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Preparing for climate change impacts on freshwater ecosystems (PRINCE): literature review and proposal methodology

Science Report: SC030300/PR

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Authors:

Dr Kieran Conlan[1]
Professor Stuart Lane[2]
Professor Steve Ormerod[3]
Trevor Wade[1]

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Research Contractor:

[1] Cascade Consulting
Enterprise House, Manchester Science Park, Lloyd Street North, Manchester M15 6SE. Tel: 0161 227 9777

Environment Agency's Project Manager:

Dr Harriet Orr, Environment Agency Research Fellow (Climate Change), Hysed, Geography Department, Lancaster University, Lancaster LA1 4YB

Collaborator(s):

[3]Cardiff University, Department of Biology, Ecological Processes
[2]Durham University, Department of Geography

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Project record Preparing for climate change impacts on freshwater ecosystems (PRINCE): Literature review and proposal methodology

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Steve Killeen
Head of Science

Executive Summary

Introduction

Preparing for Climate Change Impacts on Freshwater Ecosystems (PRINCE) was commissioned by a consortium led by the Environment Agency in partnership with English Nature and the Countryside Council for Wales.

The key objectives of the project are to:

i) review information and understanding of the implications of climate change for freshwater ecosystems;
ii) inform a wide range of policies about ecosystem management implications;
iii) communicate an improved understanding of climate change effects;
iv) apply this to predictions of consequences in selected ecosystems.

The fundamental approach is to determine potential climate change impacts on freshwaters, from the perspective of both the physico-chemical (abiotic) drivers of ecosystem function together with the process-based ecosystem (biotic) interactions that influence habitat, assemblage and/or species functioning and dynamic evolution.

PRINCE addresses climate impacts on freshwater ecosystems and will provide information for UK Agencies on the likely future impacts and implications for the management of freshwaters. The approach incorporates responses to changes in the mean and variability of the future climate (both high and low rainfall, river flows, etc.).

Climate change scenarios

A number of general circulation models (GCMs) are currently available to simulate future climate, each of which gives differing results, particularly at regional scales. To date, the UK Climate Impacts Programme (UKCIP) 2002 scenarios have been applied most widely, and for consistency form the basis of the current analysis. It is acknowledged, however, that these scenarios tend towards drier future predictions relative to other GCMs. With this in mind, data provided by UKCIP demonstrate that under medium–high emissions there will be more hot and dry summers, with over half of the summers by the 2080s similar to the hot, dry summer of 1995.

The significance of this for freshwater ecosystems is that drought conditions would be far more prevalent, particularly in south-east England. There would also be a shift to a greater proportion of wetter winters such as that experienced in 1994/1995, particularly in the north and west, although the shift is less pronounced than the projected increase in summer drought. Groundwater recharge would also be reduced and flows in groundwater-dominated rivers are projected to decrease throughout the year. The proposed research will use downscaled scenarios projected for the 2020s and 2050s to explore impacts on selected ecosystems.

Ecosystem typology

A freshwater ecosystem typology was developed for the PRINCE project to group freshwater environments that may have similar climate sensitivities. The typology drew on international and national regulations and typologies where relevant. Three classes were identified:

- **rivers and streams**: upland and lowland headwaters, middle and lower river reaches, and predominantly groundwater-fed rivers (chalk or sandstone);
- **standing open waters and canals**: lakes (eutrophic to dystrophic), ditches, ponds, canals and reservoirs;
- **wetlands and washlands**: grazing marsh and improved grasslands on floodplain, lowland raised bog, fens, reedbeds and wet woodlands.

Key non-biological factors that influence ecosystem dynamics

- Potential changes in climate are likely to influence the aquatic ecosystem in two ways, by episodic pulsed effects (i.e., changes in the frequency, duration and magnitude of extreme events) and by progressive change in average conditions.

Thermal effects

Many freshwater species are sensitive to the water thermal regime and often have a limited range of temperature tolerance. Thermal regime is also critical to the life cycle of a wide range of aquatic organisms, and often creates the spatial separation of closely related species. Along with features such as velocity or depth preference, thermal regime has been a major influence on the evolved traits of organisms. Thermal regime may also influence:

- biotic vital rates (metabolism, respiration and/or production), often in conjunction with water velocity and temperature-dependent concentrations of dissolved oxygen;
- life cycles and inter-relationship of a wide range of organisms from the invertebrates, amphibians, fish and birds that are controlled by temperature and/or by the seasonal hydrograph;
- linkages, dispersal or migrations across ecosystems, for example between marine systems and freshwaters by long-distance migrants (Atlantic salmon, eel, shad), or across watersheds during inter-basin dispersal flights by invertebrates;
- introduction, survival and population dynamics of exotic organisms, such as non-native species.

Water quantity effects

As well as effects on annual flow regimes and individual event hydrographs, changed water quantity will also modify velocity profiles, hydraulic character, water levels, inundation patterns, the cyclical changes in the amount of available water and changes in wetted perimeter. All of these can influence local habitat conditions, availability and connectivity. In addition, the frequency, timing and magnitude of extreme events of both low and high discharge will change markedly in future climates. The consequences for habitat configuration, availability and quality will depend both on local conditions and on the habitat distribution or adaptability of affected organisms.

Water-quality effects

Most potentially significant water-quality effects will be associated with hotter summers and lower rainfall. During these the reduction in dissolved oxygen carrying capacity and higher biochemical oxygen demand (because the effluents will be diluted less and there will be more algal and/or microbial growth) are likely to combine to produce increased risk of deoxygenation. This may be particularly important in middle and lower river reaches and standing waters, in which re-aeration may be reduced. Increased macrophyte growth encouraged by higher water temperatures and non-limiting nutrient supply could further reduce oxygen levels and pose threats to fish and invertebrates. It is likely that changes in sediment mobility driven by climate change will affect the transport and deposition of a wide range of solutes and pollutants. Higher flows and lower water temperatures should combine to limit water-quality impacts in winter, although increased storminess may increase run-off of contaminants into watercourses.

Solar radiation effects

Solar radiation impacts directly on the timing of specific events (phenology). Changes in solar radiation dictate the time sequences in, for example, algal growth and rates of production. These responses can have implications for higher parts of the food web, such as zooplankton and fish. There is potential for food supplies to become out of sequence when, for example, young fish hatch before their food supply is available.

Ecological sensitivity to climate change

All of the freshwater ecosystems considered are sensitive to climatic change, either directly (e.g., through temperature and rainfall-mediated effects) or indirectly through hydrological, water quality or competing ecological impacts. Sensitivity of freshwater ecosystems to climate change will depend on two major components. Firstly, on the magnitude, frequency (return period), timing (seasonality), variability (averages and extremes) and direction of predicted climate changes. Secondly, on the sensitivity and resilience of the ecosystem, habitat and/or species to that change. In general, ecosystems at the extreme range of their supported environmental conditions will be most affected. For example, these include changes in distribution for organisms sensitive to:

- increased temperature in low-temperature habitats of restricted distribution (e.g., cold-water stenotherms at high altitude);
- major increase in drought frequency (e.g., salmonids that require spawning and juvenile habitat in upland tributaries, species within lowland raised bog in the south-east);
- increased flood and erosion risk in flood-sensitive habitats (e.g., floodplain grazing marsh and reedbed).

It is difficult to rank freshwater habitats in terms of sensitivity as their resilience to the multi-factoral stresses on each will not be the same and may combine in differing ways. However, it is clear that habitats that are already at the extremes of hydrological connectivity, such as headwaters, ditches and ephemeral ponds, are sensitive to changing climatic conditions. The most resilient may be middle and lower river reaches and large lakes that have some buffering against climatic influence through greater water depths. Heavily managed water bodies, such as reservoirs and canals, will be influenced, but their ecology is controlled by factors other than climate (depending on how the resource is deployed). There is currently insufficient knowledge to identify how the habitats will respond to multiple pressures.

From this review a qualitative assessment of relative sensitivity to climate change suggests the following (in order of most to least sensitive): upland and lowland headwaters; lowland raised bog; ephemeral ponds; groundwater-fed rivers; ditches; all lakes; fens; floodplain grazing marsh and grassland; lowland rivers; middle rivers; wet woodlands; reedbeds; reservoirs and canals (for which operational management is a key consideration).

Proposed ecosystem modelling framework

Prediction of future climate change impacts on freshwater ecosystems will require simulation of climate-mediated conditions on a range of complex aquatic physico-chemical and biological processes. A review of the available techniques identified two suitable modelling approaches:

- empirical and/or statistical methods that relate biological and non-biological factors;
- deterministic modelling methods that predict changes in non-biological factors to assess changes to selected habitats, communities and/or target species.

The next phase of the PRINCE project will:

i) explore the potential for climate change effects on selected freshwater ecosystems;
ii) secondly, establish the utility of each of the modelling approaches for wider application to UK ecosystems.

Locations at which to trial the approaches have been determined largely by data availability for model calibration and verification.

Initial studies will test the influence of climate change on ecosystem drivers, including hydrology, hydrochemistry, sediments and thermal regime. The upper Wharfe catchment (Yorkshire) will be modelled using the deterministic CAS-Hydro framework. The Flood Estimation Handbook (FEH) approach to flow derivation will also be applied to compare results from CAS-Hydro with simpler statistical methods. Climate scenarios will be produced for the 2020s and 2050s using a range of GCMs to capture uncertainties that result from the choice of climate model and the downscaling approach.

CAS-Hydro outputs will be used as inputs for empirical models of in-channel macroinvertebrate dynamics and to estimate useable habitat availability for macroinvertebrates and fish. The resultant climate-driven changes to the aquatic environment will be analysed to assess implications for the relevant assemblages. A second study will use FEH-derived data as inputs to an empirical model of macroinvertebrates for the upper Tywi (mid Wales), using the long-term Cardiff University dataset.

Outputs of the studies will include a better understanding of climate-induced change on flows and sediment behaviour in upland headwaters and middle rivers, and associated impacts on floodplains and aquatic ecosystems.

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1 Introduction

1.1 Purpose of the report

Preparing for Climate Change Impacts on Freshwater Ecosystems (PRINCE) (Ref: SC030300 X1-045/3) has been commissioned by a consortium led by the Environment Agency and including English Nature and the Countryside Council for Wales (CCW). The contract is being undertaken by:

- Cascade Consulting (lead consultant);
- Department of Geography, Durham University;
- Department of Biology, Ecological Processes, Cardiff University.

This document is the Project Record. It is intended as a working document to inform the Environment Agency project management team and enable engagement with the Project Steering Group.

The Project Record provides a desk-based investigation of the sensitivity to key drivers (especially climate change) of an inventory of freshwater ecosystems in England and Wales and the causal relationships between the hydrological, physicochemical and ecological interactions and consequent ecosystem changes that may result from climate change effects.

The report proposes a number of potential process-based modelling approaches for a shortlist of sensitive habitats in freshwater ecosystems. Following review by the Project Steering Group and discussion at the meeting of 26 January 2005, three habitats or representative assemblages/species have been agreed and will be taken forward to Phase 2 of the project to develop and apply appropriate process-based models. For the other sensitive ecosystems identified, research proposals have been framed for potential future studies (currently beyond the scope of this project).

This document will compliment the Environment Agency Science Report. The Science Report and a Technical Summary will be prepared on conclusion of the project.

1.2 Aims and objectives

The key objectives of the project as defined in the Specification are to:

- '… review information and understanding of the implications of climate change for freshwater ecosystems;
- inform a wide range of policies;
- communicate an improved understanding of climate change effects; and
- apply this to predictions of consequences.'

The fundamental approach is to determine the potential for climate change impacts on freshwaters, from the perspective of both the physico-chemical (abiotic) drivers of ecosystem function and the process-based ecosystem interactions that could influence habitat, assemblage and/or species functioning and dynamic evolution.

The assessment will require a modelling framework to be produced, with the ecosystem drivers (principally hydrology, geomorphology and water quality) and ecological processes developed from a clear understanding of the scientific linkages. Where linkages are unknown or unclear, there may be the opportunity to build on existing field measurement programmes or develop new studies to investigate specific process interactions.

All aquatic ecosystems of concern in England and Wales will be reviewed in the initial analysis of scientific data, including rivers, lakes, canals, riparian wetland, etc. However, at this stage the testing will be restricted to modelling three freshwater ecosystems.

The project specification issued with the Tender Documentation is included as Appendix 1.

1.3 Introduction to PRINCE

Potential impacts of climate change have been explored in the British Isles for a range of activities and receptors. These include water resources, flood defence and biodiversity assessments (e.g., the UK MONARCH programme; the European Union's (EU) Climate, Hydrochemistry and Economics of Surface-water Systems (CHESS; Boorman, 2001); Climate impacts on Biodiversity Action Plans relevant to Scotland (Brooker, 2004); Foresight Future Flooding (Office of Science and Technology, 2004), etc.). Many of these studies have been underpinned by the Department of Environment, Food and Rural Affairs' (Defra's) climate change scenarios (UKCIP02). These are detailed, for example, in the UK Climate Impacts Programme 2002 climate change scenarios for flood and coastal defence (R&D Technical Report W5B-029/TR, dated April 2003, on http://www.defra.gov.uk/environ/fcd/pubs/pagn/Climatechangeupdate.pdf).

These and other international studies have identified a range of potential risks to both terrestrial and aquatic ecosystems. In the UK, under all greenhouse gas emissions scenarios the climate will become warmer with increased potential for high summer temperatures, wetter winters and greater likelihood of heavy winter precipitation. In certain scenarios there will be increases in summer low river flows and increased incidence of drought (see Section 2). Relative sea level will continue to rise around most of the UK's shoreline, with extreme sea levels experienced more frequently (Hulme *et al.*, 2002). Despite the uncertainties inherent in future emissions scenarios and even greater uncertainty in the short term between model predictions, there is growing acceptance that the temperature rise of the 20th century has been accompanied by the fastest rate of ecosystem change for 1000 years (van Vliet and Schwartz, 2002).

To date, there has been a lack of data and research on the long-term behaviour of rivers and relatively little work on future climate impacts on lakes and other wetlands. A raft of recent major EU projects (e.g., Euro-limpacs (impacts of global change on European freshwater ecosystems), CLIME (climate impacts on freshwater lakes), RECOVER2010 (recovery of acidified freshwaters), SWURVE (sustainable use of water), ACCELERATES (vulnerability of agro-ecosystems to environmental change) and AquaTerra (integrated soil and water research)) should help to address some of these needs (Wilby *et al.*, in press). Work in the UK in recent years by UK Water Industry Research (UKWIR) has also added considerably to the collective understanding of climate change, with a particular emphasis on catchment hydrology and water quality (Research Topic CL/06) and the implications for water industry infrastructure (Research Topic CL/04). However, the majority of the effort to date has focused on the physico-chemical aspects of the hydrological cycle, with relatively little research into the potential for ecological effects.

PRINCE therefore responds to a specific research need identified in the MONARCH programme to take a different approach to exploring climate impacts on freshwater ecosystems and to provide guidance to UK Agencies on the likely future impacts and implications for the management of freshwaters. The approach proposed incorporates an assessment of the potential variability of the climate (both high and low rainfall, flows, etc.) as well as the more general (averaged) climatic changes. The programme will take a strategic approach to define the framework for predicting climate change impacts, with specific modelling of identified vulnerable ecosystems.

Defra is proposing to undertake a review and update of the Department for Transport, Local Government and the Regions (DETR) and Ministry of Agriculture, Food and Fisheries (MAFF) publication *Climate Change and UK Nature Conservation* by 2006. This will involve looking at the vulnerability to climate change of all Biodiversity Action Plan (BAP) priority habitats and freshwater species, which PRINCE will also inform.

