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Abstract

This report presents the review carried out by the Centre for Ecology and Hydrology (CEH), part of the Natural Environment Research Council, of climate change impacts on riparian habitats and the potential for mitigation of these impacts through adaptation and restoration. The review focuses primarily on results from Europe.

The main deliverable, a peer-reviewed journal paper reviewing methods to provide climate change proof innovative, adaptive and cost effective riparian wetland management strategies, will be submitted for publication later in 2014.

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Draft manuscript

A review of climate change proof innovative adaptive and cost-effective riparian wetland management (mitigation, adaptation, and restoration) strategies to counteract negative effects of climate-driven changes on riparian wetlands ecosystems, with special focus on effects of temperature changes, changes in water level fluctuations, changes in nutrient loading and their interactions.

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Abstract

Riparian habitats provide a range of valuable functions, ranging from fertile areas for food and fuel production, natural flood storage, nutrient and carbon storage and cycling, and provision of habitat for various plant and animal species. Each of these functions is fundamentally driven by hydrological regime and whilst many have in built resilience to cope with short term hydrological variation, a long term hydrological trend may have greater potential to reduce the efficiency of one or more of these functions. Developing climate-proof strategies through mitigation, adaptation and restoration is likely to become an increasingly important research area and this review aims to establish the current knowledge base, so that future activities can be identified and planned.

Introduction

River systems are regarded as the most threatened ecosystems on the planet (Malmqvist and Rundle, 2002) and there has been rapid decline in the biodiversity of their associated ecosystems over the past 30 years (Jenkins, 2003). Multiple factors affect these systems, some of which are a direct result of human activity (e.g. drainage, abstraction, land use change) whilst others are less directly linked to human activity (e.g. changing climatic patterns). As such, some factors are seen as readily reversible, using tools such as government policy, regulatory and management practices to address issues of abstraction or drainage. The global scale issue of changing climate is rather more difficult to address at the site or individual country level, and future-proofing these sensitive habitats will need to focus on adaptation and restoration. (Schneider *et al*., 2011).

The extensive body of climate modelling indicates spatially and temporally varying future conditions. In Europe, annual mean temperatures are likely to increase more than the global mean increment, with the largest winter warming occurring in northern Europe and the largest summer warming occurring in southern Europe (Junk, 2013). Annual precipitation is expected to increase in northern Europe (driven largely by an increase in winter rainfall), but decrease in southern Europe (Junk, 2013). Summer droughts are also expected to become both more frequent and intense.

The development of the floodplain over many years provides an integrated picture reflecting many biophysical factors including geology, topography and land cover. The development of floodplain biota is influenced both by these long timescale processes and short timescale processes such as the magnitude, duration and timing of flood events (Schneider et al, 2011). So whilst the biota may be well adapted to deal with short term variability in the hydrological regime, it may be poorly equipped to deal with an unusually rapid, and ongoing change in climate. This review looks specifically at strategies to address mitigate the negative effects of this rapid and ongoing change.

Changes to the hydrological, temperature and nutrient regime of European floodplains

Under natural conditions wetlands located on floodplain areas (riverine wetlands) are subject to inundation from the river when it is in flood (Old and Thompson, 2008). The characteristics of flood pulses, such as their frequency, duration and magnitude, are controlled by the regime of the river which in turn can be modified by land use change. Low order upland streams experience numerous flood peaks and the flood pattern is irregular since these catchments respond rapidly to local precipitation. Heavy local rainfall over the catchment of such a stream can therefore result in the relatively rapid inundation of nearby wetlands. In contrast, within larger catchments, flood patterns are more seasonal and the impacts of individual precipitation events are less evident (Baker *et al.*, 2009).

On a much smaller scale the flooding of freshwater environments such as lowland wet grasslands is also partly due to high winter water levels within the ditches which cross them. Flow patterns of inundation within floodplain wetlands are often complex. In many riverine wetlands it is rare for initial inundation to occur directly over the riverbank or levee. Instead, floodwater often enters wetlands via relic floodplain features such as former channels or ditch networks that become connected to adjacent river channels during periods of high water. It is often only in the later stages of a flood event, when discharges are highest, that water enters the wetland directly over the riverbank (Old and Thompson, 2008). Similarly, within wet grasslands, the first flooding from ditches often occurs as water first intercepts shallow, small-scale drainage features which link the ditches with more remote areas (Thompson *et al.*, 2004). The hydrologic regime of wetlands is, in fact, extremely sensitive to changes in both precipitation and evapotranspiration patterns as well as to anthropogenic activity resulting in hydroperiod regime shifts (Foti *et al.*, 2012). Natural flooding processes have been modified for the vast majority of the UK's floodplain wetlands through the construction of embankments and other flood control infrastructure. It has been suggested that climate change will lead to increased river flooding through the 21st Century (e.g. Kay *et al.*, 2006; Reynard *et al.*, 2001).

