A review of the potential impacts of climate change on surface water quality

P. G. WHITEHEAD , R. L. WILBY , R. W. BATTARBEE , M. KERNAN & A. J. WADE

A review of the potential impacts of climate change on surface water quality

P. G. WHITEHEAD[1], R. L. WILBY[2], R. W. BATTARBEE[3], M. KERNAN[3] & A. J. WADE[1]

1 *Aquatic Environments Research Centre, University of Reading, Reading RG6 6AB, UK*
  p.g.whitehead@reading.ac.uk

2 *Geography Department, Loughborough University, Loughborough, Leicestershire LE11 3TU, UK*

3 *Environmental Change Research Centre, University College London, Gower Street, London WC1E 6BT, UK*

Abstract It is now accepted that some human-induced climate change is unavoidable. Potential impacts on water supply have received much attention, but relatively little is known about the concomitant changes in water quality. Projected changes in air temperature and rainfall could affect river flows and, hence, the mobility and dilution of contaminants. Increased water temperatures will affect chemical reaction kinetics and, combined with deteriorations in quality, freshwater ecological status. With increased flows there will be changes in stream power and, hence, sediment loads with the potential to alter the morphology of rivers and the transfer of sediments to lakes, thereby impacting freshwater habitats in both lake and stream systems. This paper reviews such impacts through the lens of UK surface water quality. Widely accepted climate change scenarios suggest more frequent droughts in summer, as well as flash-flooding, leading to uncontrolled discharges from urban areas to receiving water courses and estuaries. Invasion by alien species is highly likely, as is migration of species within the UK adapting to changing temperatures and flow regimes. Lower flows, reduced velocities and, hence, higher water residence times in rivers and lakes will enhance the potential for toxic algal blooms and reduce dissolved oxygen levels. Upland streams could experience increased dissolved organic carbon and colour levels, requiring action at water treatment plants to prevent toxic by-products entering public water supplies. Storms that terminate drought periods will flush nutrients from urban and rural areas or generate acid pulses in acidified upland catchments. Policy responses to climate change, such as the growth of bio-fuels or emission controls, will further impact freshwater quality.

Key words climate change; water quality; rivers; catchments; lakes; estuaries; ecology; hydrochemistry

Une revue des impacts potentiels du changement climatique sur la qualité des eaux de surface

Résumé Il est maintenant admis qu'un certain changement climatique d'origine anthropique est inévitable. Les impacts potentiels sur l'alimentation en eau ont fait l'objet de nombreuses attentions, mais peu de connaissances sont disponibles sur les changements associés en termes de qualité de l'eau. Les changements prévus en termes de température de l'air et de précipitations pourraient affecter les écoulements des rivières et par conséquent la mobilité et la dilution des substances polluantes. Une augmentation des températures de l'eau affectera la cinétique des réactions chimiques et, par combinaison avec les dégradations de la qualité, l'état écologique des hydrosystèmes. Une augmentation des écoulements aura pour conséquences des changements dans la puissance des cours d'eau et donc aussi des charges sédimentaires qui pourront altérer la morphologie des rivières et le transfert de sédiments vers les lacs, ce qui aura des impacts sur les habitats hydrobiologiques dans les systèmes lacustres et fluviatiles. Cet article fait une revue de tels impacts à travers l'exemple de la qualité des eaux de surface du Royaume Uni. Des scénarios largement admis de changement climatique suggèrent des sécheresses plus fréquentes en été, ainsi que des crues éclairs générant des écoulements incontrôlés des zones urbaines vers les cours d'eau et les estuaires récepteurs. L'invasion d'espèces exogènes est hautement probable, de même que la migration d'espèces au sein du Royaume Uni par adaptation aux changements de température et de régime d'écoulement. Des écoulements diminués, des vitesses réduites et par conséquent de plus grands temps de résidence de l'eau dans les rivières et les lacs vont augmenter le potentiel de blooms algaux toxiques et diminuer les niveaux d'oxygène dissous. Les cours d'eau de plateau pourraient subir une augmentation de la teneur en carbone organique dissous et un renforcement de la couleur, nécessitant d'agir au niveau des stations de traitement de l'eau pour prévenir les sous-produits toxiques entrant dans les réseaux d'alimentation en eau potable. Les événements pluvieux qui mettent fin aux périodes de sécheresse chasseront brutalement des nutriments des zones urbaines et rurales, ou génèreront des signaux acides dans les bassins versants acidifiés de plateau. Les réponses au changement climatique en termes de politique publique, comme l'accroissement des bio-carburants ou le contrôle des émissions, auront un impact d'autant plus important sur la qualité des eaux douces.

Mots clefs changement climatique; qualité de l'eau; rivières; lacs; estuaires; écologie; hydrochimie

INTRODUCTION

The 2007 Conference of the Parties to the United Nations Framework Convention on Climate Change, in Bali, and the latest IPCC Report (2007) confirmed the consensus amongst scientists

and policy makers that human-induced global climate change is now occurring. However, there is less certainty about the magnitude of future temperature changes and how these will drive precipitation, evaporation and hydrology at regional scales. Nonetheless, climate model scenarios provide the best available information for assessing future impacts of climate change on the water quality and ecology of surface water bodies (Kundzewicz *et al.*, 2007; Bates *et al.*, 2008).

The *Freshwater* chapter in the IPCC Fourth Assessment Report (Kundzewicz *et al.*, 2007) was unable to consider the impacts of climate change on water quality in great detail, but this topic is attracting growing attention. For example, the EU Euro-limpacs Project (www.eurolimpacs.ucl.ac.uk) is a multi-partner, €20-million research project investigating impacts on rivers, lakes and wetlands across Europe (Battarbee *et al.*, 2008). A wide range of laboratory and field experiments, data analysis and process-based modelling is being undertaken to evaluate potential impacts of climate change. This research, plus activities elsewhere (Jones & Page, 2001), is raising many important questions (see Appendix), including:

– How will climate change impact river flows and, hence, the flushing of diffuse pollutants or dilution of point effluents?
– In what ways might more intense rainfall events affect nutrients and sediments loads in urban drainage systems, rivers, lakes and estuaries?
– How might rising temperatures combined with water quality changes affect freshwater ecosystems?
– How might the carbon balance and recovery of acidification be affected in upland catchments?

This paper provides a review of water quality seen through the lens of anticipated impacts in the UK. The material is organised as follows. First, we review potential impacts on *surface* water bodies, such as rivers and lakes, in terms of their hydrological regimes, hydromorphology, nutrient status, mobilisation of toxic substances and acidification potential. Second, we review long-term changes in the water quality for specific aspects of freshwater environments, such as estuaries and urban areas. These sections are followed by a review of recent water quality model developments and examples of modelling approaches utilised in the Euro-limpacs project, including the treatment of uncertainty. The question of uncertainty is important, as predictions years into the future, based on uncertain GCM model outputs and uncertain process model parameters, will inevitably be uncertain themselves. Assessing this uncertainty is important. However, as was shown in the acid rain projects in the 1980s, uncertain models can provide some extremely useful information for planners and managers (Cosby *et al.*, 1986). A testament of this is that those models have correctly predicted the broad recovery of acid lakes and rivers across northern Europe (Wright *et al.*, 2005). Finally, the indirect consequences to water quality of wider climate change policies affecting land and water management or emission reductions are considered.

HYDROLOGY, WATER QUALITY AND THERMAL REGIMES

A review of surface water quality cannot be undertaken without considering changes in hydrological regimes. The UKCIP02 scenarios (Hulme *et al.*, 2002) suggest that winter precipitation in the UK could increase by 10–20% for a low-emissions scenario, and by 15–35% for a high emissions scenario, by the 2080s. The largest changes are predicted in the south and east of England, and the smallest in northwest Scotland. In the summer, the pattern is reversed and almost the whole of the UK could become drier, with precipitation decreases of up to 35% under a low-emissions scenario, and 50% or more under a high emissions scenario. Marsh & Hannaford (2007) has shown that summer precipitation has already fallen to some extent (Fig. 1). Furthermore, the frequency of extreme events is also predicted to increase, with two-year (return period) winter precipitation event intensities estimated to become between 5% (low emissions) and 20% (high emissions) heavier by the 2080s. These changes in precipitation have been used to simulate changes in flow across the UK (Limbrick *et al.*, 2000; Arnell, 2003). More recently, Romanowicz
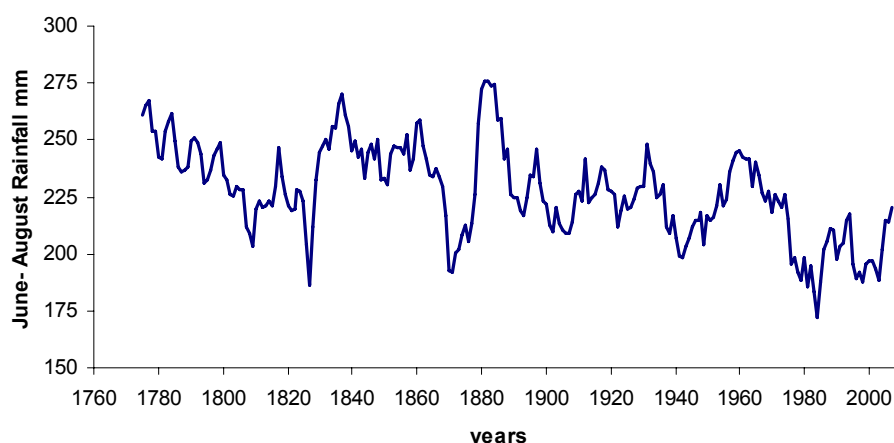
Fig. 1 Summer (June–August) rainfall totals (mm) showing long-term decline (source: Marsh & Hannaford, 2007).
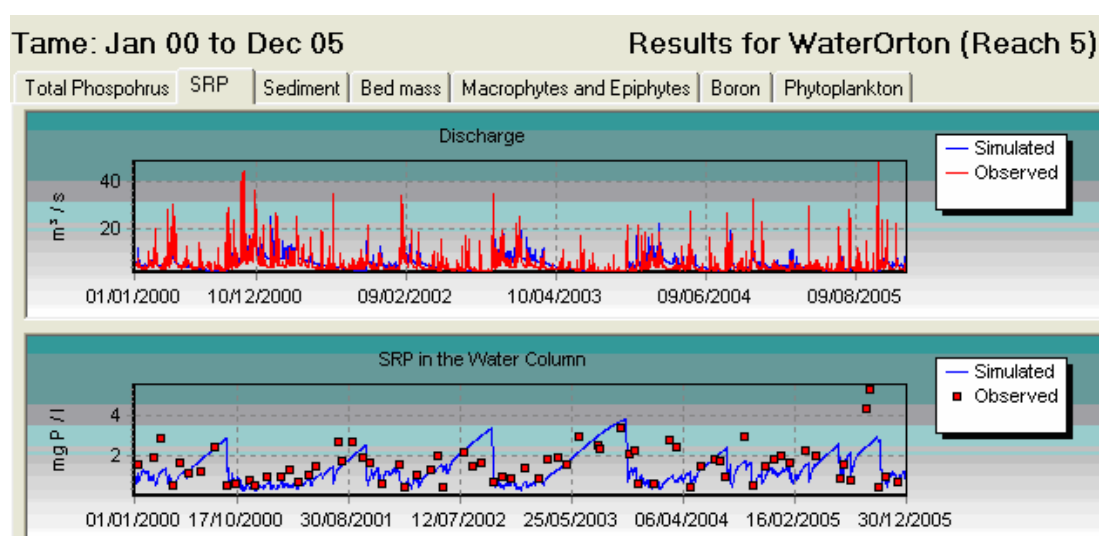

Fig. 2 Flow and phosphorus simulation and observed data for Water Orton on the River Tame in Birmingham, UK.

et al. (2006) modelled changes in river flow for a range of catchments, under different climate model projections. They conclude that, by the 2020s, flows in winter could increase by between 4 and 9%, and that summer flows could decrease on average by 11%, but this could range between 1 to 32% depending on the catchment location, land use, soils, geology, and model uncertainty.

Lower minimum flows imply less volume for dilution and, hence, higher concentrations downstream of point discharges such as wastewater treatment works (WTWs). This could affect efforts to improve water quality standards or meet EU Water Framework Directive (WFD) objectives to restore and protect freshwater ecosystems. For example, Fig. 2 shows the inverse relationship between phosphorous levels and flow in the River Tame, downstream of Birmingham, during summer months. Phosphorus increases significantly in summer months as flows fall. This is a direct consequence of reduced dilution of WTW effluents. Under climate change, natural headwater flows in summer could be lower, thereby providing less dilution and higher concentrations.