1.4 Legislative and policy drivers

Legislative requirements and national policy initiatives provide a framework for the research, recognising that any significant changes to ecosystems in response to climate change are likely to have implications for both pre-existing and future legislative and policy directions.

Key legislative and policy drivers considered in this context include:

- Ramsar Convention (1971)
- EU Water Framework Directive (2000/60/EC)
- EU Habitats Directive (92/43/EC)
- EU Birds Directive (79/409/EC)
- Wildlife and Countryside Act (WCA; 1981, 1985)
- Countryside and Rights of Way (CROW) Act (2000)
- UK Biodiversity Action Plan (BAP)

Each legislative or policy driver has specific requirements for the protection and/or enhancement of defined ecosystem types (see also Section 3.1). For example, the EU Water Framework Directive is an overarching requirement to achieve ‘good ecological status’ in all water bodies (other than heavily modified or artificial bodies, where ‘good ecological potential’ will be required). The EU Habitats Directive has specific requirements to protect identified habitats and species in designated areas (called Special Areas for Conservation (SAC), with a similar raft of requirements for nature conservation within the UK WCA and CROW legislation.

If climate change is predicted to change the composition and/or distribution of protected habitats and species, the current approach to defining nature conservation areas with static boundaries (e.g., cSACs and SPAs) may need to be reconsidered or updated in response to the future change in ecosystem dynamics. This is already being seen to some extent for coastal wetlands on the south coast (e.g., the Solent) where saltmarsh and mudflat are reducing through sea level rise and coastal squeeze. The extent of water-dependent habitats notified for their conservation importance that require protection is illustrated by their distribution in Wales (*Figure 1.1*). These locations were identified either from those:

- that hold priority aquatic species living in, or dependent on, surface waters for the whole or part of their life cycle (e.g., freshwater pearl mussel);
- that consist of surface water (e.g., oligotrophic waters; *Ranunculus* habitat), depend on frequent inundation by surface water or depend on groundwater levels (e.g., alluvial alder wood, wetlands, blanket bog, fens).

The sites are not only geographically extensive, but also they occupy both upland and lowland locations where future climate regimes will differ markedly from now. Other regions will be similar. In this respect, all protected freshwater sites may be affected by climatic change.

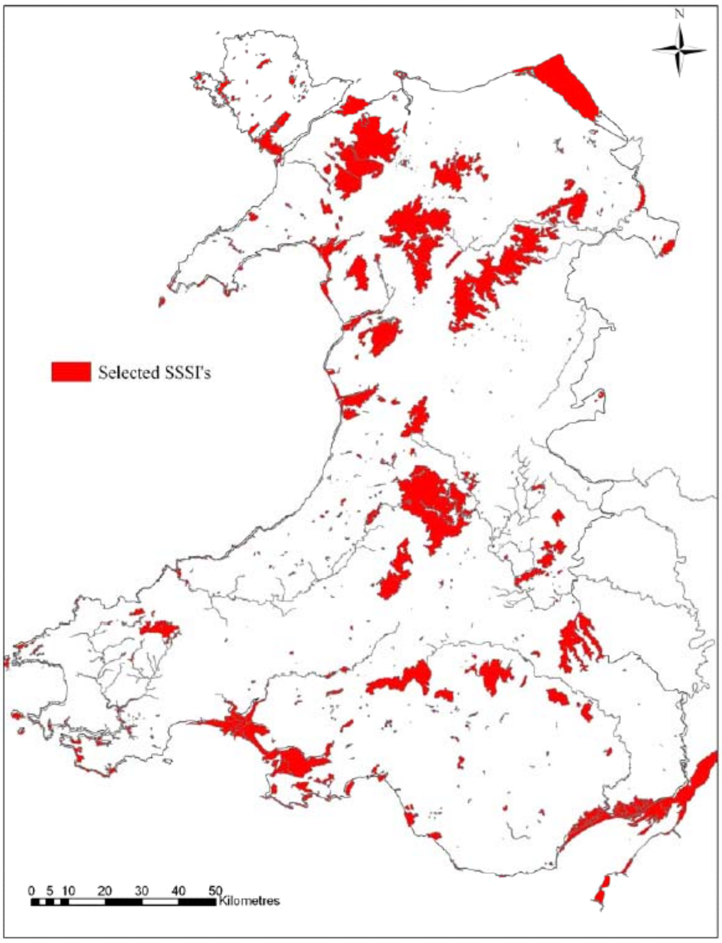

Figure 1.1 Examples of water-dependent habitats notified for protection under conservation legislation in Wales (after Reynolds *et al.*, 2004).

1.5 Study constraints

The project has been promoted through the need to develop improved understanding of the potential implications of climate change on sensitive freshwater ecosystems. There are many gaps in the present state of knowledge of the interactions between the environmental processes influenced by climate change and their implications for freshwater habitats and species. Similarly, it is widely accepted that there are few good-quality long-term ecological datasets that could establish a realistic baseline description and from which future scenarios could be developed. Study tasks have therefore been challenging and have been undertaken in consideration of a number of constraints:

- The initial project has resource constraints; development will necessarily focus on the modelling of a limited number of systems indicated by the literature review. Development of modelling approaches in this first iteration will be constrained by the budget available.
- Paucity of long-term high-quality datasets on ecosystem variables and understanding of the pathways by which key environmental drivers influence habitat viability limits the range of systems that currently can be investigated by modelling.
- A number of modelling approaches may be applicable. The study will evaluate the appropriateness of the most promising methods, allowing future concentration on a preferred set of techniques. It will not be possible in the first iteration to undertake the full range of possible approaches.
- Groundwater-dominated catchments continue to present challenges. Groundwater inflows and quality (that may be required as input terms for catchment hydrological models) are difficult to predict at a suitable resolution for ephemeral systems using existing groundwater models. At present the computational requirements of groundwater models has precluded their use for the high-resolution catchment-scale simulations that would be required to interface with the fully distributed catchment model likely to be used in this study.
- Confidence in the correct range of potential future climate change outcomes remains elusive (see, e.g., Jenkins and Lowe, 2003), and potentially require sensitivity testing (and therefore a greater range of scenarios) with the consequent budgetary implications.

However, having stated the above constraints, the modelling framework proposed for Phase 2 should represent a significant step forward in our understanding of the interactions of climate change and freshwater ecosystem function. The findings of the literature review, the recommendations for further work to fill the scientific and technical gaps identified, and the outputs of the modelling programme will all add to the scientific knowledge base. Further, the development of the climate change drivers and habitats matrices will provide a useful framework within which future climate change research in this area could be concentrated.

1.6 General framework for the assessment

It is proposed to undertake the prediction of potential climate change on freshwater ecosystems within the framework described in the UK Climate Impact Programme (UKCIP) and the Environment Agency report on *Climate Adaptation: Risk, Uncertainty* *and Decision-making* (Willows and Connell, 2003). As noted above, the findings of the proposed research will eventually feed into the decision-making process on the measures necessary to adapt to climate change, and it is therefore sensible to position the research in an accepted decision-making framework. The general approach is shown in *Figure 1.2*.

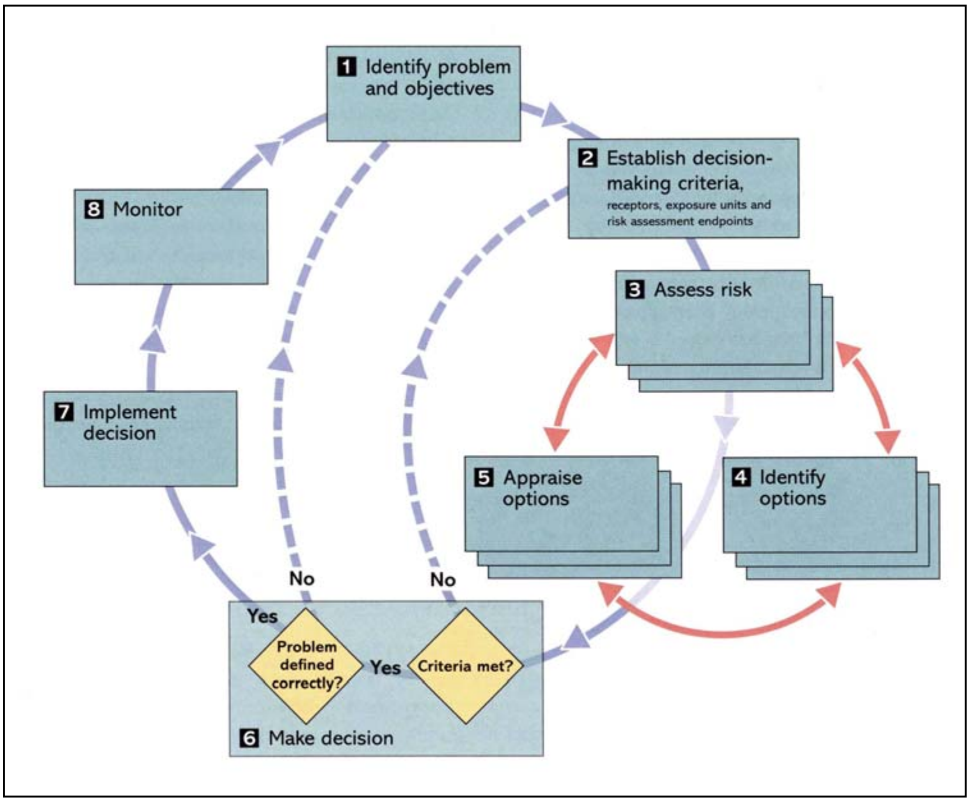

Figure 1.2 Framework to support decision-making in the face of climate change risk (taken from Willows and Connell, 2003).

It is anticipated that this research will inform boxes 1 to 3 (*Figure 1.2*), culminating in recommendations for option assessment. The options may range from changes in policy and/or legislation to counter adverse effects of climate change, through to specific guidance when implementing projects (e.g., flood management schemes). This Project Record is intended to identify the problem (box 1), from which the objectives and decision-making criteria (box 2) can be developed, leading to the initial risk assessments of three habitat types (box 3).

An important component of the studies will be to recognise the risks and uncertainties inherent in studies of this type, which are explored more fully in Section 2.

The end point for this Project Record phase will be to:

i) establish if there are potential problems for freshwater ecosystems from climate change;
ii) specify the objectives of the research (and, potentially, of the more comprehensive climate adaptation framework (boxes 1 to 8 rather than 1 to 3, which is the remit of these studies));

iii) identify decision-making criteria (receptors and risk assessment endpoints);
iv) specify studies to undertake risk assessment.

The risk assessment will be established through the general principles outlined in Willows and Connell (2003). A tiered approach will be adopted based on a risk prioritisation, with an initial screening phase (Tier 1) and generic quantification (Tier 2) reported in this Project Record (see *Figure 1.3*). The Tier 3 quantitative risk assessment will form the basis of the subsequent modelling activities.

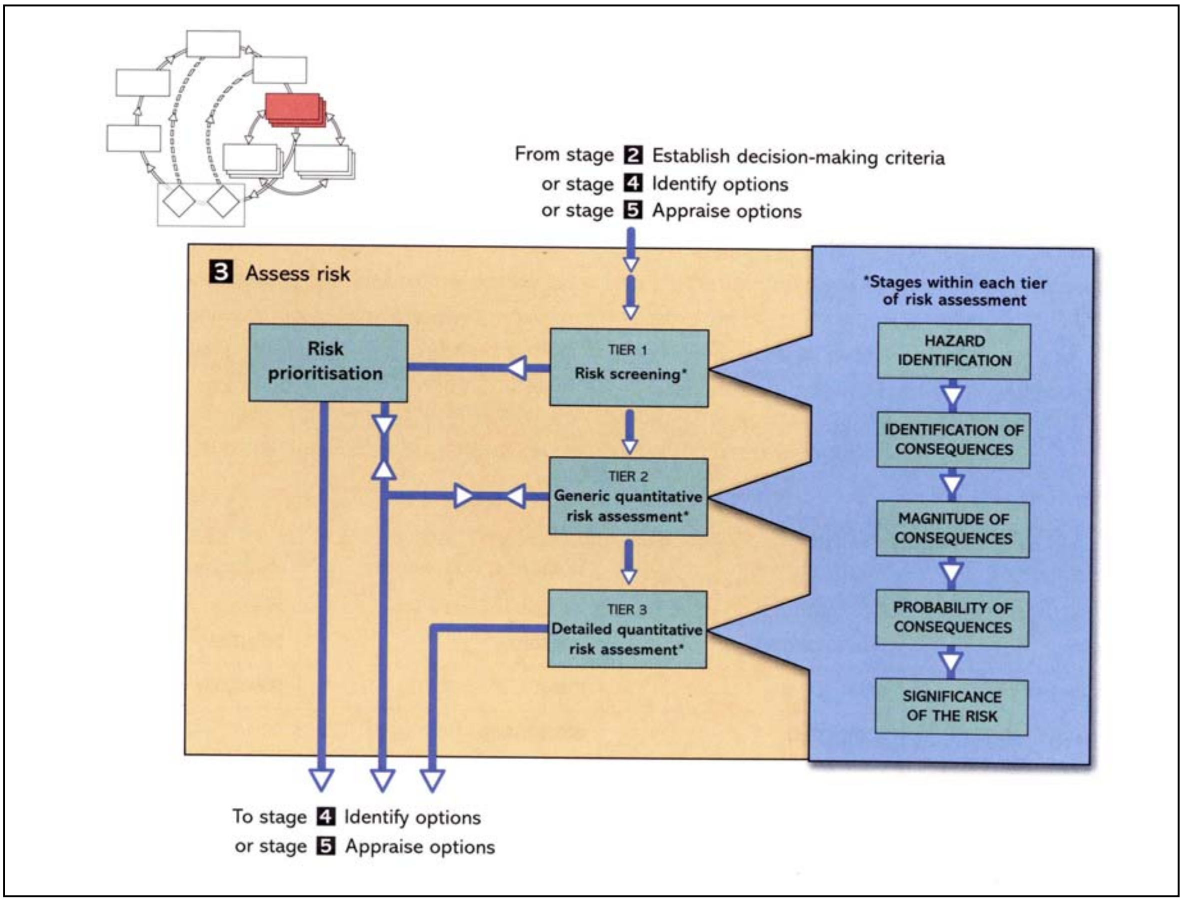

Figure 1.3 Overview of the stages within, and purposes of, each tier of risk assessment (taken from Willows and Connell, 2003).

1.7 Report contents

Section 1 of the Project Record introduces the objectives of the study and defines the risk-based framework for exploring climate change impacts on freshwater ecosystems through the PRINCE programme, within which the research will be based. Section 2 describes the range of future climate change scenarios being considered in the UK at present and the range of climatic and hydrological outcomes, together with the uncertainties that surround the projections. Possible regional variability in climatic parameters is described, with a proposal for the approach to be adopted for the PRINCE research.

Section 3 establishes the freshwater ecosystem typology proposed for the PRINCE programme. The typology is specific for climate change research and incorporates, where practicable, the various ecosystem typologies currently being applied in a UK policy context, including the Water Framework and Habitats Directives and UK BAP system. The typology provides a framework for the research within which abiotic and biotic influences can be described and the relative sensitivities of the sub-types to climate change can be identified.

Section 4 gives a brief overview of the key abiotic factors influenced by climate change, and describes the inter-relationship of each parameter for the three hydrological domains identified in Section 3 (rivers and streams, standing open waters and canals, wetlands and washlands). When combined, the outputs of Sections 3 and 4 provide the strategic framework to analyse all UK freshwater ecosystems that may be subject to climate change impacts. The framework can be considered as a matrix of hydrological domains, with a number of sub-types within each domain, which can be assessed against the key abiotic sources and pathways of potential climate change (such as temperature, flow, etc.).

