°N 1.384°W) comprises c. 10 ha of riparian wetland which is bordered to the east by a 600 m stretch of the River Lambourn (Fig. 1). The Westbrook Channel divides the wetland into northern and southern meadows. The site is located 13 km downstream from the ephemeral source of the River Lambourn at Lynch Wood, Lambourn (51.512°N, 1.529°W), the perennial head of which is situated 6–7 km downstream of the source at Maidencourt Farm (51.481°N, 1.464°E). The river drains the Chalk of the Berkshire Downs and is characterised by a large baseflow component. The baseflow index and mean discharge of the Lambourn at Shaw, the nearest gauging station 5 km downstream of the observatory, are 0.96 and 1.73 m³ s⁻¹, respectively (Marsh and Hannaford, 2008).

The wetland owes its designation as a Site of Special Scientific Interest (SSSI) and Special Area of Conservation (SAC) to the presence of Desmoulin's whorl snail (*V. moulinsiana*) and the MG8 vegetation community (*Cynosurus cristatus* – *Caltha palustris* grassland) of the UK National Vegetation Classification (NVC) (Rodwell, 1991). The site was managed as flood pastures and water meadows until the middle to late 20th century (Everard, 2005). Maps dating to the 1880s show a characteristic network of predominantly linear conduits. Most of these channels have since infilled naturally and are absent from current maps although the relic drainage network is still evident in the topography. Current management efforts concentrate on the river, where periodic cutting of instream macrophyte growth is carried out to maintain flood conveyance and lower water levels (Old et al., 2014).

A previous field campaign using three-dimensional (3D) electrical resistivity tomography (ERT) (Chambers et al., 2014) revealed a complex subsurface architecture. This comprises bedrock Chalk, overlain by a discontinuous layer of highly weathered 'putty' chalk (Younger, 1989), then gravels and peat. The peat and gravels are considered to have good hydraulic connectivity, with head boundaries in the River Lambourn and Westbrook broadly controlling water levels across the wetland (Chambers et al., 2014; Old et al., 2014; House et al., 2015a, 2015b). The putty chalk acts as a low permeability confining layer to the Chalk aquifer. Leakage occurs between the Chalk and gravels where the putty chalk is absent causing localised increases in water levels, which occur mainly in the north meadow (House et al., 2015a, 2015b). The relationship between the river and underlying gravels involves components of groundwater flow both parallel and transverse to the river, and with both influent and effluent behaviour (Lapworth et al., 2009; Allen et al., 2010).

The site instrumentation network and monitoring schedule are detailed in House et al. (2015b). Briefly, the network contains piezometers installed in the peat (P), gravel (G), and chalk (C) (Fig. 1). Stage boards are located along the River Lambourn (L1, L3–L7) and Westbrook (W1–W3), with a stilling well at L2. An automatic weather station (AWS) is installed in the south meadow.

3. Methodology

3.1. Simulation of baseline conditions

A hydrological model of the CEH Lambourn Observatory was produced using the integrated MIKE SHE/MIKE 11 modelling system, which simulates the major components of the land-based phase of the hydrological cycle (Graham and Butts, 2005). A detailed description of the MIKE SHE model of the site is provided by House et al. (2015b). For this study, the model area was discretised using a 5 m × 5 m grid, producing 4261 computational cells.

Table 1
Summary of relevant studies using hydrological models to assess the impacts of climate change on wetland ecology.

Source	Wetland type and location	Number/type of scenarios	Resolution/grid size	Ecological impacts
Johnson et al. (2005)	Prairie potholes, central USA	3/equilibrium scenario combinations of temperature and precipitation	Regional	Under a drier climate habitat for waterfowl would shift spatially
Acreman et al. (2009)	Wet heaths/raised mires and riparian, various locations, UK	1/UKCIP02 medium–high emissions scenario 2080s	Regional	Reduced summer rainfall and increased evaporation with put stress on plant communities in late summer and autumn
Barron et al. (2012)	Coastal, Perth Basin, Western Australia	3/outputs from 15 GCMs wet, medium and dry scenarios 2030s	500 m × 500 m	Impacts not uniform but could cause a threat to water-dependent ecosystems
Candela et al. (2009)	Groundwater-fed, Majorca, Spain	2/HadCM3 medium–high (A2) and medium–low (B2) emissions scenarios 2020s	250 m × 250 m	Aquifer recharge reduction could cause loss of wetland habitat
Herrera-Pantoja et al. (2012)	Groundwater-fed fen, various locations, East Anglia, UK	1/UKCIP02 high emissions scenario 2020s, 2050s and 2080s	250 × 250 m and 50 × 50 m	Decline in water levels could cause loss of species with small tolerance to dry conditions
Thompson et al. (2009)	Lowland wet grassland, Elmley Marshes, southeast UK	4/UKCIP02 low emissions, medium–low emissions, medium–high emissions and high emissions scenarios 2050s	30 × 30 m	Lower water levels result in loss of some grassland species and reduced suitability for wading birds such as lapwing and redshank
Carroll et al. (2015)	Blanket bog, various locations, UK	1/UKCP09 intermediate scenario 2011–2080	10 × 10 m	Falling water tables could cause 56–81% declines in crane fly abundance, and 15–51% declines in specialist predatory birds by 2051–2080

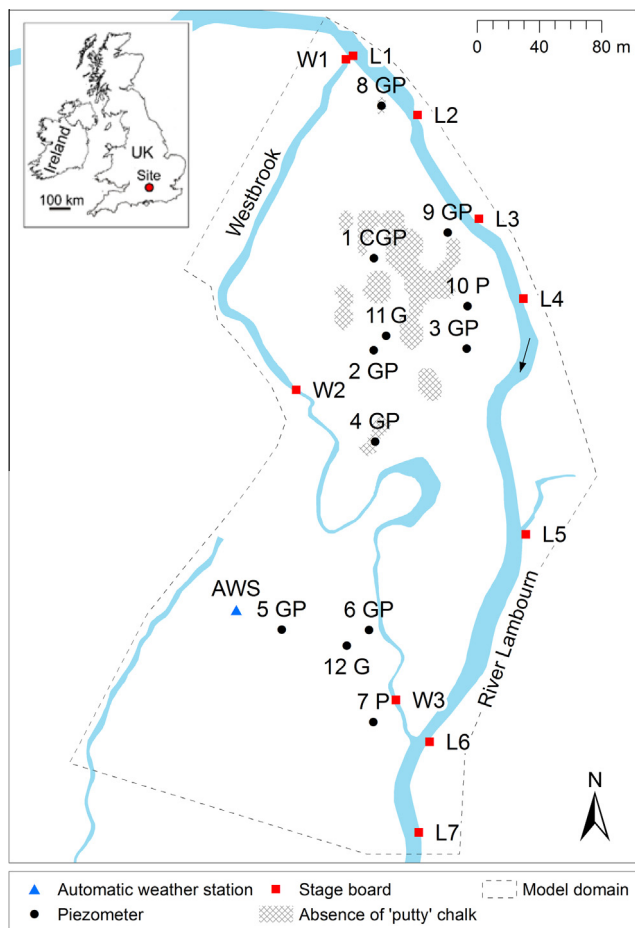


Fig. 1. The CEH River Lambourn Observatory, showing the instrumentation network with chalk (C), gravel (G) and peat (P) piezometer locations, MIKE SHE model domain, and horizontal extent of absences in highly weathered 'putty' chalk.

The computational time for each model run was approximately 30 min.

Instream macrophyte growth and its management were represented by manipulating channel bed roughness (Manning's n).

Weed cuts on 01 May 2013, 16 July 2013, 21 May 2014 and 23 July 2014 signified rapid decreases in channel bed roughness, which otherwise increased gradually during the growing season. Inflows for the upstream channel boundary were derived from a relationship between monthly measurements of discharge at L1 (Fig. 1) and corresponding flow at the downstream Shaw gauging station. The downstream boundary was set to follow stage observations at L7. Numerical errors in solving overland flow were reduced through specification of the Explicit Numerical Solution method, which calculates flow based on individual cell heads. The Manning's n roughness coefficient for overland flow was varied during calibration. Unsaturated flow was calculated in a reduced number of cells, subsetted automatically within MIKE SHE based on initial depth to groundwater. Saturated hydraulic conductivity and effective saturation were parameterised through calibration.