Reduced dilution effects will also impact on organic pollutant concentrations, with increased biological oxygen demand (BOD) and, hence, lower dissolved oxygen (DO) concentrations in rivers. Cox & Whitehead (2009) show that, under a range of UKCIP scenarios, DO in the River

Thames will be affected in the 2080s by enhanced BOD, and by the direct effects of temperature which reduces the saturation concentration for DO. Cox & Whitehead (2009) demonstrate that the effects are not large, and that this would not be an issue for ecosystems under normal circumstances. However, in the Thames, occasional algal blooms in some summers are a feature of the river ecology; the frequency and intensity of these may increase. When an algal bloom occurs, there are large diurnal variations in DO and, on poor-quality rivers, low oxygen levels can be exacerbated by pollution events during summer low-flow conditions. Williams *et al.* (2000) show that large diurnal variations in streams dominated by macrophytes occur, and thus pollution events in such streams could also generate low DO levels.

A recent study investigated BOD, DO, nitrate, ammonia and temperature in rivers, but there were insufficient data to adequately calibrate and validate the model (Conlan *et al.*, 2007). However, a model sensitivity analysis did illustrate the links between climate change and water quality. As expected, under reduced flows in summer, BOD and phosphorus levels would increase, whereas ammonia levels would fall due to higher nitrification rates. This gives rise to increased nitrate concentrations as ammonia decays to nitrate. The authors concluded that there could be enhanced growth of algal blooms in rivers and reservoirs which could affect DO levels and water supply. Also, with increased storm events, especially in summer, there could be more frequent incidences of combined sewer overflows discharging highly polluted waters into receiving water bodies, although there could be benefits in that storms will also flush away algal blooms.

The most immediate reaction to climate change is expected to be in river and lake water temperatures (Hassan *et al.*, 1998; Hammond & Pryce, 2007). River water temperatures are in close equilibrium with air temperature and, as air temperatures rise, so will river temperatures. There has already been a 1–3°C temperature rise over the past 100 years in large European rivers such as the River Rhine and the River Danube (EEA, 2007a). Small streams have shown an increase in winter temperature maxima in Scotland (Langan *et al.*, 2001), and there have been large increases in temperature reported for water courses in Switzerland at all altitudes (Hari *et al.*, 2006). There have been two sudden shifts in river temperatures, in 1988 and 2002, following changes in air temperature. Abrupt water temperature rises could have important implications for some aquatic organisms, if species are unable to adapt at the same pace. Furthermore, a recent study of long-term trends in UK surface waters revealed marked regional variations, with the greatest rates of change in the south and east (Fig. 3).

Most chemical reactions and bacteriological processes run faster at higher temperatures. In addition, temperature controls the growth rates of phytoplankton, macrophytes and epiphytes,
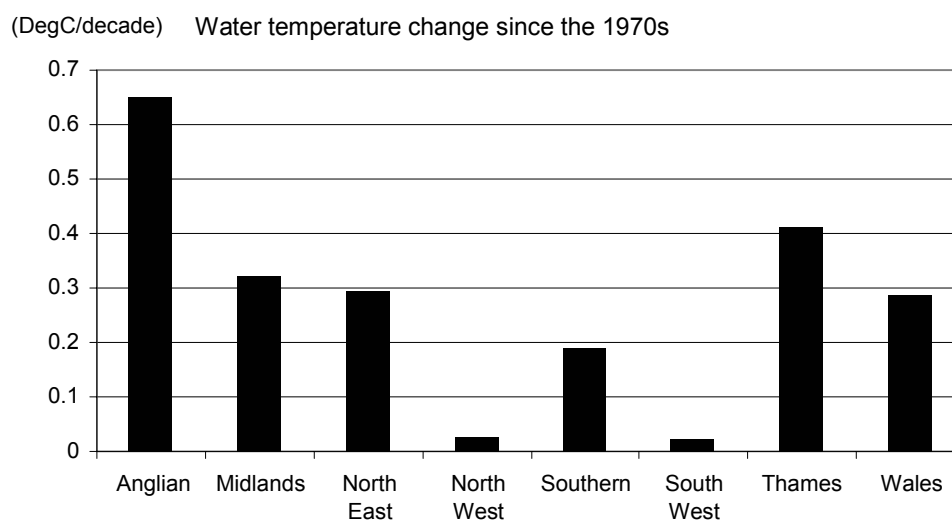

Fig. 3 Regional variations in UK surface water temperatures since the 1970s. Data adapted from Hammond & Pryce (2007).

making freshwater ecosystems sensitive to rising temperatures (Whitehead & Hornberger., 1984; Wade *et al*., 2002b). Water temperatures also regulate the behaviour of aquatic organisms, such as fish migration, and the timing of emergence and abundance of insect populations at different life-cycle stages (e.g. Davidson & Hazelwood, 2005; Durance & Ormerod, 2007). This has implications in that meeting WFD objectives and reference conditions for the restoration and improvement of the ecology of streams could be more difficult under future climate change (Wilby *et al*., 2006a).

HYDROMORPHOLOGY AND ECOLOGY

Climate change is expected to have far-reaching consequences for river regimes, flow velocity, hydraulic characteristics, water levels, inundation patterns, residence times, changes in wetted areas and habitat availability, and connectivity across habitats (Brown *et al*., 2007). More intense rainfall and flooding could result in increased loads of suspended solids (Lane *et al*., 2007), sediment yields (Wilby *et al*., 1997), *E. coli* and contaminant metal fluxes (Longfield & Macklin, 1999) associated with soil erosion and fine sediment transport from the land (Leemans & Kleidon, 2002).

In many parts of Europe, hydromorphology is a key factor controlling ecosystem behaviour. Alterations to river forms through channel straightening, loss of connectivity with flood plains, weir and dam construction, and loss of riparian vegetation also impact on river ecology. Under the WFD there is a requirement to reverse some of these changes and restore the ecology of rivers and lakes towards their natural states. However, climate change may act against restoration, making it difficult, if not impossible, to return to the previous ecosystem status (Orr & Walsh, 2006), as illustrated in Fig. 4. Changes in climate could affect sediment transfer, channel morphology and inundation frequency, thereby altering ecosystems at both catchment and habitat scales (Verdonschot, 2000). The impact of low flows on biotic communities has been studied extensively in the River Lambourn, UK (Wright *et al*., 1982). In this case, drought has a deleterious effect on aquatic ecology with *Ranunculus* being smothered by epiphytic algae (Wade *et al*., 2002b). Drought also significantly damages macro-invertebrates, although recovery can be fast (Ladle & Bass, 1981).

Extreme events could have significant impacts on upland rivers, releasing higher concentrations of sediments by erosion and re-suspension, thereby creating new or disturbed habitats downstream. However, this can be beneficial to upland stream ecology with natural formations of pool and riffle sequences and a wider range of habitats, such as meandering side channels, larger dead zone areas and deeper sediment zones to support aquatic life. An extensive study in German rivers shows that habitat restoration may be enhanced by the effects of a more variable flow regime (Hering *et al*., 2009).

The PRINCE project (PRINCE: preparing for climate change impacts on freshwater ecosystems) specifically addressed the potential impacts of climate change for selected UK freshwater ecosystems (Conlan *et al*., 2007). It was shown that changes in climate could influence aquatic ecosystems through episodic pulsed effects (i.e. changes in the frequency, duration and
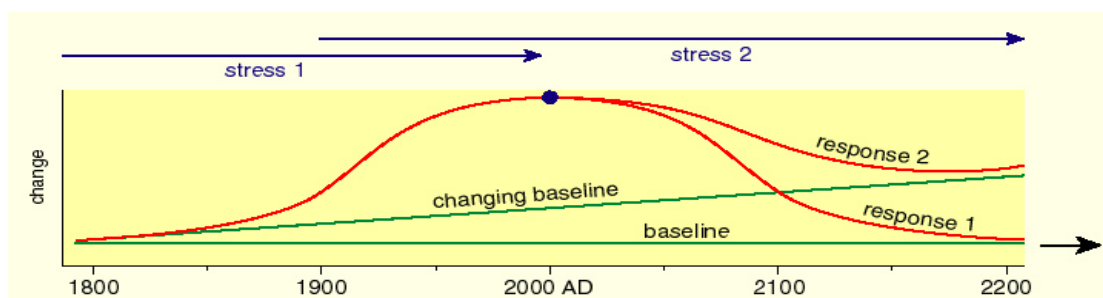

Fig. 4 Conceptual diagram illustrating ecosystem response to increasing and decreasing stresses in relation to climate change, expressed as a changing baseline (source: Battarbee *et al*., 2005).

magnitude of extreme events) and by progressive change in ambient conditions. Many freshwater species are sensitive to the water temperature regime as they are cold-blooded and many have a limited range of thermal tolerance. Thus, changes in the temperature regime could have significant effects on the life cycle of a wide variety of aquatic organisms. Temperature effects could combine with changes in water velocity and DO to affect the life cycles and inter-relationships of organisms such as invertebrates, amphibians, fishes and birds. In addition, there may be impacts on dispersal or migratory patterns across ecosystems, for example, between marine systems and freshwaters by long-distance migrants (Atlantic salmon, eel, shad), or across watersheds during inter-basin dispersal flights by invertebrates; and via the introduction, survival and population dynamics of exotic organisms.

Hotter summers and lower rainfall could increase the risk of deoxygenation. This may be particularly prominent in middle and lower river systems and standing waters, where re-aeration can be limited. This could be compounded where plant growth has been encouraged by higher water temperatures and non-limiting nutrient supply, leading to low levels of oxygen and possible threats to fish and invertebrates. Conversely, higher flows should improve oxygenation in winter, although increased storminess could increase discharges of contaminants such as herbicides, pesticides and nutrients into watercourses.

In summary, climate change could affect: (a) the magnitude, frequency (return period), timing (seasonality), variability (averages and extremes) and direction of predicted changes of flow and water quality; and (b) the sensitivity and resilience of the ecosystem, habitat and/or species to those changes. Habitats that are already in vulnerable stream sections, such as headwaters, ditches and ephemeral ponds, could be the most sensitive to changing climatic conditions.

NUTRIENTS AND EUTROPHICATION

Nutrient loads are expected to increase under climate change (Bouraoui *et al.*, 2002). However, assessing the impacts on eutrophication is not straightforward as eutrophication occurs as a result of the complex interplay between nutrient availability, light conditions, temperature, residence time and flow conditions (Jeppesen *et al.*, 2005). However, it is possible to assess the impacts of climate change on individual components contributing to eutrophication via field or modelling experiments. As has already been discussed, temperatures will rise, favouring increased growth rates of algae (Whitehead & Hornberger*,* 1984), especially cyanobacteria. Flow rates in summer could fall, thereby increasing the residence time of water in controlled reaches, as is typical for many lowland rivers, and in lakes. Increased residence times increase growth potential of algae, enhance the settling rate of sediments, and reduce water column sediment concentrations. This in turn reduces turbidity so that improved light penetration can enhance algae growth.

Meanwhile, nutrients released from agriculture or from sewage treatment works could be less diluted due to the reduced flows in summer. Whitehead *et al.* (2006b) simulated these combined effects on the River Kennet in terms of projected nitrate and ammonia concentrations. As shown in Fig. 5, nitrate concentration increases over time as higher temperatures increase soil mineralisation. This is particularly significant under high flow conditions following a drought. Whilst this was a theoretical modelling exercise, similar responses have been observed in the field. For example, Fig. 6 shows nitrate concentrations in the Thames at Days Weir at the termination of the 1976 drought (Whitehead & Williams, 1982). Nitrate-N concentrations rose from 4 to 18 mg/L as nitrates were flushed from the Thames catchment. Increased frequency of flushing events is expected under some climate change scenarios, and this extra nitrogen could enhance eutrophication in receiving water bodies. This may be more important for nutrient-poor upland rivers and lakes, and could be significant for estuary and coastal systems that ultimately receive the extra nutrients.

As part of Euro-limpacs, a set of mesocosm (small artificial lake) experiments have been established to simulate climate change impacts (Moss *et al.*, 2003*,* 2004). These controlled environments show that growing seasons are extended by increased temperatures, as are growth
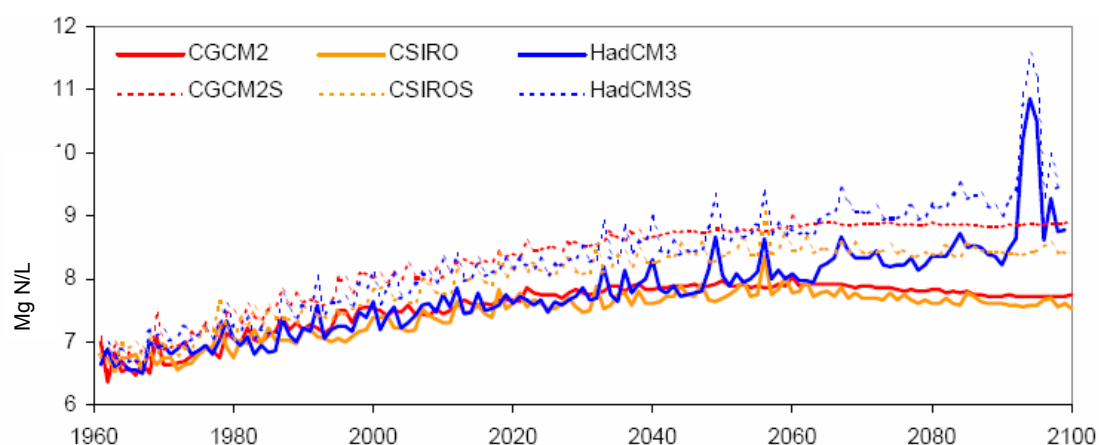
Fig. 5 Nitrate-N concentrations (fifth percentiles shown as dotted lines) over the 21st century under a range of GCMs (A2 emissions) for the upper Kennet River system.