The review of freshwater ecosystems vulnerable to climate change is reported in Section 5. The section describes the current research efforts in this area and the availability of -literature related to climate change, before establishing the range of freshwater ecological receptors that may be influenced by climate change. The review then determines for each of the three hydrological regimes and habitat sub-types the research that has been undertaken and, where possible, the relevance to climate change outcomes. It has established that the research programme currently adopted has many gaps, and fundamental science relating to the interactions of ecological communities (abiotic and/or biotic) is scarce. The range of freshwater ecosystems is ranked according to potential risks from climate-induced change, using the collated knowledge from the literature search and applied expert judgement.

Section 6 defines the current state of knowledge in process-based modelling of freshwater ecosystems. The review identifies a range of potential modelling approaches and describes the advantages and disadvantages of each before determining the key issues to consider when assessing ecosystems for model application.

The proposed modelling approaches are suggested in Section 7. A modelling framework is described that includes two modelling approaches to be applied to a limited number of freshwater ecosystems. The purpose of the research is firstly to identify potential climate change impacts on the studied ecosystems, and secondly to derive, if appropriate, the ‘proof of concept’ for each of the modelling approaches. If the approaches are suitable they could be used subsequently on other ecosystem types or at different geographical locations to give a more thorough assessment of potential climate-mediated changes (spatial differences in ecosystem types and climate scenario outputs may be considered in the future).

The appendices specify the Terms of Reference for the project (Appendix 1) and supplementary information to support the typology definitions (Appendix 2).

2 Climate change scenarios

2.1 Key climate variables for freshwater ecosystems

It is generally accepted that climatic conditions, in association with ecological interactions, are key drivers of ecosystem processes and dynamic evolution. The key climate variables of concern to ecosystem function are:

- rainfall (precipitation)
- air temperature
- solar radiation (cloud cover)
- wind speed
- humidity.

These variables are consistent with the suggested suite for the study of climate change risks in Willows and Connell (2003), and are available as outputs from the climate change models under consideration. The variables may have a direct (e.g., temperature) and/or consequent (indirect) effect (e.g., temperature, rainfall, etc., on soil moisture) on key hydrological and landscape features of concern for aquatic ecosystems.

The relationship between the climatic variables and the wider suite of physicochemical variables that may influence ecosystem response are described in Section 4.

2.2 Uncertainty in climate change predictions

There is wide acceptance that anthropogenic activity is leading to global changes in climate through increased concentrations of greenhouse gas (IPPC, 2001). Although there is general consensus that the climate in the UK is changing and that the rate of change is likely to be influenced by ongoing anthropogenic activity, it is not possible to predict with any certainty the likely future climate in the next 50-100 years.

There remains substantial and possibly growing uncertainty of the most likely climate change outcome over the medium to long term (Jenkins and Lowe, 2003; Murphy *et al*., 2004). In particular, the uncertainties will preclude firm predictions. They stem from uncertainties over trends in emissions, uncertainties over the scientific understanding of the climatic processes involved, uncertainties about how these will disaggregate regionally and uncertainties generated by natural variability. Moreover, projected outcomes from equally probable futures diverge substantially in the longer term. Exemplifying these uncertainties, future model projections for the 2080s include:

- global temperature over the next 100 years increasing by between 1.5 and 5.5°C;
- summer mean temperatures for the UK increasing by 3°C to >8°C depending on location;
- global mean precipitation increasing by between 2 and 7 per cent;
- winter precipitation for the UK increasing by 1-60 per cent and summer precipitation changing from less than –30 per cent to more than +4 per cent, depending on the models used;

- global average increases in sea level of 10-90 cm, while thermal expansion alone could increase the North Sea level relative to the UK by 0-50 cm.

It is clear, therefore, that although climate change is happening, the absolute definition of what the changes may mean in terms of climatic conditions, and hence impacts on freshwater ecosystems, must be treated with caution. The study is being progressed in consideration of these uncertainties, with a view to sensitivity testing the outcomes to establish the range of possible futures (see Section 2.5).

2.3 Current climate change models in the UK

Of the many climate change models currently being used or developed internationally, four global climate models (GCMs) are currently being used for the UKWIR strategic water resource planning project. These models are noteworthy as they have archives of daily variables for two or more emissions scenarios from 1961 to 2100:

- UK Hadley Centre HadCM3 – from which the UKCIP 2002 scenarios were developed;
- Canadian CGCM2;
- Australian CSIRO model;
- European Max Plank Institute (MPI) ECHAM4 model.

Each of the models uses a number of different underpinning assumptions to drive its climate simulations, which results in different predictions of summer and winter climatic conditions. At the most generic of levels the Hadley model predicts the driest summers, followed by dry summers in ECHAM4, with increasingly wetter predictions from the CSIRO and CGCM2 models. All of the models agree that winters will be wetter, although the magnitude of change remains unresolved. Significant scientific research and debate continues into the ways of converting model output and the most suitable suite of model scenarios to use, and it will be some time until a consensus is reached on the most reliable predictive scenarios to use.

In the UK, the majority of the studies to date have used outputs from the suite of Hadley models, which have been used to derive the UKCIP 1998 (Hulme and Jenkins, 1998) and more recent 2002 climate change scenarios (Hulme *et al*., 2002). A number of greenhouse gas emissions scenarios have been simulated that correspond to potential future conditions ranging from low to high emission levels. Outputs have been derived for time-slices of 30 years centred on the 2020s, 2050s and 2080s. Clearly, the further into the future that the predictions are modelled the greater the level of uncertainty in the predicted outputs (see, e.g., Hulme *et al*., 2002). Also, although often used for UK climate change scenario development, the UKCIP02 scenarios do not capture the full range of possible climate futures.

2.4 Regional and seasonal variability in predicted climate

Accepting the limitations to the predictive outcomes stated above, work to date in the UK indicates that there will be both regional and seasonal variability in the rate and influence of climate change. Having considered the caveats on the confidence that can be ascribed to using single CGM outputs to simulate potential future climate (e.g., Jenkins and Lowe, 2003), it is illuminating to consider the future climate outcomes that may be produced. To frame a relatively severe potential outcome, the UKCIP02 medium–high emissions scenario has been used, projected to the 2050s, to provide an indication of the sensitivity of freshwater ecosystems vulnerable to climate change in this study.

Relevant abiotic factors associated with climate are introduced in Section 4 as potential changes in rainfall, air temperature (as mean and diurnal range), humidity, wind speed and solar radiation (as cloud cover). These factors are all addressed by UKCIP (Hulme *et al.*, 2002), including regional and seasonal perspectives, which are included in *Figure 2.1*.

These modelling outputs provide a general indication of the potential regional and seasonal changes. This is reasonable for general shifts in ecological function, but the extremes may be more significant for habitat change and species distributions.

The UKCIP02 scenarios provide a considerable amount of new information on future changes in daily climate, which is important for analysing possible changes in extreme events. These include the number of ‘extremely’ warm days, changes in number of ‘intense’ rainfall days, changes in precipitation for a range of return periods (2-20 years) and changes in daily mean wind speeds for a range of return periods (2-20 years). A selection of the extremes is included for the two future time-slices in *Figure 2.2*. It should be recognised that the predicted weather in *Figures 2.1* and *2.2* are based on simulations for the HadCM3 model, which is acknowledged to simulate the driest summers of the four GCM models outlined above.

To put these potential futures into context, data provided by UKCIP are presented in *Table 2.1*, which compares projected temperature and precipitation from the 2020s to 2080s (using the medium–high emissions scenario) relative to recent hot, warm and dry years, and wet winters. The figures demonstrate that with the medium–high scenario, an increasing occurrence of hot and dry summers could be expected, with over half of the summers by the 2080s similar to the drought summer of 1995, and in all cases warmer summers by 2080. The significance of this for freshwater ecosystems is that drought conditions would be far more prevalent, in terms of reduced availability of water, higher ambient temperatures, etc., particularly in the south-east. There would also be a shift to a greater proportion of wetter winters, such as that experienced in 1994/1995, particularly in the north and west, although the shift would be less pronounced than the increase in summer drought. Groundwater recharge would also be reduced and flows in groundwater-dominated rivers are projected to decrease throughout the year (Arnell, 2003).

Table 2.1 UKCIP medium–high emissions scenario projections of the potential for increases in summer and winter extreme weather conditions.

	2020s	*2050s*	*2080s*
Mean temperature			
A hot ‘1995-type’ August (+3.4^{o}C)	1	20	63
A warm ‘1999-type’ year (+1.2^{o}C)	28	73	100
Precipitation			
A dry ‘1995-type’ summer (37% drier than average)	10	29	50
A wet ‘1994/1995-type’ winter (66% wetter than average)	1	3	7

2.5 Proposals for use of climate change scenarios

Consideration has been given to the range of potential future climates and the relative lack of confidence in the prediction of which is the most likely. The Environment Agency is currently considering this issue and undertaking sensitivity tests, based on four climate models (HadCM3, ECHAM4, CSIRO, CGCM2) of relevance to the UK climate.

Use of the range of existing climate models, available emissions scenarios, future time-slices and available temporal downscaling techniques could provide an indication of this uncertainty when applied to a geographical setting (i.e., location and topography) in the UK. Such an approach to explore the envelope of uncertainty of catchment hydrology and water temperature would be very data intensive. Furthermore, interpretation of the results would not provide a clearer understanding of which climate future would be most probable and would not demonstrate the relative sensitivity of catchment hydrology and hence ecology to climate change drivers.

It is proposed therefore to use a suitably downscaled single emissions scenario from a single GCM, projected for two future time-slices, the 2050s and 2080s. Advice will be sought from the Environment Agency on which scenario would be most suitable, based on their ongoing internal research. Downscaled data will be produced for the specified catchment location using the tool described in Wilby *et al*., (2005). Sensitivity testing during model application will be used to understand the model behaviour and to estimate model uncertainty for the prediction of climate change effects.

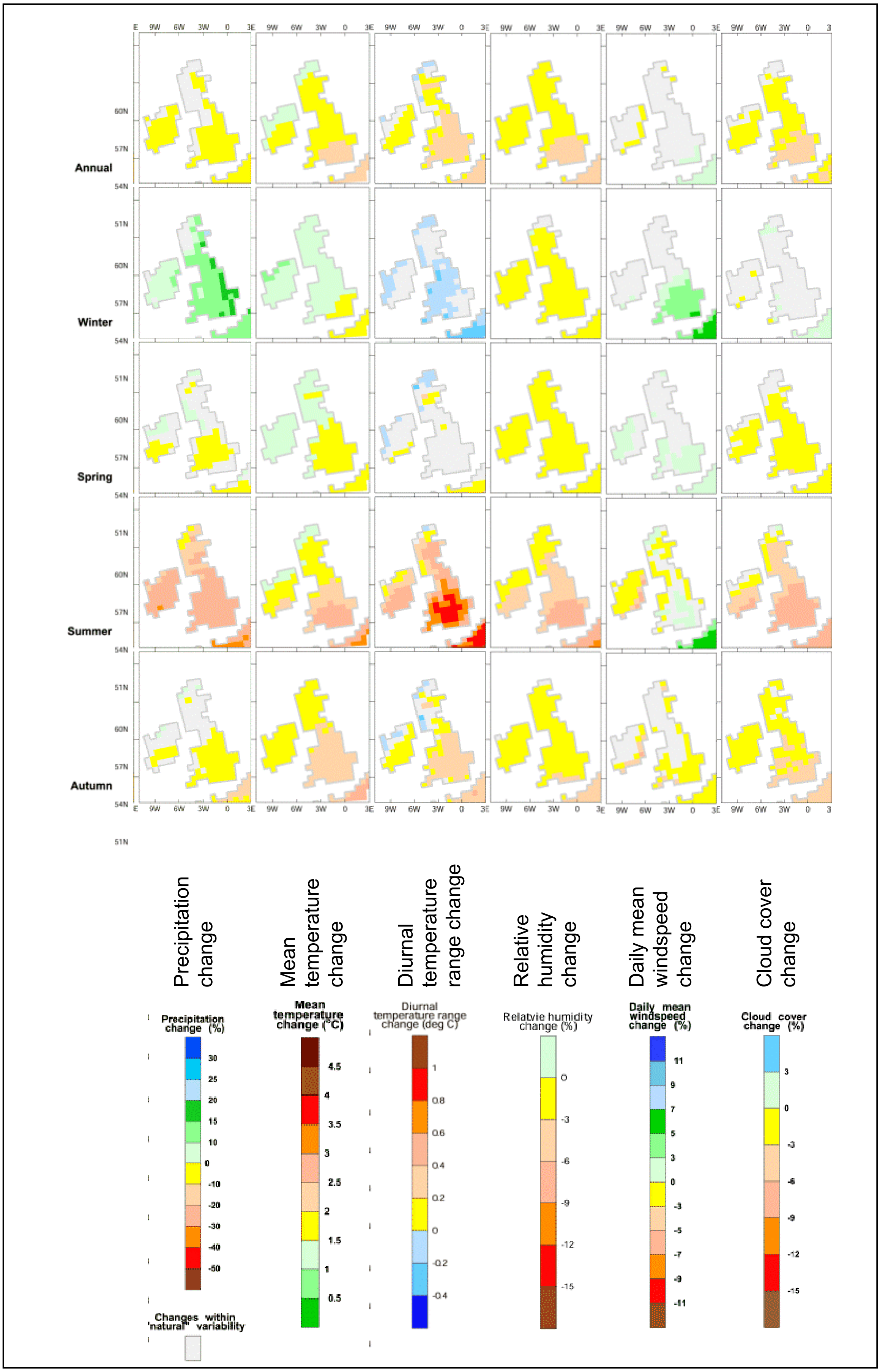

Figure 2.1 **Predicted climate change average datasets reproduced from Hulme *et al.* 2002, showing the UKCIP02 medium-high emissions scenario for the 2050s timeslice, including seasonal variation**

