Project no. GOCE-CT-2003-505540

Project acronym: Euro-limpacs

Project full name: Integrated Project to evaluate the Impacts of Global Change on European Freshwater Ecosystems

Instrument type: Integrated Project

Priority name: Sustainable Development

Deliverable No. 144

Euro-limpacs Chapter in *The Water Framework Directive – Ecological and Chemical Status Monitoring*

(changed from ‘Submission of articles to Commission for use in RTD series’)

Due date of deliverable: Month 30
Actual submission date: Month 48

Start date of project: 1 February 2004  
Duration: 5 Years

Organisation name of lead contractor for this deliverable: UCL

<table>
<thead>
<tr>
<th>Dissemination Level (tick appropriate box)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>PU</strong></td>
</tr>
<tr>
<td><strong>PP</strong></td>
</tr>
<tr>
<td><strong>RE</strong></td>
</tr>
<tr>
<td><strong>CO</strong></td>
</tr>
</tbody>
</table>

Revision: Final
This deliverable comprises a draft chapter, co-authored by Euro-impacs work package leaders entitled ‘Freshwater ecosystem responses to climate change: the Euro-impacs project’. This has been submitted for inclusion in the book ‘The Water Framework Directive - Ecological and Chemical Status Monitoring’, edited by Philippe Quevauviller, Ulrich Borchers, Clive Thompson & Tristan Simonart to be published by Wiley as part of the “Water Quality Measurements” Series. This version is currently in revision.
Introduction

Although GCMs vary in their projection of future climate change all are in agreement that significant warming will occur within this century, principally as
a result of a continued rise in the concentration of greenhouse gases, especially carbon dioxide (IPCC 2007).

Against this concern it is important to examine the potential effects of future climate change on the functioning of the earth system. Here we consider the potential impact of climate change on freshwater ecosystems (streams, lakes and wetlands) using in particular recent results from the EU-funded project “Euro-limpacs: global change impacts on European freshwater ecosystems” (http://www.eurolimpacs.ucl.ac.uk). The principal questions concern how climate change might affect the structure and function of freshwater ecosystems, how climate change might interact with concurrent trends in pollutant loading and land-use, and what might be the implications of the projected responses for environmental policy and management.

Addressing such questions requires a large-scale collaborative and integrated research programme employing a range of complementary methodologies. Euro-limpacs combines:

(i) experiments in the laboratory and field under controlled climate conditions using realistic predictions of future climate (over the next 50 to 100 years);
(ii) analysis of long-term observational physical, chemical and biological records to identify current trends, assess the role of climate change in explaining ecosystem change over recent decades and provide calibration and/or verification time-series for model output;
(iii) palaeolimnological reconstructions to extend time-series, assess natural variability, and identify processes acting over longer (decadal) time-scales;
(iv) space-for-time substitution to enable data from sites and regions with warmer climates today to be used to make predictions for cooler systems; and
(v) process-based modelling to explore mechanisms and run scenarios based on catchment-scale projections of climate, land-use and pollution change.

In this chapter we summarise some of the results obtained to date from the Euro-limpacs project and draw upon the wider literature to illustrate changes that have taken place or might take place in the future as a result of global warming. First we consider some of the direct consequences of climate change on lakes, streams and wetlands. We then describe research designed to assess the interactions between climate warming and other stressors (hydromorphology, eutrophication, acidification, and toxic substances). Finally we consider the relevance of the results for policy making and for the management of freshwaters and describe modelling and other tools that need to be developed to aid decision making.
2. Direct impacts of climate change on aquatic systems

Projected changes in temperature together with associated changes in precipitation and pressure systems will have far-reaching direct impacts on the physical, chemical and biological characteristics of many if not all freshwater ecosystems in Europe and the wider world.

Long-term time-series from freshwaters already show clear evidence for change associated with warming, especially increasing stream and lake surface water temperatures (Fang & Stefan 1999, Livingstone 2003), hypolimnetic warming in large lakes (Dokulil et al. 2006), decreasing ice-cover in northern and high altitude lakes (Magnusson et al. 2000), changes in seasonality of phytoplankton (Catalan et al. 2002), an extension to growing seasons and increases in lake productivity (Blenckner et al. 2007) changes in the geographical range of taxa e.g. Odonota (Hickling et al. 2005, Brooks et al. 2007) and threats to cold stenothermal taxa (e.g. Griffiths 2007). In addition recent results from the EU CLIME project have shown how the extremely hot summer of 2003 caused significant changes in the temperature, stratification and hypolimnetic oxygen concentration of deep lakes in Switzerland (Jankowski et al. 2006), and hot summers are likely to be more frequent in future particularly over central Europe (IPCC 2007).

In Euro-impacts there are many studies involving the direct impact of climate change on lakes, rivers and wetlands (see www.eurolimpacs.ucl.ac.uk). Here we present three examples with respect to surface water temperature, the melting of rock glaciers and the potential impact of increased wind stress on lakes.

2.1 Surface water temperature trends

Some of the best long-term temperature data-sets are for the Swiss rivers and lakes. Hari et al. (2006) have collated and analysed data for river and stream temperatures measured at 25 stations in Switzerland. They show annual running mean temperatures based on daily means calculated from the original measurements made at intervals of 1 minute. The river temperatures show a high degree of regional coherence on inter-annual and inter-decadal time-scales, implying a common, coherent response to regional climatic forcing on these time-scales. The absolute differences in water temperature from river to river are primarily a result of the general decrease in water temperature that occurs with increasing altitude, but the degree of coherence also decreases somewhat as the mean altitude of the catchment area of the sampling station increases (and is disproportionately low under the influence of glaciers or hydro-electric power stations). On time-scales exceeding the inter-decadal time-scale, a coherent warming can be seen to have occurred at all altitudes. This coherent warming reflects a corresponding long-term increase in regional air temperature. Much of the long-term water temperature increase occurred as an abrupt increase between two approximately stationary periods from 1978-1987 and 1988-2002. This abrupt shift reflects a similar shift in air
temperature and appears to be related ultimately to a shift in the North Atlantic Oscillation.

2.2. Hydrochemical response to the melting of rock glaciers

Chemical changes in rivers and lakes are more difficult to relate to climate warming than physical changes. For example, the rise in DOC observed from sites across Europe and parts of North America ascribed to climate change (Freeman et al. 2001) has now been shown to be associated with recovery of acidified sites from acidification (Monteith et al. 2007). However, some long-term chemical trends are difficult to explain except as a result of warming. An example of this in the Euro-impacts project is the work of Thies et al. (2007) in the Central European Alps.

Remote high altitude lakes are sensitive indicators of environmental and climate change. In particular, a substantial rise in solute concentration has been observed at Rasass See (2682 m, Italy), a high alpine lake in a catchment of metamorphic rocks. During the past two decades conductivity has increased by a factor of 18, while the most abundant ions, magnesium, sulfate and calcium, have reached 68-fold, 26- and 13-fold concentrations, respectively (Fig. 1) (Thies et al., 2007). The pronounced change in lake water composition is most likely caused by solute release from rock glacier outflows draining into the lake. This effect is expected to intensify with increasing air temperature and subsequent enhanced melt processes. In addition to major ions, unexpectedly high nickel concentrations (243 µg l$^{-1}$) have been found recently in Rasass See. The values exceed the limit for drinking water by one order of magnitude and cannot be related to catchment geology leaving the source of nickel still unclear.

Similar but less intense processes have been observed at Schwarzsee (2796 m, Austria), where electrical conductivity rose by a factor of three during the past two decades (Fig. 1) and nickel concentrations were just above the detection limit. This is attributed to the lower impact of glacial melt-water compared to Rasass See, where the area of active rock glaciers is larger and situated at a lower elevation.

There are still few data available from solute concentrations in outflows of active rock glaciers and their potential impact on freshwater systems. Rock glaciers are widespread in high mountains around the world (Humlum, 1998) and as global climate models predict a continuous rise in air temperature until the end of this century, high mountain freshwaters may become increasingly affected by solute release from active rock glaciers.

2.3 Increased wind stress on lakes

In the climate change debate most attention is given to the probable impacts of future change in temperature and precipitation. However, lakes are also sensitive to changes in wind conditions. In response to the predicted increase in mean geostrophic winds in northern Europe as a result of the northward
shift of cyclone activity a whole–lake mixing experiment is being carried out in Euro-limpacs to assess lake response to increased input of mixing energy. In this experiment which uses a submerged propeller to effect the mixing the focus is on thermocline manipulation, biogeochemical cycling (including Hg) and physical modelling using the myLake model (Saloranta & Andersen 2007). The results to date show that the impact of experimental mixing on Halsjärvi causes a depression of the thermocline to the extent that might be expected as a result of increased wind stress. The consequences of such a deepened thermocline include an increase in the volume of the mixed epilimnion, a decrease in the area of anoxic surface sediments, a longer ice-free season and an increase in the resuspension of littoral sediments.

3. Interaction between climate change and the hydromorphology of streams and rivers

In many parts of Europe hydromorphological alteration is the main stressor affecting rivers. Alterations include channel straightening, dam construction, disconnection of the river from its floodplain and destruction of riparian vegetation. Future climate change will introduce further stresses on channel hydromorphology including the combined effect of changes in precipitation and climate-induced changes in land-use patterns. These in turn may cause changes in catchment hydrology that will affect sediment transport and channel morphology, inundation frequency and extent, altering river ecosystems at both catchment and habitat scale (Verdonschot 2000).

The changes that may occur may not always be negative for biodiversity. As illustrated in Figure 2 we can hypothesise that climate change may cause biodiversity loss through intensification of land-use or through a more variable discharge regime. Alternatively improvements could occur if as a result of increased flooding human activity is withdrawn from floodplains generating near-natural habitat structures.

The Euro-limpacs project includes studies of the relationships between climate change, flow regime, channel morphology and the distribution and biodiversity of taxa in different hydromorphological conditions at sites across Europe. In particular it focuses on mountain streams comparing braided and non-braided reaches and meandering lowlands streams comparing catchments spanning the north-south and west–east climate gradient in Europe.