Fig. 6 Nitrate-N at Day's Weir on the River Thames during 1974, 1975 and 1976, showing increased concentrations when the drought ended (source: Whitehead & Williams, 1982).

Fig. 7 Effects of lake changing temperatures on algal blooms in Bassenthwaite Water, England (source: Elliott *et al*., 2005).

rates of algae and zooplankton. Oxygen concentrations fall as temperature reduces saturation levels and increased nutrient levels enhance respiratio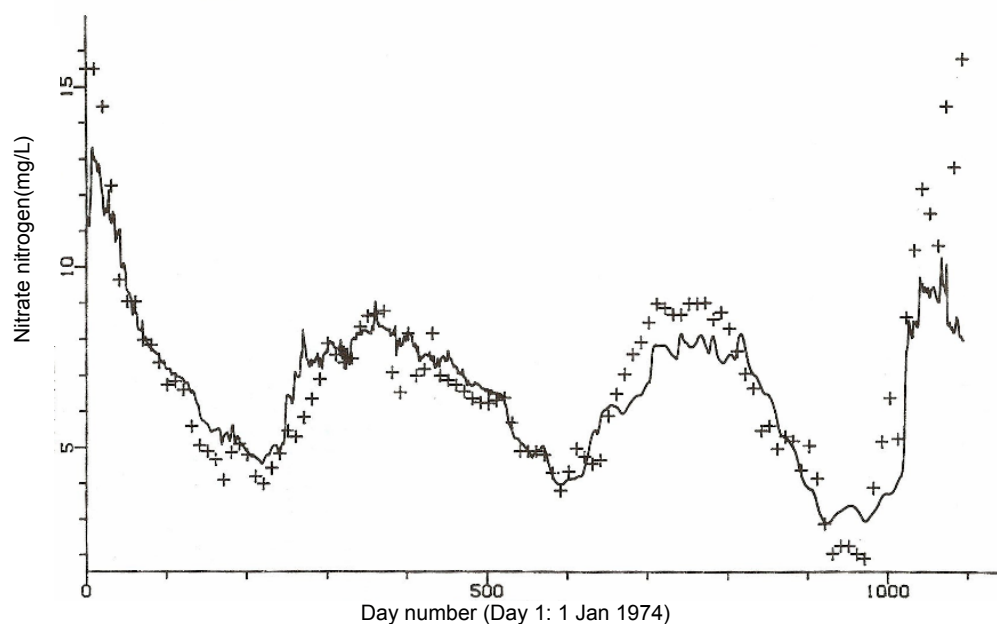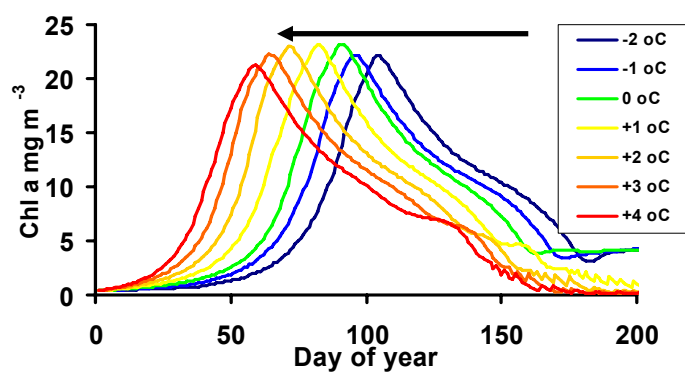n. This could, in turn, lead to increased risks of fish deaths, even for tolerant species. The ecology of the mesocosms changed significantly with exotic species out-competing native species. Future projections suggest that oxygen levels may decline and cyanobacteria blooms may become more extensive. The findings from these meso-cosm experiments could have implications for lowland rivers, as well as for shallow lakes, where water levels are controlled by weirs and where there can be long residence times in summer. Van Doorslaer *et al.* (2007) have also shown that zooplankton evolution can occur over relatively few generations, raising the possibility that ecosystems might maintain their current structure and functionality by adapting to temperature increases. Whilst one or two species might achieve sufficient rates of change, it seems unlikely that whole ecosystems could evolve in parallel.

Lake ecosystems respond to changes in inflow volumes, water quality and water temperature, as well as to changes in thermocline behaviour and residence times (George *et al.*, 2007). Numerous studies have highlighted the links between the winter North Atlantic Oscillation (NAO) and coherent responses in lake water temperature, ice conditions and spring plankton phenology across Europe (e.g. Blenckner *et al.*, 2007). Higher wind speeds could reduce lake stability, and enhance mixing of nutrients (George *et al.*, 2007). Conversely, higher temperatures lengthen the period of thermal stratification and deepen the thermocline (Hassan *et al.*, 1998). Shallow lakes may be particularly susceptible to climate-induced warming, changes in seasonal mean residence times (George *et al.*, 2007) and nutrient loads (Carvalho & Kirika, 2003). Climate impact assessments typically show associated changes in ecosystem functioning (Table 1), such as earlier blooms (Fig. 7) or increased concentrations of planktonic algae (e.g. Arheimer *et al.*, 2005; Komatsu *et al.*, 2007). A sensitivity study of phytoplankton in Loch Leven, Scotland, UK, showed larger responses to increases in phosphorus loads than water temperature (Elliott & May, 2008).

Table 1 Potential climate change impacts on shallow lakes in terms of target species, nuisance species, invading species, water transparency, carrying capacity and biodiversity. Adapted from Mooij *et al.* (2005).

(i)	Lower numbers of several target species of birds.
(ii)	Stabilized cyanobacterial dominance in phytoplankton communities.
(iii)	More serious incidents of botulism among waterfowl and enhanced spread of mosquito-borne diseases.
(iv)	Benefits to invasive species.
(v)	Stabilized turbid, phytoplankton-dominated systems, thus counteracting restoration measures.
(vi)	Destabilized macrophyte-dominated clear-water lakes.
(vii)	Increased carrying capacity of primary producers, especially phytoplankton, thus mimicking eutrophication.
(viii)	Higher trophic levels as a result of enhanced primary production.
(ix)	Negative impact on biodiversity linked to the clear water state.
(x)	Effect on biodiversity by changing the disturbance regime.

TOXIC SUBSTANCES

Although many of the most toxic substances introduced into the environment by human activity have been banned or restricted in use, many persist, especially in soils and sediments, and either remain in contact with food chains or can be remobilised and taken up by aquatic biota (Catalan *et al.*, 2004; Vives *et al.*, 2005). High levels of metals (such as mercury, Hg, and lead, Pb) and persistent organic pollutants such as polychlorinated biphenyls (PCBs) are present in the tissue of freshwater fish in arctic and alpine lakes (Grimalt *et al.*, 2001; Vives *et al.*, 2004a). This attests to the mobility and transport of these substances in the atmosphere (Carrera *et al.*, 2002) and their concentration in cold regions (Fernandez & Grimalt, 2003). Biomagnification within aquatic systems with long food chains can elevate concentrations in fish to levels lethal for human con-sumption. The major concern with respect to climate change is the extent to which toxic sub-stances will be remobilised and cause additional contamination and biological uptake in arctic and alpine freshwater systems as water temperatures rise. Storm events and flooding might also

increase soil and sediment erosion and lead to the re-mobilisation of metals and persistent organic compounds (Grimalt *et al.*, 2004a,b; Rose *et al.*, 2004). In the case of Hg, changing hydrology in Boreal forest soils may lead to the enhanced production of methyl mercury (Meili *et al.*, 2003; Munthe, 2008).

In Europe, mountains and remote ecosystems are directly influenced by temperature changes and are subject to the accumulation of persistent organic pollutants (POPs) (Grimalt *et al*., 2001). Rivers and lakes in these environments provide information on the transfer mechanisms and impact of these compounds in headwater regions. Accumulation patterns depend on diverse aspects such as the time of their introduction into the environment (Gallego *et al*., 2007). Polybromodiphenyl ethers (PBDEs) in fish from Pyrenean lakes showed higher concentrations at lower temperatures, as predicted in the global distillation model. Conversely, no temperature-dependent distribution of POPs has been observed in vertical lake transects, neither in the Tatra Mountains (Central Europe) nor in fish from high mountain lakes distributed throughout Europe (Vives e*t al*., 2004b). Concentrations of PCBs in fish show significant temperature correlations in all these studies.

In the UK, research on toxics has focused on upland lakes exposed to air pollutants from long-range transport or in pesticide use and transport in lowland systems. In particular, detailed studies at Lochnagar in Scotland have shown high concentrations of both trace metals and trace organic compounds in sediments (Rose *et al*., 2001; Yang *et al*., 2002) and fish (Rosseland *et al*., 2007), and Rose *et al*. (2004) have argued that increased storminess in future might cause the remobilisation of trace metals from catchment soils. Bloomfield *et al*. (2006) have undertaken a review of climate change impacts on pesticides in surface and groundwaters and conclude that changes in temperature, rainfall intensity and seasonality will affect pesticide release and transport. However, long-term land-use change driven by climate change may result in significant changes in pesticide use and release into rivers and lakes.

In addition, leaching of heavy metals from old mining tailings, or in discharges from abandoned mines, can cause local breaches of quality standards. Simulations of the impact of climate changes in northern England show decreased surface contamination through dilution by cleaner sediment from hillslopes unaffected by mining activity (Coulthard & Macklin, 2003). Discharges of polluted water from mines depend on the extent of groundwater rebound (Adams & Younger, 2001).

ACIDIFICATION AND DOC IN THE UPLANDS

Reductions in sulphur emissions since the 1980s have initiated the recovery of many European streams and lakes that have been subject to acidification (Wright *et al*., 2005). Models such as MAGIC (Model of Acidification In Groundwaters) successfully predicted this slow recovery (Cosby *et al*., 1986) and some studies warned of future problems associated with increased N deposition and climate change (Wilby, 1993; Wright *et al*., 1995; Whitehead *et al*., 1997; Monteith *et al*., 2000). Climate variables that could affect acidification include higher temperatures, increased summer drought, wetter winters, reduced snow pack, concomitant changes in hydrological pathways, and increased occurrence of sea-salt deposition events. Intense rainfall and wetter winter conditions favour acidic episodes (Evans *et al*., 2008; Wright, 2008) as does rapid melt of snow packs (Laudon & Bishop, 2002). Acid pulses can, in turn, cause fish kills and loss of invertebrate species (Kowalik & Ormerod, 2006).

Droughts can further exacerbate acidification by creating lower water tables, aerobic conditions and enhanced oxidation of sulphur to sulphate (Wilby, 1994; Dillon *et al*., 1997). Acid anions are exported during subsequent storm events, along with heavy metals (Tipping *et al*., 2003). Peat catchments are particularly vulnerable to climate change as they have significant stores of sulphur which could be released following summer droughts (Aherne *et al*., 2006). Nitrogen is another source of acidification in upland catchments as nitric acid is a strong acid anion that can be flushed after droughts (Whitehead *et al*., 1997; Adamson *et al*., 1998; Curtis *et al*., 2005; Wilby
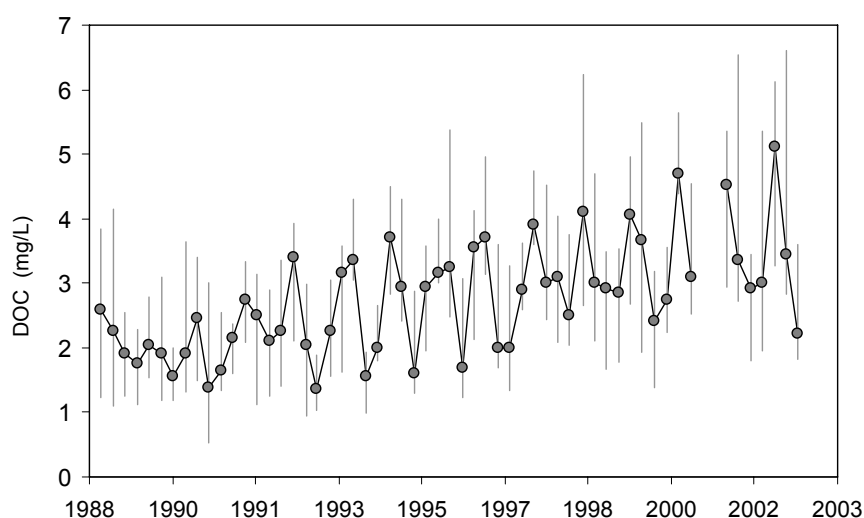
Fig. 8 Trends in DOC across the UK. Source: CEH Acid Waters Monitoring Network.

e*t al.*, 2006b). The Norwegian CLIMEX study (Wright *et al.*, 1998; Wright & Jenkins, 2001) showed significant mineralization of nitrogen following increases in temperature and CO_2, and this switched a small catchment from being a nitrogen sink to a nitrogen source.