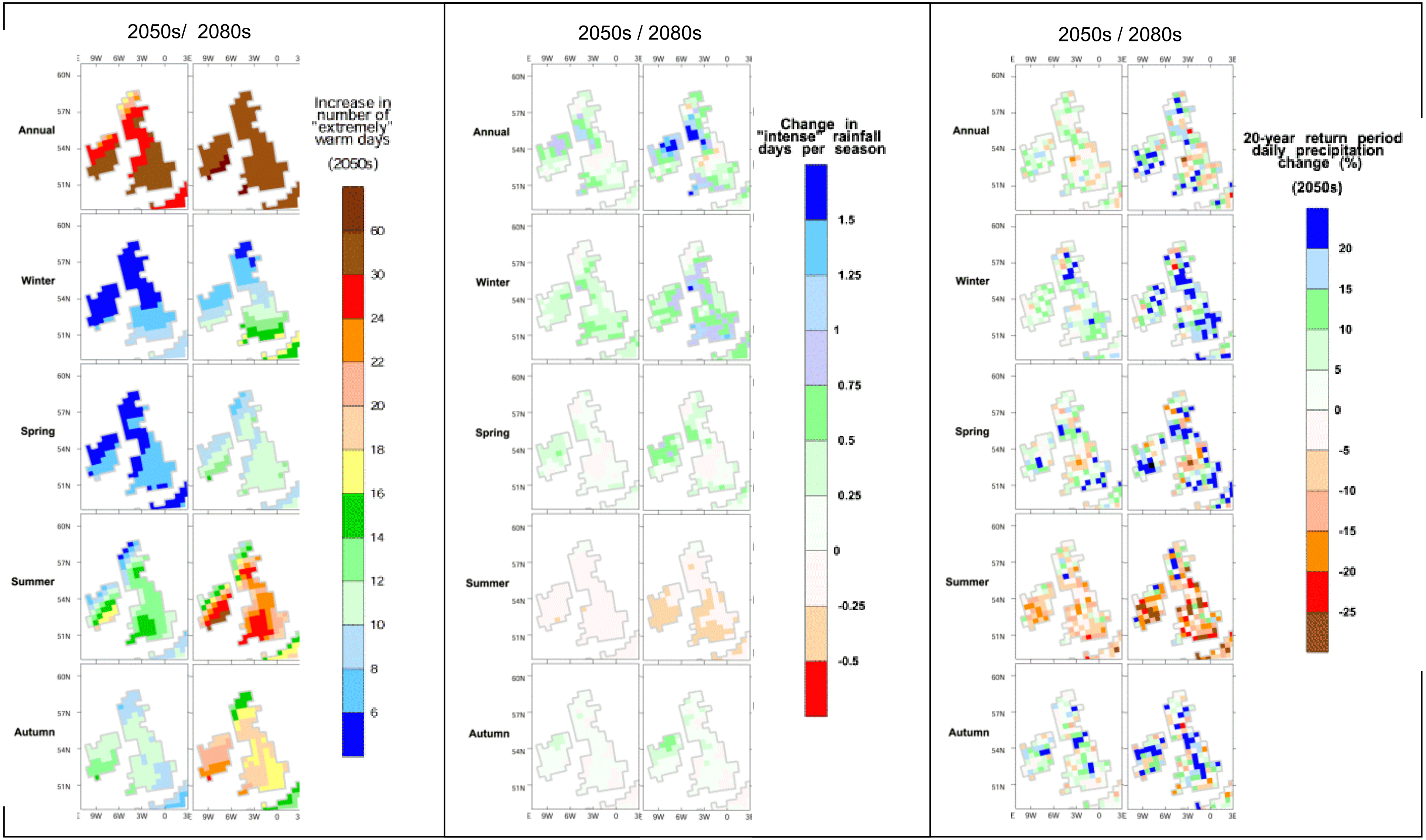

Figure 2.2 Predicted climate change daily weather and changes in extremes datasets reproduced from Hulme *et al.,* 2002, showing the UKCIP02 medium-high emissions scenario for the 2050s and 2080s timeslices, including seasonal variation

3 Establishing a freshwater ecosystem typology

One task of the study is to review the sensitivity of all freshwater ecosystems in England and Wales, within a biodiversity policy context where applicable. A freshwater ecosystem typology has been developed specifically for the PRINCE programme to provide the framework, drawing on relevant international and national regulation and typologies where relevant.

The typology has been adopted to ensure that outputs are both policy and management focused. This is important, as the differentiation into hydrologically relevant systems has to reflect how the systems are managed as well as their fundamental functioning. For example, reservoirs are separate from lakes as their management will be different and may require different adaptation strategies in the future. To arrive at a typology a number of existing classifications have been reviewed and, where possible, integrated. In some cases it has been necessary to adopt different types of classification, for example the rivers are physically based (i.e., location – headwaters to lower reaches) whereas the lakes are trophic/nutrient based (dystrophic to eutrophic). The typology will allow a systematic analysis of potential climate change impacts from which modelling approaches can be defined, if available. Gaps will also be identified for which no predictive methods are currently available.

3.1 Review of existing typologies

The review of ecologically focused legislation, policy and guidance for England and Wales identified three relevant existing ecosystem typologies, which include freshwater habitats.

3.1.1 Corine biotope classification

The Habitats Directive (Council of the European Communities, 1992; established in UK legislation through the Conservation (Natural Habitats, &c.) Regulations 1994) has the aim (Article 2) of *'contributing towards ensuring biodiversity through the conservation of natural habitats and of wild fauna and flora*' [in the European Union]. To achieve this aim habitats are prioritised according to conservation value for maintenance or, where appropriate, restoration to favourable status in their natural range, including through the designation of special areas of conservation. A range of natural habitat types is identified in Annex 1, and a range of species identified in Annex 2 of the Directive. The typology is based on the Corine biotope classification (Wyatt *et al.*, 1991), developed as a framework to compare habitats across Europe. It is a classification of biotopes as units of land with a recognisable ecological character. The typology has several hundred classes distributed between a number of higher categories. In some cases there are lists of constant and preferential species, whereas in others only a broad description is provided. The Corine biotope classification is a composite of previous classifications and has now been replaced by the EUNIS classification (see Section 3.1.2).

3.1.2 EUNIS habitat classification

EUNIS is the European Nature Information System, developed for the European Environment Agency (EEA) and the European Environment Information Observation Network (EIONET) and is their current habitat classification system. The EUNIS habitat types classification is a comprehensive pan-European system to facilitate the harmonised description and collection of data across Europe through the use of criteria for habitat identification; it covers all types of habitats from natural to artificial, from terrestrial to freshwater and marine.

Habitat type is defined within the EUNIS classification as 'plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors operating together at a particular scale.' The EUNIS classification system is hierarchical, running to five levels, and largely incorporates the Habitat Directive Annex 1 habitats (Davies and Moss, 1999-2002). Extracts of the EUNIS classification are included in Appendix 2.

3.1.3 Water Framework Directive typologies

The UK Technical Advisory Group on the Water Framework Directive (WFD-UKTAG) has prepared guidance to identify and characterise UK water bodies (including lakes, rivers, estuaries, coastal waters and groundwater) according to their physical characteristics. These guidance documents are designed to facilitate consistent implementation of the Water Framework Directive (Council of the European Communities, 2000) in the UK.

The typology for rivers (WFD-UKTAG, 2003a) uses WFD Annex 2 System A. This is based in Illies (1978) eco-regions, with descriptors and boundaries between the types fixed. There are three descriptors – altitude (three categories), catchment area (four categories) and geology (three categories – calcareous, siliceous and organic). The application of this typing system to the river network in England, Wales and Scotland provided a typology map of Great Britain that identified 18 significant types.

System A types are not necessarily biologically meaningful because of the limited range of factors included in the simple typology and the incomplete knowledge of how biology is determined by geography and/or physical conditions. The broad ecological relevance of the Great Britain river typology has been demonstrated (WFD-UKTAG, 2003a) by cross-checking against higher-level River Invertebrate Prediction and Classification System (RIVPACS) macroinvertebrate groups and macrophyte National Vegetation Classification (NVC). However, the organic river types showed wide ranges of community types with little or no differentiation biologically using the data available.

For this study there is little value in sub-typing the river hydrological domain into 18 sub-categories, as there would be insufficient resolution of the potential difference in climate change induced impacts between each. o some extent the descriptors used in the WFD typology will be incorporated at a generic level (e.g., altitude – headwater, middle, etc.; geology – groundwater, surface water fed).

A review and update of the proposed typology for the present study is recommended once the findings of the preliminary research have been identified. The greater definition provided by the WFD river typology is likely to be useful to progress some of the studies into more vulnerable habitats (e.g., size of river, catchment character, etc.) once a clearer picture of sensitivity to change and the 'proof of concept' for the approach has been established.

The draft typology for lakes (WFD-UKTAG, 2003b) uses WFD Annex 2 System B. Differentiation between types involves using 'the values for the obligatory descriptors and such descriptors, or combinations of descriptors, as are required to ensure that type-specific biological reference conditions can be reliably derived'. The draft UK typology uses alkalinity boundaries as a surrogate of geology for the catchment drainage water (five freshwater categories) and depth (two categories – ≤3 m, >3 m). An altitude descriptor (three categories) and a size descriptor (three categories, smallest >1 ha) are suggested, but not adopted as part of the core typology. The application of this typing system to the lake system in England, Wales and Scotland provided a typology map of Great Britain that identified 12 core types.

WFD-UKTAG has adopted this approach to lakes as expert judgement suggests that the proposed typology is more likely to explain biological variation than System A. There are currently insufficient biological data available to demonstrate this. The PRINCE typology adopts an alternative approach (using nutrient status) to that advocated for the WFD typology, as the purposes of each are different. As with the river typology, the differentiation into 12 sub-types is currently too detailed for the proposed approach. It is suggested, however, that incorporation of the key attributes (e.g., geology and depth) should be considered for later stages of the programme, should potential impacts of climate change be demonstrated.

3.1.4 UK BAP broad and priority habitats

A classification of terrestrial and marine habitats for the UK and the surrounding seas was published in the report of the UK Biodiversity Steering Group (Department of the Environment, 1995) as a framework for reporting on biodiversity in the UK (arising from the 1992 Convention of Biological Diversity (Rio Convention)). The scheme has been subject to review to ensure that the whole of the land surface of the UK and the surrounding sea to the edge of the continental shelf is covered. This resulted in a revised list of 27 mutually exclusive Broad Habitats that cover all the land area of the UK. A further review is on-going, due to report in 2005, with the following criteria for Broad Habitats (BRIG, 2003):

- **comprehensive** – all of the habitats types of the UK should be described within the classification;
- **exclusive** – the habitat types should be discrete to ensure that there is a 'once only fit' in the classification for each habitat encountered in the field;
- **structured** – the classification should provide a framework for organising and presenting the priority habitats that are the focus of action plans;
- **nested** – priority habitats should fit into only one broad habitat type;
- **measurable** – broad habitats should be easily recognisable, have a measurable surface and physical or biological features that are clearly characterised and wherever possible can be selected from existing systems for data collection;

- **consistent** – there should be consistency in the division of the broad habitats. The classification should not sub-divide some ecological units more finely than others.

The Broad Habitat classification aims to provide a comprehensive framework for surveillance of the UK countryside that is compatible with other widely used habitat and land cover classifications. Within the Broad Habitat type are nested a range of Priority Habitats for action plans. The series of Priority Habitats is a subset of semi-natural vegetation types for which co-ordinated conservation action across the UK is required, rather than a comprehensive list of habitats. The Broad Habitat classification provides a way to set Priority Habitats in context and a system to identify gaps and emerging new priorities in the list of Priority Habitats. Priority Habitats are identified using the following criteria:

- habitats for which the UK has international obligations;
- habitats at risk, such as those with a high rate of decline, especially over the past 20 years, or that are rare;
- habitats that may be functionally critical (i.e., areas that are part of a wider ecosystem, but provide reproductive or feeding areas for particular species);
- habitats that are important for priority species;
- a demonstrable conservation benefit from having a Habitat Action Plan (HAP).

3.1.5 Aquatic assemblages and species

It is recognised that protection through the designation of rare and/or threatened habitats and species remains a cornerstone of nature conservation, as it enables the targeting of actions and measures to meet specific objectives. However, the freshwater ecosystem typology required for PRINCE should consider the wider ecosystem approach, and identify the full range of existing and potential future habitats, communities and species regardless of their level of protected status.

This is supported by the philosophy defined in the Water Framework Directive (Council of the European Communities, 2000) that uses taxonomic groups to define biological quality elements for different freshwater systems. For river and lakes the Water Framework Directive lists:

- phytoplankton
- macrophytes and phytobenthos
- benthic invertebrate fauna
- fish fauna.

For artificial and heavily modified surface water bodies the relevant biological quality elements are those applicable to the natural surface water category most closely resembled (e.g., for a reservoir, the biological quality elements for a lake are appropriate).

In addition to these defined assemblages, PRINCE should consider additional taxonomic groups as appropriate. These may include assemblages and species that use freshwater ecosystems and that may be vulnerable to climate change, including certain amphibians, mammals, birds, large invertebrates and zooplankton. Other issues may include the future influence and spread of parasites and diseases. Individual species for which sensitivity to climate change is described in the literature are reconciled against this general structure for the PRINCE framework. Appropriate species designations are taken from:

- Habitats Directive (92/43/EEC) Annex II species (and English Nature Life in UK Rivers project);
- UK BAP Species;
- WCA 1981;
- CROW Act 2000.

3.2 Tailoring a freshwater ecosystem typology to the objectives of PRINCE

To ensure full coverage of freshwater ecosystems in England and Wales, the UK BAP Broad Habitat classification is preferred as the cornerstone of the typology. From the complete current list of Broad Habitat types (see *Table 3.1*), both standing open water and canals (Code 13) and rivers and streams (Code 14) are freshwater ecosystems.

Table 3.1 UK BAP Broad Habitat classification.

Code	*Broad Habitat Type*	*Code*	*Broad Habitat Type*
1	Broadleaved, mixed and yew woodland	14	Rivers and streams
		15	Montane habitats
2	Coniferous woodland	16	Inland rock
3	Boundary and linear features	17	Built up areas and gardens
4	Arable and horticultural	18	Supralittoral rock
5	Improved grassland	19	Supralittoral sediment
6	Neutral grassland	20	Littoral rock
7	Calcareous grassland	21	Littoral sediment
8	Acid grassland	22	Inshore sublittoral rock
9	Bracken	23	Inshore sublittoral sediment
10	Dwarf shrub heath	24	Offshore shelf rock
11	Fen, marsh and swamp	25	Offshore shelf sediment
12	Bogs	26	Continental shelf slope
13	Standing open water and canals	27	Oceanic seas

The use of Broad Habitats in the typology provides a framework in which to address specific significant habitats that are potentially vulnerable to climate change. For flowing and standing open water aquatic systems, the UK BAP Broad Habitats resembles the Environment Agency–English Nature–CCW use of hydro-ecological domains in their internal guidance to assess the hydrological requirements of habitats and species.

However, a further hydrological domain is required to encompass habitats subject to regular inundation from river systems and those influenced by a water table at or above ground level for most of the year; which have been termed ‘wetlands and washlands’. The proposed freshwater ecosystem typology therefore includes three hydrological domains – rivers and streams, standing open water and canals, and wetlands and washlands.

A suitable set of specific habitats meshed within each of the three broad hydrological domains is required to establish their vulnerability to climate change. A selection of UK BAP Priority Habitats is identified in *Table 3.2*, as these are UK-specific and at an appropriate level of detail. However, the current list of UK BAP Priority Habitats does not address the full spectrum of freshwater habitats that may be vulnerable to climate change, and additional non-designated habitats will have to be considered.

Within the Broad Habitat class *rivers and streams*, only one Priority Habitat is identified – chalk rivers. Similarly, within the Broad Habitat class *standing open water and canals*, only eutrophic standing waters, mesotrophic standing waters and aquifer-fed naturally fluctuating waterbodies are identified. A wider review of the list of Priority Habitats was undertaken through their definitions, from the descriptions agreed by the UK Biodiversity Group and published in volumes two and five of the second tranche of action plans (UK Biodiversity Group, 1998, 1999). The relationships between the Broad Habitat classes and Priority Habitats are included in Appendix 2. The wider review yielded fens and reedbeds (from Broad Habitat class *fen, marsh and swamp*), lowland raised bog (from Broad Habitat class *bog*), floodplain grazing marsh (from Broad Habitat class *improved grassland*) and wet woodland (from Broad Habitat class *broadleaved, mixed and yew woodland*).

Additional habitats potentially vulnerable to climate change were included after a review of the Habitats Directive Annex 1 habitats and the EUNIS habitat classification. The relationships between the Broad Habitat types and the Habitats Directive Annex I habitat types, reported in Jackson (2000), was reviewed in the development of the typology for this study (see Appendix 2).

Table 3.2 Proposed freshwater ecosystem typology, based on a framework of UK BAP Broad Habitats and featuring UK BAP Priority Habitats and selected other significant habitats.

