Report

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SC090021 Wetland vision: adapting freshwater wetlands to climate change

Task 2. Literature review

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Key messages

Wetland vision and climate change.

- UKCIP 2009 scenarios suggest significant changes in precipitation and temperature (and hence evaporation) which are likely to have impacts on the water balance of wetlands by 2050.
- Impacts are likely to be greater in the south-east than the north and west.
- Scenarios are probabilistic, ideally requiring 10,000 models to cover different possibilities for each time slice.
- No national assessment of the impacts of climate change of groundwater is available.

Wetland functioning.

- Some wetlands are probably already on the margin of their hydrological range and thus very vulnerable to climate change.
- The hydrological regime of many wetlands is very changeable, care is needed to identify significant long-term change within this natural variability.
- Biological change is often driven by changes in extremes, which are very uncertain
- There is new emphasis on conservation of wetland functions and services rather than target species, sufficient information may not be available to assess the impact of climate change on ecosystem functions and services.
- Good data and models are available to undertake assessments at different levels of detail; the limits are the interactions between different scales.
- Water regime requirements of many plant communities, invertebrates, birds and fish known.

Models and data.

- Good data and models are available to undertake assessments at different levels of detail; the limits are the interactions between different scales.
- Water regime requirements of many plant communities, invertebrates, birds and fish known.
- Trigger points for archaeological remains are well-understood, especially in peat wetlands.
- There is little evidence that small wetlands are more or less vulnerable.
- Landscape/catchment connectivity may be key to wetland resilience and recovery.

Wider issues

- The highly managed character of many wetlands make assessments of climate change more complex.
- Climate change may lead to alteration of infrastructure, such as flood defences.
- Future competing water uses under different scenarios may be fundamental in determining adequate water for wetlands.
- Some wetland surface features may be created in new areas where climate is suitable, however, it is not possible to create aquifers or soil stratigraphy and the archaeological records contained within existing wetlands.
- Reduced anthropogenic pressures may allow wetlands to survive climate change

1. Objective

The overall objective of this project is to develop a toolkit consisting of a tiered set of methods to assess the impact of climate change on wetlands and use this to provide a framework for assessing wetland adaptation strategies. This document provides a review of key literature to support the research.

2. Scope of review

The project steering group agreed that the literature review should be a small element of the project focusing on climate change impacts on wetlands of England Wales through hydrological change. However, relevant literature relating to wetlands beyond England and Wales and including other impacts of climate change, such as increased CO_2 levels or temperature rises, should be included where possible. This document provides an initial scoping of the literature. The Steering Group agreed that the review should be finalised at the end of the project to allow incorporation of other literature that arises during the course of the project.

3. Background

Wetlands are critically important ecosystems. Many large areas of functioning wetlands provide catchment-wide benefits including regulating water quality and quantity, ameliorating high and low flows thereby moderating the extremes of floods and droughts, assisting in aquifer recharge, storing carbon, as well as providing important habitats for wildlife and attractive landscapes for people. Smaller functioning wetlands can also provide high value habitats and, if linked, increase the potential for species migration under climate change. The continuing presence and functioning of wetlands will be of enormous benefit to society against the backdrop of environmental change. Hydrology is the single most important component of wetlands; it is the presence of saturated conditions for all or part of the time that separates them from terrestrial or fully aquatic systems (Mitsch and Gosselink, 1993). Wetlands will thus be impacted by climate change through changes in rainfall, river flow/flooding regimes, groundwater levels and increased air and water temperatures, which will result in modified evaporation.

4. Factors affecting the impacts of climate change on wetlands

It is widely accepted that soil water table level regime is a dominant control on plant communities in wetlands (Silvertown *et al*., 1999). Wheeler and Shaw (1995) identified three main environmental gradients that determine the dynamics of a wetland system and are thus key elements in conceptual understanding:

(a) acidity: ranging from acid sites on peat soils/substrates to base-rich (e.g. those which are fed by water coming from a Chalk aquifer);
(b) fertility (availability of nutrients, primarily N and P): ranging from oligotrophic to eutrophic;
(c) hydrological regime: ranging from highly variable water level (such as floodplains within flashy catchments) to more stable water levels fed by groundwater.

Consequently, it is the combination of hydrological regime and water quality that determines the ecological character of many wetlands. At North Meadow, on the upper Thames floodplain for example, annual inundation from the river is important not only because it saturates the wetland, but also because it bring nutrients to the soil. A number of studies has collated water level and water quality requirements of wetland species and vegetation in Great Britain (Newbold and Mountford, 1997, Elkington *et al.*, 2001, Gowing *et al.*, 2002, Wheeler *et al.*, 2005); for some wetlands critical values depend on soil type. Poff *et al*. (2002) recommend that the best way to consider the impact of climate change on wetlands is through the hydroperiod *i.e*. the patterns of water depth and the duration, frequency and seasonality of flooding. Related approaches include the use of Sum Exceedence Values (Gowing *et al*., 1997, 2002).

There is a range of factors influence the impact of climate change upon wetlands (Table 1). Heathwaite (1993) highlighted the importance of landscape position and differing hydrological sources for wetland sensitivity to climate change. Acreman (2005) and Acreman and Miller (2007) also identified landscape location and associated water transfer mechanisms as primary factors in assessing hydrological impacts (including climate change but also those associated with, for example, abstraction) on wetlands. Wheeler *et al.* (2009) incorporated landscape location into their framework for wetland assessment. This concept has been used in other countries, for example, Winter (2000), recognised six hydrological landscape positions for wetlands in the USA.

Table 1. Factors affecting the impact of climate change on wetlands

Factor	Notes	Reference
Acidity	Related to water source e.g. high pH from Chalk groundwater, low pH from rainfall	Wheeler and Shaw (1995), Brunning (2001)
Fertility/ nutrient status	Nitrogen, phosphorus	Wheeler and Shaw (1995)
Water table level		Newbold and Mountford, 1997, Silvertown *et al*., (1999), Elkington *et al.* (2001), Gowing *et al*. (2002), Wheeler and Shaw (1995), French (2009)
Water supply mechanism	Rainfall, river water, groundwater	Acreman (2005) and Acreman and Miller (2007), Schot and Winter (2006)
Landscape location	Often drives water supply mechanism	Heathwaite (1993), Winter (2000), Acreman (2005), Acreman and Miller (2007)
Soil moisture		Gowing *et al*. (1997), Toogood *et al* (2008)
CO_2 concentration		Schippers *et al.* (2004)
Temperature		Reading (1998), Reading & Clarke (1999)

Brunning (2001) assessed the quality of nine archaeological sites in the Somerset Levels which contained waterlogged archaeological remains as part of the MARISP (Monuments at Risk in Somerset's Peatlands) project. He concluded that the state of preservation of the archaeological resource is best in soils with a low pH and where a significant overburden of peat and/or clay is present. The presence of an alluvial clay covering was shown to augment the quality of preservation of archaeological remains within the underlying peat. Whilst this project makes no explicit reference to the impact of climate change on the preservation of archaeological remains in the Somerset Levels and Moors, the consequences of climate change can be inferred from this report.

Schot and Winter (2006) proposed that groundwater–surface water interactions constitute an important link between wetlands and the surrounding catchment. Wetlands may develop in topographic lows where groundwater exfiltrates. This water has its functions for ecological processes within the wetland, while surface water outflow from the wetland may provide water downstream. Wetlands may also receive inflowing surface water, which may become relatively stagnant giving rise, if the unleying substrate is relatively permeable, to groundwater recharge. This transition of surface water to groundwater provides groundwater resources for human and ecological purposes further down the groundwater basin. Groundwater–surface water interactions in wetlands thus play an important role with respect to spatial and temporal availability of both surface water and groundwater in the entire basin. Understanding (natural) groundwater–surface water interactions may help water resources managers deal with such issues as flood mitigation, groundwater exploitation, and biodiversity conservation, in a more integrated and sustainable manner.

