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Impacts of socio-economic and climate change scenarios on wetlands: linking water resource and biodiversity meta-models

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Abstract A meta-modelling approach has been adopted to link simulations of low and high water flows with simulations of suitable climate space for a selection of fen and bog species with differing drought and flood tolerance. The linked meta-models were used to examine the impacts of socio-economic and climate change scenarios on wetlands in two contrasting regions of the UK. The hydrological model shows that low and high flows are sensitive to climate change and to the regional distribution of abstractions and discharges. Where there are large changes in urbanisation, flows are more sensitive to socio-economic change. The changes in high flows have little impact on the species selected, but changes in low flows result in a number of areas becoming marginal or unsuitable due to dryness. At the regional scale, adaptation options appear to be limited and mostly involve, for surface water-influenced wetlands, increased water imports (either directly through increased non-consumptive water demand or indirectly through river augmentation), which may not be consistent with the socio-economic scenario or be feasible. This paper shows, therefore, that changes in hydrological regime are important for the future of wetlands and that these may depend as much on the future socio-economic situation as the projected changes in climate.

1 Introduction

Wetland ecosystems are important because of the wide range of services which they perform, such as water regulation and purification (Millennium Ecosystem Assessment 2005) and carbon storage (Gorham 1991). Mitsch and Gosselink (1993)

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argue that hydrology is probably the most important determinant of the establishment and maintenance of specific wetland communities and wetland processes. Wetlands in Europe have already been identified as at risk from climate change through higher temperatures, greater evapotranspiration and altered precipitation amounts and patterns changing the hydrological regime (Hartig et al. 1997). While wetland species are adapted to varying degrees of flooding and have different drought tolerances, changes in regime will lead to alterations in the species composition and functioning of wetlands (Crawford 1983).

Pressures on wetlands are likely to be mediated through changes in hydrology, direct and indirect effects of changes in temperature and land use change (Conlan et al. 2007). A warmer future climate is expected to lead to an increase in the length of the growing season, so that soils return to field capacity later in the autumn and start drying out sooner in the spring (Holman 2006). There is a general tendency for greater seasonality of flows in the UK, becoming lower in the summer and higher in the winter, although this relationship differs spatially and between different climate change scenarios (Arnell and Reynard 1996; Holman et al. 2005a). Gorham (1991) states that climate warming is expected to lead to a drying of wetlands, through alterations in water level, and this would be the main driver of ecosystem change.

The inherent changeability of wetland communities, resulting from spatial and temporal variability in water supply (Tallis 1983) and the differences between species in adaptation potential may be the key factors in the response of wetland communities to climate change. Information on the tolerances of species to waterlogging and drought has improved such that they are now broadly understood and are well known for some species. Subjective scales of these tolerances have been devised by Ellenberg et al. (1979), Ellenberg (1991; revised for Britain by Hill et al. 1999) and Grime et al. (1988). These provide an indication of the requirements of species based on mean water tables. Studies trying to establish correlations between water tables and plant distributions have found that there can be considerable variability within and between sites and, according to Wheeler (1999), the most discriminating variables for the occurrence of plant communities, are the mean highest and lowest groundwater levels during the growing season.

Recharge of local and regional groundwater systems, the position of the wetland relative to the local topography, and the hydraulic gradient of larger regional groundwater systems can be critical factors in determining the variability and stability of moisture storage in wetlands in climatic zones where precipitation does not greatly exceed evaporation (Winter and Woo 1990). Changes in recharge external to the wetland may be as important to the fate of the wetland under changing climatic conditions as the change in direct precipitation or evaporation on the wetland itself (Woo et al. 1993). However, levels of water flow are important in influencing water tables in many wetlands, in that under, or around the time of high flow conditions water tables are potentially high and vice versa. This has implications for the growth of species and rooting zone conditions, as flooding can lead to aeration stress, while drought can lead to water supply issues, both of which can affect the structure of wetland communities (Silvertown et al. 1999). These can have consequences for nutrient availability and soil temperature and, if sustained, implications for wetland management.

The research described in this paper was part of the RegIS2 project on *Regional Climate Change Impact and Response Studies in East Anglia and North West England.*

The RegIS2 project developed and applied a methodology for stakeholder-led, regional climate change impact assessment across the four major sectors driving landscape change (agriculture, biodiversity, coasts/floodplains and water resources). A previous study (RegIS) assessed impacts of socio-economic and climate change scenarios on the same sectors using detailed process-based models (Holman et al. 2005a, b). However, the models had very long run-times and thus the number of scenarios that could be analysed was small. Replacing the detailed models by meta-models, or reduced models, which endeavour to capture the major systematic effects shown in the full models (Audsley et al. 2008), means that numerous runs can be undertaken by the user and the influence of different input parameters within scenarios can be analysed in greater depth. Therefore, the principal objective of the RegIS2 project was to develop a series of linked meta-models within a user friendly interface (the 'Regional Impact Simulator') to provide stakeholders with the flexibility to investigate impacts and adaptive responses of relevance to themselves (Holman et al. 2008).

In this paper, linkages between the water resources meta-model (Henriques et al. 2008) and the biodiversity meta-model (Holman et al. 2008) are described. The water resources meta-model is used to examine the effects of climate and socio-economic change scenarios on high and low river flows in two contrasting regions of England: East Anglia and the North West. The biodiversity meta-model is used to analyse possible consequential impacts, and options for adaptation, for a selection of species representing fen and blanket and raised bog communities. The wetland habitats under discussion, in common with most others, have suffered large losses due to a variety of causes, especially competing land uses and changing water availability. These pressures are likely to continue and the impacts on these habitats and their component species could be exacerbated by climate change. Thus, there could be profound effects on the structure and diversity of the affected wetlands (Gorham 1994), with important implications for habitat and species conservation.

2 Methodology

2.1 Case study regions and habitat/species selection

Detailed descriptions and maps of the two regions are given in Holman et al. (2008) and Audsley et al. (2008). They are described briefly in this section in relation to the habitats chosen for study. The selection of habitats and species was undertaken in conjunction with local stakeholders in East Anglia and North West England (see Holman et al. 2008 for details of the stakeholder engagement process). The strategy for habitat selection involved focusing on habitats which interact with the hydrological model and are of conservation importance in each region. Species selection followed a protocol to ensure that different taxa and dominant, characteristic and threatened (sensitive/rare) species were modelled (Table 1). All three habitats are listed in Annex 1 of the EC Habitats and Species Directive and are UK Biodiversity Action Plan (BAP) habitats. A number of their component species have equivalent designation and thus are of significance to conservation.

The East Anglia case study region is situated in eastern England and encompasses the administrative districts of Norfolk, Suffolk and Cambridgeshire. The region has

Table 1 Species selected for study for fens and blanket and raised bogs habitats

Species Latin name	Species common name	Category
Fens		
Glyceria maxima	Reed sweet grass	Dominant
Phragmites australis	Common reed	Dominant
Epipactis palustris	Marsh helleborine	Characteristic
Ranunculus scleratus	Celery-leaved buttercup	Characteristic
Valeriana dioica	Marsh valerian	Characteristic
Vertigo moulinsiana[a]	Desmoulin's whorled snail	Rare
Blanket and raised bogs		
Empetrum nigrum	Crowberry	Dominant
Eriophorum vaginatum	Hare's-tail cotton grass	Dominant
Molinia caerulea	Purple moor grass	Dominant
Sphagnum cuspidatum	–	Dominant
Calluna vulgaris	Heather	Characteristic
Myrica gale	Bog myrtle	Characteristic
Rhynchospora alba	White-beaked sedge	Characteristic
Betula pendula	Silver birch	Characteristic on drier areas
Betula pubescens	Downy birch	Characteristic on drier areas
Coenonympha tullia	Large heath butterfly	Rare
Listera cordata	Lesser twayblade	Rare

[a] Biodiversity Action Plan species

a relatively homogeneous low-lying topography with elevations ranging between 0 and 130 m above sea level. The climate is relatively dry and temperate with average annual precipitation ranging from 550 to 750 mm. The main land uses in the region are arable cropping (cereals, sugar beet, vegetables and potatoes) and intensive grassland. Over 70% of UK fens occur in East Anglia and this also represents a large proportion of the European resource (English Nature 1996). Many of the fens in East Anglia have additional national and international conservation status, for example, the Fens in Cambridgeshire and the Broads (Norfolk and Suffolk) are both Special Areas of Conservation (SACs) and Wicken Fen (Cambridgeshire) is a National Nature Reserve, a Site of Special Scientific Interest, a SAC and it is protected under the RAMSAR Convention. It is thought that 97% of wetlands in the Fens have disappeared since 1650, with 40% of this being since 1930 (Thomas et al. 1981). In East Anglia, the habitat is currently threatened by reduced water levels and inundation in coastal areas. Also where there is a lack of management, succession can lead to replacement by scrub and woodland. In the south of the region, many of the fens are low-fertility sites now surrounded by agricultural landscape, leading to problems of water quality (Suffolk County Council 2000). Fens, together with swamp and marsh, have been identified as being of low to medium vulnerability to climate change in the UK, with those in southeast England being particularly vulnerable (Hossell et al. 2000). A number of fen species have also been identified as potentially vulnerable to climate change due to summer soil water deficits (Berry and Butt 2004; Holman et al. 2005a).