Although there are very few specific studies on the effects of droughts on wetlands, it is generally accepted that the following impacts may result from drought events; reduction in area of open water (to eventual complete drying), increasing soil moisture deficit leading eventually to a (sometimes irreversible) change in soil structure, oxidation of organic soils, hardening of the soil surface (Dollar *et al*., 2012). There can also be a concentrating effect on pollutants dissolved in the water, as the water body evaporates.

Schneider *et al*., (2011) carried out a Europe wide modelling study of hydrological regimes in floodplains and the results suggest a spatially variable picture (Table 1).

Table 1. Summary of modelled changes to the hydrological regime of European rivers with significant floodplain. (*Adapted from Schneider et al. 2011)*

	Predicted Change			
Hydrological metric	**Increase/Earlier**	**Decrease/Later**	**No Change**	**Signal Unclear**
Change in flood volume	Ireland, Scotland and Western England	Central and Eastern Europe. High altitude upper reaches of Rivers Turia (Sisterna Iberico), Cinca and Garonne (Pyrenees), Dordogne, Loire and Lot (Massif Central), Rhone, Rhine, Enns, Mura, Drava, Isar, Inn and Po (Alps), Neretva (Dinaric Alps), Maritza (Rila).		France, Spain, and Benelux countries. Also Rivers Thames and Derwent in the UK.
Change in duration of overbank flows	Ireland, Scotland and Western England.	Central and Eastern Europe, as well as mountainous areas.		France, Spain, Benelux countries as well as Rivers Thames, Elbe, Havel, Warta and Narew,
Change in timing of floodplain inundation	Eastern and Northern Europe. Many rivers in Southern and Central Europe.	Some rivers in northern Italy.		

Pollutants

Floodplains are a sink and memory for pollution that reaches the river system via wastewaters, surface waters or atmospheric deposition (Lair *et al.*, 2009). Presently, as much as 79% of the riparian area of European rivers is intensively cultivated (Tockner and Stanford, 2002) and it is possible that intensive agricultural use of floodplain soils can severely decrease the retention capacity for organic contaminants and heavy metals through excessive aeration (tillage) and subsequent loss of soil organic matter (Lair *et al.*, 2009). While ecotoxicity can be an important integral indicator for contaminant loadings the vitality of soil organisms is often controlled by other factors such as pH, organic matter, clay and nutrient contents (Lair *et al.,* 2009). In some floodplains, current anthropogenic nutrient inputs are likely to be offset by a decreasing nutrient input associated with a diminishing flood pulse effect (Antheunisse *et al*. 2006).

Anticipated ecological response to changes in hydrology, temperature and nutrient.

Impacts related to flooding

Increases or decreases in flood volume will cause increases or decreases in floodplain area respectively. So the habitat at the current floodplain margins are likely to undergo a shift towards wetter or drier (Schneider *et al*., 2011). Similar changes in duration will cause shifts towards either communities with more resilience to inundation or those better suited to short duration inundation, and changes in timing of inundation, to which some species are very sensitive, is also likely to affect habitat conditions (Wheeler, 2004). Changes in climate and land management have a strong and quantifiable predictable impact on vegetation patterns and species distribution (Foti *et al*., 2012).

Research has been undertaken to establish links between surface wetness / flooding and wading birds such as lapwing (*Vanellus vanellus*) and redshank (*Tringa totanus*). This is due to the large declines in populations of these species which has been attributed to the loss of suitable habitat such as lowland wet grassland (e.g. Ausden *et al.*, 2001; Green and Robins, 1993). A behavioural link between the distribution of waders and surface wetness has been demonstrated (e.g. Eglington *et al.*, 2008). For example, work undertaken with within the Elmley Marshes, part of the North Kent Marshes, has demonstrated that the probability of a particular part of the marshes being occupied during the breeding season (April–June), as well as the density of lapwing and redshank, increases with flood extent and the number of wet rills and hollows (Milsom *et al.*, 2000, 2002). Feeding rates of both species are also higher in rills which are wet in May compared to those which are dry (Milsom *et al.*, 2002). This may be due to effects of prolonged inundation on vegetation cover, the availability of aquatic invertebrates within pools of water, the concentration of soil macroinvertebrates relatively near the soil surface or the more penetrable nature of wet soil. (Old, *et al*., 2008).