French (2009) monitored the impact of gravel extraction on the water table on an alleviated landscape and the consequences for the archaeological remains. He determined that the impact of deep drainage was dramatic both upstream and downstream from the gravel pit. During the restoration phase, the water table upstream and downstream from the pit was quickly restored.

In addition to hydrologically driven changes, other aspects of climate change can alter wetlands. For example, increased concentrations of carbon dioxide can alter species composition in some wetlands, independently of any hydrological or temperature change, with the most responsive species out-competing the less responsive species (Arp *et al*. 1993). Similar trends have been predicted using a dynamic model to investigate the effect of atmospheric carbon dioxide increase on plant growth in freshwater ecosystems (Schippers *et al.,* 2004). The direct impact of temperature on wetland animals, and especially the dynamics of breeding, has been investigated in detail through the example of the Common Toad (*Bufo bufo*), using a 20 year study of a breeding population in a pond in southern England (Reading 1998; Reading & Clarke, 1999). This research not only showed how the arrival of toads at the breeding pond was correlated with the mean daily temperature over the 40 days immediately preceding, but also that early breeding was associated with warm winters. In addition, the duration of the tadpole stage was negatively correlated with the date that the first spawn appeared, and indeed the tadpole stage lasted up to 30 days longer in early spawning years than in late ones.

5. Specific knowledge of water regime requirements of wetlands

Good knowledge is available on the water regime characteristics of some wetland features, such as plant communities, invertebrates, birds and archaeological remains (Table 2). Some key characteristics are discussed below.

5.1 Inundation frequency

Floodplain grassland
High inundation frequencies promote target species of floodplain grassland (Hardtle *et al*., 2006) and increased flooding leads to rapid change in plant community composition (Toogood and Joyce, 2009). Community composition is highly dynamic with respect to soil moisture status (Toogood *et al.,* 2008*).* Flooding caused an overall reduction in species richness of a floodplain grassland and an increase in biomass production, which were only partly reversed after ten years. A future increase in flooding frequency might be detrimental to species richness in floodplain grasslands (Beltman et al., 2007). Flooding causes changes not only to the established vegetation, but also to the seed-bank composition, favouring wetland species (Bekker and Oomes, 1998). Species composition can recover rapidly from short duration flooding (Sera and Cudlin, 2001). Community composition at quasi equilibrium demonstrates that floodplain-meadow species have distinct hydrological niches (Silvertown *et al.*, 1999). These niches have been quantified for 99 of the more common meadow species (Gowing et al., 2002). Community types (as defined by NVC) also show clear segregation in “hydrological niche space” (Gowing *et al*., 2005).

Soil fauna
Earthworm abundance and biomass is usually reduced by extensive flooding at a large spatial scale locally by 80–100%. Some invertebrates can survive short periods of flooding (Ausden *et al.*, 2001) and others can survive shallow floods if there are sufficient variations in local topography to afford nearby refugia of relatively higher ground. However, prolonged and deep flooding is not attractive to either waterfowl or waders and will greatly reduce the density of invertebrates present in any area. Where controlled flooding is applied at a large scale, recolonisation by annelid worms from unflooded refuge sites is not likely to occur.

Therefore, the time interval between two flooding events should exceed the development time from cocoon to adult of the earthworm and enchytraeid species present. To maintain viable populations of annelids, especially in spring when they serve as food for wetland birds, a new inundation in the recovery period (2–3 months for enchytraeids and about half a year for earthworms) should be avoided or kept short (Plum & Filser, 2005). Soil invertebrate biomass (and hence potential to support breeding-wader populations) declines within a few years of water levels being raised (Ausden and Hirons, 2001). Subsequent colonisation by anoxia tolerant species does occur but biomass levels do not recover to those of hydrologically-managed grasslands that avoid anoxia.

Table 2. Specific knowledge of water regime requirements of wetlands

Factor	Impact	Reference
Frequency of flooding	Floodplain grassland vegetation	Hardtle *et al*. (2006)
	Invertebrates (e.g. earthworms)	Ausden *et al.* (2001), Plum and Filser (2005)
Timing of flooding	Bird feeding and breeding	Poiani (2006), Thomas (1980), Beintema (1987)
	Vegetation species composition	Sera and Cudlin (2001)
	Vegetation species richness	Beltman *et al.* (2007)
	Seed bank composition	Bekker and Oomes (1998)
Duration of flooding	Vegetation communities	Baptiste et al (2006)
	Invertebrates (e.g. earthworms)	Ausden *et al.* (2001)
Depth of flooding	Through light transmission – vegetation	Vervuren *et al*. (2003), Blom *et al* (1994)
	Waterfowl	Thomas (1982)
Phosphorus level	Vegetation species richness	Baptise et al. (2006)
Nitrogen level	Vegetation species diversity	Van Oorschot et al. (2008)
Oxygen level	Earthworms mortality	Plum and Fliser (2005)
	Archaeological remains	Smit *et al*. (2006)
Temperature	Fish kills	Lindenschmidt *et al*. (2009)
pH	Archaeological remains	Smit *et al*. (2006)

5.2 Timing

Timing and duration of a flooding event are considered the main components for the survival of flooding (Vervuren *et al*. 2003). Timing (*i.e*. season) will influence water temperature as well as plant growth stage. Poiani (2006) found that aquatic birds tend to disperse as an immediate response to inundation, but they will then gather on flooded areas where food sources are abundant in order to breed. Large concentrations of birds tend to be prominent as floodwaters recede and both adult and young birds concentrate at the remaining water-bodies.

Many wader species are also attracted to standing water on grasslands in winter. In addition to inundated areas, adjacent saturated soils, such as on wet grasslands, provide important feeding zones for waders, because soil invertebrates are forced closer to the surface as the water table rises. The higher water table levels also increase the penetrability of the soil that aids bird species, such as curlew and snipe, which probe for their prey (Green, 1986). Whilst high water tables are attractive to wading birds, standing water causes the death of many soil-dwelling invertebrates. This can result in short-term benefit to the birds, but prolonged flooding can greatly reduce the food supply available to feeding waders with consequences for the long-term use of a site by these birds (Ausden *et al*., 2001). Increases in flooding of grassland can be beneficial to species of conservation concern in many floodplain areas (Ausden & Hirons 2002), but the timing of flooding, the underlying soil type and the flooding history are all important in determining the impact on the soil invertebrate community in any given area (Ausden *et al*., 2001). Winter flooding of previously unflooded areas greatly reduces the soil macroinvertebrate prey of many breeding bird species, largely as a consequence of invertebrates vacating flooded areas.

In spring and summer, almost all waterfowl species nest on dry land, preferably along land/water edges (Thomas, 1980). Breeding numbers would therefore tend to be low wherever flooding is widespread and in areas with a low edge/water surface area ratio. Too much open water is not beneficial. Where flooding does extend over large areas in summer, shallow floods are more beneficial than deep floods, particularly for dabbling ducks which require water depths of less than 30cm to feed (Thomas, 1981). Intermittent out-of-bank flooding is likely to be the most detrimental to breeding waterfowl, resulting in the destruction of nests and lost clutches.

Waders are ground nesting birds and, in general, the greatest densities of breeding waders will occur in wet grasslands where the water table is high (Beintema, 1987). However, the optimum conditions usually equate to a water table 20-30cm below the surface in early March (Beintema, 1983) and where wet conditions are restricted to shallow drainage channels or rills (Milsom *et al.*, 2002). Extensive flooding during the breeding season will actually remove breeding habitat for waders and major intermittent floods will destroy nests, clutches and young birds.