Dissolved organic carbon (DOC) concentrations have doubled across the UK since the 1980s (Monteith *et al.*, 2000; Evans *et al.*, 2001, 2005; Freeman *et al.*, 2001; Worrall *et al.*, 2003, 2004) (Fig. 8). The reasons for this are not yet clear. Freeman *et al.* (2001) proposed a climate-related enzyme latch mechanism that can release carbon from wetland soils. However, there is also strong circumstantial evidence that reduced sulphur deposition has an influence on DOC trends (Monteith *et al.*, 2007). There is no doubt that hydrological conditions affect DOC export, with lower DOC concentrations during drought and greater concentrations during high flows (Hughes *et al.*, 1997). Water colour is correlated with DOC, so as DOC increases water colour will be increasingly brown. Whilst colour *per se* is not a public health issue, the chlorination processes at water treatment plants generate by-products, such as trihalomethanes, which are carcinogens (Chow e*t al.*, 2003). DOC modelling by Futter *et al.*, (2007) suggests that warmer, wetter climates could lead to higher levels of surface water DOC, but there remain large uncertainties due to the complex dynamics and biochemical processes controlling soil carbon flux.

ESTUARIES

Major UK estuaries, such as the Humber, already receive the treated discharges from industry and the sewage infrastructure, and localised sources of contaminants from past ore mining and agricultural activity (Oguchi *et al.*, 2000). Rising sea levels, salinity and water temperatures pose further threats to estuarine ecosystem structure and functioning. However, anticipating and managing the impacts on estuarine environments requires understanding of not only the hydro-dynamics of the estuary itself, but also regional influences due to changes in fluvial flows, effects of storms, surges, changes in sediment sources and sinks, wind and ocean circulation patterns (Sündermann *et al.*, 2001; Scavia *et al.*, 2002). These effects are superimposed on a range of other human influences including habitat destruction, coastal engineering, dikes, river regulation and land drainage, all of which can affect sediment and nutrient circulation in estuaries.

To date, attention has been paid to monitoring the effects of rising water temperatures and nutrient loads on existing challenges, such as eutrophication and severity of hypoxia. For example, Preston (2004) reports a warming of the Chesapeake Bay estuary (USA) by 0.8–1.1°C since the mid-20th century. Historically, winters with more frequent wet/warm weather patterns have been followed by greater phytoplankton biomass occurring later in the spring, covering a larger area and

extending farther seaward in the estuary (Miller & Harding, 2007). Higher rates of primary production have also been observed in the Hudson River estuary (USA) during dry summers when freshwater discharges are lower and residence times, stratification and depth of the photic zone increase (Howarth *et al*., 2000). Conversely, more frequent peak discharges can increase scour, and alter sediment patterns and, hence, the morphology of estuarine mouths (e.g. Fox *et al*., 2001). Furthermore, Strufy *et al*. (2004) report a downstream shift of the salinity gradient and concentrations of nutrients (but greater total nutrient loads) associated with a threefold increase in annual discharge of the upper Schelde estuary, Belgium/The Netherlands.

Assessment of future climate change impacts on estuaries requires a multidisciplinary approach that addresses the interplay between environmental, engineering and socio-economic responses (Schirmer & Schuchardt, 2001). Geographical information systems (GIS) have been the preferred tool for integrating regional climate change scenarios with data on land cover, elevation, soil type and habitats. For example, Osterkamp *et al*. (2001) examined the impact of rising sea levels, higher mean temperatures and changing winter/summer precipitation on the biotypes of the Weser estuary (Germany). By the year 2050, a rise of the mean tidal high water is expected to result in an expansion of the area occupied by frequently flooded reeds. However, projected reductions in summer flows could increase residence times and enable salt water to penetrate further upstream (Schirmer & Schuchardt, 2001). In comparison, Justic *et al*. (2005) report large variations in the frequency of hyopixa in the Gulf of Mexico due to projections of the Mississippi River discharge, nitrate flux and ambient water temperature under four different climate change scenarios.

URBAN AREAS

Built environments are already "hot spots" of environmental change (Grimm *et al*., 2008). Areas of impervious surface cover alter the hydrology and geomorphology of drainage systems, whereas municipal and industrial discharges increase loads of nutrients, heavy metals, pesticides and other contaminants in receiving surface water courses (Paul & Meyer, 2001; Clark *et al*., 2007). In addition, groundwater beneath industrial conurbations may be contaminated by microbiological agents, nitrogen, chlorinated and hydrocarbon compounds, and metals originating from the overlying land complexes or leakage from sewerage systems. Urban land uses also affect the patterns and rates of recharge to underlying aquifers as evidenced by detailed surveys for Birmingham (Ford & Tellam, 1994), Doncaster (Morris *et al*., 2006) and Nottingham (Barrett *et al*., 1999), in England, UK.

It is widely recognised that urban populations, infrastructure and institutions will come under increased pressure with climate change (Ruth & Coelho, 2007). Anticipated risks involve cooling of urban areas, urban drainage and flood risk, security of water resources supply, and outdoor spaces (including air quality and habitats) (Wilby, 2008). Potential impacts on urban water quality will be driven largely by changes in short-duration rainfall intensity overwhelming drainage systems, as well as rising sea levels affecting combined sewerage outfalls. The former could result in greater incidence of foul water flooding of domestic property, or uncontrolled discharges of untreated sewage with concomitant impacts on ecosystems. The summer 2007 flooding in England highlighted the extent to which water treatment works may, themselves, be vulnerable to flooding. Similarly, infrastructure located on low-lying coastal sites may be threatened by coastal erosion and/or inundation.

The assumption of stationarity of rainfall properties for hydraulic infrastructure design is no longer tenable, so retrofitting or upgrading of drainage capacity may be required (Denault *et al*., 2006). Adaptation responses must accommodate potential changes in both the frequency of extreme precipitation events of given magnitude, and changes in the severity of events with given return periods. For example, under a doubled CO_2 climate-change scenario the 1 in 100-year flood becomes a 1 in 10-year event for Canberra, Australia (Schreider *et al*., 2000). However, detailed hydrological modelling for urban areas continues to be confounded by a lack of high-resolution

(space and time) climate-change scenarios (Grum *et al.*, 2006; Wilby, 2007). Indeed, many climate models treat built areas as vegetated surfaces, whilst realistic simulations of localised, high-intensity, summer precipitation events have yet to be realised (see Fowler & Ekström, 2009). Water quality impacts associated with rising air and water temperatures may be more confidently projected. A beneficial consequence of rising temperatures could be improved performance of water treatment works.

To date, there have been only a handful of studies that explicitly consider water quality impacts in urban areas under climate variability and change. For example, Chang (2004) reports a 50% increase in mean annual nitrogen loads for the Conestoga River basin, Pennsylvania, USA, by 2030 under a scenario of concurrent urbanization and warmer/wetter climate. Burian *et al.* (2002) use an integrated modelling framework (comprising of an urban air quality model, an urban runoff model, and a water quality model) to investigate the ancillary benefits to water quality of reducing NOx, VOC and ammonia emissions. Air emission reductions during the dry season had a negligible impact on algal concentrations or nocturnal DOC in the Ballona Creek watershed, Los Angeles, USA. However, emission reductions in the wet season led to a 16% reduction in storm-water loads.

The large uncertainty in sub-daily regional rainfall scenarios, combined with the magnitude of some projected changes, suggest that traditional engineering measures alone are unlikely to be the solution (Ashley *et al.*, 2005). Developers and design teams are already being encouraged to incorporate more "green space" into their plans, to counter urban heat island effects, reduce flood risk, improve air quality and enhance habitat availability/connectivity (GLA, 2005). Sustainable urban drainage systems could confer benefits in terms of both reduced flood risk and local water quality improvements. Others contend that reducing the risks posed by climate change requires a mix of conurbation-scale strategic planning and neighbourhood-level urban design solutions (Lindley *et al.*, 2006). However, it should be recognised that some adaptations (such as re-use of grey water) could have implications for water quality (such as lower volumes and higher concentrations of sewage) (Rueedi *et al.*, 2005).

MODELLING FRAMEWORKS

The need for a catchment-scale approach to freshwater ecosystem management is recognised by the WFD, where the basic unit of management is referred to as the "river basin district". Wilby *et al.* (2006a) reviewed potential interactions between climate change and the WFD, and highlighted the effects of changing flow, water velocity, hydromorphology and water quality on aquatic ecosystems and, therefore, the ability to meet WFD objectives for restoration and protection. Complex interactions between aquatic and terrestrial systems can be explored using integrated modelling at the catchment scale. Models will need to represent climate, soil, land use, lakes, rivers and coastal waters so that the responses of whole catchment systems can be simulated and the models used to assess the impacts of alternative catchment management decisions. To date, the majority of modelling studies have addressed only hydrological effects (IPCC, 2007). However, projects such as Euro-limpacs are placing greater emphasis on the water quality and ecosystem responses of contrasting catchments (Table 2).

Applications

The most widely-used models in the Euro-limpacs project are the INCA (Integrated Catchment Model) suite. The first INCA model (INCA-N) was used to investigate the response of rivers to changes in nitrogen inputs and catchment nitrogen metabolism (Whitehead *et al.*, 1998a,b; Wade *et al.*, 2002a). Since then, the INCA framework has been extended to phosphorus (INCA-P), particulates (INCA-SED); dissolved organic carbon (INCA-C) and mercury (INCA-Hg). Another dynamic model being used in Euro-limpacs is the Mike11-TRANS model. This has a suite of ecological sub-models linked to a hydrodynamic model, and has been used in conjunction with the

Table 2 Principal areas used in the Euro-limpacs project for catchment-scale modelling.

Country	Study area	Area (km^2)	River	Lake	Wetland	Additional key issues*
Austria	Piburger See	2	✓	✓		Eutrophication
Finland	Savijoki	15.4	✓		✓	Eutrophication, Sed.
	Simojoki	3160	✓	✓	✓	Acid., N sat.
	Tueronjoki	439	✓	✓	✓	N sat, Eutrophication
France	Garonne-Adour	56 500	✓	✓	✓	Eutrophication
Norway	Bjerkreim	685	✓	✓	✓	Acid., N sat., C
	Tovdalselva	1 855	✓	✓	✓	N sat, Eutrophication
Romania	Lower Danube wetland	210			✓	Eutrophication
Spain	La Tordera	124	✓		✓	Eutrophication
Sweden	Gårdsjön	0.005		✓		Acid., N sat., C
	Svartberget	0.5	✓	✓		Acid., Hg
UK	Conwy	590	✓		✓	Acid., Eutrophication, C
	Kennet	1 030	✓		✓	Eutrophication, Sed.
	Lambourn	263	✓		✓	Eutrophication
	Endrick/Falloch	781	✓	✓		Eutrophication
	Tamar	917	✓		✓	Eutrophication
	Wye	4 140	✓	✓		Acid., Eutrophication, C

* Acid.: acidification; C: carbon; N sat.: N saturation; Sed.: sediment.

rainfall–runoff model NAM (Nitrogen Ammonia Model) to simulate nitrogen fluxes in lowland Danish catchments (Andersen *et al.*, 2006). Another development of existing models is the use of a landscape-based mixing model (PEARLS: Prediction of Acidification and Recovery on a Landscape Scale) coupled with the acidification model MAGIC (Cosby *et al.*, 1986) to simulate the recovery from acidification of the Conwy in North Wales, UK, a large heterogeneous river basin (Evans *et al.*, 2005).

The INCA-N model was used in one of the first attempts to model outcomes of different adaptation strategies to address the rising nitrate concentrations under climate change (Whitehead *et al.*, 2006b). Precipitation scenarios were downscaled from three GCMs for the River Kennet in southern England for the period 1961–2100. The scenarios yield different effects in detail, but all show a general increase in nitrate concentration due to enhanced microbial activity. Figure 9 shows the results of applying different interventions on this overall trend under one precipitation scenario (downscaled from HadCM3).