Vertical and horizontal hydraulic conductivities of the peat and gravels were employed as calibration terms. The model was also sensitive to vertical hydraulic conductivity in the Chalk, which formed a manual calibration parameter in the initial model build as it predominantly affected bias. This was finalised at 0.00438 ms^{-1} , an order of magnitude above the horizontal hydraulic conductivity taken from the literature as $4.4 \times 10^{-4} \text{ ms}^{-1}$ (Younger, 1989). Horizontal extents of the putty chalk (Fig. 1) were represented as a discontinuous 1 m layer at the top of the Chalk bedrock. Hydraulic conductivity was specified as $1 \times 10^{-6} \text{ ms}^{-1}$ in line with the literature (Younger, 1989), although model performance did not vary below this value. Gaps in the putty chalk were allocated the same hydraulic conductivity as the Chalk. For the Chalk aquifer, head boundaries were based on observations from piezometer 1C. These were adjusted to differences in elevation along the model boundary by linear interpolation. The assumption that the boundary condition in the chalk varies with terrain elevation was considered reasonable given the site's riparian position and relatively flat topography. Gravel boundaries were set to a constant flux gradient of 0.003 m in the north and south, following the topographic gradient, with the remaining boundaries defined as zero flow. Zero flow boundaries were assigned around the peat and putty chalk layers where lateral flow was assumed to be minimal due to low hydraulic conductivities.

The periods 01 February 2013–01 December 2013 and 01 December 2013–01 October 2014 were used for split sample calibration and validation, respectively. This was based on compar-

isons between simulated and observed head elevations in the peat and gravel piezometers installed at the site. Calibration and validation of channel stage was based on comparisons between MIKE 11 simulated stage and observations from stage boards. An automatic multiple objective calibration was performed based on the shuffled complex evolution method (Duan et al., 1992; Madsen, 2000, 2003). Model performance statistics comprised the root mean square error (RMSE) for goodness of fit and the absolute value of the average error for bias. The calibration problem is solved by defining a single objective function that aggregates the different objective functions into a single statistic (Madsen, 2003). The auto-calibration routine was run until convergence criteria were met, in this case when the minimum relative change in the aggregated objective function was less than or equal to 0.01. This required 88 simulations with a computation time of approximately 2 days. Manual adjustment of calibration parameters further improved model performance which was assessed using the Pearson correlation coefficient (R), the Nash–Sutcliffe coefficient ($R2$) (Nash and Sutcliffe, 1970) and RMSE. Model performance was classed as very good or excellent in most cases (Table S1 and Fig. S1 of the electronic supplementary material). Overall mean values for RMSE, R and $R2$ were 0.063 m, 0.92 and 0.75, respectively.

3.2. Simulation of climate change

Climate change scenarios were derived for the 2080s using datasets from the Future Flows and Groundwater Levels project (Jackson et al., 2011; Prudhomme et al., 2012). These include 11-member ensembles of 1 km gridded time series projections (1950–2098) of precipitation, PET, and groundwater levels for Great Britain based on the UKCP09 Hadley Centre's HadRM3-PPE run under the medium emissions (SRES A1B) scenario (Murphy et al., 2009). The Met Office Hadley Centre's Regional Climate Model HadRM3 represents parameter uncertainty through model variants with different climate sensitivity, defined as the equilibrium mean surface temperature change resulting from a doubling of the atmospheric CO₂ concentration (IPCC, 2014). However, emission scenario uncertainty is excluded. HadRM3-PPE consists of an ensemble of eleven members of HadRM3 used to dynamically downscale HadGM3 global climate model outputs. The ensemble comprises one unperturbed member and 10 members with different perturbations to the atmospheric parameterisations (HCCPR, 2008). Climate sensitivities for each ensemble member along with the scenario run id plus the RCM run id and descriptive id used by the Met Office Hadley Centre are summarised in Table 2. Three scenarios (H, J and K) have climate sensitivities above the likely range of 2–4.5 °C estimated by the IPCC. Outputs from HadRM3-PPE are provided at a 25 km grid resolution. Due to differences in scale between local hydrological processes and modelled atmospheric

processes from the RCM, a bias correction and spatial downscaling procedure was applied to these outputs to obtain the Future Flows precipitation and PET projections (Prudhomme et al., 2012). PET time series were calculated using the Penman–Monteith equation using projected values of the equation's meteorological components. A British Geological Survey (BGS) ZOOMQ3D regional groundwater model of the Chalk aquifer of the Marlborough and Berkshire Downs and south-west Chilterns (Jackson et al., 2011) was used to provide the Future Flows projections of changes in groundwater levels (Haxton et al., 2012). It was not possible to drive the entire chalk boundary with predictions from the regional model as the grid was too coarse at 500 m × 500 m.

Model inputs of precipitation, PET, groundwater elevation and river discharge were perturbed for each climate change scenario using a delta factor approach (Wilby and Harris, 2006; Thompson, 2012). The baseline simulation comprised the combined calibration and validation period 01 February 2013–01 October 2014. Although this is a relatively short period, constraints were imposed by data availability and the approach replicates those used elsewhere (e.g. Thompson et al., 2009). Monthly percentage differences between the ensemble reference period (1961–1990) and the future period (2071–2098) were applied to each variable. This approach assumes that climate variability does not alter and provides no information on changes in event frequency and distribution (Chiew et al., 1995; Graham et al., 2007). However, it enables a robust comparison of average outcomes and has been widely used in hydrological studies of climate change (e.g. Arnell and Reynard, 1996; Limbrick et al., 2000; Kamga, 2001; Arnell, 2004; Thompson et al., 2009; Jackson et al., 2011).

Monthly delta factors for precipitation (%), PET (%) and groundwater level (m) were extracted from the relevant 1 km grid square of the Future Flows dataset for the HadRM3 ensemble. In the absence of extant delta factors for discharge for the study location from the Future Flows dataset, a rainfall–runoff model was developed for the Lambourn catchment at Shaw. This was developed using MIKE NAM, a deterministic, lumped model describing, in a simplified quantitative form, the behaviour of the land phase of the hydrological cycle (DHI, 2009). Following model calibration, climate change delta factors for discharge were derived by running the NAM model with catchment averaged precipitation and PET under each of the 11 HadRM3 ensemble members. These factors, expressed as a percentage, were subsequently applied to the original stream inflows used within the MIKE SHE model that were based on the relationship between discharge immediately upstream of the model area and at the Shaw gauging station.

Daily precipitation for the NAM model of the 234.1 km² Lambourn catchment was obtained from the CEH-GEAR dataset (Keller et al., 2015) which provides 1 km gridded estimates of daily and monthly rainfall for Great Britain and Northern Ireland derived from the Met Office national database of observed precipitation. Monthly PET totals (subsequently disaggregated to a daily time step assuming an even distribution through the month) were taken from the Met Office Rainfall and Evaporation Calculation System (MORECS), based on the Penman–Monteith equation and providing UK-wide coverage at a 40 km² grid square resolution (Thompson et al., 1981). Spatially uniform time series of both precipitation and PET were derived from the mean of those cells for the two datasets falling within the catchment. Calibration and validation of the NAM model was based on comparisons between daily observed and simulated discharge at the Shaw gauging station for the equally split period 1963–2012 (Fig. S2 of the electronic supplementary material). An automatic multiple objective calibration routine was based on agreement between mean simulated and observed runoff along with the root mean square error. Adjusted parameters included maximum water content in the surface and root zone storage, the overland flow runoff coefficient, time

Table 2
Climate sensitivities, run ID and model variant name for the HadRM3-PPE ensemble of Climate Projections (after HCCPR, 2008).

Run ID	Climate sensitivity	RCM run ID	RCM name
A	3.53485	afgcx	HadRM3Q0
B	2.58475	afixa	HadRM3Q3
C	2.81543	afixc	HadRM3Q4
D	3.43839	afixh	HadRM3Q6
E	4.39594	afixi	HadRM3Q9
F	3.89523	afixj	HadRM3Q8
G	4.44284	afixk	HadRM3Qk
H	4.88248	afixl	HadRM3Q14
I	4.54486	afixm	HadRM3Q11
J	4.79648	afixo	HadRM3Q13
K	7.11014	afixq	HadRM3Q16

constants for interflow, routing overland flow and routing base-flow, and the root zone threshold values for overland flow, inter flow and ground water recharge. As with the MIKE SHE model of the Lambourn Observatory, performance was classified following the Henriksen et al. (2008) scheme (see Table S1 of the electronic supplementary material). RMSE in this case was deemed excellent if below $0.5 \text{ m}^3 \text{ s}^{-1}$. Calibrated values for RMSE, R and R2 were $0.441 \text{ m}^3 \text{ s}^{-1}$, 0.89 and 0.80 respectively.