3.1 Climate induced changes in flow regimes

Climate change alters the dominant pattern of precipitation which in its turn changes run-off, discharge regimes and hydraulic conditions in streams. There has been considerable previous work during the 1970s and late 1990s assessing impacts of extreme flows on the biotic communities in the Lambourn chalk stream (e.g. Wright et al. 1981, Wright & Symes 1999, Wright et al. 2004). During both periods the river experienced sustained low-flow
episodes (1976, 1997) and during the latter period there was also a phase of prolonged exceptionally high flows (2000/01).

About 90% of the R. Lambourn discharge is from the groundwater and only 10% is from direct run-off. This means that the flow regime is relatively predictable and the catchment rarely, if ever, experiences spate flooding. The crucial phase in the annual hydrograph is the winter/spring recharge of the aquifer. A particularly dry winter, e.g. 2003/04, leads to prolonged low flow conditions through the following summer, with consequences for the in-stream flora and fauna. In drought years the usually dominant macrophyte *Ranunculus* tends to be restricted due to smothering by epiphytic algae and silt. *Callitriche* and other marginal emergent vegetation replaced the *Ranunculus* as the dominant vegetation under such conditions. Drought events appeared to have a more deleterious impact on macroinvertebrates than high flow events but in both cases the River Lambourn communities recovered within a year or less (Wright et al. 2004).

Overall macroinvertebrate taxon richness tended to be greater in high-flow years. Many of these responses were influenced by changes in substrate composition and coverage of macrophyte beds in the river. *Ranunculus* thrives in sustained high flow conditions, and is capable of achieving a large biomass in the channel which provides excellent habitats and refuges for invertebrates, and also traps and supports food resources. *Ranunculus* is generally a favoured habitat for Gammaridae, Baetidae, Ephemerellidae, Simuliidae, Rhyacophilidae and Hydropsychidae. Large macrophyte beds in the channel also increases the probability of exceeding bank-full discharge and flooding of adjacent land. This in turn dissipates the damaging energy of high flows, protecting the stream biota.

The key factor however is whether the magnitude, frequency and timing of extreme discharge events in the R. Lambourn are likely to change under future climate change scenarios. Further research within Euro-limpacs will try to answer this question.

### 3.2. Climate induced changes in mountain streams

A potential consequence of increased flood intensity in mountain streams is the removal of sediment stored in flood plains and the development of multiple-channel streams. To assess whether an increase in multiple-channel reaches leads to an increase in habitat and in species diversity single- and multiple-channel sections of several streams have been studied. In a paired-site study of mountainous streams in Germany seven multiple- and seven single-channel sites (14 sections) were compared. The hypothesis that multiple-channel sections have larger and more diverse habitats and more diverse biota was tested. Various hydromorphological parameters (shoreline length, channel features, current velocity, water depth, substrate) were recorded and data for floodplain vegetation, riparian ground beetles (Carabidae) and benthic macroinvertebrates were collected. Table 1 summarises the results. The hydromorphological diversity of the multiple-channel sections is higher and is closer to the reference condition (Jähnig et
al., 2007a). Habitat diversity is also higher and cross-sectional changes in stream sediments are more dynamic but the effects of either substrate or the overall stream sections on the macroinvertebrate community are not detectable. At best they display a general tendency towards improvement (Jähnig et al., 2007b). However, taxonomic richness is clearly increased in floodplain vegetation and riparian ground beetles, with floodplain vegetation reacting most strongly (Hering et al., in prep.). The stronger reaction of floodplain vegetation and ground beetles is mainly due to the generation of several additional habitats, such as gravel bars. Reasons for the close similarity of macroinvertebrate communities from single- or multiple-channel sections include the influence of large-scale catchment pressures, the relatively short length of restored sections and lack of potential re-colonisers. Other reasons explaining the different responses between organism groups include differences in dispersal abilities and source populations. Despite the mixed effects observed on organisms, the higher habitat diversity and the more dynamic hydromorphological environment of the multiple channel sections are likely to have positive impacts on ecological quality.

Table 1: Comparison of hydromorphological und biological indices in single- and multiple-channel sections. * = U-Test significant p < 0.05.

<table>
<thead>
<tr>
<th>Number / Share</th>
<th>Median single</th>
<th>Median multiple</th>
<th>Minimum single</th>
<th>Minimum multiple</th>
<th>Maximum single</th>
<th>Maximum multiple</th>
</tr>
</thead>
<tbody>
<tr>
<td>Meso habitats (channel features) *</td>
<td>3</td>
<td>9</td>
<td>2</td>
<td>8</td>
<td>7</td>
<td>12</td>
</tr>
<tr>
<td>Micro habitats (substrate)</td>
<td>10</td>
<td>10</td>
<td>8</td>
<td>10</td>
<td>11</td>
<td>11</td>
</tr>
<tr>
<td>Vegetation units *</td>
<td>4</td>
<td>6</td>
<td>2</td>
<td>5</td>
<td>6</td>
<td>7</td>
</tr>
<tr>
<td>Floodplain vegetation species *</td>
<td>80</td>
<td>125</td>
<td>60</td>
<td>73</td>
<td>101</td>
<td>173</td>
</tr>
<tr>
<td>Floodplain vegetation genus *</td>
<td>62</td>
<td>86</td>
<td>47</td>
<td>59</td>
<td>71</td>
<td>115</td>
</tr>
<tr>
<td>Floodplain vegetation families *</td>
<td>28</td>
<td>35</td>
<td>25</td>
<td>32</td>
<td>35</td>
<td>41</td>
</tr>
<tr>
<td>Carabidae species</td>
<td>5</td>
<td>12</td>
<td>2</td>
<td>9</td>
<td>17</td>
<td>15</td>
</tr>
<tr>
<td>Carabidae genus</td>
<td>4</td>
<td>7</td>
<td>2</td>
<td>6</td>
<td>10</td>
<td>8</td>
</tr>
<tr>
<td>Riparian Carabidae (%) *</td>
<td>29</td>
<td>75</td>
<td>7</td>
<td>67</td>
<td>51</td>
<td>95</td>
</tr>
<tr>
<td>Macroinvertebrate species</td>
<td>91</td>
<td>96</td>
<td>77</td>
<td>79</td>
<td>111</td>
<td>111</td>
</tr>
<tr>
<td>Macroinvertebrate genus</td>
<td>68</td>
<td>69</td>
<td>57</td>
<td>60</td>
<td>83</td>
<td>83</td>
</tr>
<tr>
<td>Macroinvertebrate families</td>
<td>44</td>
<td>46</td>
<td>40</td>
<td>42</td>
<td>53</td>
<td>53</td>
</tr>
</tbody>
</table>

3.3 Influence of climate change on meandering lowland streams

A potential consequence of increased precipitation intensity in lowland streams is the increase in hydromorphological dynamics. An increase in flood frequency and intensity will cause an increase in flow regime dynamics and in a higher streambed instability. To assess whether a more dynamic discharge regime leads to a higher bed instability and to an in-stream biodiversity loss macroinvertebrate communities between natural, semi-natural and canalised stream sections in middle course of a Dutch lowland stream were compared. A
total of 51 samples collected between 1981 and 2006, of which 35 were from the natural, 7 from the semi-natural and 9 from the canalised stream section, were analysed. In 2004 a hydrological restoration took place in a branch of this stream causing a hydrological stabilisation downstream of the junction. A total of 238 macro-invertebrate taxa occurred in the samples.

The ordination technique Detrended Canonical Correspondence Analysis (CANOCO 4.5 for Windows; Ter Braak & Smilauer 2002) was used successively to compare the macroinvertebrate communities at the three hydromorphologically different stream sections (natural, semi-natural and canalised morphology) and the two hydrologically different periods in time (dynamic and ‘stable’ hydrological regime). The data (Fig. 3) show that there was a strong effect of morphology and a lesser effect of hydrology upon the in-stream macro-invertebrate assemblage.

4.0 Interactions between climate change and eutrophication

There have been major successes in Europe in reducing eutrophication problems but eutrophication still remains a most serious environmental problem, especially for shallow lakes and sites where diffuse pollution sources dominate nutrient loading. Eutrophication, like climate change, affects the whole of an ecological system. Nutrient effects pervade all the trophic levels, just as temperature change affects the metabolic rates of all organisms and hence growth, reproduction, respiration and nutrient recycling. Likewise climate change will affect ecosystem processes as well as potentially eliminating species whose niche limits are exceeded. In systems with hundreds of components and many interacting processes, predicting the consequences of simultaneous changes in climate, nutrient loading and biodiversity may well be impossible in any precise way.

The Euro-limpacs project focuses especially on eutrophication in shallow lakes and streams. It includes (i) attempts to find analogues of how shallow lake and stream systems function across latitudes from the warm temperate to the arctic; (ii) experiments on parallel natural stream and wetland systems in Iceland that differ in temperature within a small area; (iii) artificial tank systems simulating shallow lakes to investigate changes in ecosystem processes, and (iv) analysis of long-term data sets and palaeoecological reconstruction to separate the effects of nutrient change and climate change. This latter task is particularly difficult although there have been some successes with the simpler of sub-systems such as the open-water plankton. Multivariate correlation analysis of data from Loch Leven, Scotland (L. May, unpublished data), has demonstrated some changes in diatom community composition that may be more reliably attributed to a warming trend than to the reduction in phosphorus loading to the lake.

4.1 Geographical comparisons
Lake systems differ in general ways with latitude though exact analogues among future systems and current systems may not exist (Fox, 2007). The warmer systems of the tropics and Mediterranean-type warm-temperate zones have long growing-seasons and have not been disrupted so completely by the last glacial period than systems in the glaciated temperate and polar regions. Warm-zone lakes have a more species-rich fish fauna that tends to combine omnivory with high fecundity and frequent reproduction (Meerhof et al., 2007a,b) compared with cold temperate lakes.