Hydrological Domain	*Habitat*
Rivers and streams	• Predominantly baseflow-fed rivers (chalk streams and sandstone rivers) • Headwaters (upland catchments) • Headwaters (lowland catchments) • Middle reaches of river • Lower reaches of river
Standing open water and canals	• Lakes: dystrophic • Lakes: oligotrophic • Lakes: mesotrophic (UK BAP Priority Habitat) • Lakes: eutrophic (UK BAP Priority Habitat) • Canals • Ditches • Ponds • Reservoirs
Wetlands and washlands	• Fens (UK BAP Priority Habitat) • Reedbed (UK BAP Priority Habitat) • Lowland raised bog (UK BAP Priority Habitat) • Improved grasslands on floodplains • Floodplain grazing marsh (UK BAP Priority Habitat) • Wet woodland (UK BAP Priority Habitat)

For each of the three hydrological domains and 19 habitats proposed for use in the PRINCE freshwater ecosystem typology (*Table 3.2*) a brief description is included in Appendix 2. Further information on the seven UK BAP Priority Habitats included in the typology, including individual HAPs, are available on-line (UK BAP, 2004).

4 Key abiotic parameters influenced by climate change

Freshwater ecosystems are controlled by a range of interacting abiotic and biotic processes that determine habitat availability, extent and quality, and the diversity and abundance of species present. From an understanding of these processes, the influence of climate change on freshwater ecosystems can be developed, and its relative significance can be compared to other natural and anthropogenic factors.

This section describes the key abiotic parameters that influence freshwater ecosystems and that can be influenced by climate change. The purpose is to identify the possible climatic variables that need to be investigated as part of the literature review (see Section 5), from which to develop an understanding of their relative importance to ecosystem function. Having established their relative importance from the literature and expert judgement, modelling approaches can be considered to simulate the potential changes in abiotic sources and pathways, their interaction with defined ecosystem receptors and possibly the interaction of the supported communities (see Section 6).

4.1 Approach to description of freshwater ecosystem effects

An approach to identifying key freshwater ecosystem processes and their interaction was developed through an adapted source–pathway–receptor model. This facilitates identification of the key abiotic and biotic ecosystem interactions and the potential impacts of climate change, and their consequent direct and/or indirect influence on habitats and communities.

4.1.1 Drivers

The main driving forces of environmental change on aquatic ecosystems include climate, landscape (including geology and physical geography) and direct and indirect anthropogenic impacts (including physical change, e.g., flood management measures, land management and pollution incidents). For this study, the key driver under investigation is anthropogenic climate change.

However, it is recommended when considering adaptation responses later in the research programme that the relative significance of climate and climate change on the vulnerability of freshwater ecosystems be viewed in the context of the potentially overarching significance of other anthropogenic influences (e.g., land use change).

4.1.2 Sources

For the climate change driver the relevant sources of pressure on the environment are potential changes in rainfall, air temperature, humidity, wind speed and solar radiation. These factors are all addressed by UKCIP02 in their future climate scenarios (Hulme *et al.*, 2002).

4.1.3 Potential pathways

For the sources and/or pressures on the environment relevant to climate change identified above, a wide range of inter-related abiotic (physicochemical) and biotic (biological) interactions are possible for freshwater ecosystems. Whereas the range of sources and consequent pressures are common across the freshwater ecosystems, their potential influence in terms of a change in state (e.g., change in floodplain inundation compared to variation in lake level) will be specific to individual freshwater habitats, both regionally and locally.

The figures in this section focus on abiotic pathways, although ecologically mediated habitat and community interactions may be at least as important. These relationships are considered in greater detail in Section 5.

4.1.4 Sensitivity of receptor and ecosystem response

The degree of sensitivity of a freshwater ecosystem to potential changes in the abiotic and biotic state of the environment, together with the nature and extent of the modification, can result in ecosystem change. The resulting ecosystem response to environmental change becomes manifest as a beneficial or adverse effect in the extent, viability and quality of freshwater habitats and their dependent communities and species.

However, the scientific knowledge base from which to predict the ecosystem response to abiotic factors is relatively poorly understood. Progress in this study will require agreement of a number of fundamental assumptions about the relationship between ecological function and the supporting ecosystem dynamics (e.g., short-term change in wetted areas for some species may directly influence recruitment, through density dependent mortality in remaining habitat). Biotic interactions (e.g., behavioural adaptation, competition, etc.) are often less well understood and are likely to require further study prior to developing process representations suitable for modelling.

4.2 Abiotic interactions in freshwater ecosystems

A preliminary illustration of the key abiotic factors influenced by climate change that govern the dynamic evolution of freshwater ecosystems is given in *Figures 4.1a* to *4.1c*. Of particular significance are the range and relationships between the potential sources–pathways–receptors used as indicators of potential change (hatched marking on *Figure 4.1*) for the ecosystem sensitivity review in Section 5. *Figures 4.1a* to *4.1c* have been developed through an iterative process throughout the reporting period, informing and being informed by the literature review. They are based on current scientific knowledge and expert judgement.

Figures 4.1a to *4.1c* are illustrative at present and act as a framework for assessing the likely interactions and changes within the three hydrological domains in response to climate change. It is accepted that there will be interactions not currently illustrated or are given sufficient prominence. Significantly, the interactions between ecological communities are poorly understood and not included in detail, although emerging science in this area is considered within the literature review in Section 5.

(a)

(b)

(c)

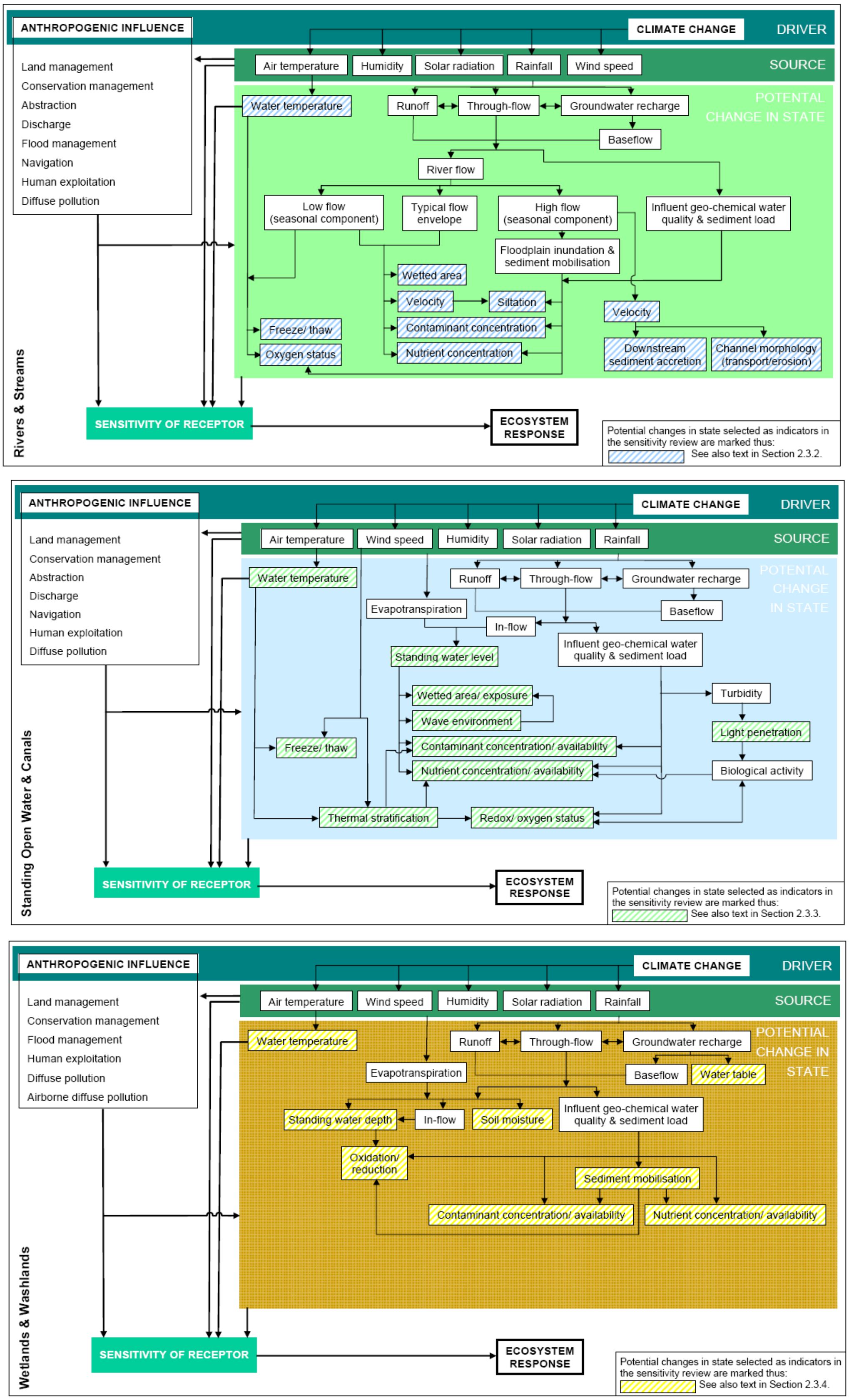

Figure 4.1 Generic framework to describe abiotic interactions for (a) rivers and streams, (b) standing open waters and canals, and (c) wetlands and washlands.

It is suggested that each of *Figures 4.1a* to *4.1c* be reviewed in the next phase of the programme if and/or when specific studies on individual hydrological domain sub-types (e.g., eutrophic lakes, fens, etc.) are taken forward, with the frameworks updated for specific circumstances as necessary.

4.3 Summary of key abiotic factors that influence ecosystem dynamics

Consideration of the key abiotic factors subject to influence through climate change, and that may consequently impact on freshwater ecosystems, has established a range of sources and pathways with the potential to lead to responses in the three hydrological regimes of concern (*Table 4.1*).

Table 4.1 Key potential changes in sources and pathways listed for each hydrological domain.

	Hydrological Domain		
	Rivers and streams	*Standing open water and canals*	*Wetlands and washlands*
Potential changes in sources and pathways	• Water temperature	• Water temperature	• Water temperature
	• Wetted area	• Standing water level	• Water table
	• Velocity	• Wetted area/exposure	• Standing water depth
	• Contaminant concentration	• Wave environment	• Soil moisture
	• Nutrient concentration	• Contaminant concentration	• Oxidation/reduction
	• Downstream sediment accretion	• Nutrient concentration	• Sediment mobilisation
		• Thermal stratification	• Contaminant concentration
	• Channel morphology (transport/erosion)	• Redox/oxygen status	• Nutrient concentration
		• Light penetration	
	• Freeze/thaw	• Freeze/thaw	
	• Oxygen status		

The framework for the literature review (described in Section 5) therefore links the key abiotic factors that may influence a change in ecosystem dynamics caused by climate change, described in this section, with the key habitat types (within the three hydrological domains – rivers and streams, standing open waters and canals, and washlands and wetlands) identified in Section 3.

5 Freshwater ecosystems vulnerable to climate change

5.1 Introduction

The project has been commissioned against a backdrop of increasing concern about the possibility of rapid changes in the aquatic ecosystems of England and Wales in response to climate change. These changes should be viewed in the context of the dynamic evolution of the landscape and its supported ecosystems, which is responding to an underlying long-term shift in climate and a more recent shift in the industrialised and agricultural landscapes of the past 250 years. Over long time scales climate has a fundamental influence on the distribution, dispersal, structure and function of habitats and species (Andrewartha and Birch, 1954; Parmesan and Galbraith, 2004). In the British Isles the current distribution of habitats and species is largely a function of the dynamic evolution of the landscape since the end of the last ice age, some 15,000 years ago, and of the increasing anthropogenic impacts from 5000 years ago. Key influences have been the gradual warming of the wider ecosystem, the separation of the greater land mass of the British Isles from mainland Europe through a sea level rise, and changes in moisture levels (e.g., Hendon and Charman, 2004). The ecosystems of the British Isles have largely been determined through a process of immigration and spread from continental Europe.

Climate and land-use change resulting in habitat modification are two of the main drivers for changes to broad ecosystems. The process of dynamic evolution continues, with many pressures (including substantial human-induced activities) that lead to changes in both the range of ecosystems and their relative diversity. Three main climate-induced evolutionary processes may influence the distribution of habitats and species in the British Isles:

- immigration/introduction (recent introductions often referred to as alien species);
- change in distribution, composition and functioning of habitats and species within the landscape, either through migration into new niches or out-competition and/or unsuitable conditions in existing ecosystems;
- extinction through loss of habitats and species that become unsuitable to existing available conditions.

Potential changes in climate are likely to influence the aquatic ecosystem in two ways, by *episodic pulsed effects* (i.e., changes in the frequency, duration and magnitude of extreme events) and by *press* effects (i.e., progressive change in average conditions). The interaction of abiotic factors with habitats, communities and species, and the integrative responses of each ecosystem component are explored within the literature review.

This report represents the first iteration of the research programme. Given the scale of potential climate change impacts and the breadth of freshwater ecosystems that may be affected, there will inevitably be omissions in the review of the literature and the possible modelling approaches. We suggest that this Project Record be used as a first stage in the development of a research framework, and any omissions or gaps noted by other researchers can be incorporated in later iterations.

5.2 Evidence for climate change impacts on ecosystems

Although climate change will have far-reaching impacts across all domains of environmental management, some of the most marked will be on environmental resources – and especially ecosystems. This includes the effects of climatic change alone, effects in synergy with other sources of global change (e.g., altered biogeochemical cycles; ; Hatch and Blaustein, 2003; Totten *et al*., 2003) and as a cause of other negative impacts, such as through increasing the successful colonisation of invasive species (Buckland *et al*., 2001). While there will be marked effects on the conservation of scarce organisms (Araujo *et al*., 2004) through changes in their distribution (Thuiller, 2003), or through multiplied extinction rates (Leemans and Eickhout, 2004; Thomas *et al*., 2004), there will also be major consequences for the widespread, often abundant, organisms that are major contributors to ecosystem processes (Lohrer *et al*., 2004). Although there is still considerable uncertainty, consequences for ecosystem goods and services are highly likely (Chapin *et al*., 2000; Wessel *et al*., 2004), particularly if future global mean temperature increase above 3-4°C (Hitz and Smith, 2004).

5.2.1 Direct observation

Real evidence of unidirectional trends in aquatic ecosystems in relation to climate is scarce (Daufresne *et al*., 2003), and much knowledge about the potential effects is speculative, inferential or conceptual. However, a number of recent, wide-ranging studies identify the general changes in ecological distributions described above, which have started or are predicted to occur as a result of climate change. These include the observations of change in US habitat and species distributions in over 40 papers on *observed* ecological changes in North America (Parmesan and Galbraith, 2004):

- sufficient studies now exist to conclude that the consequences of climate change are already detectable within US ecosystems;
- the timing of important ecological events, including the flowering of plants and the breeding times of animals, has shifted, and these changes have occurred in conjunction with changes in US climate;
- geographic ranges of some plants and animals have shifted northward and upward in elevation, and in some cases have contracted;
- species composition within communities has changed in concert with local temperature rise;
- ecosystem processes such as carbon cycling and storage have been altered by climate change;
- the findings that climate change is affecting US biological systems are consistent across different geographical scales and a variety of species, and these US impacts reflect global trends;

- the addition of climate change to the mix of stressors already affecting valued habitats and endangered species will present a major challenge to future conservation of US ecological resources;
- in the future, range contractions are more likely than simple northward or upslope shifts;
- reducing the adverse effects of climate change on US ecosystems can be facilitated through a broad range of strategies, including adaptive management, promotion of transitional habitat in non-preserved areas and the alleviation of non-climate stressors.