3.3. Assessment of ecological impacts of hydrological change

Simulated peat water table levels for both the baseline and each climate change scenario were compared to the water level requirements for the MG8 community which, as described above, contributes to the site's scientific and nature conservation status. These water level requirements are defined as monthly water table depth zones (Wheeler et al., 2004, 2009). Fig. 2i shows the requirements for the MG8 community as defined by Wheeler et al. (2004), with green areas indicating desirable conditions, amber representing tolerable conditions should the water table fall within these zones for limited periods, and red indicative of intolerable conditions. Analysis centred on establishing the favourability of current (baseline) water table conditions to supporting this community and whether climate change-related modifications to water tables are likely to cause a shift in hydrological conditions which could have implications for MG8 species.

Simulated peat water levels were also compared to the hydrological requirements of the conservation relevant Desmoulin's whorl snail. Tattersfield and McInnes (2003) suggested that optimal conditions for the snail occur where water levels are continuously above ground level, fluctuating between 0.6 and 0.0 m in winter and summer respectively. Suboptimal, yet tolerable, conditions exist where water levels fluctuate between 0.2 m above ground in winter and 0.2 m below ground in summer. If water levels drop below ground level in winter and are more than 0.4 m below the surface in summer the snail is unlikely to be present. These suggested conditions were used to define monthly ranges of desirable, tolerable and intolerable water levels for the Desmoulin's whorl snail in the same form as those used for the

MG8 vegetation community (Fig. 2ii). This enabled the same approach for defining the suitability or otherwise of baseline and scenario water level regimes for this individual species.

4. Results

4.1. Climate change impacts on model hydrometeorological inputs

Monthly delta factors for the hydrometeorological time series used to drive the MIKE SHE model of the Lambourn Observatory are summarised in Fig. 3 for each of the 11 ensemble member scenarios as well as the ensemble mean. Drier summer and wetter winter months are evident from the precipitation and PET change factors although the magnitude and duration of changes vary between ensemble members (Fig. 3i and ii). The ensemble mean shows increases in precipitation between October and March and decreases during the months April–September (Fig. 3i). November has the largest increase and greatest range of changes while the largest projected mean decline is in August. Delta factors for PET are positive in every month for all of the ensemble members (Fig. 3ii). The largest increases occur in late summer whilst the smallest changes are in mid to late winter. The inter-ensemble member range is particularly large in the latter half of the year when delta factors are the largest.

The discharge delta factors for the ensemble mean suggest declines in river flow throughout the year (Fig. 3iii). These are largest in October, whilst the declines are smallest in March. Of the 11 individual ensemble members only three show increasing discharge at any time of the year. The remaining eight members project declines in discharge throughout the year. The ensemble mean delta factors for Chalk aquifer groundwater levels generally show an increase, especially over the late winter months. Exceptions to these increases occur in August and October when there is no change, and September when small declines in groundwater level are projected. There is considerable variation in the delta factors for the individual ensemble members with only one showing an increase in level throughout the year, eight showing year round declines and the remaining two demonstrating higher levels in the first half of the year and declines thereafter.

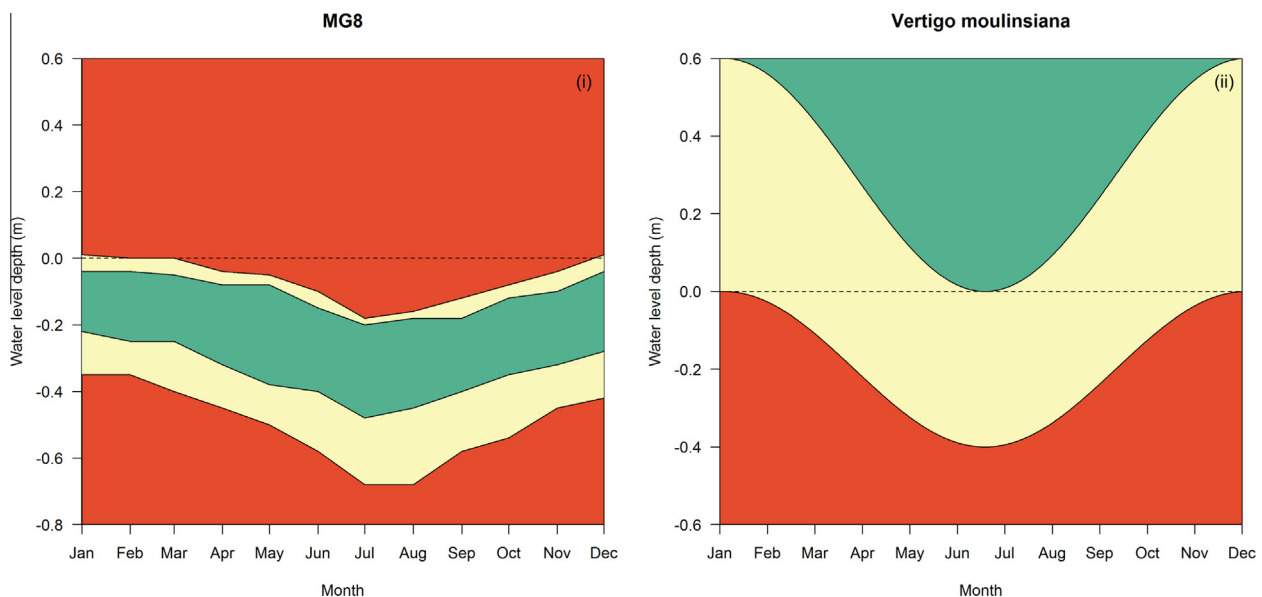


Fig. 2. Water level requirements for the MG8 vegetation community (after Wheeler et al., 2004) and Desmoulin's whorl snail (after Tattersfield and McInnes, 2003). Red – intolerable; amber – tolerable for limited periods; green – desirable. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

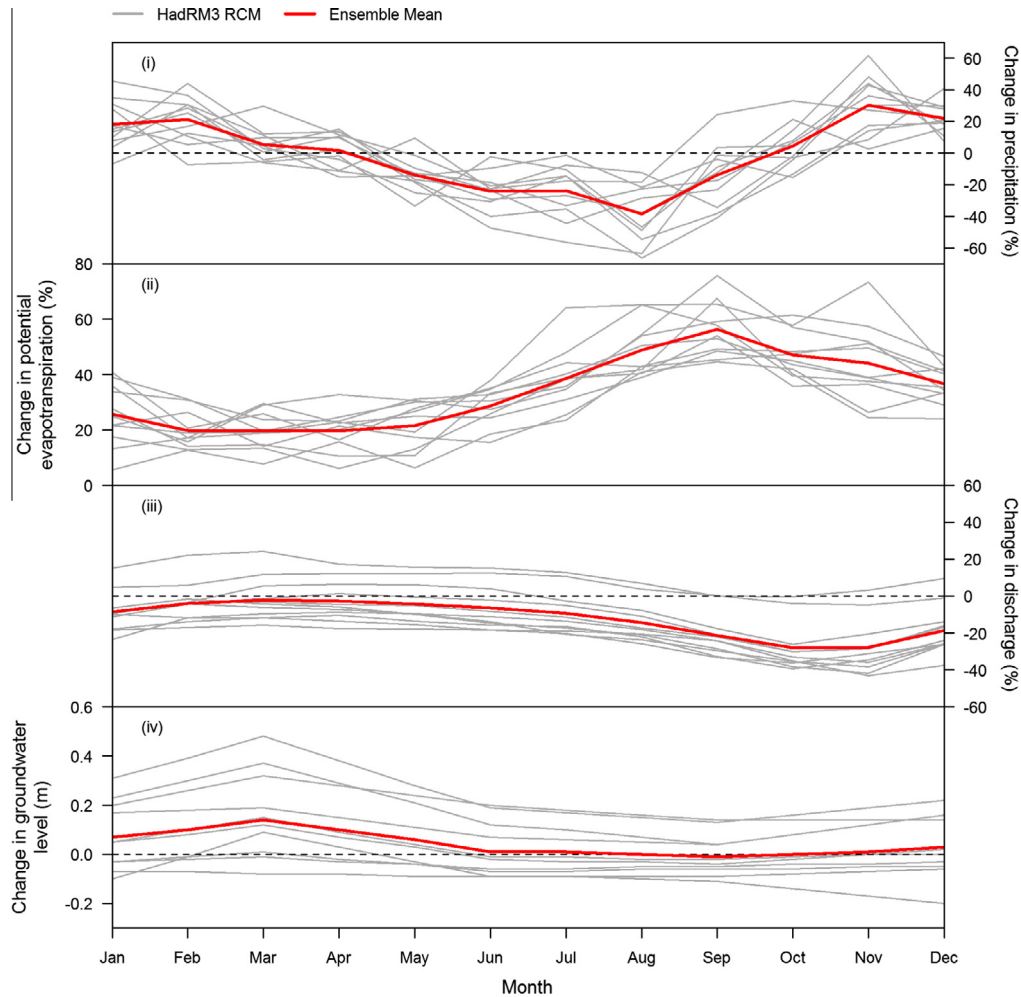


Fig. 3. Projected monthly climatic changes for the 2080s by ensemble member and mean: (i) precipitation, (ii) potential evapotranspiration, (iii) river discharge and (iv) groundwater level.