Processes involving fish operating through the food-web are very important in determining how shallow lakes function (Scheffer et al., 1993). In warmer, lower latitudes fish predation on zooplankton is very intense, with the result that the zooplankton community is depauperate in those large species and individuals that are associated with efficient grazing on phytoplankton. Waters thus tend to be more turbid and more frequently dominated by algae such as Cyanobacteria with high temperature optima for growth. Increased nutrient load thus is associated strongly with increased algal crops and there is a greater risk of loss of littoral plant-dominated communities and their associated biodiversity (Moss et al. 2004; Gyllstrom et al. 2005).

Further north, there are shorter growing seasons, lower temperatures and a more limited fish fauna, still in the process of recolonising from southern refuges following the last glaciation. A mismatch in the earlier seasonal growth of zooplankton and the slightly later hatch of zooplanktivorous fish often leads to intense grazing on algae in the spring by a zooplankton community then untrammelled by fish predation. In turn this creates clear water and allows early growth and development of plant communities. The established plant community can then survive the potentially increased growth of algae in summer through provision of refuges for zooplankton grazers against the increasing threat of predation by young-of-the-year fish.

Northwards into the arctic regions (Jeppesen et al., 2001) the influence of fish further decreases as growth seasons and fish reproduction become much reduced, and fish faunas are even poorer in species. Large zooplankters are able to keep waters clear, helped also by lower nutrient loading from the catchment in the often-drier climates of the tundra and boreal forest regions. Euro-impacts has been creating the evidence for these latitudinal scenarios using food-web studies involving stable isotope measurements and experiments.

4.2 Evidence from experiments

Experimental studies are still very few (Liboriussen et al. 2005; McKee et al. 2002, 2003). They suffer from the inherent problem that an experiment set up with controlled treatments of water chemistry, sediment, temperature and nutrient loading may give divergent results in its replicates, or when carried out on different occasions, because of the consequences in a complex system of very small random differences in starting conditions and subsequent weather. Some general lessons are emerging, however. Growing seasons are extended by increased temperature, the timing of growth peaks of algae and
zooplankton are altered. The reduced oxygen concentrations consequent on increased temperature, as well as increased respiration of biomass as nutrient levels are increased, may lead to fish deaths even of quite tolerant species. Figure 4 shows recent results from a UK Euro-limpacs mesocosm experiment. The experimental tanks contained a well-developed aquatic plant community and were rich in phosphorus. In a randomised block design, tanks were either heated by 4°C above ambient and given one of three levels of nitrogen fertilisation. All tanks were initially stocked with a resilient fish, the threespined stickleback, *Gasterosteus aculeatus*. With a 4°C rise, there was a significant fall in both maximum and minimum oxygen concentrations recorded during three 24-hr sets of measurements in late June and July 2007. Nutrient addition also resulted in reduction in both maxima and minima. With either treatment, concentrations fell dangerously low during the night and in the warmed treatments, fish populations did not survive. Plant communities were also affected by warming. Exotic species, usually introduced from warmer regions, where there is a greater biodiversity, tend to proliferate over native species. Among native species, nutrients and warming together may lead to the predominance of floating plants, like lemns (duckweeds) and warming by 4°C has been shown to increase the whole community respiration to total gross photosynthesis ratio of the systems by about 17% (D. Atkinson, B. Moss, C. Whitham, R. Moran & H. Feuchtmayr, unpublished data). This has major implications for positive feedback effects on carbon dioxide concentration in the atmosphere because of its relevance to the vast areas of wetlands in northern North America and Eurasia.

**4.3. Future projections**

We can combine these generalisations into predicted scenarios, at present for lakes, and eventually for streams as Euro-limpacs data accumulate. We assume in Europe a future of increased temperature, reduced summer rainfall (IPCC, 2007) with nutrient additions to lakes most likely to be maintained as human populations and dietary aspirations increase, despite the best efforts at control under the Water Framework Directive. Experience has shown that many years are likely to elapse before the effects of nutrient control become significant.

Growing seasons will lengthen, algal biomass will be encouraged by warm, sunny weather, hypolimnetic oxygen conditions may deteriorate, and cyanobacterial blooms may become more extensive. For shallow lakes food chain effects may exacerbate these problems, especially by a shift to more intense fish predation and reduced zooplankton populations in a warmer climate. Fish kills will, however, become more frequent followed by a temporary abatement of the symptoms of eutrophication caused by build up of zooplankton biomass. Exotic warm water species of fish and water plants will proliferate with vigorous growth that will result in near monocultures, especially where fish kills allow greater zooplankton populations to persist. Where the exotic, thermally well-adapted common carp (*Cyprinus carpio*) has been introduced, we might expect it to reproduce effectively at northern latitudes where presently it often fails. Carp cause increased damage to plant
communities, associated with more turbid water, a consequence of both disturbance of sediment as it feeds and of its mobilisation of sediment phosphorus into the water (Moss et al. 2002).

On mainland Europe we might expect the steady movement northwards of warm-adapted systems, but almost certainly at rates far slower than the wave of temperature change. On islands like the UK, where immigration of new species is barred by the sea this will not occur and fish kills will remove many native species from all but the northerly regions. However, there is a counter-risk that commercial pressures will result in introduction of warm water species of plants and especially of fish, for the water garden and angling trades. Such species tend to be vigorous and tolerant of a wide range of conditions and may displace native species that have survived the warming.

One possible mitigating factor is that evolutionary change, in the smaller organisms at least, such as certain species of zooplankton, has been shown by work in Euro-limpacs (Van Doorslaer et al. 2007) to be quite rapid and thus existing systems might maintain some of their current structure and functioning as temperatures increase. This would, however, demand a rapid and synchronous set of evolutionary changes in a large number of different organisms, and this seems inherently unlikely on current ecological experience.

Effects on rivers and streams are even more difficult to predict. Flowing waters are subject to eutrophication no less than lakes but their systems are driven much more by hydrological changes and the physical structures of their systems (see above) and only secondarily by nutrient supply. Fish predation is also less powerful in structuring the communities though it certainly has an influence. Nonetheless, large beds of weed in lowland rivers and algal growths in slowly flowing stretches of floodplain rivers are greatly influenced by nutrient supply and fish communities have similar effects to those in lakes. Only in the highly erosive, turbulent waters of the hills is it likely that combinations of temperature increase and current or changed nutrient load will not have many consequences for users of the water. Eutrophication still remains Europe’s most serious freshwater problem and in almost all situations, we expect climate change to make it a more difficult problem to control.

5. Interactions between climate change and acidification

Acid-sensitive freshwaters are found in upland and mountainous areas in which soils and overburden are poor in readily-weathered minerals. Decades of deposition of acidifying sulphur (S) and nitrogen (N) compounds (acid deposition) have resulted in widespread and chronic acidification of sensitive freshwaters across Europe, with loss of fish populations and other biological effects. Since the mid-1980s reduction in emissions of S and to a lesser extent N in Europe has resulted in recovery beginning to take place in lakes and streams (Stoddard et al. 1999, Wright et al. 2005). A critical question being addressed by the Euro-limpacs project is to what extent is this recovery threatened by future climate change. Climate variability over many time-
scales (daily, weekly, seasonal, yearly, decadal) can affect surface water chemistry and biology and thus the acidification and recovery processes. Some key projected climate changes with the potential to impact water acidity are:

i) Higher temperatures  
ii) Increased incidence and severity of summer drought  
iii) Wetter conditions during winter  
iv) Increased frequency and magnitude of winter high flows  
v) Reduced snowpack  
vi) Increased occurrence of sea-salt deposition events

Both indirect effects through impacts on the terrestrial catchments and direct effects on the water bodies themselves are important. Substantial inter-year variability in water chemistry is linked to climatic fluctuations. These include fluctuations in sea-salt deposition, with higher loadings associated with a peak in the NAO Index causing short-term acidification in the early 1990s (Evans et al. 2001). Nitrate and sulphate concentrations are also strongly influenced by climatic factors. The stability of invertebrate communities in acid-sensitive streams is sensitive to variations in the NAO Index (Bradley and Ormerod 2002).

5.1 Acid episodes and climate

Short-term climate events can cause ‘acid episodes’ with biological damage and set back the recovery process. Acidic episodes are crucially important in terms of stream biota, as it is the severity of chemical extremes, rather than average conditions, which typically determines biological damage such as fish kills (Baker et al. 1990, Hindar et al. 1994) and loss of invertebrate species (Kowalik and Ormerod 2006). Episodes can be caused by a variety of different drivers, including high rainfall events, snowmelt, sea-salt deposition events, sulphate flushes after droughts, and nitrate flushes after freezing events (Davies et al. 1992, Wright 2007, Evans et al. in press). A consistent feature of all these drivers, however, is that they are associated with some form of climatic extreme. And scenarios of future climate generally project increased frequency and severity of extreme events.

A modelling study at the Afon Gwy monitoring catchment at Plynlimon, Wales indicated that the severity of high-rainfall driven acid episodes is declining in magnitude as S deposition is reduced (Evans et al. In press). In areas subject to large annual snowpack accumulations snowmelt events are a major cause of acid episodes (Laudon and Bishop 2002, Laudon et al. 2004). Projected decreases in snowfall will reduce the influence of snowmelt on runoff chemistry.

5.2 Sulphur and climate

In wetland areas anaerobic conditions in water-saturated soils lead to S storage, via reduction to organic S compounds and inorganic sulphides. Droughts conditions lower the peat water-table, allowing oxygen to enter the
soil and the re-oxidation of reduced S compounds to sulphate (Dillon et al. 1997). This can generate extreme levels of acidity, which if flushed from the soil can cause major acid episodes in runoff, together with mobilisation of toxic metals (Tipping et al. 2003). A long-term assessment of sulphate-driven acid episodes at Birkenes, Norway, indicated that the severity of drought-induced acid episodes has decreased since the 1970s as the rate of S input has declined (Wright 2007), but a modelling study for an Ontario wetland catchment by Aherne et al. (2006) suggested that repeated drought events, even at current levels of drought frequency, would be sufficient to severely retard recovery from acidification. Repeated droughts, together with reduced S deposition, will lead to the gradual depletion of peat S stores (Tipping et al. 2003), but in more polluted regions this process may take many decades. Peat catchments containing large stores of anthropogenic S must therefore be considered highly sensitive to a projected increase in the frequency and severity of summer droughts, which could lead to the destabilisation of these stores, and consequently to an increased incidence of biologically damaging post-drought stream acidification events.