The North West England case study region encompasses the administrative districts of Cumbria, Lancashire, Merseyside, Greater Manchester and Cheshire. The topography of the region is much more heterogeneous than East Anglia with elevations ranging from 0 to 900 m above sea level. Uplands in the north and west are dominated by extensive grazing whilst land use on the lowlands in the west and south of the region includes intensive livestock production and the large urban conurbations of Manchester and Liverpool. The climate is wetter than East Anglia with average annual precipitation ranging from 650 to 3,200 mm. In North West England, internationally important blanket bogs are found in the north (Cumbria) and north east (north Pennines) of the region and they support populations of a number of listed breeding birds. In the UK, 94% of lowland raised bog has been lost since the nineteenth century, largely due to conversion to high grade agricultural land and in the North West the largest remaining areas are in Cumbria in the north of the region (Regional Biodiversity Steering Group 1999).

2.2 Climate and socio-economic change scenarios

Four climate change scenarios developed on behalf of the UK Climate Impacts Programme (UKCIP) and known as the UKCIP02 scenarios (Hulme et al. 2002) have been utilised: 2020s Low emissions, 2020s High emissions, 2050s Low emissions and 2050s High emissions. They are all based on the high-resolution regional climate model (HadRM3) from the Hadley Centre for Climate Prediction and Research. The Low and High emissions scenarios were used in order to capture the effects of uncertainties in future greenhouse gas emissions. Increases in mean annual temperature, relative to 1961–1990, range from 0.3 to 1.1°C by the 2020s and 0.5 to 2.6°C by the 2050s for the UK. In all seasons, and for all scenarios, there is a northwest to southeast gradient in the magnitude of the climate warming over the UK, the southeast consistently warming by at least several tenths of a degree Celsius more than the northwest. Changes in total annual precipitation, relative to 1961–1990, range from −4% to +3% by the 2020s and −9% to +7% by the 2050s. In winter, precipitation increases in all regions, relative to 1961–1990, with the largest changes of up to 23% occurring in the east. Conversely, in summer precipitation decreases across virtually all of the UK with the largest changes of up to −32% occurring in the south.

Four socio-economic scenarios (SES) based on regionalised versions of the UKCIP SES (Shackley and Deanwood 2003), which are themselves derived from the SRES emissions scenarios developed by the Intergovernmental Panel on Climate Change (IPCC; Nakićenović et al. 2000), have been utilised: (1) *Regional Enterprise* (RE) which suggests vibrant, semi-autonomous regions, keen to promote and maintain their distinctive qualities in a highly competitive world. Water resource supply–demand balance deficits are met through new supply options and water transfers. Privately funded nature reserves are developed but are regarded as ‘wildlife gardening’ by some; (2) *Global Market* (GM) which is based on the pursuit of high and sustained growth within a global context. Water resources are seen as a marketable commodity, with transfers used to meet deficits. There are pressures upon biodiversity from tourism in some areas; (3) *Regional Stewardship* (RS) with an emphasis on recognising and conserving regional assets, accepting that this might result in a significantly reduced level of economic growth and even a contraction of the economy in some respects. The concept of national water transfers is rejected, rather the onus is placed on demand-side reduction. Biodiversity policy is focused on preserving and improving existing and traditionally found biodiversity assets throughout the landscape; and (4) *Global Sustainability* (GS) where global approaches to achieving sustainable development take precedence over regional responses. Water resources are seen from a national perspective, the aim being an equitable sharing of the resource according to need through a new national water network. Biodiversity policy promotes an increase in the overall areas of protected habitats with financial incentives from the European Union.

2.3 Model descriptions

The hydrological and biodiversity models were developed to examine the impacts of climate and socio-economic change on high and low river flows and on the availability of suitable climate space for species respectively. They have been integrated into an interactive tool for stakeholder use (the *Regional Impact Simulator*, available to download from www.ukcip.org.uk; Holman et al. 2008), that also includes agricultural (Audsley et al. 2008) and coastal and flooding meta-models (Mokrech et al. 2008; Richards et al. 2008). This integration allows interactions between the sectors represented by the different models to be explicitly linked and for adaptation options in response to specific outputs to be explored, through changing relevant parameters. Spatially explicit inputs and outputs are presented on a 5 × 5 km grid for the two case study regions.

2.3.1 Hydrological meta-model

Modelling naturalised high and low flows The naturalised flow duration curve was simulated for every catchment in East Anglia and the North West using an improvement of the approach of Gustard et al. (1992) as presented by Henriques et al. (2008) and Henriques (2007). The simulations were based on the distribution of soil types, average annual precipitation and actual evapotranspiration derived from the distribution of land uses, including cropping (from the linked agricultural land use meta-model; Audsley et al. 2008) and urbanisation (from the SES), for the different time slices and scenarios.

In order to estimate the 'real' (or non-naturalised) flows with a probability of exceedance of 5% and 95%, a simple methodology was derived based on the regional abstraction availability and non-consumptive water demand. $Q5$ flows refer to the flow exceeded 5% of the time, i.e. high flows, whilst $Q95$ flows correspond to the flow exceeded 95% of the time, i.e. low flows. Data on the future distribution of abstractions, supply, sewerage networks, Sewage Treatment Works (STW) and industrial discharges were not available to undertake a spatially explicit assessment.

Modelling the effects of abstraction on low flows Regional water supply can be provided from imports from outside the region, reservoirs and abstractions. The water available for abstraction at a catchment scale from surface and groundwater was estimated from the naturalised flow duration curve after allowing for the environmental flow requirements of the river ecosystem as given by the Regional Environment Priority (REP) from the SES (Henriques et al. 2008). This is similar to the approach of the Environment Agency's Resource Assessment Management framework (Environment Agency 2002).

The estimated naturalised flows are affected by reservoirs and abstractions, namely surface water abstractions which represent approximately 50% and 90% of the water supplied in East Anglia and the North West, respectively (Defra 2005). Those impacts were assumed negligible for $Q5$. The naturalised $Q95$ is only impacted by the 'unconstrained' abstraction which is, by definition (Environment Agency 2002), the water that is allowed to be abstracted without reducing the $Q95$ by more than a given percentage that depends on the environmental flow requirements of the river (Henriques et al. 2008). To meet the regional demand, water was assumed to be abstracted equally impacting all the catchments, assuming that the surplus is transferred within the region. The impact of abstractions on reducing the naturalised $Q95$, A (m^3/s), is given by:

$$A = \mathrm{UNC}, \ \text{if } \mathrm{Demand}_{\mathrm{region}} > \mathrm{UNC}_{\mathrm{region}} \tag{1a}$$

$$A = \mathrm{UNC} \times \frac{\mathrm{Demand}_{\mathrm{region}}}{\mathrm{UNC}_{\mathrm{region}}}, \ \text{if } \mathrm{Demand}_{\mathrm{region}} < \mathrm{UNC}_{\mathrm{region}} \tag{1b}$$

Where,

UNC	is the unconstrained abstraction availability for a given catchment (m^3/s),
$\mathrm{UNC}_{\mathrm{region}}$	is the unconstrained abstraction availability in all the catchments of the region, assumed to be constant throughout the year (m^3/s), and
$\mathrm{Demand}_{\mathrm{region}}$	is the regional water demand (m^3/s)

Modelling the effects of returns on high and low flows Not all of the water supplied to satisfy the regional water demand is consumed. It was assumed that 95% of the water used for domestic and industrial/commercial purposes (calculated from the SES) and 100% of leakage from the mains network (Henriques et al. 2008) is returned to the environment and that irrigation (simulated by the linked agricultural meta-model; Audsley et al. 2008) is 100% consumptive. The water returns to the region, $\mathrm{Returns}_{\mathrm{region}}$ (m^3/s), assumed to be constant throughout the year, were calculated from:

$$\mathrm{Returns}_{\mathrm{region}} = \mathrm{Leakage} + (\mathrm{IC} + \mathrm{Dom}) \times 0.95, \ \text{if Demand} < \text{Supply} \tag{2a}$$

$$\mathrm{Returns}_{\mathrm{region}} = \frac{\mathrm{Supply}}{\mathrm{Demand}} \times \left(\mathrm{Leakage} + (\mathrm{IC} + \mathrm{Dom}) \times 0.95\right), \ \text{if Demand} > \text{Supply} \tag{2b}$$

Where,

Leakage	is the leakage in the region (m^3/s),
IC	is the industrial and commercial water demand in the region (m^3/s),
Dom	is the domestic water demand in the region (m^3/s),
Supply	is the regional water supply (m^3/s), and
Demand	is the regional water demand (m^3/s).