The effects of the delta factors on total annual baseline precipitation and PET as well as mean annual discharge and groundwater level for the complete hydrological year (01 October 2013–30 September 2014) of the simulation period are displayed in Table 3. The scenarios can be divided into two groups: those with precipitation minus PET above 100 mm which correspond to A–E with a mean climate sensitivity of 3.35 °C; and, those with precipitation minus PET below 100 mm (F–K with a mean climate sensitivity of 4.95 °C). The former are characterised by relatively higher precipitation and groundwater level, smaller increase in PET and either increases or decreases in mean discharge. Members of the second group have, on the whole, smaller increases (declines for scenario I) in precipitation and larger increases in PET. Mean discharge and groundwater level tend to decline although some individual members provide exceptions to these general trends. Total inflows to boundary conditions are also shown in Table 3. Percentage changes under each scenario are within the same order of magnitude as changes in precipitation. A multiple regression comparison with precipitation and PET yields a good relationship with $R^2 = 0.72$.

4.2. Climate change impacts on wetland hydrology

4.2.1. Wetland water levels

Climate change related modification to wetland water levels varies spatially and temporally (Figs. 4 and 5). Water level

Table 3

Annual baseline precipitation (P) (mm), potential evapotranspiration (PET) (mm), precipitation minus PET (P–PET) (mm), mean discharge (Q) ($\text{m}^3 \text{s}^{-1}$), mean chalk groundwater head at 1C (G) (mBGL) and total boundary inflow (mm) and changes (%) for precipitation and discharge and inflow, m for groundwater head, positive upward) in 2013/2014 for ensemble members and mean. Italicised values indicate negative changes.

Run ID	P (mm/%)	PET (mm/%)	P–PET (mm)	Q ($\text{m}^3 \text{s}^{-1}$ /%)	G (mBGL/m)	Inflow (mm/%)
Baseline	1081.4	764.3	317.1	2.37	0.31	20365.89
A	6.6	31.6	146.5	–6.53	0.11	5.2
B	2.8	26.8	142.6	–18.68	–0.08	–0.8
C	5.9	25.4	186.2	8.04	0.20	7.9
D	15.9	26.7	285.3	17.49	0.25	10.3
E	7.8	24.4	214.9	–2.48	0.17	8.0
F	0.8	35.1	57.3	–13.08	0.02	2.4
G	1.2	36.2	53.6	–11.32	–0.08	–0.7
H	3.4	36.8	72.3	–16.23	–0.04	0.3
I	–2.8	35.5	15.8	–20.38	–0.04	0.6
J	5.5	38.8	79.4	–8.91	0.03	2.5
K	5.1	38.1	80.6	–16.77	–0.08	–1.3
Mean	4.7	32.3	121.3	–8.08	0.04	2.8

responses fall into three spatial groups: locations in the north meadow that are characterised by upwelling groundwater (North – Upwelling), locations in the same part of the wetland where such upwelling is absent (North – no upwelling), and locations in the south of the wetland (South) where inter-ensemble member vari-

ability is small in comparison to northern parts of the wetland, especially during periods of relatively high water. Hence, Figs. 4 and 5 show simulated wetland water levels at selected locations that are characteristic of these groups (piezometers 2, 5 and 9 in Fig. 4; piezometers 4, 10 and 6 in Fig. 5), for the baseline scenario, each of the 11 ensemble member scenarios and the ensemble mean. Simulated wetland water levels are shown for all locations in Figs. S3 and S4 of the electronic supplementary material.

The largest inter-ensemble member range in simulated levels occurs towards the end of October 2013 and corresponds to low flow conditions whilst the smallest range corresponds to the high flows period of February 2014. Both non-upwelling and upwelling

locations in the north meadow have relatively large inter-ensemble member ranges, varying between 0.05 m and 0.31 m. In the south the range is smaller, varying between 0.04 m and 0.19 m. Changes in water levels for the ensemble mean are relatively small with projected water levels being close to the baseline throughout the simulation period. To illustrate, over the full simulation period the mean difference between the baseline and the ensemble mean is 0.00 m in the north, while in the south mean differences suggest a decline of -0.03 m (Table 4).

In the south meadow, the absence of periods when the water level exceeds the ground level under baseline conditions is repeated for each climate change scenario. However, in the north

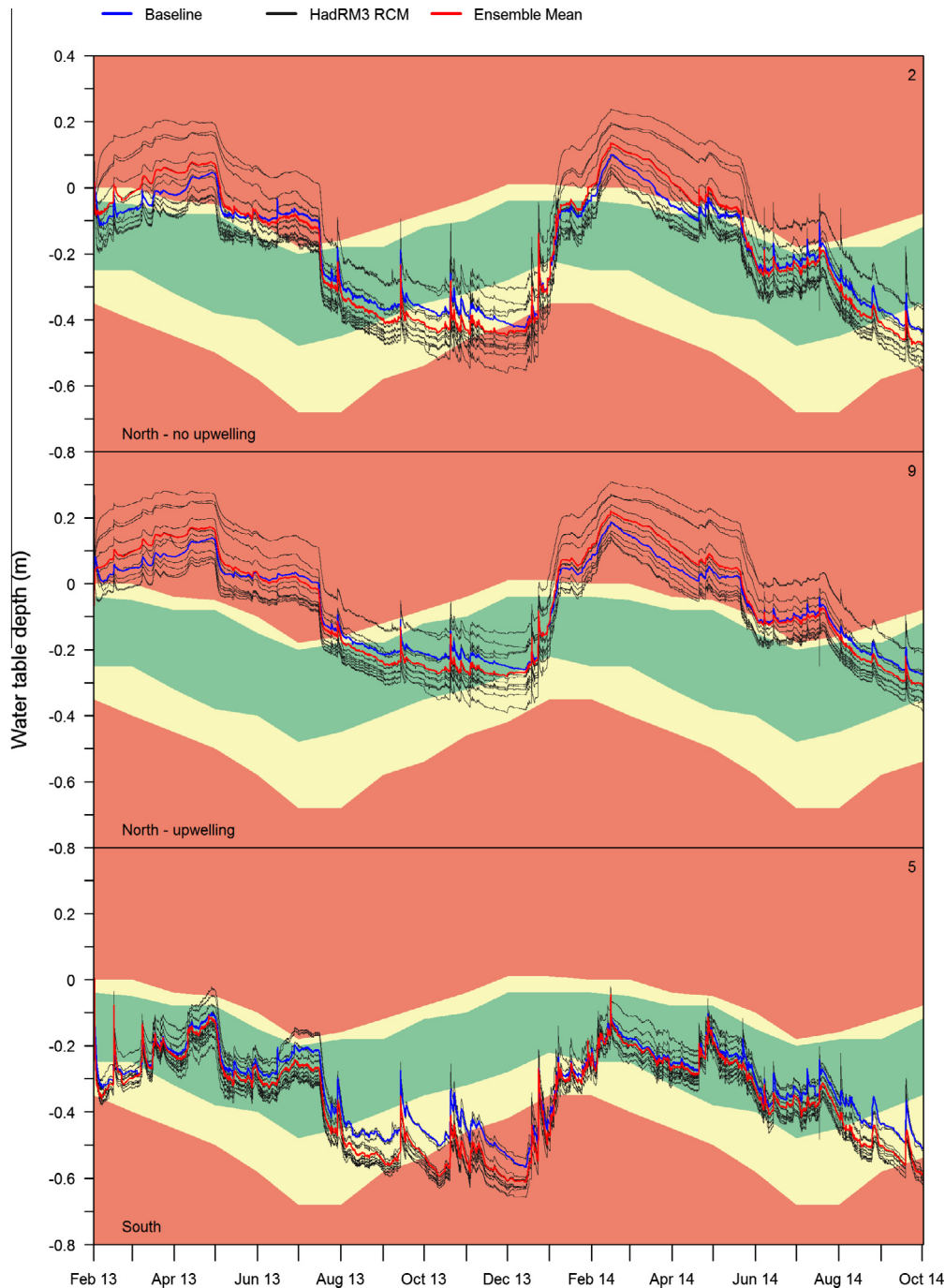


Fig. 4. Simulated baseline, projected ensemble member and mean wetland water table depths for selected locations superimposed over the MG8 vegetation community water level requirements zone diagrams. Red – intolerable; amber – tolerable for limited periods; green – desirable. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