Other species are influenced by hydrological conditions with mallard (*Anas platyrhynchus*) and Canada goose (*Branta canadensis*) exhibiting a positive association with surface wetness (Milsom *et al.*, 2000). Some flooding in April and May is therefore important for attracting waders and other birds. Optimum flood conditions would be those which create a mosaic of unflooded grassland, winter-flooded grassland and shallow pools. 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). Many wader species are also attracted to standing water on grassland in winter. The use of wet grassland by waders is determined to some extent by the level of the water table, as soil invertebrates are forced closer to the surface as the water table rises.

The height of the water table also influences the penetrability of the soil for 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 as invertebrates are forced to the surface. Ausden *et al.* (2001) quantified the response of soil macroinvetebrates to flooding as well as their ability to survive in flooded grassland and changes in abundances and physical availability for feeding waders. They demonstrated lower biomass of soil macroinvertebrates in sites with a long history of winter flooding compared to unflooded grasslands. Macroinvertebrates in the flooded sites mainly comprised a limited range of semi-aquatic earthworm species. When flooding was introduced to previously unflooded grasslands, a large reduction in soil macroinvertebrate biomass resulted. The main cause of this reduction was the vacation of the soil by earthworms soon after flooding although when artificially confined in flooded soils most earthworm species were capable of surviving periods of at least 120 days of submergence. Winter flooding also resulted in the expulsion of many overwintering arthropods. Recolonisation by soil macroinvertebrates of grassland flooded in winter was slow during the following spring so that prey biomass for wading birds was low. Research on three floodplains in the UK (Acreman *et al.*, 2008) under different soil wetness conditions, found that the largest invertebrate biomass samples (including earthworms, beetles, slugs, springtails and spiders) were collected at stations with soil moisture between 0.5 and 0.6 m3m-3, which equates to a moist soil, but not water-logged.

This is consistent with optimum conditions for earthworms used in toxicology experiments (*e.g.* Spurgeon and Hopkin, 1995). 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 with 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. For example, heavy rains during the summer of 2007 destroyed the only nest of Bittern chicks at Blacktoft Sands nature reserve, Yorkshire. The nest was either flooded or the chicks died due to starvation or hypothermia (www.wildlifeextra.com/blacktoft-sands.html). In summary, shallow flooding in winter is beneficial to many species of waterfowl and waders and lack of flooding would reduce their presence in any catchment area. 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 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. 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.

Many undisturbed lowland rivers have more gentle sloping banks that are overtopped relatively quickly, facilitating more frequent connectivity with the floodplain and thus allowing river discharge to dissipate and velocities to reduce across large areas of habitat, suitable for spawning, feeding or refuge purposes. This can result in increased diversity within the system – e.g. by providing habitats, such as oxbows and drainage ditches, for spawning or as nursery areas, that are scarce or lacking within the main river. Availability of suitable spawning habitat and nursery habitats for young fish, as well as an adequate food supply (e.g. inundated floodplain environments) during the early stages are critical for good recruitment (Nunn *et al.* 2007a, b). The habitat heterogeneity of floodplain river ecosystems is not only maintained but is often increased by erosional and depositional processes during floods (e.g. Mertes, 1997). For example, fluvial action may create fish habitats through the formation of channels, backwaters, standing water bodies and marshes. Periodic flood events maintain connectivity between river and floodplain and compensate for terrestrialisation during low flow periods (Amoros, 1991). The balance between rejuvenation and terrestrialisation processes produces a mosaic of habitats with distinct fish assemblages (Copp, 1989). For example, phytophilic species require lentic, vegetated areas, which are often only temporarily connected with the main channel. Jurajda *et al.* (2004) found long term flooding increased the abundance of phytophilous and phytolithophilous species (flooded vegetation provides food and shelter thus increasing growth and reducing predation.