In order to distribute the returns spatially within the region, the water returns were assumed to take place where the demand exists and this demand was related to urbanisation patterns, due to the higher domestic and industrial/commercial water demand and greater leakage (due to higher network pipe density) in urban areas. The contribution of returns to the naturalised river flows of a catchment, $\mathrm{Returns}_{\mathrm{catchment}}$ (m^3/s), is given by:

$$\mathrm{Returns}_{\mathrm{catchment}} = \frac{\mathrm{Returns}_{\mathrm{region}}}{(\mathrm{Urban} + \mathrm{Suburban})_{\mathrm{region}}} \times (\mathrm{Urban} + \mathrm{Suburban})_{\mathrm{catchment}} \tag{3}$$

Where, Urban and Suburban are urban and suburban land-classes in the catchment/region (%).

Derivation of 'real' Q95 and Q5 The 'real' $Q95$ and $Q5$ for the catchments were derived using Eqs. 4 and 5:

$$Q95 = Q95_{\mathrm{naturalised}} - A + \mathrm{Returns}_{\mathrm{catchment}} \tag{4}$$

$$Q5 = Q5_{\mathrm{naturalised}} + \mathrm{Returns}_{\mathrm{catchment}} \tag{5}$$

Validation A simple approach to validating the methodology of estimating 'real' flows for baseline conditions was adopted. The maximum simulated contribution of returns to catchment flows of 1.8 m^3/s is in the North West, which is less than the actual discharge of 4.1 m^3/s from the largest STWs at Manchester and similar to that of the large Carlisle and Fleetwood STWs. Moreover, given an average of 70,000 people per catchment in the North West and significant direct industrial discharges, the average simulated returns of 0.32 m^3/s for the North West corresponds to a realistic 90,000 population-equivalent per catchment, assuming a per capita flow of sewage of 300 l/day (Gray 1992). Finally, the concordance correlation coefficient (ρc; Lin 1989) was used to measure the agreement of gauged and simulated 'real' $Q95$. The sample ρc was 0.9266 for East Anglia ($n = 30$) and 0.9354 for the North West ($n = 48$) demonstrating that this simple regional model accounts adequately for the impacts of abstractions and returns.

2.3.2 Biodiversity meta-model

Modelling the effects of climate change on species' suitability The SPECIES (Spatial Estimator of the Climate Impacts on the Envelope of Species) model (Pearson et al. 2002) was used to characterise the current distribution of selected fen and bog species in Europe and to estimate their potential re-distribution under alternative climate change scenarios across Britain and the two regions. The SPECIES model is based on an artificial neural network, which integrates bioclimatic variables for predicting the distribution of species through the characterisation of bioclimatic envelopes. Integrated algorithms, including a soil water balance model, are used to pre-process climate (temperature, precipitation and potential evapotranspiration) and soils (AWC – available water holding capacity) data to derive five relevant bioclimatic variables for input to the neural network: growing degree days >5°C; absolute minimum temperature expected over a 20-year period; annual maximum temperature; accumulated annual soil water deficit; and accumulated annual soil water surplus.

The model was trained using existing empirical data on the European distributions of species to enable the full climate space of a species to be characterised and to ensure that the model does not extrapolate outside its training dataset when used to project the distribution of species under potential future climates in Britain. The data were randomly divided into three groups for training, validating and testing the neural network. The validation set ensures that the network does not over-train on the training data, thus losing its ability to generalise, while the test data was used to independently verify the prediction.

The performance of each network was statistically analysed using Cohen's Kappa statistic of similarity (k) and the Area Under the Receiver Operating Characteristic Curve (AUC). Kappa is a commonly used statistic that provides a measure of proportional accuracy, adjusted for chance agreement (Cohen 1960). k varies from 0, indicating no agreement between observed and predicted distributions, to 1 for perfect agreement. AUC is an unbiased measure of prediction accuracy calculated from the Receiver Operating Characteristic (ROC) curve (Fielding and Bell 1997). The ROC curve describes the compromise that is made between the sensitivity (defined as the proportion of true positive predictions versus the number of actual positive sites) and false positive fraction (the proportion of false positive predictions versus the number of actual negative sites). This index is independent of both species prevalence and the decision threshold for defining species' suitability. AUC ranges from 0.5 for models with no discrimination ability, to 1 for models with perfect discrimination. AUC statistics were greater than 0.9 for all 17 species, indicating very good discrimination ability. Kappa values are slightly lower, but this is to be expected as the index ranges from 0 to 1. Values were greater than 0.85 for nine species indicating excellent agreement between observed and simulated distributions, and between 0.7 and 0.85 for eight species indicating very good agreement.

Once a network is trained and validated at the European scale, it is then applied at a finer spatial resolution within the case study regions to produce a climate suitability surface. This is converted into a presence/absence distribution by applying the decision threshold which maximises agreement between observed and simulated distributions derived from the ROC curve. Further details concerning the model validation and definition of decision thresholds are provided in Pearson et al. (2002, 2004).

Results from the SPECIES model show areas of maximum potentially suitable climate space and do not account for the availability of suitable habitat nor how the dispersal ability of a species might affect its ability to colonise new areas. These limitations, along with others, to the predictive capacity of species–climate envelope modelling approaches are discussed in more detail in Pearson and Dawson (2003) and Harrison et al. (2003, 2006). They imply that the projections should be viewed as first approximations indicating the potential magnitude and broad pattern of future impacts, rather than as accurate simulations of future species' distributions (Harrison et al. 2006). An optional habitat mask showing the current distribution of fens and blanket and raised bogs was included in the integrated meta-model system to allow the user the option of constricting the climate space projections. No information was available on the future distribution of the habitats under the climate and socio-economic change scenarios.

Modelling the effects of 'real' Q95 and Q5 flows on species' suitability In order to link the estimations of 'real' $Q95$ and $Q5$ with the projections of species' potential climate space, a table was compiled (Table 2) relating the Ellenberg indicator moisture

Table 2 Ellenberg indicator values and water table requirements, and the drought and waterlogging classes defined for the selected species

Species	Ellenberg *F* value	Drought sensitivity			Waterlogging sensitivity		
		Water table (cms)		Defined class	Water table (cms)		Defined class
		Preferred min	Tolerable min		Preferred max	Tolerable max	
Fens							
Glyceria maxima	10	40	−40	2	40	100	1
Phragmites australis	10	−20	−100	1	0	50	1
Epipactis palustris	8	–	–	2	–	–	2
Ranunculus scleratus	8	−5	−20	2	20	40	2
Valeriana dioica	8	−5	−45	2	−5	–	2
Vertigo moulinsiana	–	–	–	2	–	–	1
Blanket and raised bogs							
Empetrum nigrum	6	–	–	2	–	–	2
Eriophorum vaginatum	8	−30	−50	1	0	10	2
Molinia caerulea	8	−50	−100	1	−25	0	2
Sphagnum cuspidatum	–	10	−1	3	10	30	1
Calluna vulgaris	6	–	–	2	–	–	2
Myrica gale	9	−50	−100	1	−25	10	1
Rhynchospora alba	9	−10	−20	2	0	10	1
Betula pendula	5	–	–	3	–	–	2
Betula pubescens	7	–	–	2	–	–	2
Coenonympha tullia	–	–	–	1	–	–	2
Listera cordata	6	–	–	2	–	–	2

values for Britain (Hill et al. 1999) to water level requirements of plants (Newbold and Mountford 1997). Ellenberg indicator values were originally devised for Central Europe and were not necessarily appropriate for application in a temperate oceanic area, such as Britain. Hill et al. (1999) re-calculated or estimated the values, such that they reflected the ecological niche of species in Britain and Ireland. The moisture indicator values used in this study (Table 2) range from five for moist-sites, mainly on fresh soils of average dampness, to 10 for shallow water sites that may lack standing water for extensive periods. The water level requirements of plants, however, are absolute values, usually based on the mean of over 50 water table measurements. Where no data were available from this source the species were assigned to classes based on their Ellenberg value only.