Work specifically on freshwater ecosystems has been reported from the US (Poff *et al*., 2002). Key findings of the research suggest:

- aquatic and wetland ecosystems are very vulnerable to climate change because the metabolic rates of organisms and overall rate of productivity of ecosystems are directly regulated by temperature;
- increases in water temperature will cause a shift in the thermal suitability of aquatic habitats and resident species;
- seasonal shifts in run-off will have significant negative effects on many aquatic ecosystems (but may not be as severe in the UK, where snowmelt is less of a factor);
- Specific ecological responses to climate change cannot be predicted easily, because new combinations of native and non-native species will interact in novel situations;
- increased water temperatures and seasonally reduced stream flows will alter many ecosystem processes with potential direct societal costs;
- the manner in which humans adapt to changing climate will greatly influence the future status of inland freshwater ecosystems.

5.2.2 Existing problems with the prediction of effects

There are, however, major unknowns about the potential effects of climate change on aquatic ecosystems (Kappelle *et al*., 1999), which include:

- basic outcomes for key physicochemical processes that affect organisms and ecological functions – each are large research fields in their own right (e.g., thermal regimes, system hydrology, consequences for water quality, sediment dynamics, …);
- basic spatiotemporal distributions of organisms in relation to climate both in isolation and in relation to other influences on distribution;
- migration and dispersal of aquatic organisms in responding to change, in particular with respect to limits imposed by the naturally fragmented nature of aquatic systems;
- consequences of dispersal, changes in range and likely future bottlenecks for the genetic diversity of species and the viability of populations;
- physiological tolerance of individuals and species to future change;
- effects of climate on interactions between species and the overall functional consequences within and adjacent to aquatic ecosystems;
- sum total of all these effects for ecosystem processes.

All of these effects are made even more complex in aquatic systems by virtue of the hierarchical nature of water catchments in which all changes are linked (Frissell *et al*., 1986). In other words, aquatic ecosystems will be affected not only by climate change directly, and by the physicochemical changes it engenders, but also by any landscape-scale responses that follow (e.g., Baron *et al*., 1998; Pfister *et al*., 2004). This includes constrained, adaptive or opportunistic change in land use and management. Moreover, effects will transfer in complex ways across different units in the catchment hierarchy – changes in river hydrosystems, standing waters and floodplains or wetlands will be inextricably connected. In this way, aquatic ecosystems may integrate the effects of climate change more than any other ecosystem type and may be considered among the most vulnerable of all natural resources.

5.2.3 Current UK research

In the UK the MONARCH1 programme has adopted a climate envelope approach to species climate zones and identified the migration of bioclimate space northward and the loss of some high altitude space (Box 5.1). The programme has tended to promote a predictive approach through modelling, rather than a data-driven observational approach.

Box 5.1 Extract from MONARCH1

Freshwater habitats:

- Water availability (rainfall minus evapotranspiration) is likely to increase by up to 60 mm in winter (December to February) throughout Britain and Ireland. This could lead to increased ponding and flooding. Raised bogs, wet heaths and coastal dune slacks may benefit. In summer (June to August), there is likely to be a small increase in water availability in north-west Ireland and north-west Scotland, little change in the area immediately south-east of these regions and a decrease elsewhere. This is likely to be most severe (up to 110 mm decrease) in south-east England. This reduced water availability would lead to the drying of wetland habitats with consequent changes in their species composition.
- Peat bogs, wet heaths, coastal dune slacks, drought-prone acid grassland and beech woodland could be affected adversely by the lower water availability in south-east England and, to a lesser extent, in south-east Ireland. Some chalk grassland species predicted to lose suitable climate space in the south-east could be further affected by decreased water availability.
- A local-scale hydrological model run for a site in East Anglia shows similar decreases in summer water levels, which could result in the three modelled species, the great burnet (*Sanguisorba officinalis*), adder's tongue (*Ophioglossum vulgatum*) and the celery-leaved buttercup (*Ranunculus scleratus*), experiencing suitable future climate space, but unfavourable hydrological conditions.

MONARCH2 looks more specifically at species adaptation on a local scale. Four sites have been selected on the basis of their sensitivity to future bioclimate change, conservation importance and available data – these are Hampshire, Central Highlands of Scotland, Snowdonia and Ireland. The studies explore species dispersal and change under future climate scenarios. Outputs from the research programme are expected in autumn 2006.

Freshwater ecosystems cannot be analysed effectively with the climate envelope approach alone, as was adopted for the MONARCH programme, as other processes may be equally or more important. A significant shortcoming of such an approach is also that it has not so far been expanded to include the impact of extreme events – clearly these may be more significant in water environments (e.g., extreme floods and low flows). The bioclimate approach is, however, probably most useful for species not expected to undergo rapid evolutionary change over the next century (e.g., long-lived or poor dispersal species; Pearson and Dawson, 2003).

Other climate change research is beginning to emerge from the UK and Europe that will help to define the magnitude and significance of any potential problems (*Table 5.1*).

Table 5.1 Major R&D studies into climate change underway in England and Wales.

Euro-limpacs (Environmental Change Research Centre at University College London)	MONARCH3 (Environmental Change Institute, Oxford University)
Biodiversity Adaptation in Northwest Europe to Climate change (English Nature)	FLOWCRIT – critical flow thresholds for fish species (Environment Agency)
Defra-funded fish pathogens research (Centre for Environment, Fisheries and Aquaculture Science)	Climate change and fisheries (Environment Agency) including FARRCoF (factors affecting fish recruitment)

5.2.4 Climate change analogues

Current climatic circumstances that are analogues of future conditions can provide evidence of possible future effects. Such analogues include:

- assessment of conditions in US and European regions (e.g., south-west France/north-west Spain) where climates approximate to those in British futures (i.e., space-for-time substitution);
- detailed assessment of aquatic ecological responses to extreme climatic events such as droughts and floods;
- ecological responses to natural climatic oscillations, such as the North Atlantic Oscillation (NAO);
- purposeful manipulations, for example during experiments or operations such as abstraction, regulation and thermal release.

Potential space-for-time substitution

Warm regions in maritime locations with marked seasonality in precipitation include south-west France and north-west Spain, and in both cases altitudinal ranges overlap partially with those in the UK. While the species present are often different, some eurythermal (can tolerate wide range of temperatures) taxa overlap. However, few ecological data compare wetlands, running waters or standing waters between the UK and these analogue areas. Such comparisons have the potential to aid model development and are the focus of the EU Euro-limpacs project and will not be considered in detail in this report.

Extreme events: floods and droughts

Both floods and droughts have major effects on freshwater ecosystems. Floods, while redistributing matter, energy and nutrients between aquatic and riparian ecosystems, also have major, complex consequences for aquatic organisms in extreme cases. Effects depend on timing, intensity, target organism, the availability of refugia and associated physicochemical conditions (e.g., chemistry). The effects of droughts are more readily assessed and studies into their effects underpin much of the research detailed later in this section.

Together, studies have revealed that drought is of considerable significance in rivers, lakes and wetlands (e.g., Everard, 1996; Solomon and Sambrook, 2004), but few data reveal how effects might evolve where drought occurs at the increased (and non-natural?) frequency likely in some parts of the UK in future climates.

Natural climatic oscillations: the North Atlantic Oscillation

As the relative difference in atmospheric sea-level pressure (SLP) between the Azores and Iceland, the NAO fundamentally affects atmospheric circulation across Europe, eastern North America, the Mediterranean and beyond (Hurrell, 1995; Hurrell and van Loon, 1997). Subsequent influences on freshwaters provide an indication of future winter climates, with a recent positive amplification in the NAO index accompanied by a prolonged period of warmer, wetter winters in which:

- surface temperatures during December–March increase by around 0.2-0.6°C for every unit increase in Hurrell's winter NAO index;
- precipitation varies between negative and positive NAO phases by at least 77-112 per cent of the winter average (and up to 30 per cent; Fowler and Kilsby, 2002; Barker *et al*., 2004; Hurrell and Dickson, 2004);
- rainfall intensity and extreme rainfall might have varied more strongly between seasons (Osborn and Hulme, 2002);
- run-off volume has increased in rivers during positive NAO (Bradley and Ormerod, 2001);
- upland stream temperature varies by 3-4°C between contrasting NAO phases (Elliott *et al*., 2000a; George *et al*., 2000);
- effects on nutrients, stream acid–base chemistry, metals and dissolved oxygen content (DOC) have followed changes among rainfall-derived ions, soil ion-exchange, and soil nutrient release (Monteith *et al*., 2000; Hindar *et al*., 2004; Ness *et al*., 2004);
- patterns of lake ice cover have changed at high altitudes and latitudes (Straile *et al*., 2003a);
- variations in wind speed have affected lake mixing processes with consequences for oxygen concentrations and nutrient cycling (Straile *et al.*, 2003a).

In turn, these physicochemical and climatic effects resulted in biological signals in lakes and streams that include:

- altered life cycles, with sensitive river organisms, such as salmonids and mayflies, responding with earlier emergence during positive NAO phases (Elliott *et al*., 2000a; Briers *et al*., 2003);
- subsequent effects on salmonid smolt size (Kallio-Nyberg *et al*., 2004);
- markedly reduced abundances of stream invertebrates, altered community composition and reduced year-to-year stability in invertebrate community similarity during warmer, wetter periods (Bradley and Ormerod, 2001);
- altered populations among lake autotrophs, herbivores, vertebrate predators, with consequences for food-web dynamics (Straile *et al*., 2003a; Straile, 2004);
- increased abundances of cyanobacteria (i.e., blue–green algae) following warmer, wetter, positive NAO winters (Weyhenmeyer, 2001);
- effects during subsequent summers – for example, through increased clear-water phases in some lakes in which zooplankton grazing pressure increases because of increased temperature (Scheffer *et al*., 2001).

These biological consequences of natural climate variability illustrate the potential magnitude of future climatic effects, but also the difficulties in forecasting what can be unexpected outcomes.

Purposeful manipulations: experiments and operations

In streams and rivers, much evidence for the ecological effects of discharge variation has come from examining the effects of abstraction or river regulation. Affected reaches, particularly on lowland rivers, sometimes show reduced organism diversity and abundance where flow reduction is severe (Armitage and Petts, 1992; Agnew *et al*., 2000). Some early indications of the potential phenological changes induced by altered temperature regimes also came from the examination of thermal effluents (Langford, 1975). More recently, experimental manipulations have investigated altered climatic conditions on both lakes (Moss *et al*., 2003) and streams (Hogg and Williams, 1996). In Ontario, Hogg and Williams (1996) showed how a temperature increase of 2-3.5°C decreased total invertebrate densities, particularly of Chironomidae, and also advanced adult insect emergence, changed growth rates, altered size-at-maturity and changed sex ratios in some species. In contrast, the extensive series of experiments by Moss *et al*. (2003) demonstrated that nutrient concentrations had larger effects on phytoplankton in shallow lakes than did substantial warming, with the projected effects on cyanophytes unrealised.

These data confirm the importance of experiments in testing hypotheses about climatic effects.

Differing views on the use of climate analogues

Many of the studies on climate change use surrogates from extreme flood and drought events, which are extrapolated to represent possible future climate scenarios. Although a valid approach, and one that will be used to inform the impact assessments, some of the potential climate change scenarios of the flood and drought events will be both more extreme and prevalent for longer periods (particularly droughts). The ecological impacts identified from previous extreme events may therefore indicate the direction of ecological change, but may underestimate the cumulative effect of the events over time. Process-based deterministic modelling approaches can, to some extent, mitigate for this outcome, as the changes to abiotic parameters will be predicted to give a representation of their actual values (i.e., they will allow the duration, frequency and magnitude of change in abiotic factors to be determined. Prediction of long-term changes in ecosystems as a result of multi-year drought or increased flooding magnitude and frequency will remain subject to significant uncertainty as few data on these relationships currently exist.

An alternative to this view may be that the predictions of biological responses to future climates are characterised by their own uncertainties:

- about the way individual species respond;
- about the way species are newly assembled into dynamic communities;
- about how ecological functions, such as production, decomposition and all forms of species interaction, will be affected directly by climate and as a consequence of changing community composition.

Two corollaries are that:

- considerations, forecasts and modelling based on heuristic principles driven by distinct scenarios might be more appropriate than modelling based on outputs from climate models;
- empirical assessments of aquatic ecological responses to known climatic trends, fluctuations (e.g., the NAO), extreme events (e.g., the 1990, 1991, 1995 and 2003 droughts), existing conditions elsewhere (i.e., in analogue climates) and experiments could be at least as instructive about the biological effects of future climates as are uncertain models.

The proposed modelling approach defined in Section 7 seeks to arrive at an answer to these different viewpoints. It is likely that a preferred approach will be developed that takes the positives from each (modelled versus empirical) approach.

5.3 Reported research effort into climate change and freshwater ecosystems

So far, analysis of the future ecological effects of climate change has lagged behind that for terrestrial systems. This review aims to identify potential effects, to identify gaps in understanding and to identify key areas of concern.

The terms of reference given in the project specifications for the review are:

- to identify aquatic ecosystems vulnerable to climate change – including the identification of sensitive species;
- to review the causal relationships between ecological response and hydrological, chemical and habitat factors affected by climate change.

No particular groups or types of organisms are specified, so these two general aims are kept central to the review. However, there were also substantial constraints because:

- The information available is unequally distributed across ecosystem types (*Table 5.2*) and habitats within ecosystem types (e.g., *Table 5.3*). This is particularly true with respect to aquatic organisms.
- While only a small proportion of published items on freshwaters deal explicitly with climate or climate change, a great many cover both effects and processes through which effects will be expressed (*Table 5.4*). Only a fraction could be covered during the review, and much of the literature could be identified but not reviewed in detail. However, the full databases gathered for the review are available on request.

Table 5.2 Hits on ISI 'Web of Science' on items that carry the terms 'climate change' and specific freshwater and/or wetland ecosystem types (All ISI® Journals to January 2005).

Broad Ecosystem Type	*Hits*
Lakes	1157
Rivers	1055
Streams	541
Wetlands	314
Freshwaters	280
Wetlands	314
Ponds	153
Fens	136
Marsh	119
Swamps	63
Canals	12
24 million papers searched	

Table 5.3 Hits on ISI 'Web of Science' on items that carry the terms 'climate' and specific freshwater and/or wetland ecosystem types (all ISI® journals to January 2005). The numbers in parentheses indicate the number of papers in each that involve effects on aquatic organisms.

Specific Aquatic Ecosystem Types	*Hits*	
Rivers and streams		
• Groundwater-fed rivers	2	(1)
• Headwaters (upland catchments)	9	(0)
• Headwaters (lowland catchments)	7	(2)
• Middle and lower reaches of rivers	38	(12)
Standing open waters and canals		
• Lakes	203	(70)
• Canals	68	(11)
• Ditches	33	(5)
• Ponds	230	(40)
• Reservoirs	184	(12)
Wetlands and washlands		
• Fens	73	(12)
• Reedbed	49	(25)
• Lowland raised bog	3	(0)
• Grazing marsh	1	(1)
• Wet woodland	0	0
24 million papers searched		

Table 5.4 Source research items on climate change and potential 'climate drivers' in the sample of target, leading ecological journals in the ISI® database, 1981-2004 inclusive (*n* = 24 journals with 62,000 papers qualifying).