The baseline scenario, with no intervention, shows a steady increase in concentrations of nitrate (as N). The peak towards the end of the period is due to the onset of rainfall after a simulated severe drought. "Atmospheric" represents a reduction of reactive nitrogen deposition by 50%. "Meadows" involves the construction of water meadows adjacent to the river, which are allowed to flood and remove nitrogen by denitrification. An area four times the river surface area was assumed, and this almost stabilises the nitrate concentration. "Fertiliser" (reducing N fertiliser application by 50%) is the most effective intervention, leading to a decrease in nitrate concentration, but this reduces agricultural intensity in the catchment to that of the 1950s which would seem to be an unlikely strategy. Finally a "combined" strategy is modelled in which each of the single strategies is applied at half intensity. This is not quite as effective as a 50% reduction in fertiliser, but still leads to a decrease in concentration. Further work, using a version of INCA-N modified to account for the transport of nitrate through the unsaturated zone of the underlying chalk rock, predicts that reducing fertiliser inputs today will have a short-term impact on in-stream nitrate concentrations, but a clear long-term reduction will not occur until between 2060 and 2080. This is because nitrate that has already accumulated in the chalk aquifer (Jackson *et al.*, 2007). Thus, some in-stream intervention, such as construction of water meadows, may be the best option to reduce in-stream nitrate concentrations should this be required under the Water Framework Directive.
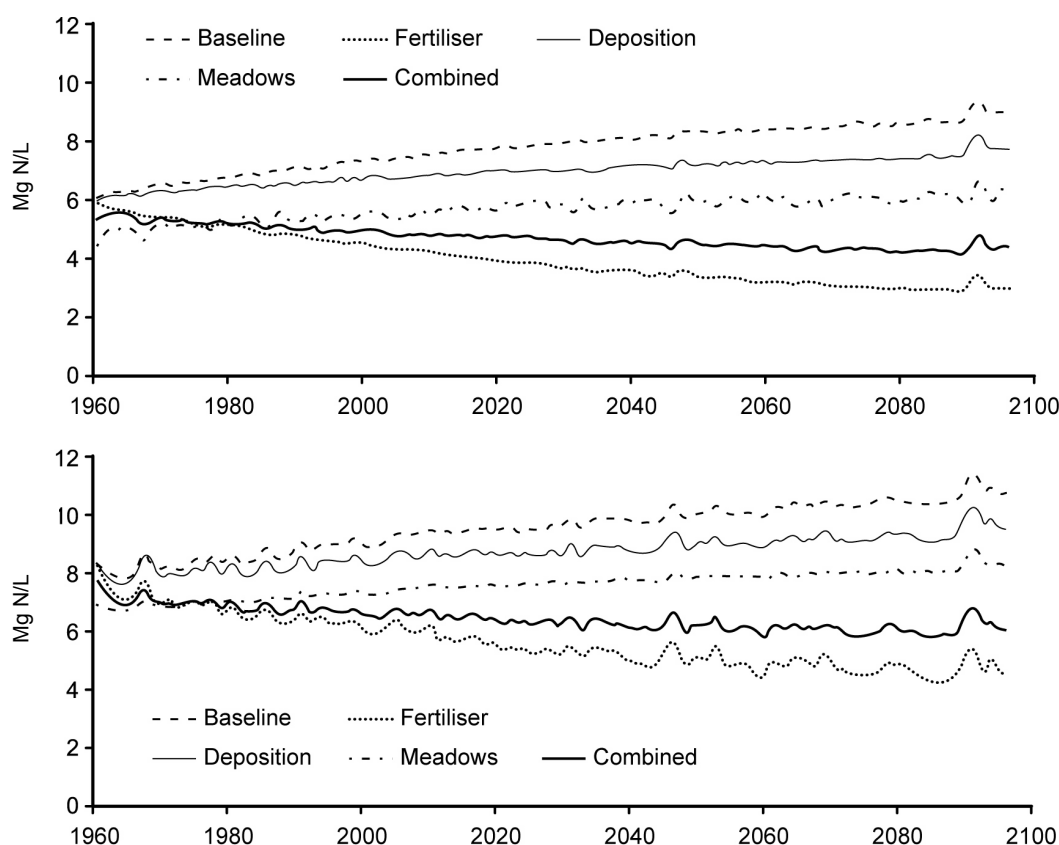
Fig. 9 Adaptation runs using HadCM3 and A2 emissions showing nitrate impacts in the upper "natural" reach (top) and in the lower "effluent affected reach" (bottom) in terms of mean annual stream water nitrate concentrations for 1961–2100. The runs represent baseline conditions, fertiliser reduction, N deposition reduction, water meadow creation, and a combined strategy (source: Whitehead *et al.*, 2007).

Model chains and integration

Modelling catchment responses often requires the correct integration of separate models addressing different components of the catchment, such as soils, vegetation, groundwater, rivers, lakes, etc., as well as models to generate driving variables such as precipitation or temperature. This integration can occur in two ways: chains of models can be produced in which the output of one model is used as input to the next, following the pathways of water through the catchment; or the component models can be integrated so that data are passed from one model to another inside the larger model. The latter is more convenient for the modeller, especially when performing large numbers of runs, but more difficult to achieve given that the component models have normally not been designed with integration in mind.

A number of chaining and integration projects are under way in Euro-limpacs as part of model toolkit development. One completed example is the chaining of various models to predict the response of the Bjerkreim River and fjord system, in southern Norway, to climate change (Kaste *et al.*, 2006). Two GCMs, namely ECHAM4 and HadAM3H, were used to project climate changes on a large spatial scale; a regional climate model (HIRHAM) to downscale these predictions to daily intervals at the catchment scale; the hydrological model HBV to translate the meteorological variables into water fluxes through the catchment; the water quality models MAGIC and INCA-N to predict nitrogen fluxes and concentrations in catchment components; and the NIVA FJORD model to predict nitrate discharges of the Bjerkreim River to the Egersund Fjord. The results driven by the HadAM3H scenario indicated the possibility of increased productivity and eutrophication in the fjord by 2080, whereas those driven by the ECHAM4 scenario did not. Irrespective of the scenario, the model chaining exercise allows the exploration of the possible consequences of

climate change on the internal dynamics of the catchment system, such as the seasonal patterns of water flow, snowmelt changes, acidification, and the possible ecological consequences.

Model uncertainty

All model predictions are uncertain to some degree, but characterising and reducing this uncertainty is one of the priorities for Euro-limpacs. The sources of uncertainty are manifold (see Jakeman *et al.*, 1993; New & Hulme, 2000; McIntyre *et al.*, 2005; Wilby, 2005). The emission scenarios used in GCMs are subject to uncertainty in predictions of population growth, economic activity and, hence, emission rates. The GCM models themselves are subject to uncertainty in process understanding, parameter values and the numerical solution of the equations. Downscaling is subject to uncertainty due to accuracy of local weather data and application of the statistical procedures required for downscaling. Also, catchment hydrochemical models are uncertain, given a lack of knowledge of detailed chemical processes and how to parameterise these. This suggests that the overall uncertainty will be so large as to render projections useless, but this is not necessarily the case. The calculation of critical loads, deposition thresholds used in pollution control policy, involves models which are combinations of 10–20 uncertain parameters. However, rather surprisingly, translating input uncertainties into uncertainty in the outputs is typically less than the summed uncertainty in the input parameters (e.g. Skeffington *et al.*, 2007). This counter-intuitive result suggests that complex behaviour patterns can reduce to surprisingly low variability in model outputs. A number of techniques for estimating uncertainty in environmental models are currently being explored by Euro-limpacs participants. For example, a Monte Carlo-based tool is being used to perform sensitivity analysis, to identify key parameters and, hence, place confidence limits about model predictions (Wade *et al.*, 2002b; Cox & Whitehead, 2004; Wilby, 2005). At the same time, less formal approaches are being used, such as applying different models to the same problem and comparing the results.

CONCLUDING REMARKS

This review is part of an ongoing process to improve understanding of the potential impacts of climate change, as well as identify the key scientific questions that need to be addressed. Although there is consensus about temperature increases, there is less certainty about the likely impacts on water quality due to changes in regional precipitation—especially due to changes in extreme events. However, it is not too early to consider the long-term adaptation options. Designs for new water supply, urban drainage and treatment systems will have to incorporate climate change effects, and new operational procedures may be required.

There could also be major implications for water quality monitoring protocols, environmental standards, compliance and reporting (Crane *et al.*, 2005). However, it could be that current water quality standards are sufficiently robust to cope with climatic change, especially as they have evolved from wide ranging toxicity studies (Crane *et al.*, 2005) and have been applied to many climatic regions across Europe.

Furthermore, efforts to mitigate climate change through reduced emissions or to adapt across different sectors could have indirect consequences for water quality. The following paragraphs identify a few examples of these linkages.

As noted above, climate change is expected to cause more hydrological extremes with enhanced drought and flooding episodes (EEA, 2007; Kundzewicz *et al.*, 2007). Adaptation to these impacts will have to be substantial, and could include new water transfer schemes to transport water to drier areas and/or new reservoirs to improve security of supply. Resulting changes in flow regimes will influence the chemistry, hydromorphology and ecology of regulated water bodies. Agricultural activity will have to adapt to longer growing seasons combined with reduced water availability, with new crops suited to drier, warmer conditions. Biofuel crops are already in demand and their intensive production could exacerbate water supply problems. Increased recycling of water in the UK, Central and Southern Europe is also an option but this

implies lower volumes and final discharges of effluents with higher concentrations. Adaptation measures to restore polluted ecosystems might involve managing water levels in wetlands, lakes and rivers, or the designation of vegetation corridors and buffer zones, all of which potentially affect water quality.

Land-use change and longer growing seasons could increase the use of fertilisers with subsequent leaching to watercourses, rivers and lakes, increasing the risk of eutrophication and loss of biodiversity (Moss *et al*., 2004). Some risks may be countered by the development of pesticides that have fewer side effects and are better targeted. Protocols involving the regulation of forestry and agriculture could reduce levels of toxics, such as mercury or PCBs. Other measures may be needed to meet WFD objectives, such as improved use of riparian/marginal wetlands, increased hedge growth and forestry to reduce sediment and nutrient transport. Harvesting of woodlands or woody crops for biofuel in upland areas removes soil base cations. This could result in enhanced soil and groundwater acidification. The interaction with atmospheric nitrogen deposition also needs to be considered as the atmospheric N deposition is a significant proportion of N budgets in catchments (Whitehead *et al*., 1998a,b).

The effects of climate change need to be considered in tandem with atmospheric pollution policies. Carbon removal technologies at power stations utilize amines, which could increase ammonia releases, thereby enhancing N deposition and, hence, both eutrophication and acidification. Conversely, the European Environment Agency (2006) has demonstrated that there could be significant ancillary benefits to human health and the environment arising from greenhouse gas emission reductions. This is because climate change policies aimed at reducing EU greenhouse gas emissions in line with the 2°C target lead to reductions of emissions of other air pollutants associated with fossil fuel combustion. The percentage of total ecosystems area in the EU receiving acid and nitrogen deposition above critical loads would decrease under both the Air Strategy scenario (2020) and the Climate Action scenario.

It is clear from the above examples that there could be significant water quality outcomes arising from a host of planned and inadvertent responses to climate change. Therefore, plans to address undesirable water quality impacts will require the integration of interventions across all sectors and institutions responsible for managing air, land and water resources.

REFERENCES

Adams, R. & Younger, P. L. (2001) A strategy for modelling ground water rebound in abandoned deep mine systems. *Groundwater* **39**, 249–261.

Adamson, J. K., Scott, W. A. & Rowland, A. P. (1998) The dynamics of dissolved nitrogen in a blanket peat dominated catchment. *Environ. Pollut.* **99**, 69–77.

Aherne, J., Larssen, T., Cosby, B. J. & Dillon P. J. (2006) Climate variability and forecasting surface water recovery from acidification: Modelling drought-induced sulphate release from wetlands. *Sci. Total Environ.* **365**, 186–199.

Andersen, H. E., Kronvang, B., Larsen, S. E., Hoffmann, C. C., Jensen, T. S. & Rasmussen, E. K. (2006) Climate-change impacts on hydrology and nutrients in a Danish lowland river basin. *Sci. Total Environ.* **365**, 223–237.

Arheimer, B., Andreasson, J., Fogelberg, S., Johnsson, H., Pers, C. B. & Persson, K. (2005) Climate change impact on water quality: Model results from southern Sweden. *Ambio* **34**, 559–566.

Arnell, N. W. (2003) Relative effects of multi-decadal climatic variability and changes in the mean and variability of climate due to global warming: future streamflows in Britain. *J. Hydrol.* **270**(3-4), 195–213.

Ashley, R. M., Balmforth, D. J., Saul, A. J. & Blanksby, J. D. (2005) Flooding in the future—predicting climate change, risks and responses in urban areas. *Water Sci. Technol.* **52**, 265–273.

Barrett, M. H., Hiscock, K. M., Pedley, S., Lerner, D. N., Tellam, J. H. & French, M. J. (1999) Marker species for identifying urban groundwater recharge sources: a review and case study in Nottingham, UK. *Water Research* **33**, 3083–3097.

Bates, B. C., Kundzewicz, Z. W., Wu, S. & Palutikof, J. P. (eds) (2008) *Climate Change and Water*. Technical Paper VI of the Intergovernmental Panel on Climate Change, IPCC Secretariat, Geneva, Switzerland.

Battarbee, R. W., Anderson, N. J., Jeppesen, E. & Leavitt, P. R. (2005) Combining palaeolimnological and limnological approaches in assessing lake ecosystem response to nutrient reduction. *Freshwater Biology* **50**, 1772–1780.