5.3 Nitrogen and climate

N in terrestrial ecosystems is tightly cycled. Any climatic event which disrupts biological cycling is likely to result in nitrate leaching. In the UK this has been most clearly observed following soil freezing events, which typically occur when the winter NAO Index is negative (Monteith et al. 2000, Davies et al. 2005). Such events are likely to become less frequent under future climate change. Nitrate flushes also occur after droughts (Adamson et al. 1998), and these events may increase in frequency. Extreme rain events may also transport nitrate directly to surface waters, where water bypasses biological sinks within the soil, e.g. as overland flow. Again, these events may be more common in future. Of greatest overall concern, however, is the long-term stability of the soil organic matter pool; as this contains most of the N accumulated over more than a century of elevated deposition. The Norwegian CLIMEX study, in which a small catchment was exposed to elevated temperature and CO₂, showed a marked increase in N mineralisation from the soil, which led to elevated nitrate leaching and effectively turning the catchment from an N sink to an N source (Wright 1998). Such a response to climate warming would clearly have grave consequences for the acidification and eutrophication of upland waters.

5.4 Dissolved organic carbon, water colour and climate

DOC represents a large part of the carbon export of many upland catchments, and is the major source of water colour. It is a significant component of the upland carbon balance, contributes significant costs to water treatment, and impacts on aquatic ecosystems by altering light regime, nutrient transport, acidity, and metal transport and toxicity. Since the late 1980s, surface water DOC concentrations have approximately doubled across a large proportion of the UK (Freeman et al. 2001, Worrall et al. 2004, Evans et al. 2005). The reasons for these increases have not been fully resolved, but there is growing
evidence that a significant, and perhaps primary, driver has been the reduction in S deposition and subsequent recovery from acidification, which has increased the solubility of organic matter (Evans et al. 2006a, Monteith et al. 2007). As an integral part of the upland carbon cycle, however, there is little question that climate-related factors also impact on DOC export (Freeman et al. 2001, Evans et al. 2006a). Droughts also appear to have a strong effect on DOC release, generally decreasing DOC concentrations during the drought period itself, with increases observed thereafter (Hughes et al. 1997, Clark et al. 2005). Overall, it is possible that climatic changes have contributed to the DOC increases observed to date, and probable that they will contribute to further DOC changes in the future. However it must be emphasised that other factors (S deposition, land management and possibly N deposition) are believed to have had as much, or more, influence on DOC trends during the last 30 years, and cannot be ignored in predicting future changes (Monteith et al. 2007).

5.5 Modelling effects of future climate change

A number of studies have attempted to predict the impact of climate change on recovery from acidification. Wright et al. (2006) used the MAGIC model to examine the sensitivity of future mean acidity to a range of projected climatic changes at 14 sites in Europe and North America. Sensitivity was highly variable both among different drivers and between sites, with climatic effects on organic acid leaching and N retention identified as those requiring the greatest focus (Figure 5). The results showed that several of the factors are of only minor importance (increase in partial pressure of CO$_2$ in soil air and runoff, for example), several are important at only a few sites (e.g. sea-salts at near-coastal sites) and several are important at nearly all sites (increased concentrations of organic acids in soil solution and runoff, for example). In addition changes in forest growth and decomposition of soil organic matter are important at forested sites and sites at risk of nitrogen saturation. The trials suggested that in future modelling of recovery from acidification should take into account possible concurrent climate changes and focus specially on the climate-induced changes in organic acids and nitrogen retention.

Research on the effect of climate on recovery of freshwaters is of direct interest in formulating new policy goals with respect to emissions of S and N compounds. Here the major research challenge is still the link between the C and N cycles in terrestrial ecosystems and how N deposition and global change will affect these cycles in the future.

6.0 Interactions between climate change and 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 re-mobilised and taken up by aquatic biota (Catalan et al., 2004; Vives et al., 2005a,b). The high levels of metals (e.g. Hg, Pb) and persistent organic pollutants (PCBs, DDE, PBDEs) in the tissue of freshwater
fish in arctic and alpine lakes (Grimalt et al., 2001; Vives et al., 2004a) attest to the mobility and transport of these substances in the atmosphere (Carrera et al., 2002; Fernandez et al. 2003; van Drooge et al., 2004) and their concentration in cold regions (Fernandez and Grimalt, 2003). For aquatic systems with long food chains biomagnification can elevate concentrations in fish to lethal levels for human consumption. The major concerns with respect to climate change for the Euro-limacs project is the extent to which increased temperatures will cause changes in the behaviour of organic pollutants in arctic and alpine freshwater systems, whether storm events and flooding might increase soil and sediment erosion and lead to the re-mobilisation of toxic substances (Yang et al. 2002), and in the case of Hg, whether changing hydrology in Boreal forest soils may lead to the enhanced production of MeHg (Munthe et al. 2001).

6.1. Accumulation of persistent organic compounds in high mountain lakes

In Europe, high mountains ecosystems are under the direct influence of the temperature-dependences of the accumulation of persistent organic pollutants (POPs) (Grimalt et al., 2001). Lakes from these environments document the transfer mechanisms and impact of these compounds in the headwater regions of Europe’s major river basins. Their accumulation patterns depend on many factors including the time of their introduction into the environment (Gallego et al., 2007). In a study of Pyrenean lakes polybromodiphenyl ethers (PBDEs) in fish from Pyrenean lakes show higher concentrations at lower temperatures, as predicted in the global distillation model (Fig. 6). Conversely, no temperature-dependent distribution was observed in vertical lake transects in the Tatra mountains (Central Europe, Fig. 6) nor in fish from high mountain lakes distributed throughout Europe (Vives et al., 2004b). The fish concentrations of PCBs examined for comparison showed significant temperature correlations in all these studied lakes (Fig. 6.1). Cold trapping of both PCBs and PBDEs concerned the less volatile congeners. These cases of distinct PBDE and PCB behaviour in high mountains probably reflect early stages in the environmental distribution of the former since they have been under secondary redistribution processes over much shorter times than the latter (Gallego et al., 2007).

The accumulation of these compounds, including PBDEs, are also subject to bioconcentration effects by organisms in high mountain aquatic systems. In a recent study of Tricoptera and Diptera it was observed that there is a non-selective enrichment of OCs and PBDEs from larvae to pupae (Bartrons et al., in press). These concentration increases may result from the weight loss of pupae during metamorphosis as a consequence of mainly protein carbon respiration and lack of feeding. The concentration increases from larvae to pupae are very relevant for the pollutant ingestion of the higher predators. The intake of OCs and PBDEs by trout are between two and five-fold higher per calorie gained when predating on the latter. Since pollutant concentration, energy reward, predation susceptibility and duration of life stage are very different between these two insect stages, and none of them is irrelevant for the incorporation of OCs or PBDEs to higher levels, bioaccumulation food-
web models need to distinguish between the two sources (Bartrons et al., 2007).

6.2 Climate change and the remobilisation of metals from polluted soils

Although the emission of many toxic substances such as DDT, PCBs and trace metals has been controlled, soils and sediments throughout Europe remain contaminated as a result of decades of toxic substance accumulation. Remobilisation of such compounds occurs not only because of temperature dependent distillation and condensation processes but also as a result of soil processes through leaching and erosion. Any change in climate, such as an increase in storminess causing increased catchment soil erosion, or increased flooding causing disturbance of floodplain sediments, is likely to remobilise toxic substances leading to changes in uptake and accumulation of these substances in freshwater food chains.

In Euro-limpacs we are examining these processes in several settings including the remobilisation of metals from upland organic soils in Scotland and the impact of flooding on the remobilisation of metals from floodplain sediments of the River Elbe. The Scottish study builds on the work of Yang et al. (2002) who showed that Hg and Pb concentrations in the recent sediments of Lochnagar, a remote Scottish mountain lake, had not declined over recent decades despite a strong reduction in the emission of these toxic metals to the atmosphere. They hypothesised that the lake was continuing to receive toxic metals due to the erosion of catchment soils that were already contaminated by historic pollution deposition. In Euro-limpacs this hypothesis is being tested by comparing the sediment records of Scottish lakes with and without eroding catchment soils, in areas with relatively high and low pollutant deposition and with relatively high and low rainfall. The concern is that any increase in winter storminess in future as a result of climate change might accelerate the transfer of pollutants from catchment soils to lakes and cause the already high levels of metal contamination in the fish tissue of remote lakes (Rose et al. 2005) to remain high or even to increase.

6.3 Climate change and mercury mobilisation

In the boreal forest zone of Northern Europe there is a concern that climate change may lead to an increase in the concentrations of methyl mercury in fish. Already thousands of lakes in Scandinavia have mercury levels in fish that exceed health guidelines of 0.5 mg/kg making them unsuitable for human consumption. In a region of Europe where climate models predict an increase in winter precipitation an increase in groundwater levels will cause more water to flow through organic-rich soil horizons where a large fraction of the soil-bound mercury is present, potentially causing direct mobilisation of mercury and methyl-mercury. Changing redox conditions and release of DOC and nutrients may enhance this process and cause an increase in methyl mercury on aquatic ecosystems (Munthe et al. 2001). In Euro-limpacs this process is being tested by manipulating precipitation and hydrology at Gårdsjön, an experimental lake site in south-west Sweden. Provisional results of the experiment show that increased soil wetness does indeed lead to anaerobic
conditions, sulphate reduction and the generation of significant amounts of methyl-mercury and total mercury in runoff.

7. Developing indicators of climate change for freshwater ecosystems

Monitoring programmes of freshwater ecosystems in Europe have traditionally focussed on water quality, either by direct chemical measurement or by using biological indicators (De Pauw & Hawkes, 1993, Knoben et al. 1995). As water quality improved additional indicator systems were developed to monitor habitat quality e.g. Raven et al. (1997). The Water Framework Directive (WFD) now requires an integrated approach to freshwater monitoring in Europe using a range of different biological groups, including phytoplankton, benthic diatoms, aquatic macrophytes, macroinvertebrates and fish either singly or together to assess the ecological status of rivers, lakes, coastal and transitional waters. According to the WFD, the ecological status of a water body is defined by comparing the present day biological community composition with near-natural reference conditions. These WFD guidelines on ecological water quality assessment have generated an urgent need to develop new or revised existing indicator systems.