Search Terms	*Rivers and Streams*	*Standing Open Waters and Canals*	*Wetlands and Washlands*
Climate change	56 (<2%)	73 (<4%)	52 (<3%)
Warming	43 (<2%)	96 (5%)	26 (<2%)
Climate	201 (5%)	304 (15%)	150 (8%)
[1]Hydro-	1,594 (44%)	821 (40%)	952 (48%)
Flow or discharge	1,971 (54%)	253 (12%)	409 (20%)
Temperature	897 (25%)	1153 (56%)	270 (13%)
Sedimentation and/or suspended sediment	657 (18%)	444 (21%)	391 (20%)
Flood	991 (27%)	199 (10%)	907 (46%)
Drought	147 (4%)	102 (5%)	204 (10%)
[2]Water quality or chemistry issues	1184	1440	523
Total references in database	3668	2068	2000

Notes:

1. Hydro-period, hydrology, etc.
2. 'Water quality' and 'chemi' were searched as general terms, rather than as specific determinands.

Even though the peer-reviewed literature on putative climate change impacts on freshwaters is *apparently* large and accelerating rapidly (
Figure **5.1**), much of the published information is based on judgement and speculation rather than evidence.

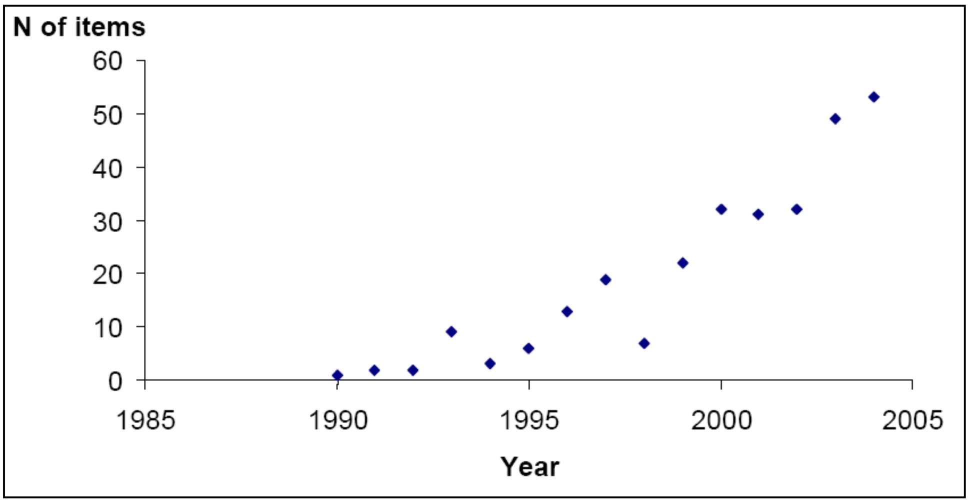

Figure 5.1 ISI Web of Science papers carrying the terms 'climate change' and 'freshwater'.

No single item in the entire ISI database (24 million papers) links the terms 'climate' with BAP species, priority species or priority habitats. Moreover, since only some of the UK's priority freshwater organisms have been the subject of substantial ecological research (Sibley, 2003; White *et al*., 2003; Carter *et al*., 2004; Foster and Beebee, 2004; Sadler *et al*., 2004; Watson and Ormerod, 2004; Watts *et al*., 2004), there is often little scope even for speculating about likely climatic effects (Hastie *et al*., 2003; Jonsson and Jonsson, 2004; Oliveria *et al*., 2004; Solomon and Sambrook, 2004). Furthermore, in few instances has climate change been seen as a priority for key freshwater habitats or species (e.g., Maitland and Lyle, 1991; Drake, 1998; Brown, 2000). This is currently a major gap in the UK's conservation planning for aquatic ecosystems.

The review has attempted to be as comprehensive as is practicable within the study constraints. A number of potential climate change induced impacts, for example on metabolic processes and survival and vectoring of pathogens, have not been reviewed in detail. Similarly, given the range and number of papers on impacts of physicochemical parameters and aquatic ecosystems it has not been possible to cover the full spectrum of potential effects. However, given that a framework has now been developed, it should be possible for researchers to focus more easily on ecosystems of interest and apply additional resource to those considered sensitive to climate change.

5.4 Literature availability and methods

Much of the information gathered for review came from published, peer-reviewed information in the ISI® database (i.e., that available to UK Universities as the 'Web of Knowledge'). The latter currently includes around 24 million individual published papers from about 8000 journal titles published between 1981 and 2004. In addition to searching this entire literature base on some general terms (e.g., *Tables 5.2* and *5.3*), more specific ecological information was derived from a representative sample of leading journals from fields in freshwater biology (n = 12) and from the journals in general ecology (n = 12) lists (*Table 5.4*). In combination, these 24 journals contained around 62,000 papers.

In all searches, no attempts were made to restrict the data to the UK since this would risk missing important generic issues. Moreover, specific geographical selection is difficult in the ISI database, since the subject location in any given published item is usually difficult to identify from the searchable material. Additionally, British contributions to the international ecological literature are now substantially outnumbered by papers from elsewhere in Europe and other continents. Where possible, however, European and British references have been given.

5.5 Vulnerable species, communities, ecosystems and functions

The location of the British islands in a largely wet maritime region characterised by substantial variation in physiographic relief means that rivers, wetlands and other freshwater habitats are a major environmental feature of the UK.

All wet habitats are linked across river basins that, over the coming decade, will be emphasised in the management for good ecological status under the EU Water Framework Directive (2000/60/EC). Brackish estuaries and an extensive, varied near-shore coastal environment are also internationally important. Climatic impacts will therefore arise on aquatic environments at a time when legislative focus and management emphasis has never been greater.

The organisms that occupy British freshwater environments are highly diverse, often functionally important and in some cases recognised for their conservation importance and sometimes also as nuisance organisms. The key groups include:

- **Viruses**: an abundant and widespread group in aquatic ecosystems of under-researched importance, but probably of major significance as organisms that are parasites of many others. In particular, they affect the dynamics of carbon, nitrogen and phosphorus by causing the death and lysis of algae and bacteria.
- **Bacteria**: universal across aquatic habitats, and involved in some of the most fundamentally important aquatic processes over a wide range of scales from the micro (exoenzyme secretion) to the macro (e.g., carbon fixation, nutrient dynamics through nitrification–denitrification, organic decomposition, sulphate reduction, formation of anaerobic hypolimnia in lakes, biomass formation and transfer, methanogenesis, …).
- **Fungi**: widespread major saprophytes involved in the decomposition of both plant and animal detritus, and hence the flow of carbon and the cycling of all major inorganic solutes. This includes material of both autochthonous and allochthonous origin (e.g., leaf litter), so that this group has major importance at the base of many freshwater food webs.

- **Algae**: a wide range of chlorophyll-bearing simple plants responsible for light–energy conversion and most carbon fixation in aquatic ecosystems. Their role at the base of food webs is fundamental and, arising from a wide array of taxa, they have major involvement in the cycling of carbon, nitrogen, phosphorus, silicon and many trace elements. As cyanophytes (blue–green algae), they also have major nuisance effects in standing waters under conditions in which nutrient concentrations and climatic circumstances interact.
- **Bryophytes**: typically, alongside the macroalgae, the major macrophytic producers in upland freshwaters, particularly streams and rivers, where many use free carbon dioxide as a carbon source. hey also provide important habitats for some invertebrates.
- **Higher plants**: many higher plant groups are represented in freshwaters, and in lowland rivers, standing waters and wetlands they provide not only major energy resources to herbivores, but also much of the structural complexity and often the integrity of aquatic systems. They also have major significance to hydraulics and system hydrology, for example in wetlands.
- **Micro-invertebrates**: originating from a wide range of groups, including protozoa, nematodes, rotifers and micro-crustacea, they form a significant feature in river benthic, lake and wetland food-webs, sequestering the energy in finely divided organic matter for use at higher trophic levels. As zooplankton, they also interact strongly with algae to affect the total plankton dynamics in standing waters.
- **Macroinvertebrates**: larger invertebrates from scores of families among the insects, particularly in their wingless aquatic stages, but also including non-insect taxa such as flatworms, segmented worms, molluscs, crustacea and hirudinea. They perform a wide range of roles in freshwaters, including the sequestering of algal production (e.g., grazers), the processing of coarse organic litter into fine litter (shredders), the accumulation of fine detritus into animal tissue (collector-gatherers), the filtering of suspended material (filter feeders) and predation on other organisms (predators). In all these respects, they are of major importance in food webs, material processing and energy transfer across trophic levels. Several species figure in the UK BAP, including a small number of aquatic insects, a range of bivalve and gastropod molluscs and the medicinal leech *Hirudo medicinalis*.
- **Fish**: globally more diverse than marine systems per unit volume of water, freshwater fish richness in Britain is less than that on the European mainland, but nevertheless around 50 species occur and occupy a wide range of habitats. These include a large proportion of species recognised under a range of conservation legislation (e.g., Ramsar Convention (1971); CITES (1975) – e.g., common sturgeon; Bern Convention (1979) – 17 species; Convention on Biological Diversity (1992) – six species; Habitats Directive (1992) – 16 species; WCA (1981/1985) – five species). The Atlantic salmon and migratory brown trout are also of major economic importance, while many other species are of major recreational significance. Also, some species are exotic introductions whose future dynamics under new climatic conditions are a major unknown.
- **Amphibians**: a further conspicuous vertebrate group in freshwaters, and also characterised by a high proportion of highly protected species whose amphibious lifestyle links them directly to climate. This group is thought to be a highly sensitive indicator of climate change.

- **Reptiles**: Often overlooked as aquatic organisms, but both native species (e.g., the grass snake *Natrix natrix*) and some conspicuous exotic introductions (e.g., red-eared terrapin) are widespread in and around some wetland habitats.
- **Birds**: also a characteristic and often abundant group along rivers, on mesotrophic standing waters and on wet grasslands and other wetlands. Several prominent species figure under both the UK BAP and the EU Birds Directive (74/409/EC), and several priority wet habitats are protected, largely for ornithological reasons.
- **Mammals**: As for birds, freshwater habitats are disproportionately important for mammals, and many species either use production originating from freshwaters (e.g., several bats) or occupy the riparian zone almost exclusively. These include two currently high-profile species, the Eurasian otter *Lutra lutra* and water vole. The water shrew *Neomys fodiens* also falls into this category.

In few specific cases, however, have any of these organisms in the UK been considered in specific research on climate change (e.g., Davidson and Hazlewood, 2005). While those species that figure explicitly in conservation legislation or as priorities for biodiversity action are most likely to be emphasised in work on future climatic effects, many of the other groups are crucially important because their abundance and general distribution makes them key contributors to ecological processes. Such processes include:

- primary production and/or carbon fixation;
- secondary production;
- community metabolism from all sources;
- detrital retention, processing and breakdown;
- nutrient spiralling and transfer within freshwaters and with adjacent systems;
- all other biogeochemical cycles in which organisms figure;
- energy and material exchange with the catchment, riparian zone and hyporheos;
- community interactions of all types – competition, predation, food-web transfers.

All of these processes will be vulnerable both to direct changes caused by climatic change (e.g., any thermal dependency in process rates, and any hydraulic dependency in retention and transfer) and to any biologically mediated changes caused to the groups of organisms outlined above.

Although in the following commentary each of the key groups may not be discussed individually, where implications for their preferred habitats are recognised (e.g., middle rivers and otter) the possibility of interaction and risk of impact on the supported assemblages and species may also occur.

5.6 Effects of climate change on freshwater ecosystems

5.6.1 General principles

In outline at least, the future potential processes through which climate change will affect freshwaters are known – direct changes in temperature, thermal regime and rainfall pattern as outlined in Section 4 will directly alter the hydrology and thermal regimes of all freshwater systems (e.g., Hauer *et al*., 1997; Rouse *et al*., 1997; Wilby, 2004). However, direct climatic effects will be translated into freshwaters through interactions with catchment and in-system processes that will result in complex, locally mediated and often synergistic indirect consequences whose final outcomes will depend on the specific circumstances in any given water body (

Figure 5.2).

The governing abiotic and biotic interactions and the ecosystem responses for each of the three hydrological domains and their sub-typologies are explored in Sections 5.6.2 to 5.6.6.

5.6.2 Potential implications of changes in abiotic factors on freshwater ecosystems

General principles relevant to abiotic interactions within freshwaters are considered here, with the more detailed interactions of specific hydrological domains described in subsequent sub-sections. The review of potential abiotic factors with the potential to influence freshwater ecosystems must consider both the change in average or typical conditions that may be encountered, together with any modification to extreme events that may widen or narrow the bioclimatic envelope.

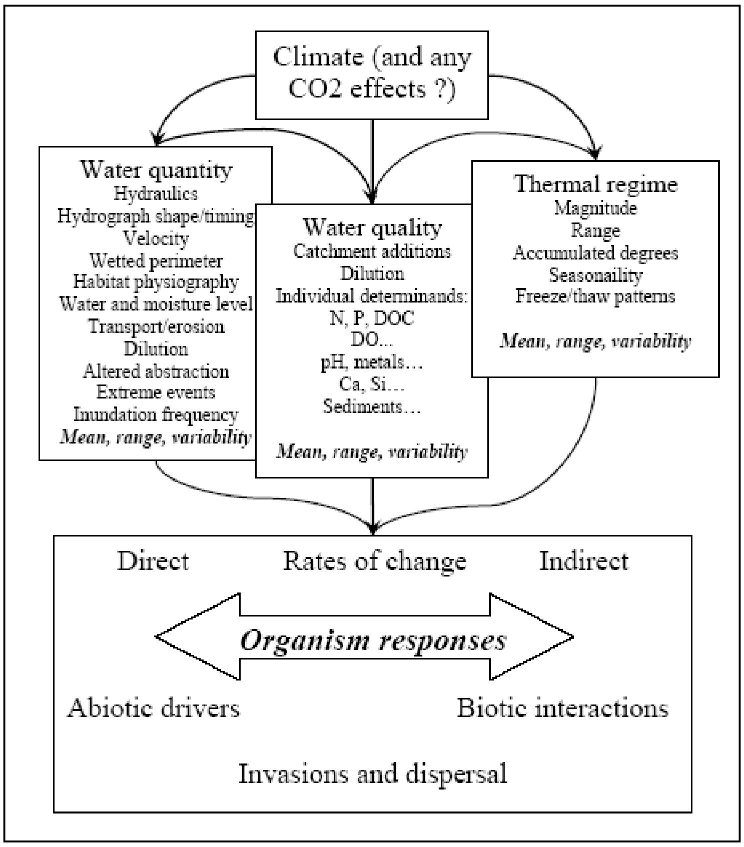

Figure 5.2 Conceptual model of climate change impacts on aquatic organisms

Water quantity: hydrology and hydraulic effects

Besides effects on annual and individual-event hydrographs, altered water quantity will be translated into varying combinations of altered velocity profiles, hydraulic character, water levels, inundation patterns, hydroperiod and changes in wetted perimeter that will change local habitat conditions, habitat availability and connectivity across habitats. In addition, the frequency, timing and magnitude of extreme events of both low- and high-discharge will change markedly in future climates. All freshwaters will be impacted, but effects associated with velocity and hydraulics will be greatest in running waters; changes in level and inundation extent and period will be most significant for lakes and wetlands.

Consequences for habitat configuration, habitat availability and habitat quality will depend both on local conditions and on the habitat distribution or plasticity of each organism. Additionally, altered hydrology and hydraulics will have major consequences for the retention, residence and processing of both matter and energy – and this could be a feature of major significance for organisms (Wallace *et al*., 1995; Muotka and Laasonen, 2002). There is good evidence, for example, that litter retention behind debris dams of wood and other material affects the standing stock of invertebrates in upland streams. Increased discharge in general, and major storms in particular, dramatically decrease retentiveness and hence impact organism densities (Wallace *et al*., 1995).