Battarbee, R. W., Kernan, M., Livingstone, D., Nickus, U., Verdonschot, P., Hering, D., Moss, B., Wright, R., Evans, C., Grimalt, J., Johnson, R., Maltby, E., Linstead, L. & Skeffington, R. (2008) Freshwater ecosystem responses to climate change: the Euro-limpacs project. In: *The Water Framework Directive—Ecological and Chemical Status Monitoring.* (ed. by P. Quevauviller, U. Borchers, C. Thompson & T. Simonart), 313–354. John Wiley & Sons Ltd, Chichester, UK.

Blenckner, T., Adrian, R., Livingstone, D. M., Jennings, E., Weyhenmeyer, G. A., George, D. G., Jankowski, T., Jarvinen, M., Aoghusa, C. N., Noges, T., Straile, D. & Teubner, K. (2007) Large-scale climatic signatures in lakes across Europe: a meta-analysis. *Global Change Biology* **13**, 1314–1326.

Bloomfield, J. P., Williams, R. J., Goody, D. C., Cape, J. N. & Guha, P. (2006) Impacts of climate change on the fate and behaviour of pesticides in surface and groundwater—a UK perspective. *Sci. Total Environ.* **369**, 163–177.

Bouraoui, F., Galbiati, L. & Bidoglio, G. (2002) Climate change impacts on nutrient loads in the Yorkshire Ouse catchment (UK). *Hydrol. Earth System Sci.* **6**, 197–209.

Brown, L. E., Hannah, D. M. & Milner, A. M. (2007) Vulnerability of alpine stream biodiversity to shrinking glaciers and snowpacks. *Global Change Biol.* **13**(5), 958–966.

Burian, S. J., McPherson, T. N., Brown, M. J., Streit, G. E. & Turin, H. J. (2002) Modeling the effects of air quality policy changes on water quality in urban areas. *Environ. Model. Assess.* **7**, 179–190.

Carrera, G., Fernandez, P., Grimalt, J. O., Ventura, M., Camarero, L., Catalan, J., Nickus, U., Thies, H. & Psenner, R. (2002) Atmospheric deposition of organochlorine compounds to remote high mountain lakes of Europe. *Environ. Sci. Technol.* **36,** 2581–2588.

Carvalho, L. & Kirika, A. (2003) Changes in shallow lake functioning: response to climate change and nutrient reduction. *Hydrobiologia* **506**, 789–796.

Catalan, J., Ventura, M., Vives, I. & Grimalt, J. O. (2004) The roles of food and water in the bioaccumulation of organochlorine compounds in high mountain lake fish. *Environ. Sci. Technol.* **38**, 4269–4275

Chang, H. J. (2004) Water quality impacts of climate and land use changes in southeastern Pennsylvania. *Profess. Geographer* **56**, 240–257.

Chow, A. T., Tanji, K. K. & Gao, S. (2003) Production of dissolved organic carbon DOC and trihalomethane precursor THM from peat soils. *Water Res.* **37**, 4475–4485.

Clark, S. E., Burian, S., Pitt, R. & Field, R. (2007) Urban wet-weather flows. *Water Environ. Res.* **79**, 1166–1227.

Conlan, K., Lane, S., Ormerod, S. & Wade, T. (2007) Preparing for climate change impacts on freshwater ecosystems, PRINCE: results. Environment Agency Science Report SC030300/SR, Bristol, UK.

Cosby, B. J., Whitehead, P. G. & Neale, R. (1986) A preliminary model of long term changes in stream acidity in south western Scotland. *J. Hydrol.* **84**, 381–401.

Coulthard, T. J. & Macklin, M. G. (2003) Modeling long-term contamination in river systems from historical metal mining. *Geology* **31**, 451–454.

Cox, B. A. & Whitehead, P. G. (2004) Parameter sensitivity and predictive uncertainty in a new water quality model, Q2. *J. Environ. Engng* **131**(1), 147–157.

Cox, B. A. & Whitehead, P. G. (2009) Potential impacts of climate change on dissolved oxygen in the River Thames. *Hydrol. Res.* (in press).

Crane, M., Whitehouse, P., Comber, S., Ellis, J. & Wilby, R. L. (2005) Climate change influences on environmental and human health chemical standards. *Human & Ecol. Risk Assess.* **11**, 289–318.

Curtis, C. J., Evans, C. D., Helliwell, R. C. & Monteith, D. T. (2005) Nitrate leaching as a confounding factor in chemical recovey from acidification in UK upland waters. *Environ. Pollut.* **137**(1), 73–82.

Davidson, I. C. & Hazelwood, M. S. (2005) Effect of climate change on salmon fisheries. Environment Agency Science Report W2-047/SR, Bristol, UK.

Denault, C., Millar, R. G. & Lence, B. J. (2006) Assessment of possible impacts of climate change in an urban catchment. *J. Am. Water Resour. Assoc.* **42**, 685–697.

Dillon, P. J., Molot, L. A. & Futter, M. (1997) The effect of El Nino-related drought on the recovery of acidified lakes. *Environ. Monit. Assess.* **46**, 105–111.

Durance, I. & Ormerod, S. J. (2007) Climate change effects on upland stream macroinvertebrates over a 25-year period. *Global Change Biology* **13**, 942–957.

Elliott, J. A. & May, L. (2008) The sensitivity of phytoplankton in Loch Leven (UK) to changes in nutrient load and water temperature. *Freshwater Biology* **53**, 32–41.

Elliott, A. J., Thackeray, S. J. Huntingford, S. & Jones R. G. (2005) Combining a regional climate model with a phytoplankton community model to predict future changes in phytoplankton in lakes. *Freshwater Biology* **50**(8). 1404–1411.

EEA (European Environment Agency) (2006) Air quality and ancillary benefits of climate change policies. European Environment Agency Technical Report no. 4/2006, Copenhagen, Denmark.

EEA (European Environment Agency) (2007a) Climate change and water adaptation issues. EEA Tech. Report no. 2/2007, Copenhagen, Denmark.

EEA (European Environment Agency) (2007b) Climate change: the cost of inaction and the cost of adaptation. EEA Tech. Report no. 13/2007, Copenhagen, Denmark.

Evans, C. D., Monteith, D. T. & Harriman, R. (2001) Long-term variability in the deposisiton of marine ions at west coast sites in the UK Acid Waters Monitoring Network: impacts on surface water chemistry and significance for trend determination. *Sci. Total Environ.* **265**, 115–129.

Evans, C. D., Monteith, D. T. & Cooper, D. M. (2005) Long-term increases in surface water dissolved organic carbon: observations, possible causes and environmental impacts. *Environ. Pollut.* **137**, 55–71.

Evans, C., Reynolds, B., Hinton, C., Hughes, S., Norris, D., Grant, G. & Williams, B. (2008) Effects of decreasing acid deposition and climate change on acid extremes in an upland stream. *Hydrol. Earth System Sci.* **12**, 337–351.

Fernandez, P. & Grimalt, J. O. (2003) On the global distribution of persistent organic pollutants. *Chimia* **57**, 514–521.

Ford, M. & Tellam, J. H. (1994) Source, type and extent of inorganic contamination within the Birmingham urban aquifer system, UK. *J. Hydrol.* **156**, 101–135.

Fowler, H. J. & Ekström, M. (2009) Multi-model ensemble estimates of climate change impacts on UK seasonal rainfall extremes. *Int. J. Climatol.* (in press).

Fox, H. R., Wilby, R. L. & Moore, H. M. (2001) The impact of river regulation and climate change on the barred estuary of the Oued Massa, southern Morocco. *Regulated Rivers–Res. Manage.* **17**, 235–250.

Freeman, C., Evans, C. D., Monteith, D. T., Reynolds, B. & Fenner, N. (2001) Export of organic carbon from peat soils. *Nature* **412**, 785.

Futter, M. N., Butterfield, D., Cosby, B. J., Dillon, P. J., Wade, A. J. & Whitehead, P. G. (2007) Modeling the mechanisms that control in-stream dissolved organic carbon dynamics in upland and forested catchments. *Water Resour. Res.* **43**(2) doi:10.1029/2006WR004960.

Gallego, E., Grimalt, J. O., Bartrons, M., Lopez, J. F., Camarero, L., Catalan, J., Stuchlik, E. & Battarbee, R. (2007) Altitudinal Gradients of PBDEs and PCBs in fish from European high mountain lakes. *Environ. Sci. Technol.* **41**, 2196–2202.

George, G., Hurley, M. & Hewitt, D. (2007) The impact of climate change on the physical characteristics of the larger lakes in the English Lake District. *Freshwater Biology* **52**, 1647–1666.

GLA (Greater London Authority) (2005) *Adapting to Climate Change: A Checklist for Development*. London Climate Change Partnership, London, UK.

Grimalt, J. O., Fernandez, P., Berdié, L., Vilanova, R. M., Catalan, J., Psenner, R., Hofer, R., Appleby, P. G., Rosseland, B. O. Lien, L., Massabuau, J. C. & Battarbee, R. W. (2001) Selective trapping of organochlorine compounds in mountain lakes of temperate areas. *Environ. Sci. Technol.* **35**, 2690–2697.

Grimalt, J. O., van Drooge, B. L., Ribes, A., Fernandez, P. & Appleby, P. (2004a) Polycyclic aromatic hydrocarbon composition in soils and sediments of high altitude lakes. *Environ. Pollut.* **131**, 13–24.

Grimalt, J. O., van Drooge, B. L., Ribes, A., Vilanova, R. M., Fernandez, P. & Appleby, P. (2004b) Persistent organochlorine compounds in soils and sediments of European high mountain lakes. *Chemosphere* **54**, 1549–1561.

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J. G., Bai, X. M. & Briggs, J. M. (2008) Global change and the ecology of cities. *Science* **319**, 756–760.

Grum, M., Jorgensen, A. T., Johansen, R. M. & Linde, J. J. (2006) The effect of climate change on urban drainage: an evaluation based on regional climate model simulations. *Water Sci. Technol.* **54**, 9–15.

Hari, R., Livingstone, D., Siber, R., Burkhardt-Holm, P. & Güttinger, H. (2006) Consequences of climatic change for water temperature and brown trout populations in Alpine rivers and streams. *Global Change Biology* **12**(1), 10–26.

Hammond, D. & Pryce, A. R. (2007) Climate change impacts and water temperature. Environment Agency Science Report SC060017/SR, Bristol, UK.

Hassan, H., Aramaki, T., Hanaki, K., Matsuo, T. & Wilby, R. L. (1998) Lake stratification and temperature profiles simulated using downscaled GCM output. *J. Water Sci. Technol.* **38**, 217–226.

Hering, D., Johnson, R. K., Kramm, S., Schmutz, S., Szoszkiewicz, K. & Verdonschot, P. F. M. (2006) Assessment of European rivers with diatoms, macrophytes, invertebrates and fish: a comparative metric-based analysis of organism response to stress. *Freshwater Biology* **51**, 1757–1785.

Hering, D., Schmidt-Kloiber, A., Murphy, J., Dahl, J., Zamora-Muñoz, C., López Rodríguez, M. J., Huber, T. & Graf, W. (2009) Potential impact of climate change on aquatic insects: a sensitivity analysis for European caddisflies *Trichoptera.* based on species traits. *Aquatic Sci.* (in press).

Howarth, R.W., Swaney, D. P., Butler, T. J. & Marino, R. (2000) Climatic control on eutrophication of the Hudson River estuary. *Ecosystems* **3**, 210–215.

Hughes, S., Reynolds, B., Hudson, J. & Freeman, C. (1997) Effects of summer drought on peat soil solution chemistry in an acid gully mire. *Hydrol. Earth System Sci.* **1**, 661–669.

Hulme, M., Jenkins, G. J., Lu, X., Turnpenny, J. R., Mitchell, T. D., Jones, R. G., Lowe, J., Murphy, J. M., Hassell, D., Boorman, P., McDonald, R. & Hill, S. (2002) Climate change scenarios for the 21st century for the UK. Scientific and Tech. Report for UKCIP, Tyndall Centre, University of East Anglia, Norwich, UK.

IPCC (Intergovernmental Panel on Climate Change) (2007) *Climate Change 2007: The Physical Science Basis: Summary for Policymakers*. Fourth Assessment Report of the IPCC (ed. by M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden & C. E. Hanson). Cambridge University Press, Cambridge, UK.

Jackson, B. M., Wheater, H. S., Wade, A. J., Butterfield, D., Mathias, S. A., Ireson, A. M., Butler, A. P., McIntyre, N. & Whitehead, P. G. (2007) Catchment-scale modelling of flow and nutrient transport in the Chalk unsaturated zone. *Ecol. Model.* **209**, 41–52.

Jakeman, A. J., Chen, T. H., Post, D. A., Hornberger, G. M. & Littlewood, I. G. (1993) Assessing uncertainties in hydrological response to climate at large scales. In: *Macroscale Modelling of the Hydrosphere* (ed. by W. B. Wilkinson), 37–47. IAHS Publ. 214. IAHS Press, Wallingford, UK.