During spring and early summer, raised water tables are of benefit to breeding waterfowl and waders. However, out-of-bank flooding would remove breeding habitat and intermittent flooding will actually destroy nests, clutches and young birds. Some birds, such as snipe will, re-nest if first nests are destroyed by floods.

In ideal conditions, floods can be very beneficial. On the Ouse Washes, in a typical year, the main winter floods come in late November or early December and leave the washes under a deep bank-to-bank flood along their entire length until March. These floodwaters drain off gradually during early spring to provide ideal conditions for wet grassland breeding species and grazing cattle (an essential management tool). During late spring, summer and autumn, small temporary and permanent pools provide the best areas for birds, but major flooding can have negative consequences. During the June 2008 floods some 600 pairs of ground-nesting waders (lapwing, snipe and redshank) lost eggs or chicks in the flooding on the Ouse Washes. These floods were also the main cause for the collapse in the Ouse Washes population of black-tailed godwits, one of the UK's rarest breeding waders. Across Cambridgshire, more than 1,600 pairs of wading birds and ducks had their nests destroyed, including more than 1,100 pairs of eight species of duck, including 12 pairs of the rare garganey.

The summer 2007 floods severely affected ground nesting waders in the Severn and Avon Vales, Gloucestershire and south Worcestershire. Lapwing nesting in wet meadows were severely hit by floods in May and June. Some that tried to produce a second clutch were flooded a second time. Many curlew chicks were drowned in the deep floodwater in late June. Redshank young tend to fledge by mid-June, many moved off the vales before floods rose in late June.

These issues have been recognised worldwide. For example Knutson and Klass (1997) found that the 1993 Mississippi floods led to a decline in many species due to poor reproductive success. However, they felt that in the long-term major flooding was important for maintaining suitable floodplain habitat for birds. Tome (2002) studied the density of meadow birds breeding in Slovenia related to three different flooding regimes and found positive effects of floods on many birds including lapwing, sky lark, marsh warbler and corn bunting.

The following provides specific examples of the timing of nesting.

- Common snipe (*Gallinago gallinago*) late March to July. Snipe build their nests of grass on the ground, often concealing them in clumps of rushes. Late flooding is a hazard, drowning nests, but snipe are persistent breeders and females made produce three or four clutches in the season before rearing young. In these circumstances the latest nests may be started in July finishing in August.

- Redshank (*Tringa totanus*) late March to May. The nests are situated on the ground on tussocks or grassy hollows. Cattle tend to be introduced only in late May or early June, to minimise trampling of the eggs.
- Lapwing (*Vanellus vanellus*) mid-March to June. Birds often nest on arable land and relocate their young to nearby wet ground with short vegetation (such as grazed pasture) in order to find suitable feeding.
- Curlew (*Numenius arquata*) April to July. These birds particularly favour upland blanket bogs, lowland raised bogs and rough pasture.

5.3 Duration

Baptiste *et al*. (2006) present a table of flood duration for different vegetation types. Lethal threshold values for inundation duration under extreme conditions were defined. The most important constraint that plants have to deal with during flooding is oxygen deficiency, especially in soils with high organic matter content. Long duration flooding causes major “setbacks” in the vegetation composition. Shortening the flood duration is, therefore, advantageous to vegetation (Baptise *et al*., 2006). Duration is the primary driver of community composition in flooded grasslands (Violle, Cudennec et al. 2005). Length of exposure is important in the recruitment of species (Coops and van der Velde, 1995). The lower boundary of species distribution on floodplains is set by duration of flooding during the growing season (van Eck *et al*., 2004).

5.4 Water depth

Deep floods are characterized by very low median light transmission levels (Vervuren *et al*., 2003). The degree to which riparian plants survive a given period of submergence is determined by light intensity. Some species show adaptation to sudden changes in flood depth (Blom *et al.*, 1994).

In winter, many waterfowl species are attracted to standing water and can feed in water depths up to 50cm (Thomas, 1982). In general, the larger the area flooded the better, especially for roosting waterfowl. However, feeding conditions are usually better for many species at the margins of flooded areas, so several smaller areas of floodwater are usually more beneficial to waterfowl than one large one. Moreover, prolonged deep flooding can make an area as unattractive to waterfowl as areas without any surface water at all (Thomas, 1976).

5.5 Water quality

Phosphorus
About 30% of the sediment and adsorbed phosphorus that enters the detention area during an extreme flood is retained within the area (Baptise *et al.,* 2006). Species richness (of floodplain grasslands) decreases significantly with increasing phosphate supply (Hardtle *et al*., 2006). Species diversity is limited by nutrient availability with N & P interacting (Janssens *et al*, 1998). Phosphorus availability in floodplain soils is a function of flooding regime rather than soil wetness (van Oorschot *et al.,* 1998).

5.6 Stagnation

With high organic content, flood water causes high oxygen demand resulting in an elevated risk of low dissolved oxygen concentrations in the storage areas that can lead to fish kills.

This risk can potentially increase if sediment oxygen demand increases due to crop residue and/or water temperatures increase (Lindenschmidt *et al.,* 2009). Depletion of oxygen also exacerbates earthworm kills (Plum & Fliser, 2005).

5.7 Water temperature

As discussed above, the aforementioned high risk of low dissolved oxygen concentration in flood storage areas that can result in fish kills can potentially increase if sediment oxygen demand increases due to increases in water temperatures (Lindenschmidt *et al*., 2009).

5.8 Management

Mowing
Both flood frequency and mowing affect species composition of temperate lowland floodplain meadows. However, flood regime was less importance than mowing regime (Gerard *et al*., 2008). Annual disturbance and/or the creation of open vegetation gaps through annual mowing are necessary in order to maintain species-rich vegetation in these systems. Mowing is effective at nutrient removal (Bakker and Olff, 1995) and favours fungal dominance in the soil microbial fauna (Donnison *et al.*, 1999). It is essential for the maintenance of species diversity in regularly flooded sites (Antheunisse *et al*., 2006).

Grazing
Plant species richness (of floodplain grasslands) decreases significantly under grazing compared to mowing (Hardtle *et al*., 2006).

Species introduction
Floods were found to be ineffective at introducing species of conservation interest to a system via hydrochory, other techniques are needed (Bissels *et al.,* 2004).

5.9 Archaeological remains

Holden *et al.* (2006) considered the trigger points of *in situ* preservation of wet-preserved archaeological remains. They discuss the hydrological, biological and chemical components of preservation and deterioration of waterlogged remains, including those in urban areas. Case studies are presented from rural (Somerset wetlands, Cambridgeshire fens, Sutton Common - South Yorkshire, Wood Hall - North Yorkshire, Pict’s Knowe - Dumfries and Galloway) and urban (Rose and Globe theatres - London, Parliament Street - York, Bergen - Norway) contexts. Smit *et al.* (2006) set standards for the monitoring of archaeological sites and remains in the Netherlands, where much of the land is below sea-level and where water tables are frequently close to the surface. They define conditions and trigger points, showing the importance of water table, soil moisture, pH and Eh for the preservation of a range of organic archaeological remains. The principal trigger points are identified as:

- Water table lowering — shifting the burial environment from one with permanent saturation to one with non-permanent saturation;
- pH: increases to above 6.5
- Redox potential: increases to above 200 mV
- Electrical conductivity: decreases to below 1000 µS cm^{-1}
- Dissolved oxygen: increases to above 5 mg l^{-1}

Douterelo *et al.* (2009) counted total bacteria and ^{14}C-leucine assimilation rates, alongside water table, pH and redox potential, at two sites in South Yorkshire: the rain-fed wetland of Sutton Common and the rain-fed raised mire of Hatfield Moors. Their work offers a methodological innovation in the assessment of the burial environment through actual bacterial counts. They found that bacterial activity always decreases with depth, and that water table, pH and redox potential correlated highly with this decrease. Whilst nutrient addition showed that nutrient availability was limiting for microbial communities in upper soil horizons, this was not the case for lower levels.