Permanent stocks of bream have been found in oxbow lakes on the River Rhine (Molls, 1999). Waidbacher (1989) found a positive relationship between hydrological connectivity and fish species richness in European aquatic floodplain habitats. Tockner *et al.* (1998) identified that fish diversity peaked in highly connected habitats on a Danube floodplain. During floods river-floodplain connectivity allows fish to disperse and take advantage of different floodplain habitats for refuge, spawning, nursery and feeding. Ward *et al.* (1999) emphasised that fish movements to floodplain spawning and nursery areas, are crucial for the recruitment and sustainability of fish populations. Furthermore, nutrient release during floodplain inundation stimulates phytoplankton and zooplankton production providing an abundant food source for newly hatched larvae (Junk *et al.*, 1989). Welcomme and Halls (2004) reviewed the influence of the hydrological regime on fisheries, and detailed that floods of greater amplitude increased the area for spawning sites, food and shelter for the fish, whilst duration influences the time available for fish to grow and shelter from predators. These principles apply to rivers globally; large floods of long-term duration increase fish species richness and abundance in temperate floodplain systems (e.g. Modde *et al.*, 1996).

Impacts related to drought. In general, wetlands supplied with water by rivers, often in conjunction with rainfall, experience a large range of wetland water levels, potentially supporting vegetation communities which have a wide range of preferred water levels (summer and winter). The minimum water level requirement for these communities is generally some depth below the surface (Wheeler *et al*., 2004).

As open water features are lost, population numbers of aquatic species (such as ducks and grebes) and of semi-aquatic species (such as Moorhen), are likely to decline and ultimately fall to zero. The reduced viability of fringing vegetation might damage food supply for a range of species such as the Reed Warbler and Sedge Warbler. The shrinkage of water bodies can increase the exposure of nests to terrestrial predators and the safety of roost sites for various species, including some non-aquatic species such as swallows and martins, may also be reduced. Reduced connectivity between water features will likely force species to disperse more widely and this may have a negative impact on species health, much of which would be associated with a reduction in availability of food, particularly aquatic invertebrates and vertebrates. This will drive a deterioration of viable feeding habitat for wetland and non-wetland birds. Dollar *et al*., 2012.

An increased concentration of water-borne nutrients and pollutants may both have a direct toxic effect and an effect due to bioaccumulation. Increased turbidity could reduce hunting success in visual pursuit predators such as grebes. Dollar *et al*., 2012. The impact of reduction in soil moisture can lead to a reduced food supply for herbivores and reduced cover for nesting, as there is reduced plant biomass accumulation. Reduced penetrability of the soil surface will have a direct effect on bird species feeding on soil invertebrates as the prey become inaccessible. This applies to species such as snipe and thrush. Dollar *et al*., 2012.

Species of permanent open water will be impacted adversely but species that are well adapted to withstand fluctuating water levels will probably experience little impact in response to seasonal drought. There may even be some benefit to ephemeral species which are adapted to complete life on exposed mud. Reduced availability of soil water may have a temporary effect on wetland species of marshes or swamps but the magnitude of this impact will be related to the extent, frequency and duration of the drought. Supra-seasonal drought would likely bring about a shift in vegetation community if the wetland water levels fall below the 'absolute minimum' water level which varies according to vegetation community (Wheeler *et al*., 2004; Wheeler *et al*., 2009; Davy *et al*., 2010). Trajectory diagrams which give detailed information about the response of vegetation communities to changing conditions are available (for example, see Wheeler *et al*., 2004).

For species which lay their eggs in permanent water bodies, a reduction in open water would decrease available habitat and might increase competition between larvae. For species which lay their eggs on land and wait for water levels to rise, a supra-seasonal drought may result in eggs dying at a rate that is species dependent. This might reduce mosquito abundance. As drought intensifies, reduced connectivity between water features may enhance larval survival since many species prefer isolated water features that may have a reduced number of predators of larvae. Adult mosquito survival rate is related to air moisture which has a relationship with soil moisture. In general, reduced moisture will decrease adult survival, though effects will be different for individual species.

During drought, soil invertebrates (such as earthworms, springtails) move to deeper soil layers or enter diapauses. This affects survival and reproduction rates resulting in longer term declines in population size. With loss of soil structure, soil microbial activity becomes inactive. This affects rates of carbon mineralisation, rates of cycling of key nutrients including nitrogen and phosphorus, potential degradation of complex organic chemicals including deposited air pollutants and pesticides, and soil food web and energy flows.

Reduced productivity of wetland plants and crops will provide less cattle feed and associated products. Similarly, the knock-on effect of reduced numbers of wetland species (both flora and fauna) will result in a more widespread reduction in food and greater pressure on dependent species. Loss of soil structure, in particular in organic soils, may permanently compromise the ability of the soil to store water. It therefore will have reduced ability to both store water in times of flood (potentially increasing downstream flood risk) and release water in times of drought.