Jeppesen, E., Sondergaard, M., Jensen, J. P., Havens, K. E., Anneville, O., Carvalho, L., Coveney, M. F., Deneke, R., Dokulil, M. T., Foy, B., Gerdeaux, D., Hampton, S. E., Hilt, S., Kangur, K., Kohler, J., Lammens, E. H. H. R., Lauridsen,T. L., Manca, M., Miracle, M. R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C. L., Straile, D., Tatrai, I., Willen, E. & Winder, M. (2005) Lake responses to reduced nutrient loading—an analysis of contemporary long-term data from 35 case studies. *Freshwater Biology* **50**, 1747–1771.

Jones, R. N., & Page, C. M. (2001) Assessing the risk of climate change on the water resources of the Macquarie River catchment. In: *Integrating Models for Natural Resources Management across Disciplines*, Part 2. *Issues and Scales.* (Proc. MODSIM 2001—Int. Congress on Modelling and Simulation) (ed. by P. Ghassemi, P. Whetton, R. Little & M. Littleboy), 673–678. Modelling and Simulation Society of Australia and New Zealand, Canberra, Australia.

Justic, D., Rabalais, N. N. & Turner, R. E. (2005) Coupling between climate variability and coastal eutrophication: evidence and outlook for the northern Gulf of Mexico. *J. Sea Res.* **54**, 23–35.

Kaste, O., Wright, R. F., Barkved, L. J., Bjerkeng, B., Engen-Skaugen, T., Magnusson, J. & Saelthun, N. R. (2006) Linked models to assess the impacts of climate change on nitrogen in a Norwegian river basin and fjord system. *Sci. Total Environ.* **365**, 200–222.

Komatsu, E., Fukushima, T. & Harasawa, H. (2007) A modeling approach to forecast the effect of long-term climate change on lake water quality. *Ecol. Model.* **209**, 351–366.

Kowalik, R. A. & Ormerod, S. J. (2006) Intensive sampling and transplantation experiments reveal continued effects of episodic acidification on sensitive stream invertebrates. *Freshwater Biology* **51**, 180–191.

Kundzewicz, Z. W., Mata, L. J., Arnell, N. W., Döll, P., Kabat, P., Jiménez, B., Miller, K. A., Oki, T., Sen, Z. & Shiklomanov, I. A. (2007) Freshwater resources and their management. In: *Climate Change 2007: Impacts, Adaptation*

and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (ed. by M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden & C. E. Hanson), 173–210. Cambridge University Press, Cambridge, UK.

Kundzewicz, Z. W., Mata, L. J., Arnell, N. W., Döll, P., Jimenez, B., Miller, K., Oki, T., Şen, Z. & Shiklomanov, I. (2008) The implications of projected climate change for freshwater resources and their management. *Hydrol. Sci. J.* **53**(1), 3–10.

Ladle, M. & Bass, J. A. B. (1981) The ecology of a small chalk stream and its responses to drying during drought conditions. *Arch. Hydrobiol.* **90**, 448–466.

Lane, S. N., Reid, S. C., Tayefi, V., Yu, D. & Hardy, R. J. (2007) Interactions between sediment delivery, channel change, climate change and flood risk in a temperate upland environment. *Earth Surf. Processes Landf.* **32**, 429–446.

Langan, S. J., Johnston, L., Donaghy, M. J., Youngson, A. F., Hay, D. W. & Soulsby, C. (2001) Variation in river water temperatures in an upland stream over a 30-year period. *Sci. Total Environ.* **265**, 195–207.

Laudon, H. & Bishop, K. H. (2002) The rapid and extensive recovery from episodic acidification in northern Sweden due to declines in SO_4^{2-} deposition. *Geophys. Res. Lett.* **29**, 1594.

Leemans, R. & Kleidon, A. (2002) Regional and global assessment of the dimensions of desertification. In: *Global Desertification. Do Humans Cause Deserts?* (ed. by J. F. Reynolds & D. M. Stafford-Smith), 215–232. Dahlem University Press, Berlin, Germany.

Limbrick, K. J., Whitehead, P. G, Butterfield, D. & Reynard, N. (2000) Assessing the potential impacts of various climate change scenarios on the hydrological regime of the River Kennet at Theale, Berkshire, south-central England, UK: an application and evaluation of the new semi-distributed model, INCA. *Sci. Total Environ.* **251/252**, 539–555.

Lindley, S. J., Handley, J. F., Theuray, N., Peet, E. & Mcevoy, D. (2006) Adaptation strategies for climate change in the urban environment: Assessing climate change related risk in UK urban areas. *J. Risk Res.* **9**, 543–568.

Longfield, S. A. & Macklin, M. G. (1999) The influence of recent environmental change on flooding and sediment fluxes in the Yorkshire Ouse basin. *Hydrol. Processes* **13**, 1051–1066.

Marsh, T. J. & Hannaford, J. (2007) The summer 2007 floods in England & Wales—a hydrological appraisal. National Hydrological Monitoring Programme. Centre for Ecology and Hydrology, Wallingford, UK.

McIntyre, N., Jackson, B., Wade, A. J., Butterfield, D. & Wheater, H. S. (2005) Sensitivity analysis of a catchment-scale nitrogen model. *J. Hydrol.* **315**, 71–92.

Meili, M., Bishop, K., Bringmark, L., Johansson, K, Munthe, J., Sverdrup, H. & de Vries, W. (2003) Critical levels of atmospheric pollution: criteria and concepts for operational modelling of mercury in forest and lake ecosystems. *Sci. Total Environ.* **304**, 83–106.

Miller, W. D. & Harding, L. W. (2007) Climate forcing of the spring bloom in the Chesapeake Bay. *Marine Ecology – Progress Series* **331**, 11–22.

Monteith, D. T., Evans, C. D. & Reynolds, B. (2000) Are temporal variations in the nitrate content of UK upland freshwaters linked to the North Atlantic Oscillation? *Hydrol. Processes* **14**, 1745–1749.

Monteith, D. T., Stoddard, J. L., Evans, C. D., de Wit, H., Forsius, M., Høgåsen, T., Wilander, A., Skjelkvåle B. L., Jeffries, D. S., Vuorenmaa, J., Keller, B., Kopácek, J. & Vesely, J. (2007) Rising freshwater dissolved organic carbon driven by changes in atmospheric deposition. *Nature* **450**, 537–540.

Mooij, W. M., Hulsmann, S., Domis, L. N. D., Nolet, B. A., Bodelier, P. L. E., Boers, P. C. M., Pires, L. M. D., Gons, H. J., Ibelings, B. W., Noordhuis, R., Portielje, R., Wolfstein, K. & Lammens, E. H. R. R. (2005) The impact of climate change on lakes in the Netherlands: a review. *Aquatic Ecology* **39**, 381–400.

Morris, B. L., Darling, W. G., Cronin, A. A., Rueedi, J., Whitehead, E. J. & Gooddy, D. C. (2006) Assessing the impact of modern recharge on a sandstone aquifer beneath a suburb of Doncaster, UK. *Hydrogeol. J.* **14**, 979–997.

Moss, B., McKee, D., Atkinson, D., Collings, S. E., Eaton, J. W., Gill, A. B., Harvey, I., Hatton, K., Heyes, T. & Wilson, D. (2003) How important is climate? Effects of warming, nutrient addition and fish on phytoplankton in shallow lake microcosms. *J. Appl. Ecology* **40**, 782–792.

Moss, B., Stephen, D., Balayla, D. M., Becares, E., Collings, S. E., Fernandez-Alaez, C., Fernandez-Alaez, M., Ferriol, C., Garcia, P., Goma, J., Gyllstrom., M., Hannson, L.-A., Hietala, J., Kairesalo, T., Miracle, R., Romo, S., Rueda, J., Russell, V., Stahl-Delbanco, A., Svensson, M., Vakkilainen, K., Valentini, M., Van den Bund, W. J., Van Donk, E., Vicente, E. & Villen, M. J. (2004) Coninental scale patterns of nutrient and fish effects on shallow lakes: synthesis of a pan-European mesocosm experiment. *Freshwater Biology* **49**, 1633–1649.

Munthe, J. (2008) Hydrological experiments at Gardsjon and release of methylmercury under wet conditions. Deliverable 107, Euro-limpacs Project Report, UVL, Sweden.

New, M. & Hulme, M. (2000) Representing uncertainties in climate change scenarios: a Monte Carlo approach. *Integrated Assessment* **1**, 203–213.

Oguchi, T., Jarvie, H. P. & Neal, C. (2000) River water quality in the Humber catchment: an introduction using GIS-based mapping and analysis. *Sci. Total Environ.* **251**, 9–26.

Orr, H. G. & Walsh, C. L. (2006) Incorporating climate change in channel typologies for the Water Framework Directive. Environment Agency Science Report SC030301/SR, Bristol, UK.

Osterkamp, S., Kraft, D. & Schirmer, M. (2001) Climate change and the ecology of the Weser estuary region: assessing the impact of an abrupt change in climate. *Climate Res.* **18**, 97–104.

Paul, M. L. & Meyer, J. L. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics* **32**, 333–365.

Preston, B. L. (2004) Observed winter warming of the Chesapeake Bay estuary 1949–2002: implications for ecosystem management. *Environ. Manage.* **34**, 125–139.

Romanowicz, R., Beven, K. , Wade, S. & Vidal, J. (2006) Effects of climate change on river flows and groundwater recharge, a practical methodology: Interim Report on Rainfall–Runoff Modelling, UKWIR Report CL/04, London, UK.

Rose, N. L., Backus, S., Karlsson, H. & Muir, D. C. G. (2001) An historical record of toxaphene and its congeners in a remote lake in Western Europe. *Environ. Sci. Technol.* **35**(7), 1312–1319.

Rose, N. L., Monteith, D. T., Kettle, H., Thompson, R., Yang, H. & Muir, D. (2004) A consideration of potential confounding factors limiting chemical and biological recovery in Lochnagar, a remote mountain loch in Scotland. *J. Limnol.* **63**(1), 63–76.

Rosseland, B-O., Rognerud, S., Collen, P., Grimalt, J. O., Vives, I., Massabuau, J.-C., Lackner, R., Hofer, R., Raddum, G. G., Fjellheim, A., Harriman, R. & Piña, B. (2007). Brown trout in Lochnagar: Population and contamination by metals and

organic micropollutants. In: *Lochnagar: The Natural History of a Mountain Lake* (ed, by N. L. Rose), Chapter 12, 253–285. Springer, Dordrecht, The Netherlands.

Rueedi, J., Cronin, A. A., Moon, B., Wolf, L. & Hoetl, H. (2005) Effect of different water management strategies on water and contaminant fluxes in Doncaster, United Kingdom. *Water Sci. Technol.* **52**, 115–123.

Ruth, M. & Coelho, D. (2007) Understanding and managing the complexity of urban systems under climate change. *Climate Policy* **7**, 317–336.

Scavia, D., Field, J. C., Boesch, D. F., Buddemeier, R. W., Burkett, V., Cayan, D. R., Fogarty, M., Hartwell, M. A., Howarth, R. W., Mason, C., Reed, D. J., Royer, T. C., Sallenger, A. H. & Titus, J. G. (2002) Climate change impacts on US coastal and marine ecosystems. *Estuaries* **25**, 149–164.

Schirmer, M. & Schuchardt, B. (2001) Assessing the impact of climate change on the Weser estuary region: an interdisciplinary approach. *Climate Res.* **18**, 133–140.

Schreider, S. Y., Smith, D. I. & Jakeman, A. J. (2000) Climate change impacts on urban flooding. *Climatic Change* **47**, 91–115.

Skeffington, R. A., Whitehead, P. G., Heywood, E., Hall, J. R., Wadsworth, R. A. & Reynolds, B. (2007) Estimating uncertainty in terrestrial critical loads and their exceedances at four sites in the UK. *Sci. Total Environ.* **382**, 199–213.

Strufy, E., Van Damme, S. & Meire, P. (2004) Possible effects of climate change on estuarine nutrient fluxes: a case study in the highly nutrified Schelde estuary, Belgium, The Netherlands. *Estuarine, Coastal and Shelf Sci.* **60**, 649–661.

Sündermann, J., Beddig, S., Huthnance, J. & Mooers, C. N. K. (2001) Impact of climate change on the coastal zone: discussion and conclusions. *Climate Research* **18**, 1–3.

Tipping, E., Smith, E. J., Lawlor, A. J., Hughes, S. & Stevens, P. A. (2003) Predicting the release of metals from ombrotrophic peat due to drought-induced acidification. *Environ. Pollut.* **123**, 239–253.

Van Doorslaer, W., Stoks, R., Jeppesen, E. & de Meester, L. (2007) Adaptive microevolutionary responses to simulated global warming in *Simocephalus vetulus*: a mesocosm study. *Global Change Biology* **13**(4), 878–886. doi:10.1111/j.1365-2486.2007.01317.x.