5.10 Functional approach

Much of the past research on wetlands has been on particular species or communities, including vegetation, birds, invertebrates and archaeological remains. More recently thinking has focused on the need to conserve functions and services of wetlands. The current National Ecosystem Assessment is following the Millennium Assessment approach in assessing the current status and recent trends in ecosystem services. However, the functions and services of wetlands and their sensitivity to climate change are more difficult to quantify than for species or communities. Maltby (2009) produced a functional assessment of wetlands which allows functions to be quantified, but its application depends on detailed site field surveys and data. The method has not yet been applied to many sites beyond the case study wetlands of the original research.

6. Wetland typologies

All wetlands are unique to some extent. However, it is useful to classify wetlands into broad types according to characteristics they have in common to aid assessment and prediction. Many wetland typologies are available and vary depending on the purpose of the classification (Table 3). Lloyd *et al.* (1993) produced a hydrological classification of East Anglian wetlands (based on water source, e.g. surface water or groundwater) as an aid to assessment of their vulnerability to abstraction. Gilvear and McInnes (1994) identified 12 types of wetland based on possible combinations of hydrological mechanisms. In recognising the importance of landscape location in influencing wetland character, Winter (2000) defined six types of wetlands in the USA: mountainous, plateau and high plain, broad basins of interior drainage, riverine, flat coastal, and hummocky glacial and dune; though some of these do not occur in the UK.

Table 3. Wetland typologies

Elements	Reference
Water source - surface or groundwater	Lloyd *et al*. (1993)
	Gilvear and McInnes (1994)
Landscape location - mountains, glacier, coastal etc	Winter (2000)
Landscape location and water source - upland, depression, valley bottom	Acreman (2005), Acreman and Miller (2007)
Landscape location, water source, acidity, fertility, vegetation	Wheeler *et al*. (2009)
Biological/hydrological	SNIFFER (2009)

Wheeler *et al.* (2009) developed a classification system of WETland water supply MEChanism types (WETMECS) that combines landscape situation, water supply mechanism, hydrotopographical elements, acidity (base-richness) and fertility. A key aim was to identify homogeneous wetland types that are supported by the same hydrological processes and thus broad classes of wetlands that would respond in a similar way to external or internal impacts. However, it was clear from the study that there are several different hydrological mechanisms which can deliver the same National Vegetation Classification (NVC) wetland vegetation community when combined with other variables, such as water quality and soil/geology type.

For each WETMEC type, the wetlands are further classified into sub-types according to such characteristics as the strength of spring discharges. Within each sub-type, there are two further categories, which define ‘ecological types’: base-status (base-rich, sub-neutral and base poor) and fertility (oligotrophic, mesotrophic and eutrophic). The base status category can be determined on site by pH measurements. The fertility category requires phytometric analyses of soil samples. For both these categories, the plant communities present may be used as a surrogate indicator. NVC community types are then related to ecological types.

As part of development an approach to hydrological impact assessment, Acreman (2005) and Acreman and Miller (2007) used landscape location and water transfer mechanisms as a means of classification, defining flatland (upland and lowland), slope, depression and valley-bottom wetlands as key freshwater types.

SNIFFER (2009) identified 11 wetland types in Scotland, which are a mix of biological and hydrological classes: wet woodland, wet grassland, seepage/flush/spring, fen, swamp, reedbed, wet heath, bog, saltmarsh, dune slacks and Machair.

For climate change studies, where the major impact is through hydrological change, classification needs to incorporate the hydrological regime, either directly, such as through water supply mechanisms, or indirectly such as through landscape location or vegetation type that indicates hydrological mechanism.

7. Structured approaches to assessing climate change impacts

UKCIP produced generic guidelines for developing a risk-uncertainty framework for incorporating climate change in decision-making (Willows and Connell, 2003). This framework describes a tiered approach to considering climate change, enabling decision-makers to recognise and evaluate the risks posed by climate change.

Tier 1: This is termed ‘risk screening’ and is generally a qualitative assessment of potential risks posed by a changing climate.
Tier 2: The second tier approach involves a generic, quantitative assessment to identify potential consequences of change. In this tier, models need to include important processes, but can have generalised form and parameters that are broadly applicable to areas or regions, as they are not intended to represent details of specific sites.
Tier 3: This third tier is a more detailed quantitative assessment to establish the magnitude and probability of the consequences of a changing climate and hence future risk for particular locations. This level of approach requires models that represent all relevant processes and parameter values at the site of interest, such as a coupled surface-groundwater modelling approach.

Acreman and Miller (2007) developed the three tier approach to produce a similar risk-based framework for wetland assessment, which proposed a sequential approach starting with simple general screening-level models and moving to more complex models if simple models cannot provide sufficiently detailed or certainty of results. Table 4 classifies assessments of the impacts of climate change upon wetlands according to a three tiered approach.

Whiteman *et al.* (2004) describe the steps required to produce a local hydrological impact assessment of groundwater-fed wetlands, including consideration of the sometimes problematic translation of hydrological impacts into ecological effects. Impact assessment in this study was defined as determining whether existing or future actions such as abstraction, can cause significant degradation of the ecological character for a particular wetland. The steps include a simple desk study of existing data, field investigations, numerical modelling, ecological impacts and stakeholder consensus. It was noted that the steps used cannot be considered independent as an understanding of the interactions between the wetland and key water sources develops as data are collected and analysis undertaken.

7.1 Qualitative (Tier 1) approaches

A wetland example of Tier 1 analysis is provided by Winter (2000) who considered the impact of climate change on wetlands using conceptual qualitative understanding of processes in different landscape locations in the USA. Burkett and Kusler (2000) also presented a qualitative assessment of the potential effects of climate change on US wetlands, in which permafrost wetlands, coastal and estuarine wetlands, peatlands, alpine wetlands, and prairie pothole wetlands were thought to be the most likely to be affected by a warmer, wetter climate and associated changes such as a rise in sea-level. Krause *et al.* (2007) developed an eco-hydrogeological framework to assess the risk of significant damage to groundwater-dependent terrestrial ecosystems in the UK and the Republic of Ireland to implement the WFD. The framework considers the variety of groundwater controls and pathways of seven wetland types and allows a specific assessment to be made of the vulnerability of different wetland types to groundwater related risks.

Table 4. Structured approaches to assessing climate change impacts on wetlands

Tier	Impact	Indicators	Location	Reference
1	Conceptual, qualitative	General	USA	Winter (2000) Burkett and Kusler (2000)
2	Water balance	Hydrology	Upwood Meadows	Dawson *et al.* (2003)
			Ouse washes	Acreman (2005)
			Eastern Graveney Marsh	"
			Seasalter Levels	
			Sheringham and Beeston Regis Commons	
			Pevensey Levels	
			Crymlyn Bog	
			Lopham and Regrave fen	
		Vegetation	GB wide	Acreman *et al.* (2009)
3	2-D model soil hydraulic	Hydrology	Somerset Levels and Moors	Armstrong *et al.* (2080)
			Halvergate Marshes	Armstrong and Rose (2001)
	Coupled surface-groundwater	Hydrology	North Kent Marshes	Thompson *et al.* (2004)
	3-D groundwater	Hydrology	Pevensey Levels	Bradford and Acreman (2003)
	?	Archaeology	Sutton Common	Chapman and Cheetham (2002)

7.2 Simple quantitative (Tier 2) approaches

Dawson *et al.* (2003) developed a simple water balance model for Upwood Meadows NNR to compare a 1961-1990 baseline using observed precipitation and evaporation data to simulated wetland water levels, with four climate change scenarios from the UKCIP98 (Low and High for both the 2020s and 2050s time slices). Results showed summer water levels which are lower, especially for the 2050s.