Most new indicator systems are based on metrics, i.e. attributes of the biotic assemblages, which reflect community composition and abundance, taxon richness, the proportion of sensitive and tolerant taxa or functional attributes (Hering et al., 2006). They are designed to assess overall ecological stress (e.g. deviation from reference conditions) or to assess single stressors, e.g. eutrophication (Kelly et al., 1995), organic pollution (Zelinka & Marvan, 1961), hydromorphological degradation (Statzner et al., 2001) or acidification (Townsend et al., 1983).

However, the indicator systems defined for the WFD have shortcomings, limiting the applicability, reliability and interpretation of the assessment results. Problems include: (i) important ecosystem types, such as wetlands and small streams and particularly at risk from climate change, are not included in the WFD; (ii) reference conditions are poorly defined or have been derived with different methodologies; (iii) scores used in the metric systems are rarely based on experimental evidence or on a thorough examination of the literature; (iv) many assessment systems are not capable of discriminating between stressor types; and (v) the system does not allow for new types of stressor. This is particularly the case for climate change. Although climate change interacts with many ‘traditional’ stressor types such as eutrophication, it adds additional stress e.g. through increases in water temperature, or change in stream discharge.

In addressing these problems the Euro-limpacs project aims to assess how current indicator systems need to be modified to be suitable for climate change monitoring. To do this it is necessary to:

- Generate hypotheses on how climate change will directly and indirectly affect freshwater ecosystems and their communities;
• Build databases that contain reliable ecological background information on species or higher taxa inhabiting different types of European freshwater ecosystem types. Information on traits, which demonstrate sensitivity to direct and indirect effects of climate change (e.g. temperature preferences, resilience to droughts) is especially important;
• Demonstrate that species or metrics change with direct or indirect effects of climate change can be incorporated into standard assessment systems.

To derive hypotheses on the effects of climate change on freshwater ecosystems around 1,000 papers have been evaluated. From these, ‘cause-effect-chains’ have been generated to categorise the effects for which indicators are needed. The literature evaluation revealed a strong bias towards abiotic components of the different ecosystem types. For example, of the 317 papers dealing with effects on lakes, 179 are related to abiotic effects, such as water temperature or water column stratification, 90 papers deal with plankton communities but less than 50 are concerned with other biota. Of the 713 papers dealing with the effects of climate change on rivers, 616 are related to abiotic effects, mainly on hydrology (397 papers). Only 97 are related to biotic communities and processes.

In Euro-impacts data on ecological characteristics and distribution patterns for six organism groups widely used in freshwater assessment, diatoms, fish and four groups of benthic invertebrates (Ephemeroptera, Plecoptera, Trichoptera and Chironomidae) have been compiled. For the invertebrates we aimed to cover the complete literature, including ‘grey’ literature such as Diploma- and PhD-theses. In total more than 8,000 literature references have been consulted. The resulting data are stored in an online database (www.freshwaterecology.info) and includes data on more than 22,000 taxa. Data on individual attributes differ in their degree of completeness. For example, for European caddisflies, completeness of the data for the individual attributes ranges from 0.2% to 99.7%. Besides the attribute ‘distribution in ecoregions’ (99.7% of all taxa classified), high proportions of taxa were also classified for the attributes ‘current preference’ (84.2%), ‘stream zonation preference’ (71.1%) and ‘substrate/microhabitat preference’ (66.9%) (Graf et al., 2006).

The sensitivity of European caddisfly species to the impacts of climate change differs greatly between ecoregions. One criterion, which leads to a high vulnerability, is “restricted distribution”. Endemic taxa are often characterised by a restricted ecological niche and limited dispersal capacity. These taxa are more severely threatened by climate change than widely distributed species, as shown for vascular plants (Malcolm et al., 2006) and as also suggested for benthic invertebrates (Brown et al., 2007). The number of endemic caddisfly species and subspecies is highest on the Iberian Peninsula (141 taxa), followed by Italy (118 taxa) and the Hellenic Western Balkan (76 taxa), while in 14 out of the 23 European ecoregions less than five endemic taxa occur (Hering et al., submitted).
Other ecological characteristics, leading to a high sensitivity of species to climate change, include the preference for springs, preference for cold water temperatures, short flight periods and restricted ecological niches. On the community level, the following attributes are expected to be affected by changing climate: the proportion of spring-preferring taxa (or the longitudinal preferences of the taxa in general), the proportion of cold-stenothermic taxa, or the proportion of taxa with a long life-cycle. By including such metrics into assessment systems the impact of climate change can better represented in overall appraisals of ecological quality (Fig. 7).

8. Implications of climate change for aquatic ecosystem restoration

Attempts to restore aquatic ecosystems in Europe are now subject to the requirements of the European Water Framework Directive that requires freshwaters to be maintained in or returned to good ecological status, a status that is defined as “slightly” different from the pristine or reference state. Whilst the concept is simple, defining the reference state is or can be problematic, and as climate change increasingly influences all aspects of the structure and functioning of ecosystems, use of the reference state as a target for restoration may become increasingly difficult. In the Euro-impacts project, therefore, we focus on how climate may affect or confound the use of reference conditions in ecological assessment. This is done first by recognising that a reference condition is not a static state, but changes over time, albeit within the confines of its ecological envelope defined as: (i) the physical conditions, such as geographic location, catchment geology and hydrology, that determine the physical constraints of the site; (ii) the biological potential biota of the site, constrained by those populations whose distributions overlap at the site; and (iii) the interactions between populations that result in complex networks and feedback loops (e.g. O’Neill 2001).

8.1 Detecting recovery: confounding effects of climate?

The reference state is dynamic, varying both chemically and biologically on seasonal, inter-annual and decadal time-scales in response to natural fluctuations in climate, internal ecological processes and random events. Human modification of river and lake catchments through land-use change and hydromorphological alteration may also alter the reference state and now there is concern that ecosystems thought to be in equilibrium with their surroundings (but see O’Neill 2001) may already be undergoing unprecedented changes due to global warming (cf. Catalan et al. 2002, Smol et al. 2005). As warming continues to increase in future to temperatures not seen before in the Holocene, new boundary conditions for ecosystem processes will be created. Reference conditions are then not only dynamic but also undergoing directional change in which the baseline is continually changing (Battarbee et al. 2005, Figure 8). It consequently prompts work into better understanding the drivers behind long-term patterns and processes, with particular focus on the analysis of long time-series from observational records, the use of sediment records from lakes (e.g. Bennion & Battarbee 2007) and the understanding of processes and functions at extant analogue systems with good ecological status.
In addition to understanding the reference state and the longer term changes in baselines that help to define restoration targets, studies designed to assess the success of restoration also need at the outset a consideration of factors, such as the choice of indicator and habitat, that maximise the chance of detecting responses to mitigation if and when it occurs. Moreover, because the success or failure of restoration is often judged as a deviation from an expected condition (target state), the ideal design should also include pre- and post-restoration monitoring (Downes et al. 2002). In Euro-limpacs we address these issues attempting to show how climate change might affect restoration strategies for different system types. Here we present three examples, on lake acidification, river hydromorphology and landscape connectivity.

8.2 Lake acidification restoration and climate change

As a result of a major reduction in the emission of acidifying S and N compounds since the 1970 in Europe (see section 5 above), acid deposition on boreal lakes and their catchments has also strongly declined. Recently, Stendera and Johnson (submitted), assessing the recovery of boreal lake ecosystems from acidification, used different indicators (water chemistry, biota), different trophic levels (primary producers and consumers), and different habitats within the lakes (pelagic, benthic) to compare the response of both acidified and reference lakes to the reduction in acid deposition.

Several of the indicators showed positive trends, whereas others showed surprisingly negative trends during the 16-year study (Fig. 9). Although decreasing acidity and the changes in phytoplankton and littoral invertebrate assemblages indicated that there are signs of recovery from acidification, other changes occurred unrelated to changes in acid deposition. For example, unexpected changes in the reference lakes occurred in phytoplankton and littoral invertebrates probably caused by climatically-related changes in water colour and temperature, and decreases in the richness of sublittoral and profundal invertebrate assemblages may be the result of climate-related change acting on habitat quality, such as ambient oxygen concentrations and temperature, rather than by changes in lake acidity. Taken together the results of the study indicated that recovery is a complex process influenced by multiple spatial and temporal factors. It demonstrates the importance of including multiple organism groups and trophic levels in studies designed to detect and understand the processes and changes associated with recovery, especially in situations where there are complex interactions between pollutant behaviour and climate.

8.3 Stream habitat restoration and climate change

Concern for the loss of stream and floodplain habitats and biodiversity over recent decades has provided the impetus for extensive programmes of stream rehabilitation and restoration in many European countries. Many techniques are being used including reforestation of the floodplain, re-meandering, removal of dams and bank supports. New, more innovative
approaches include the adding of coarse woody debris (Gerhard & Reich 2000, Gippel & Stewardson 1996), the removal of sediment deposits in floodplains and various methods to combat gullyling.

In order to make the proper choices in stream restoration, the complex spatial and temporal interactions between physical parameters, habitat diversity and biodiversity have to be understood, and consideration needs to be given to how the success of restoration schemes might be influenced by future climate-related changes in hydromorphology. When a stream has been restored, the success (increase in biodiversity) depends on the re-colonisation of the original (indicator) species. Whether these species will be able to re-colonise the restored stream depends on the distance to remaining populations, dispersal barriers in between the remaining population and the restored stream, and the dispersal ability of the species. Establishment of an invasive or non-native species may also hinder re-colonisation, and biodiversity may in general be threatened by invasive species replacing the native ones. Climate change affects all these processes and influences hydromorphology at all scales. Under changing climate conditions either the measures adopted to meet restoration targets in such schemes need to be adapted, or the targets themselves must be adjusted.