Verdonschot, P. F. M. (2000) Integrated ecological assessment methods as a basis for sustainable catchment management. *Hydrobiologia* **422/423**, 389–412

Vives, I., Grimalt, J. O., Catalan, J., Rosseland, B. O. & Battarbee, R. W. (2004a) Influence of altitude and age in the accumulation of organochlorine compounds in fish from high mountain lakes. *Environ. Sci. Technol.* **38,** 690–698.

Vives, I., Grimalt, J. O., Fernandez, P. & Rosseland, B. O. (2004b) Polycyclic aromatic hydrocarbons in fish from remote and high mountain lakes in Europe and Greenland. *Sci. Total Environ.* **324,** 67–77.

Vives, I., Grimalt, J. O., Ventura, M., Catalan, J. & Rosseland, B. O. (2005) Age dependence of the accumulation of organochlorine pollutants in brown trout *Salmo trutta* from a remote high mountain lake Redó, Pyrenees. *Environ. Pollut.* **133,** 343–350.

Wade, A. J., Durand, P., Beaujouan, V., Wessel, W. W., Raat, K. J., Whitehead, P. G., Butterfield, D., Rankinen, K. & Lepisto, A. (2002a) A nitrogen model for European catchments: INCA, new model structure & equations. *Hydrol. Earth System Sci.* **6**, 559–582.

Wade, A. J., Whitehead, P. G., Hornberger, G. M. & Snook, D. (2002b) On modelling the flow controls on macrophytes and epiphyte dynamics in a lowland permeable catchment: the River Kennet, southern England. *Sci. Total Environ.* **282–283**, 395–417.

Whitehead, P. G. & Hornberger, G. E. (1984) Modelling algal behaviour in the River Thames. *Water Res.* **18**, 945–953.

Whitehead, P. G. & Williams, R. J. (1982) Operational management of water quality in river systems. In: *Effects of Waste Disposal on Groundwater and Surface Water* (Proc. Exeter Conf., July 1982) (ed. by R. Perry), 163–169. IAHS Publ. 139. IAHS Press, Wallingford, UK.

Whitehead, P. G., Barlow, J., Haworth, E. Y. & Adamson, J. K. (1997) Acidification in three Lake District tarns: historical long term trends and modelled future behaviour under changing sulphate and nitrate deposition. *Hydrol. Earth System Sci.* **1**, 197–204.

Whitehead, P. G., Wilson, E. J. & Butterfield, D. (1998a) A semi-distributed integrated nitrogen model for multiple source in catchments INCA.: Part I – model structure and process equations. *Sci. Total Environ.* **210/211**, 547–558.

Whitehead, P. G., Wilson, E. J., Butterfield, D. & Seed, K. (1998b) A semi-distributed integrated flow and nitrogen model for multiple source assessment in catchments, INCA. Part II, Application to large River Basins in South Wales and Eastern England. *Sci. Total Environ.* **210/211**, 559–583.

Whitehead, P. G., Wilby, R. L. & Futter, M. (2006a) Impacts of climate change on hydrology, nitrogen and carbon in upland and lowland streams: assessment of adaptation strategies to meet Water Framework Directive objectives. Proceedings Durham Meeting, British Hydrological Society, UK.

Whitehead, P. G., Wilby, R. L., Butterfield, D. & Wade, A. J. (2006b) Impacts of climate change on nitrogen in lowland chalk streams: adaptation strategies to minimise impacts. *Sci. Total Environ.* **365**, 260–273.

Wilby, R. L. (1993) The influence of variable weather patterns on river water quantity and quality regimes. *Int. J. Climatol.* **13**, 447–459.

Wilby, R. L. (1994) Exceptional weather in the Midlands, UK during 1988–1990 results in the rapid acidification of an upland stream. *Environ. Pollut.* **86**, 15–19.

Wilby, R. L. (2005) Uncertainty in water resource model parameters used for climate change impact assessment. *Hydrol. Processes* **19**, 3201–3219.

Wilby, R. L. (2007) A review of climate change impacts on the built environment. *Built Environ. J.* **33**, 31–45.

Wilby, R. L. (2008) Constructing climate change scenarios of urban heat island intensity and air quality. *Environ. & Planning B: Planning and Design* **35**, 909–919.

Wilby, R. L., Dalgleish, H. Y. & Foster, I. D. L. (1997) The impact of weather patterns on contemporary and historic catchment sediment yields. *Earth Surf. Processes Landf.* **22**, 353–363.

Wilby, R. L., Orr, H. G., Hedger, M., Forrow, D. & Blackmore, M. (2006a) Risks posed by climate change to delivery of Water Framework Directive objectives. *Environ. Int.*, **32**, 1043–1055.

Wilby, R. L., Whitehead, P. G., Wade, A. J., Butterfield, D., Davis, R. J. & Watts, G. (2006b) Integrated modelling of climate change impacts on water resources and quality in a lowland catchment: River Kennet, UK. *J. Hydrol.* **330**(1/2), 204–220.

Williams, R. J., White, C., Harrow, M. & Neal, C. (2000) Temporal and small-scale spatial variations of dissolved oxygen in the rivers Thames, Pang and Kennet. *Sci. Total Environ.* **251/252**, 497–512.

Worrall, F., Swank, W. T. & Burt, T. P. (2003) Changes in stream nitrate concentrations due to land management practices, ecological succession, and climate: developing a systems approach to integrated catchment response. *Water Resour. Res.* **39**, 1177.

Worrall, F., Burt, B. & Adamson, A. (2004) Can climate change explain increases in DOC flux from upland peat catchments? *Sci. Total Environ.* **326**, 95–112.

Wright, J. F., Cameron, C., Hiley, P. D. & Berrie, A. D. (1982) Seasonal changes in biomass of macrophytes on shaded and unshaded sections of the River Lambourn, England. *Freshwater Biology* **12**, 271–283.

Wright, R. F. (1998) Effect of increased CO_2 and temperature on runoff chemistry at a forested catchment in southern Norway CLIMEX project. *Ecosystems* **1**, 216–225.

Wright, R. F. (2008) The decreasing importance of acidification episodes with recovery from acidification: an analysis of the 30-year record from Birkenes, Norway. *Hydrol. Earth System Sci.* **12**, 353–362.

Wright, R. F. & Jenkins, A. (2001) Climate change as a confounding factor in reversibility of acidification: RAIN and CLIMEX projects. *Hydrol. Earth System Sci.* **5**, 477–486.

Wright, R. F., Emmett, B. A. & Jenkins, A. (1998) Acid deposition, land-use change and global change: MAGIC7 model applied to Aber, UK NITREX project. and Risdalsheia, Norway RAIN and CLIMEX projects. *Hydrol. Earth System Sci.* **2**, 385–398.

Wright, R. F., Larssen, T., Camarero, L., Cosby, B. J., Ferrier, R. C., Helliwell, R. C., Forsius, M., Jenkins, A., Kopácek, J., Majer, V., Moldan, F., Posch, M., Rogora, M. & Schöpp, W. (2005) Recovery of acidified European surface waters. *Environ. Sci. Technol.* **39**, 64A–72A.

Wright, R. F., Aherne, J., Bishop, K., Camarero, L., Cosby, B. J., Erlandsson, M., Evans, C. D., Forsius, M., Hardekopf, D. W., Helliwell, R., Hruška, J., Jenkins, A., Kopáček, J., Moldan, F., Posch, M. & Rogora, M. (2006) Modelling the effect of climate change on recovery of acidified freshwaters: relative sensitivity of individual processes in the MAGIC model. *Sci. Total Environ.* **365**, 154–166.

Yang, H. & Rose, N. L. (2003) Distribution of mercury in six lake sediment cores across the UK. *Sci. Total Environ.* **304**(1/3), 391–404.

Yang, H., Rose, N. L., Battarbee, R. W. & Boyle, J. J. (2002) Mercury and lead budgets for Lochnagar, a Scottish mountain lake and its catchment. *Environ. Sci. Technol.* **36**(7), 1383–1388.

APPENDIX

CLIMATE CHANGE AND FRESHWATERS— KNOWLEDGE AND INFORMATION GAPS

One of the outputs of the Euro-limpacs project has been the identification of knowledge gaps. The following questions form the basis of the Euro-limpacs science programme and are at the heart of the current international research agenda that aims to improve understanding of the potential effects of climate change on freshwater quality. Such an understanding is needed to identify the adaptive actions human society might need to take to avoid the unwanted consequences of climate change.

Direct effects of climate change on freshwaters

− What will the effects of lower flows be on pollutant concentrations in rivers?

− What are the potential impacts of changing climate on catchment chemical fluxes?

− How will a changing discharge regime affect stream ecology?

− How do changing air temperature and precipitation impact on discharge patterns in glacierised and non-glacierised upland river basins?

− How have freshwater biological communities responded to natural climate variability in the past?

− What effects will changing temperature and wind patterns have on the structure of lake water columns?

− How will changes in ice-cover duration, stratification and mixing regime affect lake biota?

− How will climate change affect the hydrology of marginal wetlands?

− What effects will changing hydrology and biogeochemical processes have on plant communities, nutrient dynamics and productivity in marginal wetlands?

− How will climate change affect the quality and quantity of dissolved organic carbon release from soils?

–   What was the amplitude of natural variability of dissolved organic carbon concentrations in surface waters prior to any impacts from greenhouse gas-forced climate change?

Interactions between climate and hydromorphological/land-use change

–   How do climate, hydrology, land use and morphology interact in space and time?
–   How do these interactions affect aquatic ecosystems at the catchment scale?
–   What effect will changing hydrological conditions (both directly and through morphological change) have on stream aquatic communities at the habitat scale?
–   How will climate change affect mountain stream restoration?
–   How will climate change affect channel morphology and stability in meandering lowland streams?

Climate change and eutrophication

–   How will increasing temperatures and nutrient loading affect food-web relationships?
–   How can climate and nutrient-induced changes in food-web structure be disentangled?
–   How can food-web relationships be reconstructed using stable isotope techniques?
–   Do nutrients structure ecosystems in different ways in different climates?
–   How will climate change affect turbid phytoplankton-dominated lake ecosystems?
–   What effects will increasing temperature have on the functioning of littoral wetlands?
–   Are the predicted changes in temperature and nutrient dynamics comparable in amplitude with those currently recorded from the existing climate gradients?
–   Can palaeolimnological techniques be applied to wetlands to examine past variations in hydrology and ecology?

Climate change and acidification

–   Will changes in episodic and seasonal climatic events lead to increases in the magnitude and frequency of acid pulses in sensitive streams?
–   What are the likely ecological effects of changes in the magnitude and frequency of extreme flows?
–   How will climate change affect the setting of chemical and biological targets for surface waters recovering from acidification?
–   How can dynamic models be used to simulate scenarios combining future climate change with future acid deposition?

Climate change and toxic substances

–   How will climate change affect the loading of toxic substances to headwater systems?
–   What effects will temperature change have on the redistribution and uptake of persistent organic pollutants?
–   Will increases in precipitation enhance mobilisation of mercury and methyl mercury in soils?
–   Will climate change lead to a remobilisation of accumulated heavy metals and persistent organic pollutants from polluted soils, and their subsequent transportation into aquatic ecosystems?
–   How will changes in river discharge affect trace metal remobilisation from flood-plain sediments?

Modelling climate change effects on surface water

–   How can the impacts of climate change, land-use change and pollution be evaluated using a modelling approach?

–   How can component models be used to assess the likely affects of climate change on freshwater systems?
–   How can the uncertainty associated with component models be quantified?
–   How can socio-economic scenarios be incorporated into modelling assessments of climate change effects?
–   How can the spatial and temporal variation in the factors and processes controlling pollutant behaviour in coupled wetland–lake–river systems be simulated using models?
–   How can models be best used to assist to manage the impacts of climate change on freshwaters?

Indicators of ecosystem health
–   What chemical parameters are best suited as indicators of climate change?
–   How can functional indicators be identified to address climate change impacts on wetlands, rivers and lakes?
–   How can biological indicators of climate change be identified and can these be used to assess the response of communities to change?
–   How can the different indicator types be linked to provide a common framework for rivers, lakes and wetlands?
–   How can existing assessment and prediction methods for European freshwater systems be expanded and modified to address climate change?

Reference conditions and restoration strategies
–   What reference conditions can be ascribed to different freshwater ecosystem types across Europe?
–   How comparable are the different methods commonly used to establish reference conditions?
–   What are the errors associated with methods used to establish reference conditions?
–   How can reference conditions be used to establish restoration targets?
–   To what extent is climate change already affecting restoration success?
–   How might climate change affect both natural and human-induced ecosystem recovery?

Policy and management
–   How do current policies, protocols and socio-economic pressures influence the drivers of change on freshwater ecosystems?
–   What impact will future climate policies have on emissions and deposition of atmospheric pollutants?
–   What tools are available for policy makers and managers for planning at the catchment scale?
–   How best can stakeholders become involved in the development of tools for catchment management?
–   What kinds of socio-economic tools are needed to aid decision making?