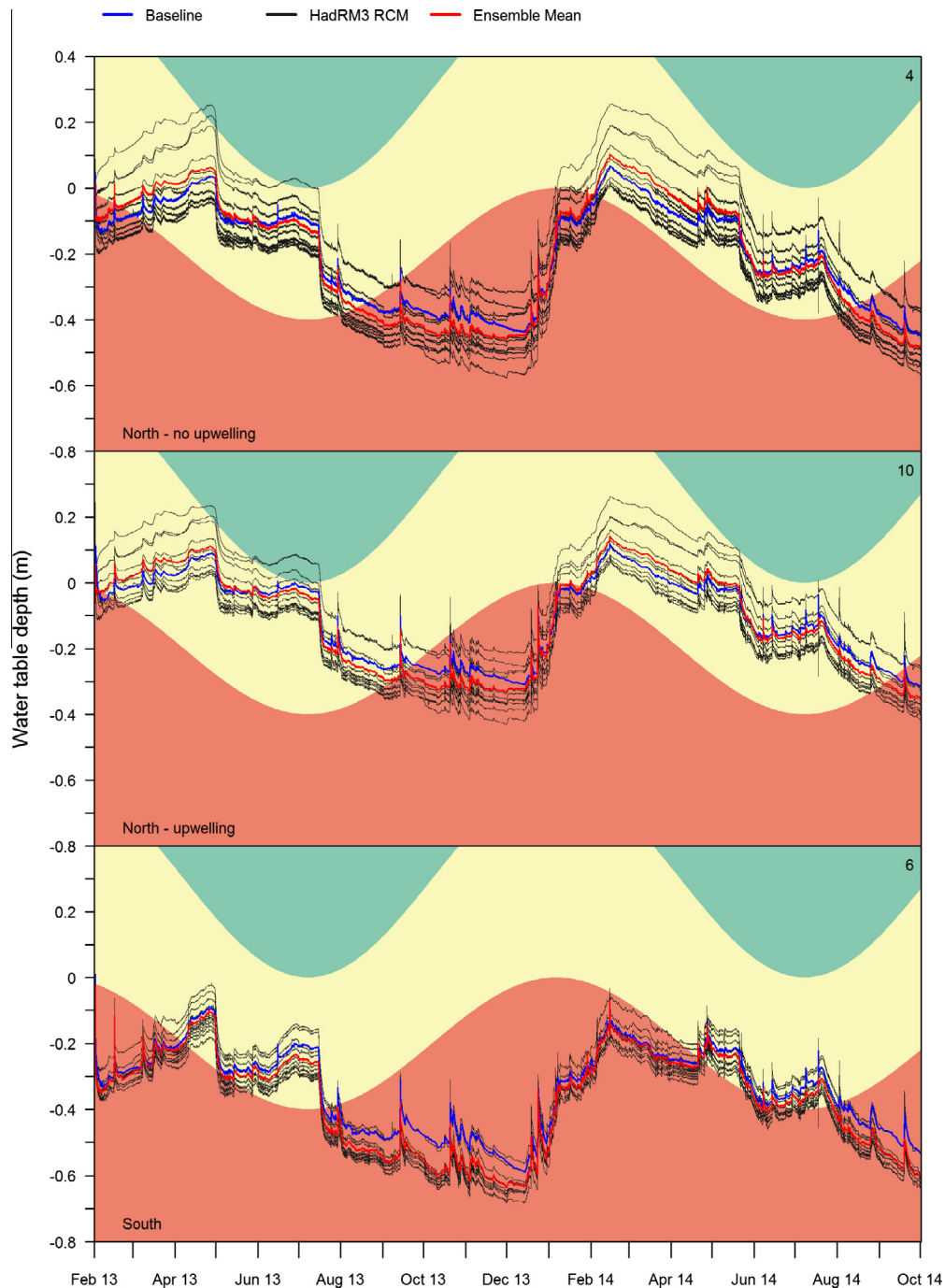


Fig. 5. Simulated baseline, projected ensemble member and mean wetland water table depths for selected locations superimposed over the Desmoulin's whorl snail (*Vertigo moulinsiana*) water level requirements zone diagrams. Red – intolerable; amber – tolerable for limited periods; green – desirable. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

meadow non-upwelling locations, some scenarios (A, C, D and E) result in an increase in the depth and duration of groundwater induced surface flooding. This can be by up to 0.2 m and the period of simulated surface water extends from 1–2 months up to 10 months. In upwelling areas, the same scenarios predict increases in the depth of standing water of up to 0.3 m deep. Conversely, other scenarios (B, G, I and K) project declines in water levels so that they are below ground level for the complete simulation period. The baseline groundwater induced flooding no longer occurs in these locations. In both north and south locations, declines in water level of up to -0.15 m from baseline are simulated during low water level periods in November and Decem-

ber 2013. In the south meadow water levels drop to around 0.6 mBGL.

The highest simulated wetland water levels for all locations are generally associated with scenario D. The lowest simulated levels for the north non-upwelling and upwelling areas are associated with scenario K. In the south meadow, B and I generate the lowest levels over the simulation period (-0.06 to -0.07 m).

4.2.2. Channel stage

Simulated stages for baseline, each ensemble member scenario and ensemble mean are shown for two locations (L2 and L4) along the Lambourn and one in the Westbrook (W3) in Fig. 6. Stages for

Table 4

Baseline mean wetland water levels (mBGL) averaged for North no upwelling, North upwelling and South locations, and channel stage (m) for the River Lambourn and Westbrook, with ensemble member and mean changes in level (m). Italicised values indicate negative changes.

Run ID	North – no upwelling	North – upwelling	South	Lambourn	Westbrook
Baseline	0.23	0.07	0.31	0.44	0.47
A	0.04	0.03	<i>-0.03</i>	<i>-0.02</i>	<i>-0.03</i>
B	<i>-0.08</i>	<i>-0.06</i>	<i>-0.07</i>	<i>-0.06</i>	<i>-0.08</i>
C	0.11	0.08	0.02	0.02	0.03
D	0.14	0.10	0.04	0.04	0.05
E	0.07	0.05	<i>-0.01</i>	<i>-0.01</i>	<i>-0.01</i>
F	<i>-0.02</i>	<i>-0.02</i>	<i>-0.06</i>	<i>-0.05</i>	<i>-0.06</i>
G	<i>-0.07</i>	<i>-0.06</i>	<i>-0.06</i>	<i>-0.05</i>	<i>-0.06</i>
H	<i>-0.06</i>	<i>-0.05</i>	<i>-0.06</i>	<i>-0.05</i>	<i>-0.07</i>
I	<i>-0.06</i>	<i>-0.05</i>	<i>-0.07</i>	<i>-0.06</i>	<i>-0.08</i>
J	<i>-0.01</i>	<i>-0.01</i>	<i>-0.04</i>	<i>-0.03</i>	<i>-0.04</i>
K	<i>-0.08</i>	<i>-0.06</i>	<i>-0.06</i>	<i>-0.05</i>	<i>-0.06</i>
Mean	0	0	<i>-0.03</i>	<i>-0.03</i>	<i>-0.04</i>

all locations are shown in Fig. S5 of the electronic supplementary material. Mean ensemble changes show a general reduction in stage which is much more apparent during periods of low flow (July 2013–December 2013 and August 2014–October 2014). For the ensemble mean the largest declines in stage occur in mid-December 2013. During periods of high flow the simulated stage for the ensemble mean corresponds much more closely to the baseline.

Only two scenarios (C and D) show overall increases through the simulation period (Table 4). The largest decreases in stage over the full simulation period in both the Lambourn and the Westbrook are associated with the B and I ensemble member scenarios (-0.06 and -0.08 m for the Lambourn and Westbrook, respectively). Stage drops to near zero at L4 and W3 in December 2013, the period associated with the largest decline in simulated stage, under scenarios B and I. During periods of high flow, simulated ensemble member stages are spread reasonably evenly on either side of the baseline with a maximum range of 0.15 m.