Mitigation and Adaptation

Text from Leyer, I., Mosner, E. and Lehmann ‘Managing floodplain-forest restoration in European river landscapes combining ecological and flood-protection issues. Ecological Applications, 22(1), 2012, pp. 240–249

Recent catastrophic flood events across Europe reveal that technical solutions impeding flood damages have only limited effects and that new approaches to flood management must include the restoration of rivers’ natural flood zones in order to reactivate the ability of floodplains to retain water and alleviate flood impacts (Dworak 2008).Therefore, there are tendencies in European river management to move from the traditional dike raising and reinforcement toward ‘‘room for rivers’’ measures to assure flood protection (van Stokkom et al. 2005). Among others, suitable measures comprise the widening of river cross sections by dike relocation, and lowering the floodplain by excavations leading to the requested increase in the water discharge capacity of the rivers. In the meanwhile, in several projects in different countries such measures are planned or already applied (e.g., Monstadt 2008).

The Dutch Government has even implemented a ‘‘room for the rivers’’ concept in national management plans (van Stokkom et al. 2005). These kinds of measures serve both flood protection (by enlarging the retention areas) and floodplain rehabilitation (by allowing a certain dynamization of hydrological, geomorphological, and ecological processes). If they are accompanied by landuse changes from agriculture to natural development, floodplain forests can profit strongly. Floodplain lowering by excavation work, creating new pioneer stages, is a possibility in regulated floodplains lacking erosion and other disturbance processes to enhance the reoccurrence of softwood forests by natural recruitment (Friedman et al. 1995). However, these measures are not self-sustaining in regulated rivers either, and need continued management input to provide bare-ground sites (Rohde et al. 2006). Additionally, there is evidence that the initial gain in discharge capacity cannot be maintained because of the combined effect of sedimentation and vegetation succession (Geerling et al. 2008). Therefore, while the ecological benefit is obvious, it remains unclear where floodplain-lowering measures are suitable from a hydraulic point of view.

The most promising concept to combine the sustainable enlargement of the discharge capacity and the rehabilitation of natural processes seems to be dike relocations. Indeed, the space for such actions is limited due to the densely populated and intensively used area behind the dikes in many European regions. However, there is a certain potential for dike-relocation measures (BMU and BfN 2009) mitigating flood risks and allowing floodplain-forest development without any negative hydraulic effects as has also been stated for North American floodplain systems (Gergel et al. 2002). These kinds of restoration and rehabilitation projects are longterm undertakings and they need a rigorous interdisciplinary and multi-sectoral approach at all governmental, regional, and local levels (Buijse et al. 2002). On the whole, there is a considerable potential to restore floodplain functions and processes while also observing and benefiting flood-protection issues. Nevertheless, due to the limited space in European river landscapes where process-inducing measures can be conducted, site-specific restoration initiatives through continued human interventions, e.g., by floodplainforest plantings, are necessary to slow down a further deterioration of riverine landscapes and habitat diversity. The approach we have presented provides a tool for an integrated river-basin management to make floodplain- forest establishment realizable.

Table 2. Example European Case Studies (from Moss and Monstadt (Eds.), Restoring floodplains in Europe. IWA Publishing, London, UK. 355 pp. ISBN: 1843390906)

Name	Area	Aim	Objectives	Techniques
The Rheinvorland Sud project, located near Rastatt in south-west Germany.(Kruse, 2008).	~ 660 ha	To improve the ecological and hydrological conditions within the alluvial area of the Rhine.	Develop and implement various measures to improve the discharge rate within the floodplains	• Reconnecting 'dead' arms with the main river • Improving the permeability of roads and footpaths • Lowering existing dams
Restoration of an alluvial forest and ancient meander on the Garonne. (Hagemeier and Klaphake, 2008)	~ 100 ha	Biodiversity conservation in combination with other targets (e.g. recreation, angling)	Improvement of the ecological quality of the forest	• Restrict the area in which new plantations could be established • Largely not to touch the existing monoculture poplar forests • Not to establish a 'pure' natural alluvial forest, but a semi-natural mixed forest • Reconnection of the ancient meander to the river • Upgrading of the area for recreational purposes.
The Long Eau floodplain restoration project. (Moss, 2008)	~ 20 ha	The restoration of floodplain storage to enhance flood protection and improve wildlife habitat in the river channel, banks and floodplain		• Setting back of floodbanks • Construction of new floodbanks • Re-routing of existing drains

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