The two sources were used to divide the species into classes indicating low (1), medium (2) or high (3) drought or flooding sensitivity, as Ellenberg indicator values are on an arbitrary scale and in both cases there is not a direct relationship between flows and water tables. Also species' ecological requirements may vary in different parts of their range and according to local conditions and thus a broad classification was appropriate. The species' water level requirements were linked to the outputs from the hydrological meta-model by examining the range of 'real' $Q5$ and $Q95$ values under the climate change scenarios and attributing thresholds for percentage changes in these values from the baseline to the plant tolerance classes (Table 3).

Table 3 Species sensitivity thresholds for drought and waterlogging related to percentage change in 'real' $Q95$ and $Q5$, respectively

Species' sensitivity		Thresholds (% change)	
Class	Description	Stressed	Unsuitable
Drought sensitivity (related to $Q95$)			
1	Tolerant of drought	−35 to −50	<−50
2	Some drought tolerance	−25 to −35	<−35
3	Little drought tolerance	0 to −25	<−25
Waterlogging sensitivity (related to $Q5$)			
1	Tolerant of waterlogging	35 to 50	> 50
2	Some waterlogging tolerance	25 to 35	> 35
3	Little waterlogging tolerance	0 to 25	> 25

3 Results

The intention of the Regional Impact Simulator was to develop a user friendly and interactive software tool, which stakeholders could use to assess the implications of climate and socio-economic change for themselves. The user can interactively examine different scenarios, and the effects of individual model input parameters. The value of the Simulator (and the meta-models described within this paper) is not therefore limited to the restricted number of model runs described herein.

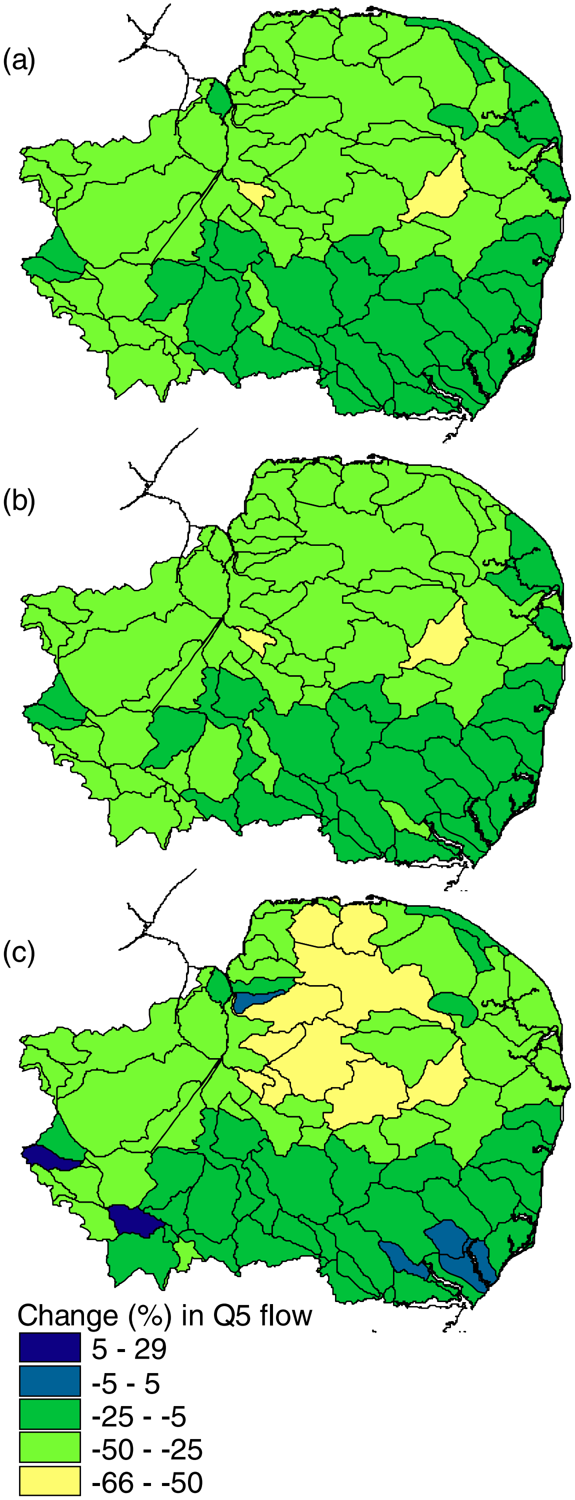
3.1 Changes in low and high flows

Changes to the 'real' high ($Q5$) and low ($Q95$) flows within each catchment are a consequence of the changing climate, the changing landscape (both urban and rural) and the changing water demand patterns of society consistent with the allocation of flows for abstraction.

In the 2050s High RS scenario, urbanisation in East Anglia uniformly decreases by 6% (compared to the baseline) and water returns are reduced because of decreased water consumption arising from a reduced population and lower per capita usage. Comparing the naturalised and 'real' $Q5$ flows (Fig. 1), it is apparent that the 'real' $Q5$ is mainly driven by the climate and landscape with very little influence from reduced returns. Flows decrease by up to 25% in the south and east of the region and by up to about 50% in the north and west. A slightly different situation occurs with the naturalised and 'real' $Q95$. The decrease in naturalised $Q95$ has a similar spatial pattern to that shown for the naturalised $Q5$. Elements of this spatial pattern are still visible in the 'real' $Q95$ (Fig. 2), however, local changes occur resulting from relative changes in returns and the unconstrained abstraction availability, that is also reduced under RS because of the high REP in this scenario. In catchments with significant urban areas (where the greatest absolute changes in urban areas and population occurs), the 'real' $Q95$ decreases compared to the naturalised $Q95$ as returns decrease more than the abstractions. In catchments with high water availability, 'real' $Q95$ increases as abstractions decrease more than returns.

Urbanisation increases in the GM scenario in the 2050s are spatially heterogeneous, with some catchments experiencing very large increases in urbanisation of more than 100% (relative to the baseline). Under the 2050s High GM scenario, the magnitude of change of 'real' $Q95$ from the baseline is extremely high in catchments experiencing very large new urban development associated with new towns, and the regional variability of the change of 'real' $Q95$ is higher than under the RS

Fig. 1 Percentage change from baseline in **a** naturalised *Q*5 for the 2050s High RS scenario, **b** 'real' *Q*5 for the 2050s High RS scenario and **c** 'real' *Q*5 for the 2050s High GM scenario for the sub-catchments in East Anglia



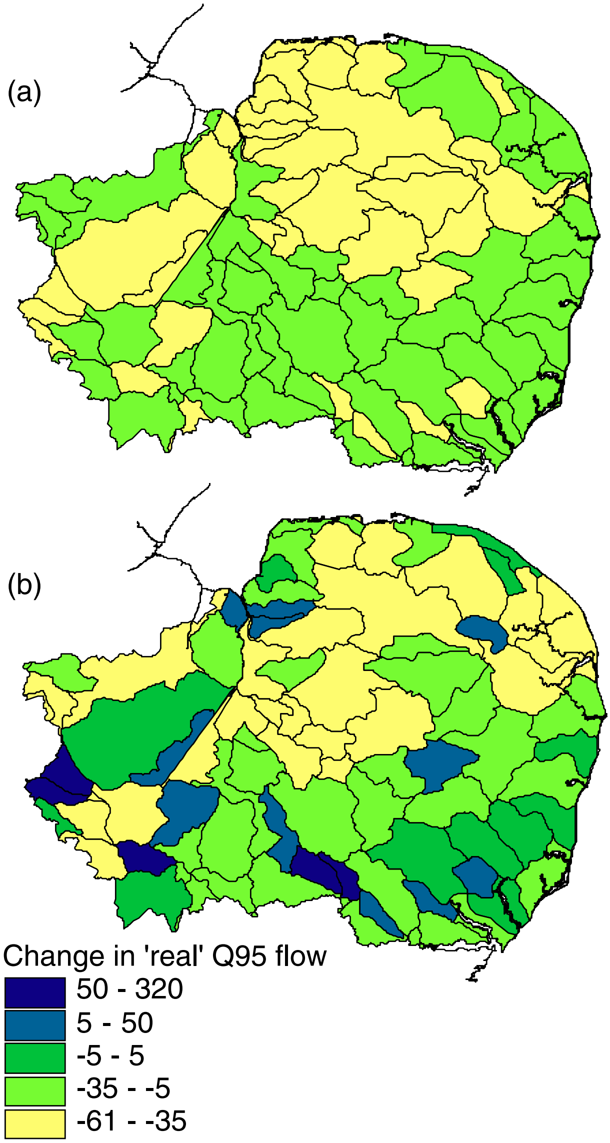


Fig. 2 Percentage change from baseline in **a** ‘real’ *Q*95 for the 2050s High RS scenario and **b** ‘real’ *Q*95 for the 2050s High GM scenario for the sub-catchments in East Anglia

socio-economic scenario (Fig. 2). This is due to the consequent concentration of returns in the catchments experiencing increases in urbanisation. Moreover, in these catchments, the impacts of the socio-economic changes are clearly evidenced in the increased (relative to the baseline) ‘real’ *Q*95 due to increased returns resulting from the higher population and per capita water usage.