4.2.3. Groundwater upwelling

Simulated groundwater flow from the Chalk aquifer in areas where the putty chalk is absent shows a strong seasonality in the baseline, ensemble member and mean scenarios between winter wet periods and summer dry spells (Fig. 7). Rapid increases occur during and immediately after weed cuts. A mean scenario increase is evident throughout the simulation period. This is accentuated during periods of high flow (February 2013–May 2013 and January 2014–May 2014), with the largest increases in the ensemble mean occurring in March 2014.

Of the individual ensemble members, three scenarios (A, C, D and E) show increases throughout the simulation period. Scenario D displays the largest increases in March 2014, also the period of greatest inter-ensemble variation. The smallest inter-ensemble range occurs in September 2013 during the low flow period. Only a single scenario (B) shows decreases in groundwater flow over the full simulation period, although it is only scenario G that results in negative flow, or recharge, in December 2013.

4.3. Ecological impacts

4.3.1. Vegetation community

Inspection of simulated wetland water levels against a backdrop of water depth zones for the MG8 vegetation community reveals that under baseline conditions water levels in the North – no upwelling and South locations are, on the whole, within the desirable or tolerable ranges for MG8 vegetation (Fig. 4 – results

for all piezometer locations are shown in Fig. S3 and Table S2 of the electronic supplementary material). They are, however, often close to the boundary of the intolerable zone suggesting that current conditions are approaching the limit for this community. Water levels in the South fall into the lower intolerable zone for 6.6% of the simulated period. This occurs in December 2013 and coincides with the lowest simulated water levels (Table 5). In the North – no upwelling locations simulated baseline water levels extend into the higher intolerable zone during peak periods in April 2013, June 2013 and February 2014. These periods account for up to 16.0% of the total simulation period. At other locations groundwater upwelling elevates water levels so that they are above the tolerable range for much of the period, dropping to tolerable conditions for between 6 and 12 months during the summer low periods.

The ensemble member ranges show that the potential effects of climate change on MG8 vegetation differ across the site. However, in nearly all scenarios there is a shift towards more prolonged intolerable conditions for this particular vegetation community. In the locations that experience groundwater upwelling the higher groundwater levels within the underlying chalk for some scenarios push the highest wetland water levels further out of tolerable limits. The durations of the periods when water levels are in the upper intolerable zone therefore increases for scenarios A, C, D and E. However, for most scenarios the lower levels at other times of year now extend into the tolerable conditions for a larger proportion of the simulated period. In North – no upwelling locations the upper range of changes increases both the magnitude and duration of water levels falling within the intolerable zone. For the C, D and E ensemble member scenarios water levels are within this zone for as much as 8–10 months. At the other extreme, the lower levels associated with some ensemble members increase the occurrence of tolerable rather than desirable conditions, and pushes levels into the intolerable zone through the October–December 2013 low period. In the south meadow the projected increases in water levels for scenarios C and D could be beneficial for the MG8 vegetation community since the water table moves into the desirable zone. However, all of the other scenarios predict a decrease from desirable to tolerable levels, with a longer duration inside the tolerable zone of up to 4 months.

4.3.2. Desmoulin's whorl snail

Examination of simulated baseline and scenario water levels against the water level requirements of Desmoulin's whorl snail shows that in North – no upwelling locations both baseline and scenario simulated water levels are, for most of the simulation period, within the tolerable zone, while in the South they predominantly fall into the intolerable zone (Fig. 5 – for all piezometer locations see Fig. S4 and Table S3 of the electronic supplementary material). The increases in water levels for 4 out of the 11 scenarios (A, C, D and E) simulated for North – no upwelling locations have the potential to improve conditions for the snail, with predicted increases in the duration of tolerable conditions ranging between +5.0% (A) and +11.6% (C) (Table 6). Only two scenarios (C and D) show water level increases into the desirable zone. Where scenarios display lower water levels through the year the duration of tolerable conditions decreases, especially for scenario B where the duration of the period when water levels are within the intolerable zone increases by +17.4%.

In South locations, where baseline and ensemble member water levels do not intercept the ground surface, conditions approach tolerable on few occasions. For the baseline these are at the high points between April and June 2013 and again between May and August 2014. The largest increases in scenario water levels do little to improve conditions for the snail, only slightly extending the duration of tolerable conditions for the scenario with the largest

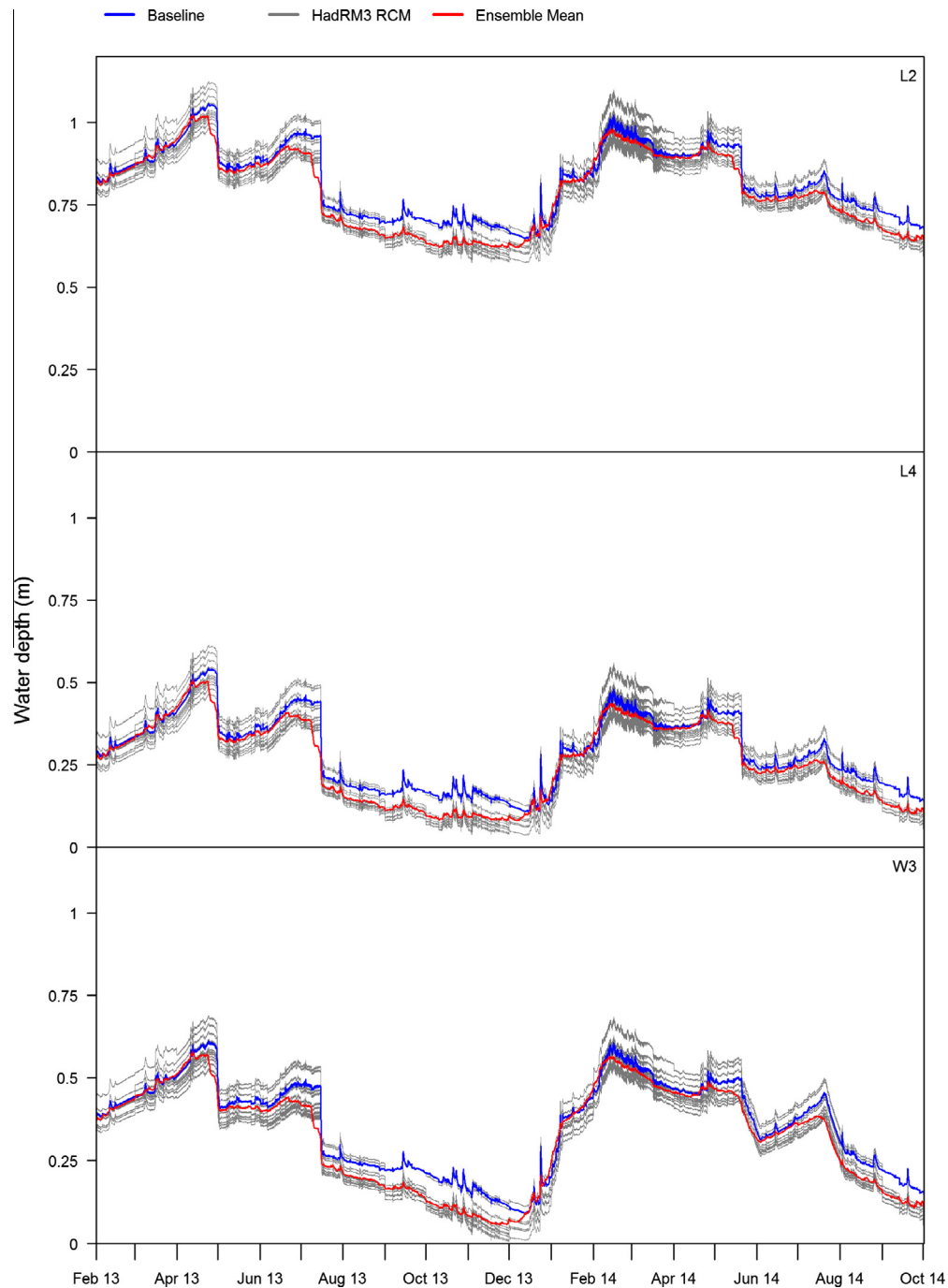


Fig. 6. Simulated baseline, projected ensemble member and mean channel stages for selected locations.

increases (D) by +8.7%. Scenarios with the largest declines in water levels (B, I) cause water levels to extend further into the lower intolerable zone, increasing duration by up to 9.8% (3.25 months).