Acreman (2005) collated a series of simple water balance studies for wetlands across England and Wales, including the Ouse Washes, Eastern Graveney Marsh and Seasalter Levels, Sheringham and Beeston Regis Commons, Pevensey Levels, Crymlyn Bog, Lopham and Regrave fen to show their use in assessing the impacts of abstraction, droughts and flood control. Acreman *et al*. (2009) also used simple models to predict climate change impacts on rain-fed and river margin wetlands across Great Britain.

Johnson *et al.* (2005) undertook water balance modelling of prairie potholes using WETSIM to predict representative water levels for future climate change scenarios. Such regional modelling can assist in directing funds for wetland restoration to areas which are likely to be hydro-ecologically viable under a changing climate in the long-term. Kont *et al*. (2007) synthesised long term monitoring data showing varying wetland climate change effects over a large region. Zhang and Mitsch (2005) investigated hydrological processes of four different flow-through created wetlands using water budget model. The model includes surface inflows and outflows, precipitation, evapotranspiration, and groundwater seepage. An integrated systems approach was developed using STELLA 7.0; its dynamic interface level control features (e.g. buttons and switches) offering a wide range of capabilities.

French (2009) monitored the impact of a gravel extraction on the water table on an alleviated landscape, and the consequences for the archaeological remains. Monthly rainfall, soil moisture deficit, groundwater level, water temperature, pH, conductivity, redox potential and dissolved oxygen wet measured. The impact of water abstraction was monitored up to 1.5 km from the gravel pit. The study determined that the impact of deep drainage was dramatic both upstream and downstream from the gravel pit, with the water table lowered by 5 m near the pit, and by 2 m at least 1500 m downstream from the gravel pit, and 500 m upstream from the pit. During the restoration phase, the water table upstream and downstream from the pit was quickly restored.

7.3 Complex quantitative (Tier 3) approaches

Many two-dimensional models, based on land drainage concepts and basic soil physics theory (Youngs *et al*, 1989; 1991), have been developed to simulate the water table levels in wetlands. Examples include FDRAIN on the Somerset Levels (Armstrong *et al*., 1980; Armstrong, 1993) and DITCH (an adaptation of FDRAIN) on South Lake Moor Somerset and Halvergate Marshes, Norfolk (Armstrong and Rose, 1999). Similar models have been used in the USA (DRAINMOD, Skaggs, 1982) and The Netherlands (SWATRE, Belmans *et al*., 1983).

The numerical groundwater code, MODFLOW, has been used to model water table levels within several wetlands, including: Pevensey Levels (Bradford and Acreman, 2003); Badley Moor (Gilvear *et al.,* 1993); Weston Fen (Lloyd and Tellam, 1995); Narborough Bog (Bradley, 1996); Catfield Fen, Norfolk (Gilvear *et al.*, 1997); and Great Cressingham Fen (ENTEC, 2003). A similar model AQUA3D has been employed at Thorne Moor (Bromley *et al.*, 1999) whilst MIKE II has been used to assess river levels and floodplain storage in the Derwent Ings (JBA, 2003). There are more numerous examples of similar detailed approaches overseas, including models of groundwater and cypress ponds in the Coastal Plain forest region, USA (Mansell *et al.*, 2000), Prairie wetlands in USA (Gerla and Matheney, 1996), cutover peatlands (Kennedy and Price, 2004) and blanket peat (Lapen *et al.*, 2005) in Canada. Restrepo *et al.* (1998) developed a module for MODFLOW that would simulate three-dimensional wetland flow hydroperiods and wetland interactions with aquifers and slough channels, representing flow routing, inflow and outflow of water, and evaporation from wetlands for different hydroperiods.

Cooper *et al.* (1995) assessed the effects of changes in climate on aquifer storage and groundwater flow to rivers. Their generalized aquifer/river model can incorporate spatial variability in aquifer transmissivity and is applied with parameters characteristic of Chalk and Triassic sandstone aquifers in the United Kingdom, and is also applicable to other aquifers elsewhere. The model can be run using historical time series of recharge, estimated from observed rainfall and potential evaporation data, and with climate inputs perturbed according to a number of climate change scenarios.

Thompson et al. (2004) describe the application of a coupled hydrological-hydraulic modelling system (MIKE SHE / MIKE 11) to a typical coastal lowland wet grassland (Elmley Marshes, Isle of Sheppey). This model was subsequently used by Thompson *et al.* (2009) to investigate the impacts of climate change. Meteorological inputs to the original calibrated model (calibration based on observed ditch water and groundwater levels) were forced according to predicted changes in precipitation, temperature, wind speed and radiation from

UKCIP02. Scenarios are developed for four emissions scenarios for the 2050s with two revised potential evapotranspiration (PET) time series for each scenario (1. changes in temperature, 2. changes in temperature, wind speed and radiation).

Some Tier 3 tools have been applied to wetlands with a rich archaeological resource where data on preservation is published in peer-reviewed journals: Sutton Common (South Yorkshire) and Somerset Levels and two locations where monitoring data exists but has, to date, not been published: Star Carr (North Yorkshire) and Flag Fen (Peterborough). Van de Noort *et al.* (2001) undertook hydrological monitoring and linked this to the preservation of organic archaeological remains from the Sutton Common site (probably one of the best studied sites in the UK from the perspective of the direct relationship between (rain-) water and archaeology). Sutton Common itself is a fourth century BC ‘marsh-fort’ on the edge of the early Post-glacial Lake Humber, an area with little natural drainage. Van de Noort *et al*. (2001) found that waterlogging and wet-preservation of archaeological remains are closely linked to the dynamic water table. Chapman and Cheetham (2002) used the hydrological monitoring data at the Sutton Common site to introduce the use of a vertical zonation of the *quality* of wet-preservation of archaeological remains (permanently dry, intermittently saturated and permanently saturated) and linked this to a 3D model of dynamic water table movements. This project set a benchmark for the hydrological monitoring on archaeological sites with wet-preservation, whilst the use of the three zones is a simple, but important way in which the quality of wet-preservation can be assessed.

8. Impacts of climate change on wetlands

8.1 General

The UK Climate Impact Programme (UKCIP) produced a benchmark set of climate change scenarios for the UK (Hulme *et al*, 2002) and highlighted the differential alterations to the climate across the UK; for example the changes to wetter winters and drier summers would be greatest in the South and East.

In analysis of British fens, marshes and swamp, Hossell *et al*. (2000) summarised the possible risks under predicted climate change as:

- Change in species composition to favour temperature responsive species.
- Increased risk of soligenous fens drying out in summer.
- Drought may exacerbate damage to plant species from atmospheric pollution.
- Increased pollution risk from runoff from surrounding agricultural land.

A number of studies have been undertaken to assess the impacts of climate change upon different types of wetland in Great Britain (Table 5).

8.2 Rain-fed wetlands

Heathwaite (1993) reported that the global carbon sink capacity of peatlands has decreased by 16-37% over the last 200 years. In Britain this will be exacerbated by predicted climate change and associated changes in peat water levels. The effect may not, however, be spatially consistent. Rain-fed upland wetlands may well be affected by increased erosion, which will outweigh peat regeneration due to increased precipitation. River-fed and groundwater-fed lowland wetlands may be subject to increased flooding and elevated water tables (due to sea level rise and the sinking land surface), promoting peat regeneration. Likely areas include the Norfolk Broads, Suffolk valley peats, Cambridgeshire Fens, Humberhead Levels, Derwent Ings and possibly the Somerset and Gwent Levels. Maintenance of higher water tables will minimize the effects of climate change on peatlands and preserve their existing carbon sink capacity, providing they also remain hydrologically intact.