Whilst ‘real’ *Q*5 decreases by more than 15% under the RS scenario, the ‘real’ *Q*5 under the GM scenario decreases by less and increases in some catchments by up to 33% in East Anglia (Fig. 1) for the above mentioned reasons.

It is apparent that the ‘real’ *Q*5 and *Q*95 are sensitive to the regional non-consumptive water demand which determines the returns. These regionally increase

Table 4 Non-consumptive demand, domestic water demand, industrial/commercial water demand, population and unconstrained abstraction availability for East Anglia 2050s High climate scenario and Baseline, GM and RS socio-economic scenarios

	Non-consumptive demand (Ml/day)	Domestic water demand (l/person day)[a]	Population ($\times$1,000)	Industrial/commercial water demand (Ml/day)[a]	Unconstrained abstraction availability (Ml/day)
2050HI Base	657	150	2,190.0	238	85
2050HI GM	1,035	190	3,068.3	376	181
2050HI RS	367	95	2,237.8	112	52

[a] Does not include leakage for that proportion supplied by the mains network

by 58% for the 2050s GM scenario and decrease by 44% for the 2050s RS scenario due to a change in water usage and population (Table 4). The impact on the 'real' $Q95$ of the change in returns is partially offset by a corresponding change in the unconstrained abstraction. For example, the effect of the 378 Ml/day increase in effluent returns due to the GM scenario, resulting from the higher non-consumptive water demand, is partially cancelled by an increase of 96 Ml/day in the unconstrained abstraction availability due to the very low REP under the GM future (Table 4).

The impacts of both climate and socio-economic changes are evident in the results for the 2050s High RS and GM scenarios, with the socio-economic changes being more significant in urbanised catchments. Comparing results under the 2050s Low GM and 2050s High GM scenarios reveals the impacts of climate under identical socio-economic conditions, e.g. urbanisation, domestic and industrial water demand, leakage and REP. From Figs. 2 and 3 it can be observed that although 'real' $Q95$ flows are generally reduced (compared to the baseline) under the 2050s Low GM scenario due to a decrease in annual precipitation, the flows are higher than under the 2050s High climate where precipitation decreases to a greater extent. However, in areas in the south west of the region, where urbanisation increases most, it is observed that the 'real' flows increase (compared to the baseline). This suggests that the increase in returns under the 2050s GM scenario are greater than the reduction in $Q95$ under the 2050s Low climate scenario but are less than the equivalent change under the 2050s High climate.

Figure 4 shows that 'real' $Q95$ under a 2050s High GM future in the north of North West England are greatly impacted. Whilst 'real' $Q95$ increases by up to 25% in the south, it decreases by up to 30% in the north, which reflects the heterogeneity of urbanisation and population density in the region – the effects of abstraction in the north are not offset by returns because of the low population density, whereas in the south the high population leads to returns being greater than the abstractions.

The results therefore show the significant influence of future urbanisation, population water requirements and regional environmental allocation of water on the hydrological responses of catchments. In general, climate and consequent rural land use changes in the 2050s will produce decreases in naturalised flows, the decreases being greatest under a 2050s High climate change scenario and in East Anglia. Whether the effects of the socio-economic scenario moderate or amplify these flow reductions depends on the characteristics of the scenario and the region:

- In those catchments with low levels of urbanisation (in much of the northern and central parts of the North West and much of East Anglia) and under GS and RS scenarios with low water demand, flows will be little affected by the

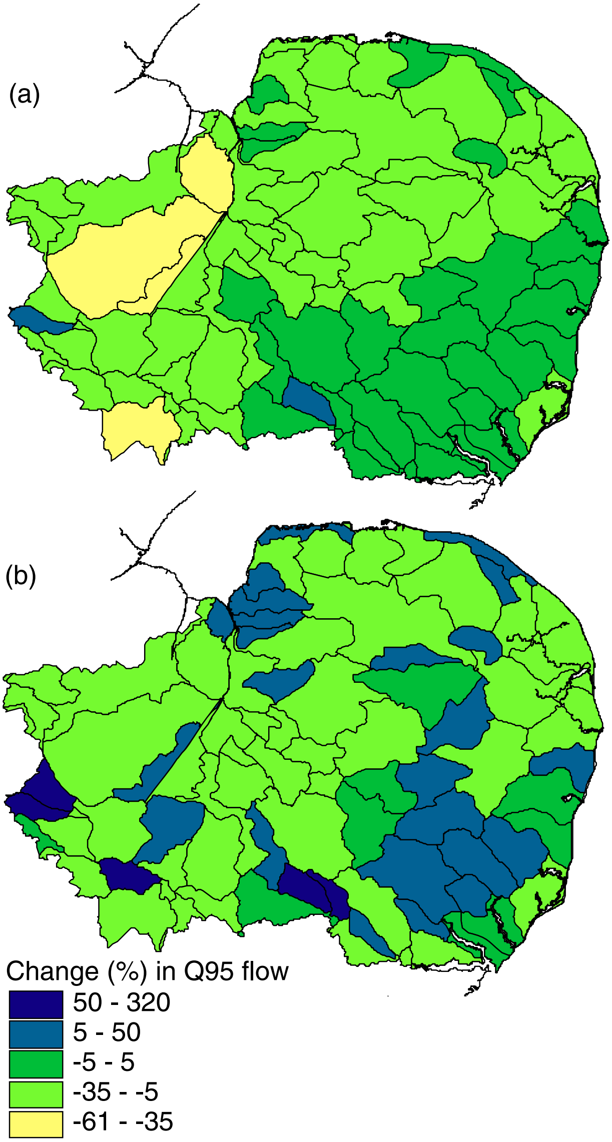


Fig. 3 Percentage change from baseline in **a** naturalised *Q*95 and **b** 'real' *Q*95 for the 2050s Low GM scenario for the sub-catchments in East Anglia

socio-economic factors, and 'real' flows will be determined primarily by climate change.

- In those catchments with low levels of urbanisation and under GM and RE scenarios in which water demand is high (especially domestic water consumption), increased abstraction allowed by a lower REP will lead to further reductions in flows.
- Finally, in those catchments with significant urbanised areas (such as the south east of East Anglia and to a lesser extent the south of the North West), the influence of urbanisation and population water requirements can be highly