For those areas where groundwater upwelling from the chalk occurs in the north meadow simulated levels indicate that, on the whole, baseline conditions are tolerable for the snail, only dipping into the intolerable zone from November 2013 to January 2014 (accounting for between 12.9% and 25% of the simulated period). The largest increases from the scenario ensemble suggest improved conditions for the snail with levels just reaching the desirable zone for short periods in June to August 2013. For scenario D water levels are within the desirable zone for 9.9% of the simulation period. Conversely the largest decreases in level from

the ensemble causes an earlier departure (from October 2013 instead of November 2013) into the intolerable zone and suggest the shift to intolerable conditions (albeit by small amounts) at the beginning and end of the simulation period.

5. Discussion

Baseline hydrological conditions and in turn the response to climate change differs noticeably over relatively short distances through the wetland. Other studies have shown similar hydrological complexity in comparable settings (Gilvear et al., 1993, 1997; Grapes et al., 2006). At these scales hydrological processes

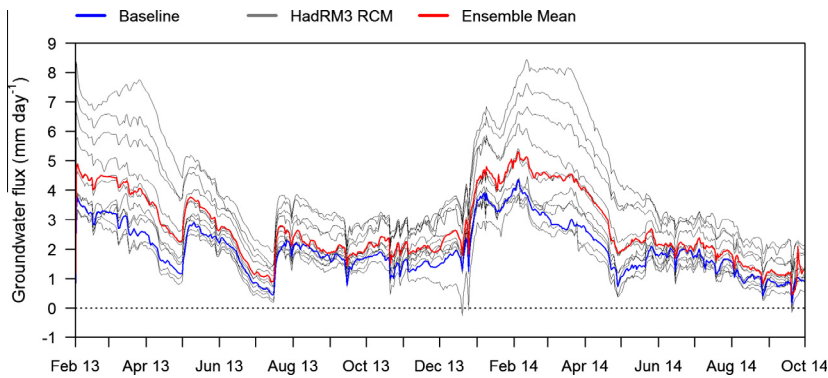


Fig. 7. Simulated baseline, projected ensemble member and mean groundwater flow from Chalk aquifer into wetland (positive = upwards).

are dominated by the interaction between groundwater and surface water, reflecting the site's position in a chalk valley bottom. Indeed, baseline results from the MIKE SHE model indicated proportional contributions to the water balance of 44.2% for surface water, 43.4% for groundwater, 6.3% for precipitation, and 5.7% for actual evapotranspiration (House et al., 2015b). Wetter winters and drier summers due to seasonal changes in scenario precipitation and year-round increasing PET have some direct influence. However, changes to wetland water levels are mostly governed by the projected changes in discharge and groundwater level. These in turn are influenced by meteorological changes occurring over the catchment and regional area. The disadvantage of a hydrological model at the site scale lies in the ability of the boundary conditions to represent flow changes from the wider area. Regional changes in precipitation and evapotranspiration would be expected to translate to comparable changes across the flow boundaries. The comparison of total boundary inflow under each scenario to precipitation shows that changes are within the same magnitude. Additionally, the good relationship between total boundary inflow, precipitation and evapotranspiration indicates that the effects of climate change are accounted for by the modelling approach.

In the south meadow the water levels are principally controlled by boundary channel stages. Scenario changes in water levels in this part of the wetland replicate the pattern of change in the River Lambourn and Westbrook. Conversely, since changes in chalk groundwater levels are larger than those for channel stage, the influence of upwelling chalk groundwater in the north meadow causes a greater projected range of scenario wetland water levels. The relatively small scenario changes in channel stage indicate the importance of regional and catchment processes in controlling water supply mechanisms for the site. The river has a high base flow index (0.96) and an ephemeral source, with the perennial head located 6–7 km downstream. Groundwater feeding the river may act as a buffer to the stresses of climate change at the catchment scale. Therefore, at 13 km downstream from the source the effects on discharge would be small. Indeed, a linear regression relationship between changes in the discharge inputs to the MIKE 11 model and the corresponding changes in the groundwater head boundary ($r^2 = 0.77$) is stronger than that for discharge and precipitation minus potential evapotranspiration ($r^2 = 0.63$) (Fig. S6 of the electronic supplementary material). Additionally, in-channel macrophyte growth is a principal control on river stage at the site (House et al., 2015b) so that the importance of discharge in controlling channel stage and corresponding water levels in the wetland may be moderate.

Declining river flow and increasing groundwater levels as indicated by the hydrometeorological projections are counterintuitive. The River Lambourn is reported as comprising 0.96 baseflow

(Marsh and Hannaford, 2008) and, at first glance, the reverse relationship would be expected. However, the proportion of this baseflow which comes from the gravel aquifer or the Chalk aquifer is unclear, as the hydrochemistry in the gravels is well mixed and displays similarity to the Chalk aquifer (House et al., 2015a). The gravel aquifer itself accounts for a down-valley component of groundwater flow, with variable hydraulic connection to the Chalk (Grapes et al., 2006), whilst the river is in good hydraulic connectivity with the gravels (Allen et al., 2010). It is possible that the two aquifers will experience differing responses to climate change, with the effect shown that the mostly gravel aquifer influenced river will display reductions in discharge, whilst the mostly separated Chalk aquifer will show increases in head.

The uncertainty contained within the projected hydrometeorological drivers for the MIKE SHE model is echoed in the water level responses across the wetland. Inter-ensemble member variations in simulated water levels differ spatially and over time, exhibiting some seasonality. In the north meadow results are split between four scenarios showing an overall increase in mean water levels throughout the simulation period and the remaining seven that show mean declines. The same directional trend is seen in summer, while in winter six of the 11 scenarios show increases in mean water levels. In the south meadow the general trend of change is negative with nine out of the 11 scenarios resulting in lower mean water levels for the complete simulation period and the summer period. In winter 8 of the 11 scenarios show lower levels, one no appreciable change and only two projecting increases.

Different hydrological impacts of climate change in distinct areas of such a relatively small site would have important implications for conservation designated species and their management. Differences in water level requirements between communities and species would necessitate separate management strategies in distinct areas of the wetland. Both the MG8 vegetation community, notable as a habitat for breeding snipe (*G. gallinago*), and Desmoulin's whorl snail, considered to be Near Threatened in Great Britain and on the IUCN red list of threatened species (Killeen et al., 2012), are currently in decline at the site (Natural England, 2012). Results of this study suggest that without any modifications to current management practices, their declines would be exacerbated under the majority of the climate change scenarios.

A comparison of the water level requirements for the MG8 vegetation community and baseline water levels simulated by the MIKE SHE model explains why only remnants of this community are currently found in the south meadow. Current conditions are infrequently desirable and often stray into the dry intolerable range. The seasonal pattern of water level lows and highs, characteristic of the area, does not match with the requirements for each species and would contribute to their current decline. It is realised that the water level requirements provided by the literature

Table 5

Percentage of full simulation period (01 February 2013–01 October 2014) simulated baseline, ensemble member and mean water levels are within each water depth zone (WDZ) for the MG8 plant community. UI, Upper Intolerable; UT, Upper Tolerable; D, Desirable; LT, Lower Tolerable; LI, Lower Intolerable.

Run ID	WDZ	North – no upwelling	North – upwelling	South
Baseline	UI	16.0	60.7	0
	UT	13.7	6.2	0
	D	57.4	33.1	62.6
	LT	12.9	0	30.8
	LI	0	0	6.6
A	UI	43.5	65.2	0
	UT	6.7	5.8	0
	D	35.7	29.0	55.2
	LT	14.0	0	33.8
	LI	0	0	11.0
B	UI	2.4	31.2	0
	UT	6.4	20.5	0
	D	64.2	45.2	37.1
	LT	22.1	3.1	48.0
	LI	4.9	0	14.9
C	UI	55.4	78.4	0
	UT	6.6	6.1	2.3
	D	32.5	15.5	61.5
	LT	5.5	0	30.3
	LI	0	0	5.9
D	UI	56.6	78.9	0.7
	UT	5.9	8.4	3.4
	D	37.4	12.8	68.4
	LT	0.1	0	22.8
	LI	0	0	4.6
E	UI	50.1	69.6	0
	UT	3.0	3.0	0.2
	D	34.0	27.3	57.5
	LT	12.9	0	33.1
	LI	0	0	9.1
F	UI	18.6	51.8	0
	UT	13.6	7.2	0
	D	45.6	41.0	41.9
	LT	21.1	0	43.7
	LI	1.2	0	14.4
G	UI	16.1	46.1	0
	UT	5.9	6.7	0
	D	51.2	38.5	41.6
	LT	18.8	8.8	44.9
	LI	8.1	0	13.5
H	UI	9.4	48.4	0
	UT	8.0	6.3	0
	D	57.7	44.1	39.6
	LT	20.2	1.1	45.9
	LI	4.7	0	14.4
I	UI	5.7	43.0	0
	UT	6.7	12.0	0
	D	65.5	44.6	35.6
	LT	17.7	0.4	49.7
	LI	4.3	0	14.6
J	UI	21.4	55.9	0
	UT	17.0	5.6	0
	D	42.0	38.5	50.2
	LT	19.4	0	37.7
	LI	0.1	0	12.1
K	UI	4.0	36.5	0
	UT	6.7	16.0	0
	D	63.6	41.7	42.8
	LT	19.4	5.9	42.1
	LI	6.3	0	15.1
Mean	UI	25.9	60.5	0
	UT	15.6	6.1	0
	D	38.6	33.4	52.4
	LT	19.9	0	34.0
	LI	0	0	13.6

Table 6

Percentage of full simulation period (01 February 2013–01 October 2014) simulated baseline, ensemble member and mean water levels are within each water depth zone (WDZ) for Desmoulin's whorl snail (*Vertigo moulinsiana*). D, Desirable; T, Tolerable; I, Intolerable.