Table 5. Impacts of climate change on wetlands in Great Britain

Type	Location	method	impact	Reference
Rain-fed	Upwood Meadows	Water balance model	Summer water levels lower, winter max stay the same	Dawson *et al* (2003)
	GB	Water balance model	Reduced water levels in late summer/autumn, greater impact in SE than NW	Acreman *et al* (2009)
River-fed	21 UK rivers	Rainfall-runoff model	More runoff in winter, less in summer	Arnell and Reynard (1996)
	15 UK rivers	Rainfall-runoff model	Increase in flood peaks in north and west, decrease in south and east	Kay *et al*. (2006)
	NW England	Rainfall-runoff model	Lower summer flows, higher winter flows	Fowler and Kilsby (200&0
	GB	River-floodplain model	Reduced water levels in late summer/autumn, greater impact in SE than NW	Acreman *et al* (2009)
	Kennet	River model	Reduced runoff volume, high proportion of flow in spring	Limbrick *et al* (2000)
Groundwater	UK	Simple aquifer/river model	South and east more sensitive than north and west; sandstone aquifers more robust than others	Wilkinson and Cooper (1993)
	Chalk and sandstone	Simple aquifer/river model	Reduced flows from Chalk, less severe reductions	Cooper *et al*. (1995)
	Chalk	2 layer groundwater model	Decrease in autumn flows	Yusoff *et al*. (2002)
	Chalk, sandstone, limestone	?	Minimum levels increased in sandstone and limestone, decreased in the chalk	Bloomfield *et al.* (2003)
	Midlands sandstone	Regional groundwater model	Decrease in recharge	Lewis et al. (2004)
	Chalk	Regional groundwater model	High recharge rate in winter but for shorter period, decline in groundwater levels	Jackson *et al*. (2010)

As noted above, Dawson *et al.* (2003) used a simple water balance model for Upwood Meadows NNR, a rain-fed wetland in Cambridgeshire They applied four climate change scenarios from the UKCIP98 (Low and High for both the 2020s and 2050s time slices). Results show that summer water levels will be lower, especially for the 2050s. There was no significant change in maximum winter water levels. Simulated water levels suggest that three wet meadow species currently found at Upwood Meadows may experience greater water stress due to water tables falling below their ideal ranges.

Knapp *et al.* (2008) developed a conceptual framework for assessing the impacts of projected more extreme precipitation regimes characterised by larger rainfall events and longer intervals between events. A conceptual “soil water bucket” model was employed to assess the impact of changes in precipitation upon three broad ecosystem types – xeric, mesic and hydric (i.e. wetland). In the case of the hydric environments, changes towards higher individual rainfall events separated by longer dry periods will lead to a reduction in the number of days when soils are anoxic. This will therefore increase the rates of some aerobic processes that will include potentially rapid rates of organic matter decomposition. Wetlands will in these circumstances be vulnerable to invasion by “upland” species.

8.3 River-fed wetlands

There have been many studies of the impacts of climate change on rivers flows to assess implications for floods and droughts. These have significant implications for river-fed wetlands, such as floodplains. Arnell and Reynard (1996) used a conceptual rainfall-runoff model, at a daily time-step, to simulate potential changes in river flow in 21 UK catchments using equilibrium and transient climate change scenarios. A baseline period of 1951-1980 was employed against which result of the climate change scenarios were compared. The equilibrium scenario is for conditions around 2050 whilst the transient scenarios are for 1990-2050. Results show that by 2050 annual rainfall increase by over 20% in the wettest scenarios and declines by over 20% in the driest scenarios. The drier areas of southern and eastern England demonstrated the greatest sensitivity to climate change whilst changes in more humid north and west were relatively small. All the scenarios showed a greater concentration of runoff in winter whilst summer flows generally decline, especially in the drier scenarios, so that river flow is more seasonal.

Limbrick *et al*. (2000) employed a semi-distributed version of the INCA model to simulated flows for the River Kennet, using observed flow (at Theale) in 1992 and 1985 respectively. Three climate change scenarios were simulated based on outputs from HadCM1 and HadCM2 for the 2050s when simulating changes in greenhouse gas concentrations and, in the case of HadCM2 cooling effects of atmospheric sulphur aerosols. Climate model results were disaggregated to 0.5°× 0.5° resolution over so uth-central England. Original daily model meteorological input data were perturbed by the monthly deviations suggested by the climate model results. Results show a substantial reduction in total annual runoff (average reduction 18.97%) Increase in seasonality was indicated with a greater percentage of flow occurring in spring and sometimes summer. It was suggested that due to increased soil moisture deficits the groundwater recharge period could be reduced whilst the ecological status of the river (e.g. for aquatic macrophytes and fish) could decline.

Kay *et al.* (2006) set-up rainfall-runoff models for 15 catchments using a simplified version of a relatively simple conceptual hydrological model (PDM, Moore, 1988). The model was directly forced with outputs from a high resolution (25 km) Regional Climate Model (RCM), HadRM3H for two scenarios, a baseline scenario (1961-1990) and a future (2071-2100) scenario corresponding to IPCC A2 SRES. The focus of hydrological model results was flood frequency. Decreases in flood peaks were shown in a number of catchments in the southeast of England, despite increases in winter mean rainfall and extreme rainfall. It was argued that this was due to elevated summer and autumn soil moisture deficits. Catchments further north and west showed increases in flood peaks of over 50% for the 50 year return period in some cases.

Fowler and Kilsby (2007) employed daily rainfall and temperature data from the multi-ensemble HadRM3H regional climate model (RCM) for a baseline (1960-1990) and future (2070-2100 using the IPCC A2 SRES scenario – UKCIP02 Medium-High) time slices. These outputs were bias-corrected using monthly factors so that the modelled monthly average for the baseline period matched the observed. The same correction factors were used for the future scenario. The bias-corrected RCM outputs were used as inputs to a simplified version of the part-physically based Arno hydrological model calibrated for catchments in northwest England. Results show that annual runoff increases slightly at higher elevations but reduces by approximately 16% at lower elevations. Runoff was predicted to become more seasonal with mean summer flows 40-80% of baseline and winter flow increasing by up to 20%. Low flows (Q_{95}) are predicted to decrease by 40-80% with important implications for river ecology as well as abstraction. High flows ($>Q_5$) increased by up to 25%, particularly at high elevations.

Although most climate models predict drier summers and wetter winters in the UK, detailed predictions of impacts on river flow regimes vary. Reynard *et al.* (2004) concluded that floods would be less severe or similar to current conditions in the future, based on outputs from the Hadley Centre global climate model. This suggests that the associated behaviour pattern of river ecosystem biota, such as movement of fish species on to floodplains (such as dace) or to breed in backwaters (such as pike) may be largely unaffected. Likewise, habitat and invertebrate food for other floodplain users such as birds (e.g. redshank (*Tringa totanus*) and lapwing (*Vanellus vanellus*)) will not be affected. Other models suggest there may be more frequent inundation of floodplains. Areas of flooding provide roosting for waterfowl. Feeding conditions are usually better for many species at the margins of flooded areas, so several smaller areas of floodwater are usually more beneficial to waterfowl than one large one. However, standing water causes the death of many soil-dwelling invertebrates. This can result in short-term benefit to the birds, but prolonged flooding can greatly reduce the food supply available to feeding waders (Ausden *et al.*, 2001).

Dawson *et al.* (2003) presented a quantitative assessment of water availability (represented by rainfall less potential evaporation) for the UK and Ireland under current (1961 – 1990) and future (UKCIP98) climate scenarios. This showed that water availability may increase in winter for all areas. Northwest Ireland and northwest Scotland could also have a small increase in water availability during the summer; other regions could have reduced or unchanged water availability during the summer. Southeast England could have particularly large reductions in water availability during the summer months. Acreman *et al*. (2009) predicted that reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities in late summer and autumn with greater impacts in the south and east of Great Britain. In addition, impacts on rain-fed wetlands will be greater than on those dominated by river inflows.