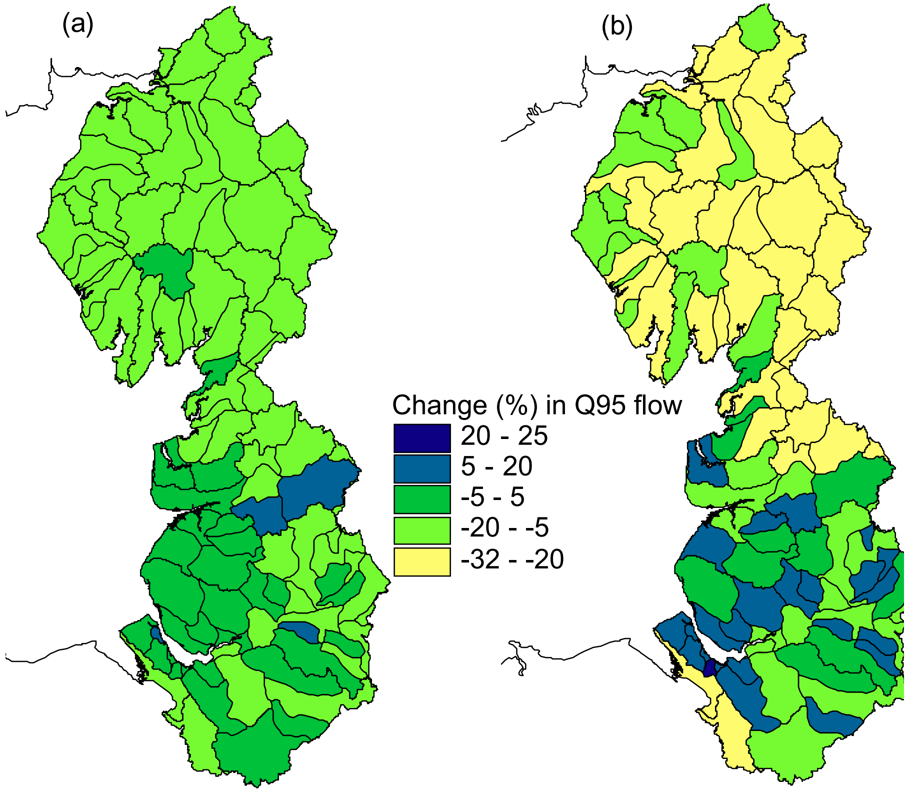


Fig. 4 Percentage change from baseline in **a** naturalised *Q*95 and **b** 'real' *Q*95 for the 2050s High GM scenario for the sub-catchments in North West England

significant depending on the SES. Decreases in low flows, as given by the naturalised *Q*95, will be partially or in some cases completely moderated by the effects of returns. High flows, as given by the *Q*5 may increase to a lesser extent than *Q*95 due to the increased returns.

Although the results show that increases in future regional water demand may benefit river flows in some catchments due to the associated increases in returns, it must be recognised that this is at the expense of water resources and flows in other catchments.

3.2 Changes in species suitability

Three of the six fen species modelled show virtually no change in climate space in East Anglia (*Phragmites australis* [Common reed], *Vertigo moulinsiana* [Desmoulin's whorled snail] and *Glyceria maxima* [Reed sweet grass]). The other three species show no gains in climate space, but significant losses of 28% (*Epipactis palustris* [Marsh helleborine]), 40% (*Valeriana dioica* [Marsh valerian]) and 56% (*Ranunculus scleratus* [Celery-leaved buttercup]) centred in the west and south of the region under the 2050s High scenario. In all cases this is likely to be due to increased summer dryness. Four of the 11 bog species show only small changes in climate space in the North West (*Eriophorum vaginatum* [Hare's-tail cotton grass], *Myrica gale* [Bog myrtle], *Rhynchospora alba* [White-beaked sedge] and *Calluna vulgaris* [Heather]). The other seven bog species show either no or small gains in climate space combined with larger losses, ranging from 11% for *Betula pendula* (Silver birch) to 86% for *Coenonympha tullia* (Large heath butterfly), under the 2050s High scenario.

The impacts of changes in high and low flows on species' suitability under the climate and socio-economic scenarios are illustrated for species with different sensitivities to drought and waterlogging stress (Table 2). For the fens, two species: *P. australis* and *R. scleratus* will be focused on, which are low and medium sensitive species respectively, as none are in the high drought or water tolerance classes. For the bogs, three species: *E. vaginatum*, *Listera cordata* (Lesser twayblade) and *Sphagnum cuspidatum* (a moss), with increasing drought sensitivity will be examined. Results for other species are similar to those being illustrated for the relevant tolerance class, but with varying losses and gains in climate space as previously described.

Application of a habitat mask to the climate space results led to only 14% (79 out of 566) of the 5 × 5 km grid cells in East Anglia being classified as suitable for either fen species, as fens are a very restricted habitat. *P. australis* shows little change under the scenarios, as it is found in a range of wetland habitats with varying water levels (Haslam 1972). *R. scleratus* is often found at the edges of ponds and ditches (Grime et al. 2007) and thus could be affected by changing flows. The modelling for *R. scleratus* shows that 12 of the 79 suitable grid cells (15%) become unsuitable under the 2050s Low scenarios and 36 (46%) become unsuitable under the 2050s High scenarios due to loss of climate space (Fig. 5). The impacts of changes in low and high flows on the suitability of *R. scleratus* are minor for the 2050s Low GM

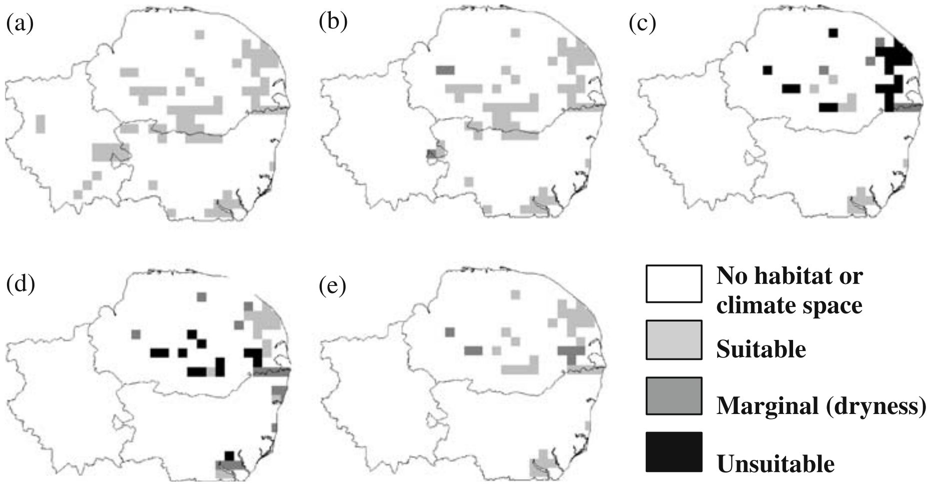


Fig. 5 Suitability for *R. scleratus* in East Anglia under various climate and socio-economic change scenarios, shown with a habitat mask: **a** baseline; **b** 2050s Low GM; **c** 2050s High GM; **d** 2050s High RS; and **e** 2050s High RE

scenario where just three cells become marginal due to dryness (or decreasing low flows). Results are more significant under the different SES combined with the 2050s High climate. Under the 2050s High GM scenario, 'real' flows are generally reduced except in the southwest of the region where urbanisation increases most causing small changes or increases in flows. This pattern is reflected in the species' suitability map where 21 cells are classified as unsuitable due to decreasing low flows (as a result of climate change and increased abstraction to meet the regional demand) and a further six become marginal in the northern half of the region. This increased abstraction, particularly in the vicinity of groundwater-fed wetlands in East Anglia, would be expected to be damaging through changed water table levels and to lead to changes in species composition (Klötzli and Grootjans 2001; Robroek et al. 2007). The suitable grid squares in the southwest remain unchanged due to increased effluent returns. The RS SES shows an overall decrease in urbanisation and flows which results in a general worsening of suitability for *R. scleratus* throughout the region including the southwest. However, suitability in the northeast of the region remains unchanged as 'real' *Q*95 flows increase in these catchments as abstraction decreases more than returns. The SES which results in the least decreases in species' suitability under a 2050s High climate is the RE scenario where urban development is projected to increase across the region. Here, only seven cells are classified as marginal due to dryness.

In North West England, species with a high drought tolerance show very little change under any of the scenarios. *E. vaginatum*, for example, particularly likes sites which are waterlogged in spring but drier in summer (Wein 1973) and can be found on different types of mire. Also it has a deep rooting system and thus it is able to withstand prolonged drought (Kummerow et al. 1988) and this is reflected in its potential tolerance of projected changes. It only loses 4% of grid cells in the northwest of the region under the 2050s High scenarios due to a retraction in climate space. Further, changes in high and low flows have no effect under the GM and RE SES and only minor effects under the GS and RS SES where four to six grid cells become marginal due to dryness.

Results for *L. cordata*, a species with a medium drought tolerance, are more significant. 14.5% of grid cells become unsuitable under the 2050s Low scenarios and 23% under the 2050s High scenarios, mostly from the southern part of the region, due to loss of climate space. Changes in low flows affect the suitability of *L. cordata* under just one SES (GM) where 29 grid cells (4%) become marginal due to dryness in the northeast (Fig. 6). This reflects the projected decreases in 'real' *Q*95 resulting from the low population density in the north of the region where abstraction is not offset by returns.