**Table 2.** Hydromorphology characteristics of the river Lahne in a single and multiple channel situation.

<table>
<thead>
<tr>
<th>factor</th>
<th>single channel</th>
<th>multiple channel</th>
<th>factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>shore length (m)</td>
<td>432</td>
<td>1408</td>
<td>+ 3.3</td>
</tr>
<tr>
<td>CV current velocity</td>
<td>0.62</td>
<td>1.1</td>
<td>+ 1.8</td>
</tr>
<tr>
<td>CV depth</td>
<td>0.59</td>
<td>0.99</td>
<td>+ 1.7</td>
</tr>
<tr>
<td>wet area (m2)</td>
<td>76</td>
<td>124</td>
<td>+ 1.6</td>
</tr>
</tbody>
</table>

In Euro-limpacs the development of hydromorphology and macroinvertebrates was studied in a multiple channel restoration of the River Lahne. Table 2 shows that the shore length, current velocity variation, depth variation and wet area increased 1.6 to 3.3 times. However, the analysis of the macro-invertebrates of both the single and multiple channel though showed no change at all in composition and abundances, illustrating the need for studies on longer time-scales and the need for restoration on a catchment rather than reach scale.

### 8.4 Landscape scale restoration and climate change

Restoration endeavours in the past have often focused on individual sites (e.g. lakes) or stream reaches, largely ignoring the importance of connectivity such as channel movement for terrestrial-aquatic linkages (e.g. river-floodplain exchanges) and biotic dispersal and ignoring the fact that connectivity may be increasingly threatened in future by the effects of climate change. Awareness that large-scale processes determine the structure and function of aquatic ecosystems and that these properties and principles need to be considered in
restoration prompted Verhoeven et al. (submitted) within the Euro-limpacs project to propose the Operation Landscape Unit (OLU) concept. The OLU is defined as “combinations of landscape patches with their hydrological and biotic connections” and used to identify a parsimonious set of landscape elements and their configuration to understand better the constraints (e.g. like fragmented landscapes) which may impede the effectiveness of restoration efforts. This implies that regional conservation strategies need to be spatially coherent, based on the relevant physico-chemical and biological processes to ensure that restoration measures are successful and sustained.

Future climate-related impacts are expected to profoundly affect hydrological cycles. Practical management of lakes and streams needs to take into account catchment scale issues and large-scale, landscape-level planning is needed if restoration projects are to be cost-effective and ecologically successful. Restoration targets need to be chosen carefully taking into consideration the difficulty of defining a fixed reference state. High quality pre- and post-restoration monitoring is needed to understand how climate may confound recovery trajectories, so that lessons from past experiences can be used to improve future plans.

9.0 Modelling the impacts of climate change on freshwater ecosystems at the catchment scale

The need for a catchment-scale approach to freshwater ecosystem management is recognised by the EU Water Framework Directive, where the basic unit of management is referred to as the “river basin district” (EU 2000). The complexity of the interactions between aquatic and terrestrial systems at the catchment scale necessitates a modelling approach also at the catchment scale. With respect to climate change, existing or new models need developing to representing 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.

The principal catchments being modelled in Euro-limpacs are shown in Table 3. The areas of these catchments vary over 7 orders of magnitude, from the 5,200 m² Gårdsjön experimental catchment to the Garonne-Adour catchments, which occupy the whole of SW France. A wide variety of climates and eco-regions are also represented.
<table>
<thead>
<tr>
<th>Country</th>
<th>Study Area</th>
<th>Area km²</th>
<th>System Components Present</th>
<th>Key Issues</th>
<th>Climate</th>
<th>Eco-region</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>River</td>
<td>Lake</td>
<td>Wetland</td>
<td></td>
</tr>
<tr>
<td>Austria</td>
<td>Piburger See</td>
<td>2</td>
<td>Eutrophication</td>
<td></td>
<td>Cool</td>
<td>Mountainous</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Continental</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Denmark</td>
<td>Gjern</td>
<td>110</td>
<td>Eutrophication, Sed.</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Odense</td>
<td>486</td>
<td>Eutrophication, Sed.</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td>Greece</td>
<td>Cheimadipta</td>
<td>35</td>
<td>Eutrophication</td>
<td></td>
<td>Med/Cool</td>
<td>Mediterranean</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Continental</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Savijoki</td>
<td>15.4</td>
<td>Eutrophication, Sed.</td>
<td></td>
<td>Sub-arctic</td>
<td>Boreal</td>
</tr>
<tr>
<td></td>
<td>Simojoki</td>
<td>3160</td>
<td>Acid., N sat.</td>
<td></td>
<td>Sub-arctic</td>
<td>Boreal</td>
</tr>
<tr>
<td>Finland</td>
<td>Tueronjoki</td>
<td>439</td>
<td>N sat, Eutrophication</td>
<td></td>
<td>Sub-arctic</td>
<td>Boreal</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>France</td>
<td>Garonne-Adour</td>
<td>56500</td>
<td>Eutrophication</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td>Norway</td>
<td>Bjerkreim</td>
<td>685</td>
<td>Acid., N sat., C</td>
<td></td>
<td>Maritime</td>
<td>Boreal</td>
</tr>
<tr>
<td></td>
<td>Tovdalselva</td>
<td>1855</td>
<td>N sat, Eutrophication</td>
<td></td>
<td>Maritime</td>
<td>Boreal</td>
</tr>
<tr>
<td>Romania</td>
<td>Lower Danube Wetland</td>
<td>210</td>
<td>Eutrophication</td>
<td></td>
<td>Cool</td>
<td>Continental</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Continental</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spain</td>
<td>La Tordera</td>
<td>124</td>
<td>Eutrophication</td>
<td></td>
<td>Mediterranean</td>
<td>Mediterranean</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sweden</td>
<td>Gårdsjön</td>
<td>0.005</td>
<td>Acid., N sat., C</td>
<td></td>
<td>Maritime</td>
<td>Boreal</td>
</tr>
<tr>
<td></td>
<td>Svartberget</td>
<td>0.5</td>
<td>Acid., Hg</td>
<td></td>
<td>Sub-arctic</td>
<td>Boreal</td>
</tr>
<tr>
<td>UK</td>
<td>Conwy</td>
<td>590</td>
<td>Acid., Eutrophication, C</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Kennet</td>
<td>1030</td>
<td>Eutrophication, Sed.</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Lambourn</td>
<td>263</td>
<td>Eutrophication</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Endrick / Falloch</td>
<td>781</td>
<td>Eutrophication</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Tamar</td>
<td>917</td>
<td>Eutrophication</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
<tr>
<td></td>
<td>Wye</td>
<td>4140</td>
<td>Acid., Eutrophication, C</td>
<td></td>
<td>Maritime</td>
<td>Atlantic</td>
</tr>
</tbody>
</table>
The most widely-used models in the Euro-limpacs project are the INCA suite of models. The first INCA model (INCA-N) was aimed at understanding the response of rivers to changes in nitrogen inputs and catchment nitrogen metabolism (Whitehead et al., 1998; Wade et al., 2002). Since then the INCA framework has been developed to encompass phosphorus (INCA-P); particulates (INCA-SED); dissolved organic carbon (INCA-C) and mercury (INCA-Hg). Other dynamic models being used in Euro-limpacs include the Mike11-TRANS model. This model is a suite of ecological sub-models linked to a hydrodynamic model, and has been used in conjunction with the rainfall-runoff model NAM to model N fluxes in lowland Danish catchments (Anderson et al., 2006). Another development of existing models (Evans et al., 2006b) is the use of a landscape-based mixing model (PEARLS) coupled with the acidification model MAGIC to simulate the recovery from acidification of the Conwy in North Wales, a large heterogeneous river basin.

9.1 Example model application

Here (Fig. 10) we provide an example of the use of INCA-N (Whitehead et al., 2006). It is the first attempt to model adaptation strategies to mitigate the effects of climate change on nitrate concentrations in rivers. The example is from the River Kennet in S England for the period 1961 to 2100, with precipitation scenarios down-scaled from three general circulation models. These generate different effects in detail, but all show a general increase in nitrate concentration due to enhanced microbial activity. Figure 9 shows the results of attempting various mitigation strategies on this overall picture using one precipitation scenario (from the HadCM3 Model). The baseline scenario (no mitigation attempted) shows a steady increase in nitrate. The peak towards the end of the period is due to a simulated 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 4 times the river surface area is 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 seems unlikely. 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 of the large mass of nitrate that has accumulated in the chalk aquifer (Jackson et al.2007). Thus, some kind of in-stream intervention by construction of water meadows may be the best option to reduce in-stream nitrate concentrations within the timescale of the Water Framework Directive. This is an interesting result and suggests that in catchments with a long residence time, pollutants in the groundwater may continue to confound attempts to improve the chemical and ecological status between now and 2015.
9.2 Model chains and integration

Modelling catchment responses often requires the integration of separate models covering components of the catchment, such as soils, vegetation, groundwater, rivers, lakes etc., and also the integration of models which predict the response of 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 much more difficult to achieve given that the component models have normally not been designed with this integration in mind. A number of chaining and integration projects are under way in Euro-limnacs 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) The models used were two GCMs (ECHAM4 and HadAM3H) to predict the effects of climate change on meteorology on a large spatial scale; a regional climate model (HIRHAM) to downscale these predictions to the catchment scale and to a daily temporal scale; the hydrological model HBV to translate the meteorological variables into water fluxes through the catchment and water status in catchment components; the water quality models MAGIC and INCA-N which use this information to predict nitrogen fluxes and concentrations in catchment components; and the NIVA FJORD model to use the predicted N discharge of the Bjerkreim river to predict annual variation of nitrate in the Egarsund fjord which receives the runoff. The results driven by the HadAM3H model indicated the possibility of increased productivity and eutrophication in the fjord by 2080, whereas those driven by the ECHAM4 model did not. Irrespective of whether either of these predictions is correct, 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.