Run ID	WDZ	North – no upwelling	North – upwelling	South
Baseline	D	0	0	0
	T	59.7	75.0	44.3
	I	40.3	25.0	55.7
A	D	0	0.9	0
	T	64.7	77.6	42.0
	I	35.3	21.5	58.0
B	D	0	0	0
	T	51.5	63.7	34.5
	I	48.5	36.3	65.5
C	D	4	8.8	0
	T	71.3	74.4	48.2
	I	24.6	16.8	51.8
D	D	4.9	9.9	0
	T	70.5	74.0	53.0
	I	24.6	16.1	47.0
E	D	0	3.5	0
	T	69.3	75.1	44.1
	I	30.7	21.3	55.9
F	D	0	0	0
	T	56.7	70.2	35.9
	I	43.3	29.8	64.1
G	D	0	0	0
	T	54.5	67.3	37.2
	I	45.5	32.7	62.8
H	D	0	0	0
	T	53.7	67.4	35.2
	I	46.3	32.6	64.8
I	D	0	0	0
	T	53.8	66.1	34.2
	I	46.2	33.9	65.8
J	D	0	0	0
	T	59.0	72.0	38.6
	I	41.0	28.0	61.4
K	D	0	0	0
	T	52.3	64.7	36.2
	I	47.7	35.3	63.8
Mean	D	0.1	1.1	0
	T	59.3	72.8	39.8
	I	40.6	26.1	60.2

(Wheeler et al., 2004) indicate the broad range of hydrological regime that gives rise to specific vegetation communities. Nevertheless, for the MG8 community there is detailed data to identify the magnitude of hydrological impact that would have an effect. Hence, the requirements are considered robust. Model results suggest that climate change is likely to push wetland water levels further into intolerable conditions that would further facilitate succession by other communities. An extended duration of waterlogging, as seen for scenarios A, C, D and E, would cause the community composition to change from grassland to mire or swamp (Gowing et al., 2002; Wheeler et al., 2004). Conversely, deeper water tables, as simulated for the other scenarios, would cause a gradual loss of characteristic, moisture demanding species such as marsh marigold (*C. palustris*), ragged robin (*Lychnis flos-cuculil*) and common spikerush (*Eleocharis palustris*). Waterlogging already occurs in the north meadow, especially around groundwater upwelling areas, where the communities S5 *Glyceria maxima* swamp and S6 *Carex riparia* swamp are prevalent (House et al., 2015a). Any increase in water levels is likely to cause an expansion of these swamp areas. Declines of the magnitude simulated by

some of the climate change scenarios (I, K) would likely have little restorative effect in upwelling areas, and in the rest of the north meadow may cause drying out during late summer and the consequent loss of wetland species in favour of tall-herb communities such as S28 *Phalaris arundinacea* tall-herb fen, OV24 *Urtica dioica*–*Galium aparine* and OV26 *Epilobium hirsutum*. In the south meadow, where remnants of MG8 still exist, the community has a better chance of recovery and expansion. Predicted levels for the climate change scenarios are on the whole within desirable or tolerable ranges, only falling outside of these when projections result in decreases through the late summer and early winter months. Management efforts for the MG8 community would therefore likely to be more productive when directed towards the south meadow.

Hydrological requirements for Desmoulin's whorl snail obtained from the literature (Tattersfield and McInnes, 2003) are uncertain, but do provide some indicative water levels which can be used to assess the potential impacts of climate change on this individual species. Survival of the snail is dependent on the maintenance of high water levels and standing water. It is clear that the areas where the snail is most likely to survive at the site are around the zones of upwelling in the north meadow. However, even in these areas and under the extreme climate change scenario, water levels rarely reach elevations that are considered desirable. Those climate change projections associated with lower water levels result in the creation of periods of intolerable hydrological conditions, even in these relatively wet areas. In the south meadow, where the simulated water levels do not exceed ground level under the baseline and any ensemble member, conditions are unlikely to support any Desmoulin's whorl snail.

As described by House et al. (2015b), and to some extent Old et al. (2014), in-channel management of macrophytes through periodic weed cutting has a substantial effect on water levels throughout the wetland. Weed cuts are carried out to increase flood conveyance, reduce riparian water levels and maintain fisheries (Baatrup-Pedersen and Riis, 2004; Nikora et al., 2008; Old et al., 2014). The hydrological implications for the site have been fully discussed elsewhere (House et al., 2015b). Substantial drops in water level are seen, with weed cuts on 16 July 2013 and 23 July 2014 having the most impact and causing water levels to fall into the lower intolerable zone for both MG8 vegetation and Desmoulin's whorl snail. Without weed cuts it is debatable whether water levels would have become intolerable.

The differences in the water level requirements of the MG8 community and Desmoulin's whorl snail suggests that there is some potential to manage the wetland for the promotion of each in different parts of the site. Managing for multiple objectives is an important consideration for areas where complicated feedbacks between hydrology and biotic communities exist, of which this study is an example. There are a range of activities which could result in changes to wetland water levels and thus have implications for ecosystem management in relation to climate change. These include groundwater abstraction, alterations to channel morphology, and vegetation management. The MIKE SHE model employed in the current study could be employed to assess the impact of a range of weed cut and other management options upon wetland water levels under both current and scenario climates. This could include a reassessment of how weed cuts are carried out by, for example, reducing its frequency in channels bordering the north meadow in order to maintain the high water levels necessary for Desmoulin's whorl snail. However, there is a delicate balance between managing for flood conveyance and the conservation of desired species, whilst also incorporating measures to account for potential effects of climate change. Additional sociopolitical aspects for the surrounding community in terms of flood

resilience and natural capital would need to be factored into any management scenario.

6. Conclusion

The simulated hydrological impacts of climate change vary considerably over relatively small distances within the wetland. This is due to differences in groundwater/surface-water interaction and water availability, and will have important implications for the management of conservation priority species. Alternative approaches, dependent on the severity of impacts, are required to promote Desmoulin's whorl snail in the groundwater influenced north of the site, and the MG8 vegetation community in the south where channel stage controls water levels. Strategies could be based around the regulation of channel stage through management of macrophyte growth. The MIKE SHE model could be used to examine the impacts of any revised management by simulating the channel under different cutting regimes.

Valley bottom wetlands within chalk catchments are discontinuous but widespread. Such groundwater-dependent wetland ecosystems are celebrated for their conservation and scientific value, providing habitats with high biodiversity. The impact of climate change, and other environmental changes such as groundwater abstraction, will have differing responses in terms of water availability. They will also have implications for other environmental factors influencing wetland ecosystems, such as nutrient supply, that are not addressed in the current study. Additionally, the water requirements of particular species and communities may themselves change with the climate, a factor beyond the scope of this study, yet which could have an influence on their maintenance or succession. Due to complex hydrological relationships between bedrock and alluvial aquifers, plus river stage and also the interplay between site scale and catchment processes, responses may vary dramatically over relatively small spatial scales. Conservation management should thus be targeted spatially to incorporate differing responses in hydrological conditions. Knowledge of the hydrological requirements of desired species, along with robust understanding of water supply mechanisms, will be essential for the development of robust, climate-change proof management approaches.

Extension of the MIKE SHE model to incorporate environmental flows in the channels could provide a means of assessing the impacts of climate change on habitat for fish and other aquatic organisms, whilst the addition of a nutrient or contaminant transport module could further support assessment of species maintenance or succession. The MIKE SHE model of the Lambourn Observatory provides a powerful tool to aid understanding of the response of the wetland ecosystem to change and to develop management strategies.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.jhydrol.2016.01.004>.

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