A study of vegetation and climate drivers of change in an Italian mire showed that bryophytes, in particular *Sphagnum* mosses, were sensitive to climatic conditions which affected water levels (Bragazza 2006). In this study, for *S. cuspidatum*, a bog pool specialist species representing the low drought tolerance class, impacts vary under both the climate and socio-economic scenarios. Suitability decreases by 9% under the 2050s Low climate and 14% under the 2050s High climate due to losses in climate space. The 2050s High GM scenario shows 11% of grid cells become unsuitable due to decreases in low flows and a further 4% become marginal (Fig. 6). This worsening in suitability is focused in the north of the region, where high water availability supports large abstraction, reflecting the same pattern of

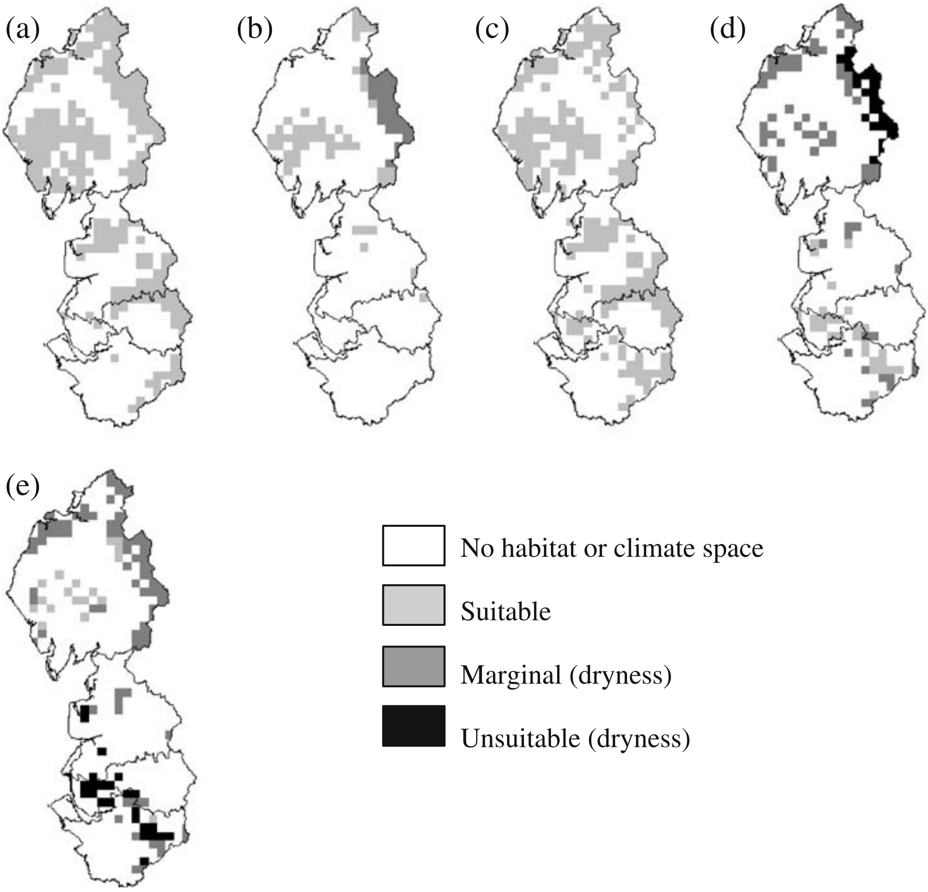


Fig. 6 Suitability for two bog species in North West England under various climate and socio-economic change scenarios, shown with a habitat mask: **a** *L. cordata*: baseline; **b** *L. cordata*: 2050s High GM; **c** *S. cuspidatum*: baseline; **d** *S. cuspidatum*: 2050s High GM; and **e** *S. cuspidatum*: 2050s High RS

changes in 'real' *Q*95 values as explained for *L. cordata*. Results for the RE scenario show a similar pattern but are slightly less severe. Alternatively, the 2050s High RS and GS scenarios project a general worsening in suitability for *S. cuspidatum* throughout North West England but the most severe impacts are focused in the more urbanised south, where 'real' *Q*95 decreases more significantly, due to reduced water consumption.

3.3 Adaptation options for improving the suitability of wetland species

The selection of a specific climate and socio-economic scenario within the Regional Impact Simulator results in default values being set for a wide range of input parameters. A user can then interactively examine the impact of these parameters by altering them individually or in combination within credible and extreme limits (see Holman et al. 2008). This allows a user to explore various adaptation options by changing relevant input parameters to produce more desirable outcomes.

Figure 7 shows the suitability for *R. alba* in North West England for the 2050s High RS scenario under different adaptation options. Results under this scenario with no adaptation classify 5 grid cells as unsuitable and 41 grid cells as marginal due to low flows, all in the drier, more urbanised south of the region. Increasing REP to the highest grade improves suitability slightly with the number of grid cells classified as unsuitable and marginal due to dryness decreasing to 2 and 29, respectively. Decreasing both domestic and industrial water demand results in an increase in the number of grid cells in the south of the region classified as unsuitable due to reduced return flows to the rivers and consequently greater flow reductions in these more

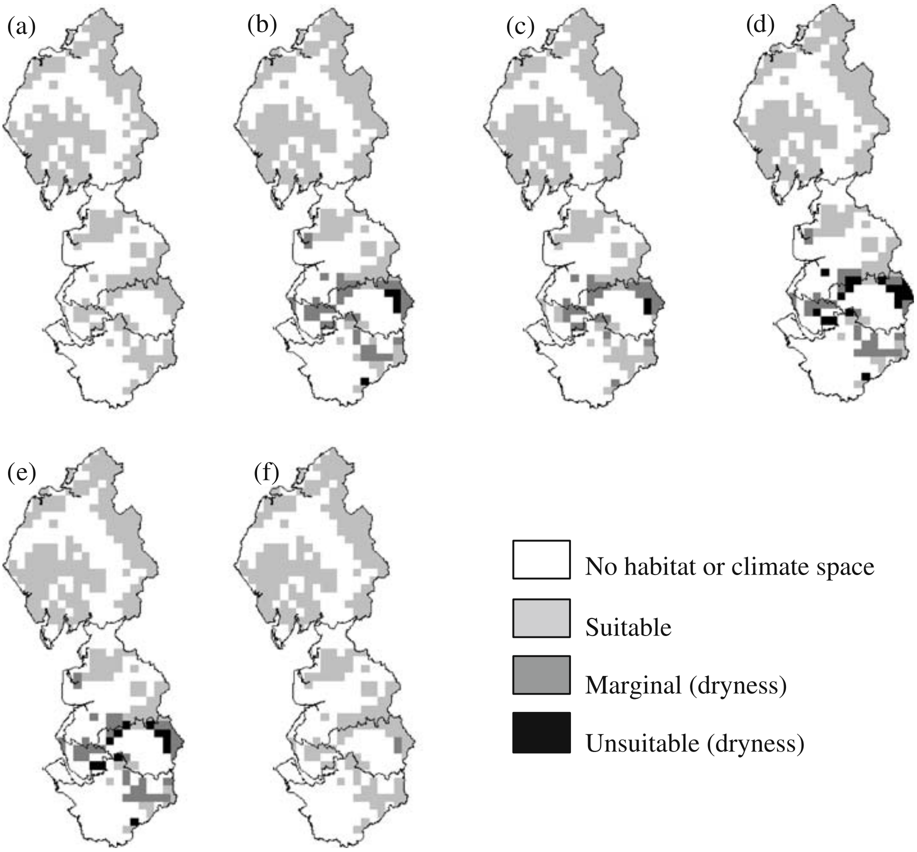


Fig. 7 Suitability for *R. alba* in North West England, shown with a habitat mask for **a** baseline; and **b**–**f** 2050s High RS scenario under different adaptation options: **b** no adaptation; **c** regional environmental concern changed from grade 2 to 1 (highest); **d** domestic water demand decreased from 95 to 75 l per person per day; **e** industrial water demand decreased from 410 to 330 Ml/day; and **f** domestic water demand increased from 95 to 135 Ml/day and industrial water demand increased from 410 to 575 Ml/day

heavily urbanised catchments. Alternatively, increases in non-consumptive demand lead to considerable improvements in suitability as more water is imported through the water supply network into these catchments to satisfy demand which in turn leads to increased return flows into the rivers at times of low flow. This seemingly counter-intuitive situation arises from two factors. Firstly, the cells classified as unsuitable and marginal due to dryness are located in the more urbanised catchments in the south of the region where effluent returns are greatest. And secondly, much of the abstraction to support this higher water demand is coming from the catchments with high water availability in the north of the region where the cells are classified as suitable. However, such options are only likely to be beneficial in reducing drought stress in surface water-influenced wetlands, and may have a detrimental effect in groundwater-dominated wetlands through the increase in abstraction needed, namely in East Anglia where approximately half of the water supply comes from groundwater. Combining an increase in REP (Fig. 7c) with an increase in non-consumptive water demand (Fig. 7f) results in all grid cells with appropriate climate space and habitat being classified as suitable. Similar improvements in suitability are predicted for the other species with medium to high drought sensitivity which had distributions in the southern part of the region.