Van de Noort *et al*. (2001) determined the rate of wetland heritage loss for England between 1950 and 2000 in three broad wetland types: upland wetlands (predominantly blanket mires), lowland peatlands and lowland alleviated wetlands. They concluded that the peatlands had suffered the most in the second half of the twentieth century, especially the upland peats, and that up to 50% of the archaeological sites that existed in 1950 were lost by 2000; the rate of loss of wetland heritage in alleviated lowlands was much less, and also less visible. The study quantified individual threats, and which of these are likely to be augmented by climate change. Loss to upland peat through accelerated erosion and 'wind-blow' was considered the greatest threat to wetlands under the climate change scenario of wetter winters and drier summers, whilst in the lowlands the increased fluctuation and/or lowering of groundwater tables and human impact in response to climate change are identified as potentially affecting the long-term wet-preservation of archaeological remains

The study by Thompson et al. (2009) using a coupled hydrological-hydraulic model (MIKE SHE / MIKE 11) of the Emley Marshes, Isle of Sheppey and predicted changes in precipitation, temperature, wind speed and radiation from UKCIP02 developed scenarios for four emissions scenarios for the 2050s with two revised potential evaporation (PET) time series for each scenario (1. changes in temperature, 2. changes in temperature, wind speed and radiation). They found progressively lower groundwater and ditch water levels with the higher emissions scenarios. Changes were particularly large using PET forced with changes in temperature, wind speed and radiation. Under these scenarios the water table rarely reached the ground surface whereas it was common under baseline conditions (the calibrated model). Flood extents were reduced accordingly. It was suggested that these hydrological changes will result in loss of specialist coastal grazing marshes vegetation species. Reduced flooding is also likely to have negative impacts on breeding wading birds, most notably redshank and lapwing.

Howard *et al.* (2008) assessed the impact of climate change on the archaeological resource of river systems and provided palaeoenvironmental evidence of the impact from past phases of relatively rapid climate change, especially from the Medieval Warm Period and the Little Ice Age, to quantify the response of rivers and the impact on the archaeological resource. They identified the following potential processes induced by climate change: increased river incision and lateral migration; increased slope channel coupling, augmenting both erosion and alluviation; increased groundwater table fluctuations; human impact responding to climate change in the form of hard engineering (realignments of rivers, levee/dike construction and more frequent dredging); and human impact in the form of soft engineering (e.g. 'construction' of washlands); changes in the agricultural regime (introduction of deeper rooting crops, water abstraction for irrigation); and the human impact in the form of new forms of leisure and tourism.

8.4 Groundwater-fed wetlands

Wilkinson and Cooper (1993) studied the effect of changes in climate on aquifer storage and river recharge using a simple model of an idealized aquifer/river system. They showed the combined influence of aquifer properties and climate change scenario on the system response. Changes in the seasonal distribution of recharge may have a critical effect on low flows in rivers supported by baseflow. However, rivers supported by slowly responding aquifers may show a considerable delay in response to climate change allowing an opportunity for water resources planning over an extended period. They concluded that Sandstone aquifer systems are more robust to future changes in climate, due to their slow response. However, benefits from changes in the management of such systems will not be gained for several years after implementation.

Cooper *et al.* (1995) assessed the effects of changes in climate on aquifer storage and groundwater flow to rivers. Their generalized aquifer/river model, which can incorporate spatial variability in aquifer transmissivity, was applied with parameters characteristic of Chalk and Triassic sandstone aquifers in the United Kingdom. Simulations of baseflow suggest large proportional reductions at low flows from Chalk under high evaporation change scenarios. Simulated baseflow from the slower responding Triassic sandstone aquifer shows more uniform and less severe reductions. The change in hydrological regime is less extreme for the low evaporation change scenario, but remains significant for the Chalk aquifer. Thus, low evaporation climate change scenarios are likely to give little change in recharge of the Triassic sandstone aquifer, and small but significant changes for the Chalk aquifer. Under high evaporation change scenarios there are significant reductions in recharge for both aquifers.

Yusoff *et al*. (2002) investigated the impacts of climate change on the Chalk aquifer in west Norfolk using a two-layer transient groundwater flow model for the future periods 2020–2035 and 2050–2065 using HadCM2-generated values. They suggested a 26% decrease in recharge in autumn for all scenarios as a consequence of the smaller amount of summer precipitation and increased autumn evaporation. Relatively little effect on summer groundwater levels (1 to 2% decrease) was predicted, but a decrease of up to 14% in autumn river baseflow volumes was possible. Two opposite trends are predicted. The Medium High (MH) emission scenario predicts first a decrease (2020) in annual groundwater levels followed by an increase (2050), while the Medium Low (ML) scenario predicts an increase followed by a decrease, with a greater range between these two figures. However both predict a fall in end of year levels.

Bloomfield *et al.* (2003) analysed the impacts of climate change on groundwater levels in three aquifer units: Chalk in Kent, Permo-Triassic Sandstone in Devon and Jurassic Limestone in Lincolnshire. They found that minimum annual groundwater table levels increased in the Sandstone and Limestone, but decreased in the Chalk, by 2080. It is unclear whether this impact is due to the character of the aquifer or to the geographical position, or both. However, it suggests that wetlands fed by Chalk aquifers in the south-east may be more vulnerable to climate change than those fed by sandstone or limestone further north or west.

Lewis *et al*. (2004) used two existing regional groundwater models, covering parts of the East Midlands and West Midlands Permo-Triassic sandstone aquifers, to investigate the potential effects of eight UKCIP02 climate change scenarios on groundwater levels and river flows. Both aquifers are heavily exploited and the results of this study have been used by one of the major UK water supply companies to help guide preparation of Water Resource Plans submitted as part of the UK Periodic Review Process. A decrease in recharge was predicted over the entire eastern (western) model area which is equivalent to around 7% (9%) of current abstraction for the Low 2020s scenario, and 13% (20%) for the High 2050s scenario. These calculated reductions are sufficiently large to warrant consideration of any potential impact on aquifer sustainability.

Jackson *et al.* (2010) predicted changes in groundwater resources of a Chalk aquifer in central-southern England for the 2080s. They used an ensemble of GCMs, a distributed recharge model and a groundwater flow model of the Chalk aquifer of the Marlborough and Berkshire Downs and South-West Chilterns. Not all of the simulations in the ensemble agree about the sign of the change, however, ten of the thirteen predict a decrease. The multi-model results suggest that the seasonal variation in the groundwater resource will be enhanced with higher recharge rates occurring during winter but for a shorter period of time. The effects of climate change are shown to depend significantly on the type of land-use. For the River Wye, which has the smallest catchment in the study, the predicted change in baseflow ranges from a 74% decrease to an 88% increase. For the Kennet, the largest catchment, this range is from a 17% decrease to a 24% increase. The ensemble average suggests that mean groundwater levels will decline at all of the boreholes considered in the study and by up to nearly three metres over the higher ground away from the rivers.

8.5 Climate change impacts on archaeology

Research into the consequences of climate change on the archaeology of wetlands is practically non-existent. The scientific monitoring of wetland archaeology is only some 15 years old, and most of the available publications focus on ways in which the quality of preservation can be monitored and assessed. Furthermore, many ongoing projects have, to date, not been published in peer-reviewed journals, and often only exist as unpublished (and generally unavailable) internal reports. Nevertheless, several of the published papers on wet-preserved archaeological remains offer explicit conclusions on the conditions by which organic archaeological remains are preserved, and from this the impact of the most likely climate change scenarios can be extrapolated. Archaeologists put much emphasis on the specificity of the site, not just in terms of its geographical setting, but also in terms of past human engagement and impact on particular locales. As a consequence, broad-based Tier 1 and site-specific Tier 3 studies appear more frequent than Tier 2 assessments of categories of wetlands.