9.3 Model uncertainty

All model predictions are uncertain to some degree, but quantifying and preferably reducing this uncertainty is one of the priorities for Euro-limnacs. The sources of this uncertainty are many (e.g. Morgan and Henrion, 1990). It might be thought that since catchment hydrochemical models can include more than 100 parameters, some of which are very imperfectly known, that the overall uncertainty will be so large as to render them essentially 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, but uncertainty propagation studies show that the uncertainty in the calculated critical loads is typically less than the uncertainty in any of the input parameters (e.g. Skeffington et al., 2007). A number of techniques for estimating uncertainty in environmental
models are currently being applied by Euro-impacts participants. Partners who have predicted the future nitrogen status of their catchments using INCA-N are using a Monte Carlo – based tool developed within the project, which performs a general sensitivity analysis (e.g. Wade et al. 2001) to identify key parameters and which puts confidence limits on model predictions. At the same time less formal approaches are being used, such as applying different models to the same problem and comparing the results. By the end of the project it is hoped we will be in a better position to assess the uncertainty in predictions of the effects of climate change on freshwaters, and to be able to identify the steps necessary to reduce it.

10. Implications of predicted climate change impacts on freshwater ecosystems for policy and management

The general lessons that can be drawn from our improved understanding of how ecosystems may respond to climate change in future have important consequences for their management and for the development of related environmental policy. The uncertainties inherent in this process need to be built into the policy and management process. These uncertainties derive not just from climate and catchment models themselves (as above), but also relate to the uncertainty in predicting future patterns in the emissions of greenhouse gases, land-use change and pollutant loading and the response of society to such changes.

In Euro-impacts we are considering these issues at the global, regional and local (catchment) scales, and engaging with a wide range of stakeholders to provide advice and to develop a decision support system (DSS) of practical value to the user community.

10.1 Socio-economic pressures and global change

In assessing the potential impact of climate change on freshwater ecosystems in future we need to take into consideration how current policies, protocols and socio-economic pressures might influence other drivers of change on freshwater ecosystems, especially how scenarios for climate change itself might alter future patterns of pollutant loading and land-use patterns. With the signing of the Kyoto Protocol, climate change itself has become the most important driving force for the reduction of pollutant emissions to the atmosphere and will lead to significant reductions in emission and deposition patterns of both acids and nutrients over and above those identified to reduce current problems of acidification. Likewise under climate change the economic costs and returns of many agricultural practices will change. However, subsidies, taxation and tradeable permits can also influence land management decisions within the agricultural sector. In Euro-impacts we are developing an economic model that integrates the impact of global change on crop yield, growing costs (e.g. fertilisers) and the impacts of economic incentives. Information from this model will be used to assess the likely impact of changes in agriculture under different global change scenarios on surface waters and marginal wetlands. An integral component of this process is the
contingent valuation of the impacts of climate change, land management and atmospheric deposition on lakes, rivers and marginal wetlands, especially the need to establish non-market values of habitat change brought about by the impacts of global change.

10.2 Analysis of policies influencing catchment management

Policies for the protection of freshwater ecosystems are implemented at the catchment scale and catchment managers need to be aware not only of policies designed specifically for water bodies but also those that influence other processes operating in catchments. This is often complicated by the fact that different responsibilities are vested in different administrative bodies that operate at different scales, including European, national and local scales and that administrative boundaries are often not coincident with catchment boundaries at these different scales. In Euro-limpecs we are drawing up an inventory of policies and international agreements affecting catchment management, systematically assessing the agendas and resources of these authorities and examining their congruence with catchment management issues, highlighting especially those situations where policies in other sectors (notably agriculture but also transport) may be in conflict with catchment management. Differences between member states within the EU are of special concern and the DSS will need to be designed to apply in different national settings. To this end workshops with national experts are being used for assessment purposes taking specific catchments including in Euro-limpecs as a focus.

10.3 Stakeholder engagement and the Decision Support System

A central element of Euro-limpecs is the involvement and integration of stakeholders to ensure that the scientific results of the project are of the most practical value to the user community. The user community includes national governments, the European Environment Agency, the European Commission, and especially Government agencies charged with the responsibility of implementing policy at the catchment scale.

The DSS itself (Fig. 11) aims to provide a framework for the integration and analysis of data from disparate sources, and which relate to a wide range of issues, into a single holistic and GIS-based analysis of catchment management in the context of climate change. It also aims to bridge the gap between science and decision making to achieve integrated catchment management and provide a mechanism by which the range of services provided by ecosystems can be evaluated in a holistic assessment, using Multi-criteria Analysis (MCA).

The DSS incorporates outputs from across the Euro-limpecs project to inform the management decisions made using the system. As the DSS is intended for application right across Europe the expert knowledge and scientific results from the project that have a generic applicability are embedded within the DSS software, in the form of models, databases or documents. However, many of the outputs from the project are detailed scientific studies that are
being carried out at a site or catchment scale and cannot often be generalised or extrapolated beyond the specific conditions of the study. This makes them unsuitable for being embedded within a DSS that is intended to be generic. However, these results are important outputs from the project and the DSS framework therefore allows for the integration of site specific data into applications of the DSS.

The resulting framework addresses specific and targeted management questions such as: (i) will climate change affect some parts of a catchment more than others? (ii) what measures should be taken to mitigate the effects of climate change? (iii) which part of a catchment should resources be targeted towards? and (iv) which measures most effectively tackle the defined problem?

The DSS is being tested at seven catchments across Europe that form a core, common set of catchments for the Euro-impacts project (see also Table 3). These are:

- Tamar (UK),
- Danube sub-basin (Romania),
- Bjerkreim/Tovdal/Vannsjo-Hobol (Norway),
- Odense (Denmark),
- Inn (Austria),
- Tordera (Spain),
- Vecht (Netherlands) and
- Cheimaditida (Greece).

These seven case-studies will, firstly, act as demonstrations of how the DSS can be applied to different types of issues and management problems and, secondly, integrate the site specific data being generated by the Euro-impacts project in these catchments within the application of the DSS. The case-studies include the Tamar catchment (UK) which is being developed as the pathfinder case-study. In the context of the application at the Euro-impacts case-study sites, much of the site-specific data comes from Euro-impacts itself but in the application of the DSS by external users once it is completed and distributed, it is anticipated that these additional site-specific data will come from the users themselves and will reflect the particular issues of concern in the user’s catchment and the availability of data from studies and models specific to that catchment.

The DSS is implemented as an extension of the ArcGIS software, which facilitates the integration of the DSS framework with existing GIS information available to water managers. The geographical area under consideration is divided into spatial units for comparison. Spatial units can be individual water bodies (in the sense used in the Water Framework Directive), water body types (rivers, lakes, wetlands) or other spatially delineated areas, such as sub-catchments, depending on the particular problem being addressed. These spatial units are represented in the DSS as shapes or linear features within a GIS layer. The Tamar example total MCA scores are given that reflect the status of the sub-catchments for 2050 for all climate and management
scenarios. The results demonstrate how the implementation of different management scenarios affects the status of different sub-catchments under climate change thereby enabling management measures to be targeted towards those parts of the catchment where intervention would be most cost-effective.

The proto-type application of the DSS to the Tamar catchment has demonstrated the functionality of this approach and the contribution it can make to strategic planning in catchment management. Its flexibility and potential for integrating modelled and empirical data from across the environmental social and economic dimensions of sustainability are key attributes of the DSS. By providing a mechanism for integrating data from disparate sources, it acts as a complementary tool to detailed, process-driven models that, because of their complexity and data requirements, can only focus on a single or small number of variables. In doing so it allows the outputs from models and empirical studies to be used in a holistic analysis of catchment management issues that takes into account the complex interactions between climate change and management actions and their effects on the environment, society and the economy.

11. Conclusions

Current mean temperatures in the northern hemisphere may now be as high if not higher than during the Medieval Period but have not yet exceeded those of the Holocene climate optimum of approximately 7,500 years ago when July temperatures were approximately 2°C higher than the twentieth century mean. Global climate models strongly suggest that temperatures in the next few decades will soon rise above this value, reaching levels that have not occurred naturally for over 100,000 years.

The impact on natural ecosystems is likely to be profound. Freshwater ecosystems are especially vulnerable to climate change, both through the direct effects of changing temperature and precipitation patterns and indirectly through climate-driven changes in terrestrial ecosystems, land-use and pollution loading. Some of the likely future impacts can be assessed from an examination of ecosystem response to past warm periods recorded in lake sediments and from long-term observational records. Additional insights can be gained from experiments designed to test hypotheses about future change under controlled conditions and from time-space comparisons where lower latitude or lower altitude systems are used as analogues for the future. Information from paleoecological studies, long-term data records, and experiments can be used to parameterise ecosystem models, which then can be driven with scenarios of future climate to project ecosystem response. The Euro-limpacs project is using all these approaches to better understand the processes controlling freshwater responses to climate change. Of special interest are the interactions with other stressors such as artificial alteration of stream and river channels (hydromorphology) and pollution from acid deposition, plant nutrients and toxic substances. A major objective is to develop models both for specific ecosystem types and processes and also for scenario assessment at the catchment scale.
Provisional results from Euro-limpacs described here indicate that future climate change will affect the structure and functioning of European rivers, lakes and wetlands. Among phenomenon likely to occur are:

(i) loss of ice-cover, and strengthening of summer stratification in lakes
(ii) reduction in river flow and lake water level, especially in southern Europe
(iii) melting of rock glaciers with associated solutes and pollutants in high mountain regions
(iv) threats to biodiversity from increased river and lake temperatures
(v) changes to channel hydromorphology affecting river and stream biodiversity
(vi) increased algal growth, an increase in piscivorous populations and a decline in hypolimnionic oxygen concentrations in eutrophic lakes
(vii) delay in recovery from acid deposition and threats to stream organisms from increased frequency and magnitude of high discharge, low pH events in upland regions of Northern Europe
(viii) change in the transport, deposition and food chain uptake of volatile persistent organic pollutants
(ix) remobilisation of toxic substances from increased flooding and storminess
(x) enhanced production of MeHg in boreal lake ecosystems

The impact of climate change on freshwaters will necessitate thorough re-evaluation of national, EU and UNECE policies related to environmental protection. For freshwater ecosystems there are potentially major implications for the EU Habitats Directive, the Urban Wastewaters Directive and the Water Framework Directive and for the UNECE Convention on Long-Range Transboundary Air Pollution (CLTRAP).