In East Anglia, adaptation options for the fen species tend to be more limited than in the North West. Individually adjusting the water planning and demand change inputs within their credible ranges produced minimal improvements in the suitability of those species in the medium drought sensitivity class. These species could be considered very vulnerable, as effective regional adaptation options are limited, and local site-specific action needs to be taken. The local adaptation options available depend on whether the site is primarily influenced by surface water or groundwater level. In surface water controlled wetlands, river level at times of low flow may be augmented using groundwater abstraction to maintain prescribed flows (Petts et al. 1999). Alternatively in groundwater-dominated wetlands, using dams to raise water levels (e.g. Mawby 2003), the local relocation of groundwater abstraction sites (e.g. Harding 1993) or hydrological isolation (as has been carried out on parts of Wicken Fen, Cambridgeshire) may be possible. Increased winter storage or increased imports from other catchments or regions may lessen abstraction impacts in the summer low flow period, although this assumes that catchments outside of the region are not being negatively impacted and that adequate winter water is available. Although peak flood flows are shown to increase (Mokrech et al. 2008), this water is often not very suitable for abstraction because of high sediment load, and the winter flows indicated by the ‘real’ $Q5$ are shown to decrease in many catchments. Therefore an assumption of widespread availability of winter water may be optimistic.

However, it is not solely hydrological conditions which affect a site’s species suitability – ditch vegetation management (e.g. Painter 1998) and water quality also can play important roles. Improved nutrient treatment processes at STW and improved sewerage network design to reduce intermittent discharges/combined sewer overflows, in the face of the increased extreme runoff events (Mokrech et al. 2008) and increased precipitation intensity (Jones and Reid 2001), will be beneficial. In addressing these wider issues, national and regional water management policies should explicitly incorporate wetland ecosystem needs, particularly those related to naturally variable flow regimes and to the linking of water quality with water quantity (Baron et al. 2002).

4 Discussion

A linked meta-model approach has been used to investigate the impacts of climate and socio-economic change scenarios on water flows and wetland species in two contrasting regions of the UK. The main advantage of the meta-model approach is the speed of the system which enables users to undertake many more analyses than was previously possible using detailed process-based models. This allows users to examine why a response happens in relation to changes in different parameters rather than answers being taken on trust from a black box modelling approach (Audsley et al. 2008). The rapid run time also enables the user to undertake detailed sensitivity or uncertainty analyses of the models and scenarios using the credible and extreme limits provided for each input parameter as well as testing the outcome of various adaptation options. In using meta-models to integrate the impacts of climate and socio-economic change on the hydrological environment, the study provides novel insights into some (but of course not all) of the regional challenges facing the sustainable management of wetland habitats within these two contrasting regions.

The main disadvantage of such an approach is inevitably the accuracy and detail of the modelling. However, the intention within the RegIS2 study was to develop a regional approach which provided spatially distributed output, but which was never intended to provide site-specific impacts or adaptation assessments for individual wetlands. There are limitations in a hydrological modelling approach to simulate all catchments within such large regions. Firstly, there is a lack of integration between water quantity and quality – urbanisation leads to increased river flows and reduced drought stress but may lead to increased nutrient stresses, especially in winter. Unless there is an equivalent improvement in stormwater management, increased urbanisation and winter rainfall will lead to increased likelihoods of combined sewer overflows leading to decreased water quality and increased nutrient loads. None of these processes are included in the hydrological models. Secondly, the modelling has been undertaken at the sub-catchment and catchment scale, meaning that average changes have to be assumed across the entire sub-catchment, yet the hydrological response of species will depend on the water levels at the point of interest. Finally, a simplistic spatial distribution of future abstractions and returns, the locations of which will have important site-specific implications, has had to be used in the absence of detailed data for future scenarios.

Limitations of species–climate envelope models are discussed in detail in Harrison et al. (2003, 2006) and Thuiller et al. (2008). The benefits and limitations of neural networks compared to other species–climate envelope modelling techniques are described in Pearson and Dawson (2003), and have been evaluated in Thuiller (2003), Segurado and Araújo (2004) and Araújo et al. (2005). The extent to which changes in climate space will reflect future distributions is speculative. For many species, other abiotic factors (e.g. habitat requirements, atmospheric CO_2 concentration), biotic interactions (e.g. competition, predation and symbiosis with other species) and individual species characteristics (e.g. dispersal ability, population dynamics) are likely to have a strong influence on future distribution patterns.

Linking the hydrological and biodiversity meta-models introduces further assumptions and limitations. Mean Ellenberg values were used to link river flows and water requirements of species which may not hold over the whole range of the species, so the values used are only typical ones (Hill et al. 1999). Hill et al. also suggest that indicator values are more sensitive to plant requirements than a selected physical variable, such as water depth. Newbold and Mountford (1997) point out that survival of species outside their preferred levels can be affected when stressed by factors like competition, so the timing of the excess water may be critical, as many wetland plants are unable to tolerate periods of extreme wetness during the growing season. Fluctuating levels and flow also may be critical, both directly and because they affect the role of nutrients. These limitations may mean that the results presented are slightly conservative.

The integrity of mire or peatland ecosystems depends upon adequate quantity, quality, timing and temporal variability of water flow (Baron et al. 2002). In linking the range of variation of water flow (based on the 'real' $Q5$ and $Q95$) to species' suitability in wetland ecosystems, it has not been possible to incorporate the importance of other future changes including water quality, timing and hydromorphological condition. In addition, both the $Q5$ and $Q95$ indicators relate to conditions in the surface water system as opposed to groundwater, although it is realistic to assume that a reduction in the naturalised $Q95$, which will principally be composed of baseflow (or groundwater contribution) will have been caused by a reduction in catchment groundwater levels, albeit not necessary of the same magnitude. Also, the reduction in river levels associated with reduced $Q95$ may have a knock-on effect on groundwater levels in nearby hydraulically connected wetlands.

The main areas for future development of the regionalised hydrological meta-model relate to the integration of water quantity with quality (especially nutrients), improved representation of abstraction availability within catchments and the representation of the effects of groundwater abstraction on wetlands. Where more detailed analysis of the impacts of future change on individual wetlands is required, spatially distributed time-variant modelling approaches which incorporate the proximity of abstraction sites with wetlands are needed (e.g. Yussoff et al. 2002). The SPECIES model could be further developed by the incorporation of input variables tailored more to the habitat under consideration and also to the regional to local scale. Also more dominant species found in similar situations within the particular wetland types could be modelled in order to understand the potential impacts of climate change on wetland function. In conjunction with the hydrological model developments, water quality could be included more explicitly in the analysis as it affects nutrient availability and thus plant growth and competition. Further testing of the species sensitivity thresholds for drought and waterlogging which were used to link the hydrological and biodiversity meta-models would be useful to ascertain uncertainty limits in this key parameter. Such refinements are important as, at regional to local scales, more specific modelling is required in order to more realistically simulate current conditions.

Water quantity is not the only pressure on biodiversity in these environments, for example, land drainage is an important cause of loss of condition for raised bogs and fens, while upland management practices, such as heather burning and gripping can adversely affect blanket bogs. The quality of these habitats, particularly blanket and raised bogs which are dependent on atmospheric inputs, can also be affected by nutrient additions through atmospheric deposition. These pressures on wetlands are not part of the modelling, but could decrease wetland habitat quality and quantity and thus exacerbate the effects of climate change. Given the level of conservation designation of the habitats and some of their species, concern about them in the future is only likely to increase.

5 Conclusions

This paper has shown that future urbanisation and population water requirements have a significant influence on the hydrological response of catchments to future changes in high and low flows, with significant differences in the latter under the different socio-economic scenarios. This could lead to potentially significant changes in the suitability of East Anglia and North West England for fen and bog species, respectively, of medium and high sensitivity to drought. Thus there could be significant changes in community composition. This would lead to concern for the conservation of both the habitat and individual species. The possibilities for adaptation are limited and, for surface water-influenced wetlands, based on increasing non-consumptive water demand and supply that could be difficult to meet, especially in East Anglia.

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