8.6 Relevant non-UK research on climate change impacts

Winter (2000) argued that wetland vulnerability to climate change is determined by hydrological landscape position. Each of six typical positions has varying proportions of atmospheric-, surface-, and ground- water inputs. Narrow riparian wetlands in mountainous landscapes and plateau / high plains landscapes are river- and groundwater-fed. Climate change vulnerability mainly depends on upstream catchment precipitation and groundwater discharge. Wetlands in gently sloping, poorly drained, upland areas (e.g. blanket bogs) are predominantly rain-fed and therefore highly sensitive to climate change. In gently sloping, poorly drained, lowland areas (e.g. coastal plains, broad riverine valleys and glacial lake plains), input from rainfall is augmented by discharge from regional groundwater systems. Groundwater flow can help maintain wetland communities during droughts, although shallower water tables will enhance evaporative losses. Wetlands developed at the base of scarps and terraces in riverine or flat coastal landscapes are predominantly fed by upslope groundwater seepage. Such flow systems are local scale, increasing wetland vulnerability compared to those fed by regional systems. Isolated closed depression wetlands in glacial and dune areas can be groundwater-fed and/or discharge to groundwater, according to relative elevation. Their sensitivity to climate change is therefore highly variable. In summary, wetlands dependent primarily on precipitation for their water supply are highly vulnerable; those dependent primarily on regional groundwater discharge are least vulnerable as large groundwater systems buffer the effects of climate change. To assess wetland vulnerability to climate change, the relative proportions of water from these sources should be determined.

Burkett and Kusler (2000) presented a qualitative assessment of the potential effects of climate change on wetlands with specific examples from the United States. Permafrost wetlands, coastal and estuarine wetlands, peatlands, alpine wetlands, and prairie pothole wetlands were thought to be the most likely to be affected by a warmer, wetter climate and associated changes such as a rise in sea-level. A wide range of potential impacts were envisaged, ranging from changes in community structure to changes in ecological function (both degradation and improvement). Climate change mitigation options were proposed for wetlands including the reduction of current anthropogenic stresses, allowing for inland migration of coastal wetlands as sea-level rises, active management to preserve wetland hydrology, and a wide range of other management and restoration options.

Kont *et al.* (2007) analysed temperature, moisture, wind, and sea level data for Estonia and the Baltic Sea region over the second half of the twentieth century and investigated the concomitant changes in wetland ecosystems. Water levels in an inland mire were found to be linked to the changing climate, with rising levels in a ridge-pool microtope, but falling levels in a ridge-hollow microtope. Coastal grasslands were found to be undergoing replacement by reed beds, scrub and woodland although this effect was due to management and other anthropogenic activities as well as climate change. This demonstrates the potential complexity of individual wetland response to climate change.

In the USA, Johnson *et al.* (2005) predicted that waterfowl breeding areas would shift to the eastern and northern fringes which are predicted to be wetter, because many wetlands in these areas have been drained, there is a need to protect and restore wetlands here to mitigate the future effects of climate change.

Banaszuk and Kamocki (2008) analysed long-term hydrological observations (water depth, river flow) combined with meteorological observations to assess how water resources within a Polish mire are linked to climatic fluctuations. It was shown that over the last three decades the climate has become milder and drier. In response, surface water inflows to the mire have decreased.The extent and duration of flooding have also both declined. Summer precipitation has declined whilst evapotranspiration has increased. As a result there has been a substantial fall in groundwater table within the mire (summer water table as much as 60 cm lower than in the past). Expansion of Phragmites is linked to large increases in evapotranspiration suggesting that large scale vegetation manipulation through cutting / mowing might offer a means of reducing the extent and duration of summer groundwater drawdowns.

Holetn *et al.* (2009) developed an ecohydrological model (SWIM), which is a continuous-time spatially semi-distributed model that can simulate hydrological processes, vegetation growth, nutrient cycling and sediment transport at the basin scale. Modelling was undertaken for the period 1951-2003 across Brandenburg, Germany, with a particular emphasis on SACs which included fenland. Three future climate scenarios for the period 2004-2055 reflecting comparatively wet, medium and dry trends were simulated. These were developed using regionally downscaled climate projections from the ECHAM 5/MPI-OM GCM. Average available soil moisture was projected to decline with a range between -4% and -15%. The most pronounced changes in soil moisture occurred in floodplains of large rivers and it was concluded that wetlands are particularly vulnerable to climate change.

9. Policies

Gearey *et al*. (2006) considered the appropriateness of archaeological policy that states a preference for *in situ* preservation over excavation. Whilst such a policy may be appropriate in dryland conditions, there is limited confidence that it is appropriate for wetland archaeology. The main problems include: the specificity of each archaeological site and landscape and the dangers in extrapolating lessons learnt from one site to another; the quality of the hydrological and biochemical monitoring of sites; lack of knowledge of trigger points in wet-preservation and the adequacy of the advice given to and by local government officers with responsibility for the historic environment.

10. Wetland feedback on climate

A review of available European carbon budget data (Byrne *et al*., 2004) concluded that most peatland types vary between a small sink and a moderate source of GHGs, principally from a substantial CH_4 emission; none show unambiguous net uptake of GHG, thus even undamaged peatlands may have a net warming effect on climate, although restored fens and bogs have a much smaller effect than that for various types of pre-restoration management. Therefore, restoration has clear benefits in global warming terms over the un-restored case, even though restored peatlands may not have a net carbon sink function.The importance of wetlands (in terms of their carbon sequestering function; especially boreal peatlands) to the global carbon cycle was also noted. Wetland drainage caused by climate change was highlighted as a source of methane and carbon dioxide to the atmosphere, which will have a positive feedback on global warming.

Acreman *et al*. (in review) explore the impacts of hydrological change on the carbon balance of the Somerset Levels and Moors. Although raising water levels may make wetlands CO_2 sinks, they may emit large amounts of methane (CH_4). Persistently wet soils create anaerobic conditions that favour the activity of methanogenic bacteria, which produce and emit CH_4 from soils. In contrast, dryer aerobic soils favour the activity of methanotrophic bacteria that oxidise CH_4 leading to soil CH_4 consumption. Least carbon was generated when water levels were around 10 cm below the surface.

11. Recommendations for conservation/mitigation

Erwin (2009) made ten recommendations for global wetland conservation under a changing climate: (1) Reduce non-climate stressors on ecosystems; (2) Protect coastal wetlands and accommodate sea level change via habitat migration inland; (3) Monitor ecosystem response to climate change / the effectiveness of management techniques; (4) Train wetland restoration scientists/practitioners; (5) Identify and control invasive species; (6) Allow for known climatic oscillations in restoration and management plans; (7) Incorporate climate change in medium/long-term planning; (8) Select and manage restoration areas appropriately (e.g. provision of migration corridors); (9) Educate non-specialists that wetland spatial 'drift' is likely; and (10) Improve understanding of climate change effects on wetland hydro-ecology at the region scale.

Burkett and Kusler (2000) also suggested reduction of current anthropogenic stresses to allow for inland migration of coastal wetlands as sea-level rises, active management to preserve wetland hydrology, and increased restoration.

As noted above, in the USA, Johnson *et al.* (2005) predicted that waterfowl breeding areas would shift to the eastern and northern fringes which are predicted to be wetter. Because many wetlands in these areas have been drained, there is a need to protect and restore wetlands here to mitigate the future effects of climate change.

It is noteworthy that in the case of wetland extension or creation, climate change impacts may be less important for archaeology because the burial environment that is re-wetted or flooded may not contain wet-preserved organic remains.

12. References

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