The CLTRAP addresses emissions of pollutant gases to the atmosphere (mainly sulphur and nitrogen), and seeks to reduce emissions such that adverse effects on ecosystems can be prevented or remedied. Work within the CLTRAP is now attempting to incorporate future climate changes and their effects and a possible revision of the Gothenburg protocol may take climate change into account.

EU polices relating to biodiversity and conservation will need to make allowance for geographical shifts in the range of species, and for changes in the nature of aquatic habitats both chemically and morphologically. This might include the need for EU countries to work more closely together by assigning conservation value at the continental rather than national scale, integrating activities to provide migration corridors, improving habitat connectivity at all scales but especially at the catchment scale and being alert to the impacts of climate change on the potentially disruptive effect of invasive alien taxa and pathogens.
For the WFD and other policies associated with environmental restoration, climate change has serious implications. In particular (i) the reference state, although valuable as a concept, may be unstable for many freshwater systems over the longer term as reference sites themselves are subject to change; and (ii) restoration targets for disturbed systems may not simply be achieved by removing stresses, as how those stresses interact or might interact with climate change in future will determine the directions in which ecosystems trend.

To understand better how future climate change will affect freshwater ecosystems in future, it will be necessary to:

- Continue to generate high resolution climate models that project probable future climate change at the regional scale and can be down-scaled to project future climate for individual catchments;
- Generate realistic scenarios for future changes in pollution and land-use that are influenced both by climate change and by societal responses to other drivers of change, especially economic ones;
- Continue to perform critical experiments both at the field and mesocosm scale that aim to test specific hypotheses in controlled conditions;
- Continue to develop system specific models to simulate probable hydrochemical and ecological responses to climate change;
- Continue to develop coupled or integrated models that are able to simulate catchment-scale responses to climate change;
- Continue to invest in high quality monitoring programmes to provide early warning of future changes and to provide long-term data-sets for model calibration and verification;
- Establish an array of appropriate chemical and biological indicators for detecting climate change effects;
- Continue to promote integration amongst the freshwater science community in Europe to maintain research capacity and enable coherent responses to emerging problems;
- Invest in central databases for hydrological, hydrochemical and hydrobiological data to safeguard data-sets, especially long time-series, and to enable model development and upscaling to the regional and continental levels; and
- Improve the interaction between water managers and freshwater scientists to enable intelligent data analysis and decision making in restoring ecosystem quality.

As the years go by the evidence that human activity is contributing significantly to global warming becomes more compelling, and as European freshwater ecosystems continue to recover from problems caused by nineteenth and twentieth century pollution, global warming will become the dominating influence on freshwater ecosystems. Some of the responses, such as the loss of winter ice-cover on northern lakes or the reduction in summer flow in southern streams and rivers can be predicted, but other responses, for example, involving interactions with future land-use change and pollutant behaviour or the effects on ecological processes are complex.
and uncertain. However, it is too late to prevent some of these responses. Indeed it is probable that many of the changes are already occurring, as outlined in this chapter. In cases where we have adequate understanding of the consequences, policies and management practice will need to be adapted quickly to accommodate the threats. In other cases there is an urgent need to continue research programmes designed to address the many uncertainties. Throughout and underpinning everything is the importance of maintaining and developing long-term monitoring networks that are able to identify trends and alert both scientists and decision makers to future threats.

12 Acknowledgements

We would like to thank all Euro-limpacs participants for contributing directly or indirectly to this chapter and apologise to those whose specific work is not included here. Euro-limpacs is an EU FP6 Integrated Project (Project no. GOCE-CT-2003-505540) and we are grateful in particular to Christos Fragakis, our programme manager, for his support. We would also like to thank Miles Irving and David Hunt for help in the preparation of the manuscript.

13. References

D. Hering et al. in prep.
D. Hering et al. submitted
Lelystad.
M. Meerhof et al. 2007b. Freshwater Biology, in press.
J. Munthe et al. 2001. Water Air and Soil Pollution Focus vol. 1, 385-393
N.L. Rose et al. 2005. In Global change and mountain regions - A state of knowledge
J.P. Smol et al. 2005. Proceedings of the National Academy of Sciences, vol., 102,
4397–402.
Environmental Change, Springer.
Windows User’s Guide: Software for Canonical Community Ordoniation (version 4.5)
W. van Doorslaer et al. 2007. Global Change Biology, in press
J. Verhoeven et al. submitted
R.F. Wright 2007. Hydrology and Earth System Sciences Discussions, vol., 4, 2945-
2973.
Figure captions

Figure 1. Conductivity, magnesium, sulphate and calcium concentrations in the water of two high alpine lakes during the past 20 years (Rasass See: red dots, Schwarzsee: open circles). Note the break in the concentration axis due to the high change in solute concentrations. (modified from Thies et al., 2007).

Figure 2. Cause-effect chain illustrating the potential impact of changes in climate on stream biodiversity.

Figure 3. DCCA ordination diagram for the hydromorphological gradient analysis with the morphological gradient indicated by black (canalised), grey (semi-natural) and open (natural) symbols and with the hydrological periods indicated by circles (pre-hydrological restoration) and triangles (post-hydrological restoration).

Figure 4. Average maximum and minimum oxygen concentrations recorded during three 24-hr experiments carried out in a mesocosm system in late June and July 2007. For each treatment there were eight replicates. Both heating and addition of nutrients significantly reduced both maximal and minimal concentrations (P<0.01). Unpublished data of C.Whitham, H.Feuchtmayr, D.Atkinson, R.Moran, I.Harvey and B.Moss.

Figure 5. Top panel: ANC (volume-weighted annual mean) concentrations in runoff at nine sites (eight in Europe, one in Canada) for the calibration year (2000) and predicted (using the dynamic model MAGIC) for the year 2030 assuming no climate change (base scenario). Bottom panel: Change in ANC predicted for the year 2030 relative to the base scenario (from Wright et al. 2006).

Figure 6. Lake-averaged muscle concentrations (n = 12-14) of selected PCB and PBDE congeners vs. reciprocal of annual mean air temperature.

Figure 7. Scheme of integrating climate change effects into a multimetric assessment system for river invertebrates. Nine metrics, which are individually assessed by comparison to reference conditions, indicate four different stressor types: Organic pollution (Saprobiic index), hydromorphological degradation (Fauna Index, [%] alk preferences, [%] gatherers, [%] stoneflies, [%] rheophile taxa), acidification (acid index), temperature (longitudinal zonation index, [%] cold stenotherms).

Figure 8. Conceptual diagram illustrating ecosystem response to increasing and decreasing stresses past, present and future in relation to climate change, expressed as a changing baseline (from Battarbee et al. 2005).

Figure 9. Conceptual diagram illustrating a comparison of expected and observed responses of macroinvertebrates (expressed as taxon richness) in acidified and reference lakes in S. Sweden to a reduction in sulphur deposition.
Figure 10. Modelled nitrate concentrations in the lower River Kennet, UK, 1960 – 2100 given various management treatments (see text) and downscaled climate scenarios from the HadCM3 GCM, medium-high emissions scenario. After Whitehead et al. (2006), with permission.

Figure 11. Conceptual structure of the Euro-impacs Decision Support System.
Fig. 1. Conductivity, magnesium, sulphate and calcium concentrations in the water of two high alpine lakes during the past 20 years (Rasass See: red dots, Schwarzsee: open circles). Note the break in the concentration axis due to the high change in solute concentrations. (Modified from Thies et al., 2007).
Figure 2  Cause-effect chain illustrating the potential impact of changes in climate on stream biodiversity.
Figure 3. DCCA ordination diagram for the hydromorphological gradient analysis with the morphological gradient indicated by black (canalised), grey (semi-natural) and open (natural) labels and with the hydrological periods indicated by circles (pre-hydrological restoration) and triangles (post-hydrological restoration).
Figure 4. Average maximum and minimum oxygen concentrations recorded during three 24-hr experiments carried out in a mesocosm system in late June and July 2007. For each treatment there were eight replicates. Both heating and addition of nutrients significantly reduced both maximal and minimal concentrations (P<0.01). Unpublished data of C.Whitham, H.Feuchtmayr, D.Atkinson, R.Moran, I.Harvey and B.Moss.
Figure 5. Top panel: ANC (volume-weighted annual mean) concentrations in runoff at nine sites (eight in Europe, one in Canada) for the calibration year (2000) and predicted (using the dynamic model MAGIC) for the year 2030 assuming no climate change (base scenario). Bottom panel: Change in ANC predicted for the year 2030 relative to the base scenario (Wright et al. 2006).
Figure 6. Lake-averaged muscle concentrations (n = 12-14) of selected PCB and PBDE congeners vs reciprocal of annual mean air temperature.
Figure 7. Scheme of integrating climate change effects into a multimetric assessment system for river invertebrates. Nine metrics, which are individually assessed by comparison to reference conditions, indicate four different stressor types: Organic pollution (Saprobiic index), hydromorphological degradation (Fauna Index, [%] alkal preferences, [%] gatherers, [%] stoneflies, [%] rheophile taxa), acidification (acid index), temperature (longitudinal zonation index, [%] cold stenotherms).
Figure 8 Conceptual diagram illustrating ecosystem response to increasing and decreasing stresses in relation to climate change, expressed as a changing baseline (from Battarbee et al. 2005).
Figure 9. Conceptual diagram illustrating a comparison of expected and observed responses of macroinvertebrates (expressed as taxon richness) in acidified and reference lakes in S. Sweden to a reduction in sulphur deposition.
Figure 10. Modelled nitrate concentrations in the lower River Kennet, UK, 1960 – 2100, given various management treatments (see text) and downscaled climate scenarios from the HadCM3 GCM, medium-high emissions scenario. After Whitehead et al. (2006), with permission.
Figure 11. Conceptual structure of the Euro-limpacs Decision Support System.