Water quality

Alterations in climatic conditions will not only affect local hydrology and hydraulic character in all types of surface freshwaters, but also will influence naturally derived water quality through interactions with catchment and in-system features (Klotz, 1991; Thies, 1994; Evans and Prepas, 1996; Bridgham *et al*., 1998; Dahm *et al*., 1998; Marion and Brient, 1998; Butturini and Sabater, 1999; Carvalho and Moss, 1999; Saunders and Kalff, 2001; Carvalho and Kirika, 2003, Pastor *et al*., 2003; Weyhenmeyer, 2004) as summarised in *Table 5.5*.

Slightly higher flows in winter in combination with typically lower temperatures suggest that water quality in most water bodies would not be at greater risk from increased rainfall or storminess, unless contaminants were associated with greater run-off potential. However, with hotter summers and lower rainfall, the reduction in dissolved oxygen carrying capacity and higher biological oxygen demands (lower dilution of effluents and increased algal and/or microbial growth) that are generally prevalent are likely to combine to produce an increased risk of deoxygenation (Everard, 1996). This may be particularly prominent in middle and lower river systems and standing waters, where re-aeration (e.g., from riffles, wind-induced turbulence, human-made structures such as weirs, etc.) is likely to be reduced. This may be compounded where macrophyte growth has been encouraged by higher water temperatures and a non-limiting nutrient supply (Wade *et al*., 2002), which leads to low levels of oxygen and possible threats to fish and invertebrates.

Consequences will arise from flow-mediated effects for transported and deposited sediments (Kelly, 1992; Matthaei *et al*., 1999; Sogon *et al*., 1999; Sweet *et al*., 2003), for a wide range of solutes, including hydrogen ions (H^+), base cations, sodium, chlorine, sulphate, bicarbonate, nitrogen, potassium, DOC, metals, silicon and for most forms of pollution.

Water quality has a major role as an influence on macro-distribution. Dissolved oxygen concentrations, acid–base status, nutrient availability, chlorine and base cations are of fundamental importance for most aquatic organisms so any climatically mediated changes in quality will have profound consequences. This is true not only aquatic organisms, but also for vegetation in semi-aquatic habitats such as wetlands (Tremolieres *et al*., 1994; Venterink *et al*., 2002).

Thermal regime

Both in running waters (Crisp and Howson, 1982; Mackey and Berrie, 1991) and standing waters (George *et al*., 2000), temperature and thermal regimes can closely follow variations in air temperature, which indicates that future temperature changes will be closely tracked by surface waters. However, local conditions complicate straightforward relationships, and in particular there can be marked thermal variability even over short spatial scales within water bodies (Webb and Walling, 1986; Webb and Nobilis, 1994; Finlay *et al*., 2001; Hannah *et al*., 2004). This is especially true for lakes, for which the critical importance of both ice cover in winter and thermal stratification at other times are major features that drive other key processes and features, for example oxygen concentration. These will require careful consideration in any model developments.

Table 5.5 Examples of pollutants in freshwaters and their potential links to climatically mediated processes.

Pollutant Category	*Altered Temperature or Thermal Regime*	*Altered Hydrological Conditions*
Acids	Sulphur and, particularly, nitrogen release from catchment soils is thermally mediated Snowmelt episodes more advanced and pronounced under rapid thaw	Acid episodes more likely under high, upland rainfall Episodes more acidifying at high discharge due to base-cation dilution Natural sea-salt episodes more likely in Atlantic storm tracks Naturally-driven sulphate episodes more likely after drought effects on sulphur-rich peat soils Acid-mine drainage more likely at high rainfall and discharge
Heat	Effects of thermal effluents further exacerbated at increased temperatures	Effects of thermal effluents exacerbated at low discharge, but moderated by increased discharge through changes in volume
Inert solids		Releases of catchment sediments more likely from all sources under high rainfall and/or high discharge, in events of increased intensity Catchment sediment released following drought conditions as soil carbon declines and other disturbances follow (e.g., wildfires; Wessel *et al.*, 2004) Major interactions with the discharge-dependent transport of adsorbed contaminants (Lepage *et al.*, 2000) and adsorbed nutrients (Dikhuis *et al.*, 1992).
Nutrients (nitrogen, phosphorus)	Nitrate leaching from upland catchments to surface and groundwaters more likely at increased temperature (Wessel *et al.*, 2004) Altered lake evaporation patterns change local nutrient concentrations in shallow systems Cyanobacterial response more likely at elevated temperature	Altered pattern of residence times alter the risk of algal developments Potential for thermal stratification of standing water bodies
Oil and oil dispersants		Risk of spillage and release greater in extreme weather events Hydrocarbon-rich run-off, for example from trunk road systems, increased at high rainfall, along with other transport-derived pollutants (Maltby *et al.*, 1995)
Organic wastes	Decomposition more rapid at high temperature leading to more rapid de-oxygenation Effects of de-oxygenation exacerbated at high temperature because of reduced saturation concentration Effects of de-oxygenation more stressful to metabolically active organisms at high temperature	Risk of spillage, leakage, run-off and storm-drain discharge greater under high rainfall Dilution reduced at reduced discharge, but increased at high discharge
Pathogens	Pathogen survival greater at higher water temperature Change in array of pathogen vectors under altered thermal regime	Contamination risk from catchment sources greater at high discharge Residence times longer and hence survival greater at low discharge
Pesticides	Use more likely and more widespread under more pest-rich futures Longer growing season and possible changes in crop types	Dilution and transport discharge dependent Linked to mobility of inert solids (see above)

In wetlands, temperatures interact closely with other important processes, including evapotranspiration, soil moisture retention, soil respiration, oxygen availability and biogeochemical cycling including exports of nitrogen, methane and dissolved organic carbon (Callaway and King, 1996; Davidson *et al*., 1998; Granberg *et al*., 2001; Pastor *et al*., 2003). All these features will be of ecological significance both internally within wetlands and in downstream habitats. More significantly, thermal elements of climate exert major controls over where wetlands persist within the wider landscape and in what form, influencing both evapotranspiration rate and *in situ* thermal regime. These are clearly aspects of major significance for the continued existence and fragmentation of wetlands in general (Halsey *et al*., 1997).

In freshwater ecosystems the majority of the communities and species are sensitive to the prevalent thermal regime for a number of reasons:

- The poikilothermic (i.e., 'cold-blooded') and often stenothermic (i.e., limited temperature tolerance) nature of many species. Not only are temperatures and thermal regimes critical to the life cycle of a wide range of aquatic organisms (Elliott, 1987, 1991; Elliott *et al*., 1994, 1996; Giberson and Rosenberg, 1994; Griffiths, 1997; Elliott and Hurley, 1998; Elgmork, 2004), their ranges reflect thermal patterns, often mediating spatial segregation between closely related species (Hildew and Edington, 1979). Along with features such as velocity or depth preference, thermal regime has been a major evolutionary influence on the evolved traits of organisms (e.g., Charvet *et al*., 2000).
- The basic dependence of poikilotherm vital rates (metabolism, respiration and/or production) on temperature, often in conjunction with current velocity and temperature-dependent concentrations of dissolved oxygen. Classic research long since showed how species have physiological tolerances (e.g., oxygen demands) that are markedly affected by both temperature and current velocity, which probably underpins much of their distributional pattern (Fox *et al*., 1934).
- The life cycles and seasonal phenology of a wide range of organisms from invertebrates, amphibians, fish and birds are proximately or ultimately controlled by temperature and/or by the seasonal hydrograph.
- The basic dependence on temperature and/or residence times of key ecosystem rates such as autotrophic production (by algae or macrophytes) or allochthonous processing (e.g., the retention and breakdown of coarse particulate organic matter such as leaf litter).
- Substantial studies on algae include those by Uehlinger (1993) and Morin *et al*. (1999), and there are reviews by Flanagan *et al*. (2003). For litter breakdown, these include Irons *et al*. (1994) and Rowe *et al*. (1996). In most cases, rates increase at higher temperature, which will result in more rapid redistribution of energy across ecosystem components.
- Linkages, dispersal or migratory movements across ecosystems, for example between marine systems and freshwaters by long-distance migrants (Atlantic salmon, eel, shad; e.g., Davidson and Hazelwood, 2005), and across watersheds during inter-basin dispersal flights by invertebrates. Both upstream and downstream movements by salmonids are affected by thermal regimes (Bohlin *et al*., 1993; McCormick *et al*., 1998, 1999; Jonsson and Jonsson, 2002) as well as by hydrograph pattern (Rustadbakken *et al*., 2004; Svendsen *et al*., 2004). Many other freshwater fish are

similarly affected (e.g., White and Knights 1997). In invertebrates, dispersal and colonisation between wetlands, lakes or across river basins requires emergence and aerial flight in which weather and climate have a direct effect (Briers *et al*., 2003). However, basic aspects of this feature of invertebrate life cycle are still poorly understood (e.g., Petersen *et al*., 2004). Dispersal not only has major consequences for meta-population dynamics, but also will affect the redistribution of species as new climates develop.

- Marked, concordant relationships across a range of freshwater organism groups between richness and temperature, altitude and latitude (Heino, 2002). In other words, climatic features appear to drive major aspects of aquatic organism diversity.
- The introduction, survival and population dynamics of exotic organisms. In altering fundamental conditions, climate change will alter the survival, dispersal, reproduction and population dynamics of nuisance organisms in aquatic systems with potential major consequences. This may, for example, include continental fish species and macrophytes.

5.6.3 Potential climate change impacts on rivers and streams

Generic issues

Abiotic interactions on rivers and streams can be either a direct or indirect result of a number of interactions. For example, change in air temperature has impacts on water temperature with direct consequences for physiological rates of the biota, but it also influences soil microbial rate for nutrient cycling and has consequences for in-channel oxygen status (Cascade Consulting, 2005). Equally, the cycling of river flow, including high and low flow (seasonality, magnitude, duration, frequency and periodicity) and the typical flow envelope are dependent on rainfall, modified by evapotranspiration, run-off, throughflow and groundwater recharge and baseflow. Alteration of the hydrological regime may also have indirect consequence on ecosystem function through changes in velocity, marginal wetted area, contaminant and nutrient dilution. Along with temperature, current velocity and hydraulic pattern have been major evolutionary factors in the habitat preferences, evolved traits and distribution of a wide range of organisms at both micro- and macro-scales. These organisms include diatoms, bryophytes, macrophytes, invertebrates and fish (Sabater and Roca, 1990 Stalnaker *et al*., 1996;; Charvet *et al*., 2000; Malmqvist, 2002). Pronounced changes in hydrology will have major consequences for nearly all groups of river organisms through direct velocity effects, but also through sediment-size distribution, vegetational structure, aggregated dead zones, hydraulic conditions, marginal habitat character (e.g., emergent vegetation), connectivity, woody debris retention, etc. (Gurnell *et al*., 2002). In certain situations the influence of flow results in indirect effects on the ecosystem, as demonstrated for the River Kennet (Wade *et al*., 2002), in which reduced drought flows are shown to increase epiphytic algal growth. This increase, in turn, leads to significant reductions in macrophyte growth (in this case the protected *Ranunculus* spp. population). Flow, in this instance, was shown to be more important than nutrient supply (phosphorus), which was non-limiting through agricultural inputs.

Implications for changes to low flows

Low flows during drought periods in UK rivers and streams have clear effects on a wide range of organisms, ranging form primary producers to invertebrates and salmonids (Elliott *et al*., 1997; Davidson and Hazlewood, 2005). Effects on fish can be particularly profound (Weatherley *et al*., 1991; Matthews and Marsh-Matthews, 2003). In low-flow events stream power generally reduces, lower sediment loads are introduced to the river channel and the remaining load tends to be deposited. In prolonged periods of low flow, which may be a consequence of climate change, emergent vegetation may encroach into the channel, impeding flow further and reducing the availability and diversity of habitats for other life forms (e.g., Johnson *et al*., 1993; Wade, 1995).

River or stream velocity may alter the channel geomorphology and physical habitat, through erosion and transport mechanisms, coarse sediment movement and downstream fine-sediment accretion. Effects reflect not only the direct effects of discharge and velocity, but also indirect effects mediated, for example, through altered hydrogeomorphology, water quality and sedimentation (Wood and Petts, 1999; Watts *et al*., 2004). This blend of indirect and direct ecological effects is supported by existing data, but the interactions can be complex (Extence, 1981; Cowx *et al*., 1984). For example, studies of aquatic macroinvertebrates have shown negatively impacted groups to include lotic forms such as glossosomatids, simuliids, ephemeropterans and plecopterans, while forms typical of slow flow and pool conditions may benefit. Aggregate, community-level responses include reductions in low flow index (LIFE) scores and average score per taxon (ASPT), but these effects have been transitory provided that the flows returned to pre-drought conditions (Brewin and Ormerod, unpublished data). In Essex and Suffolk, for example, drought effects on invertebrates were linked to the effects of discharges on nutrients and dissolved oxygen downstream of sewage works, and also possibly to the effects of abstraction, for example for mills (Parr and Mason, 2003). Indications from several drought studies are that recovery can be rapid provide flow conditions approach those in pre-drought conditions (Morrison, 1990; Ledger and Hildrew, 2001). As a consequence of these generally clear but rapid responses to change, invertebrate indicators of flow regimes are highly effective (Extence *et al*., 1999).

In the River Kennet, drought effects in 1997 reflected interactions between nutrient outflows and vegetation cover and so were restricted to certain sites where lentic taxa increased in abundance relative to lotic families (Wright *et al*., 2002). Studies of the severe droughts in the late 1980s and 1990s on the chalk-fed Rivers Test and Itchen demonstrated that macrophyte cover was particularly sensitive to flow, with filamentous algae growing to the detriment of the *Ranunculus* spp. in low-flow years (Wilby *et al*., 1998). Extensive studies by the Aquatic Environments Research Centre identified a number of flow and water-quality issues related to ecological succession within the lower river, with many improvements linked to upgraded sewage discharges in the catchment (e.g., Flynn *et al*., 2002). Flow and insolation are considered key determinants to predict macrophyte biomass and cover in the river, which will be directly influenced by future climate change. The periodicity of growth and the flow envelope will define community composition and competitive interaction. The research also demonstrated that for short-period extreme events the river is very resilient, often returning to pre-drought conditions quickly, which mirrors the findings of other UK research (Holmes, 1996). The question remains as to how resilient river systems will be in response to persistent changes in average conditions and to higher frequency and greater magnitude extreme events. Little research is available to identify the potential response patterns, although Furse *et al*. (1977) concluded that the macroinvertebrate population, although resilient to short-term drought, would in the longer term change to reflect the new environmental conditions.

In contrast to the predictable outcomes in the above discussion, research in a Berkshire chalk stream showed that the 1976 drought increased the densities of chironomids as they exploited epiphytic diatoms on macrophytes, but the 1997 drought had no similar effects because the heavy shading by marginal vegetation had reduced instream vegetation (Wright *et al*., 2003). These effects illustrate how interactions with other, largely unpredictable and local, factors might influence outcomes. It will therefore be important for future research to reflect the full range of driving processes that contribute to the dynamic modification of assemblage interactions.

Implications for changes to high flows

The influence of high-flow events on ecosystems is less well understood than that of low flows, but it is becoming recognised as a key factor in habitat and species distributions (Poff, 2002). Flooding is important for ecosystem function for two reasons. Floods lead directly to the death of organisms by physical processes such as scouring, burial or displacement into unsuitable habitats, although species tend to evolve traits to avoid flood-related mortality. Floods also create new habitat and alter resource distribution through transport and deposition of sediment, woody debris and nutrients. Many species have adapted to exploit these newly created niches.

Extreme flood events will therefore affect all aspects of river and stream function, from the geomorphological processes (large floods often introduce and sort sediments; e.g., Smith *et al*., 2003) to channel reconfiguration (e.g., Sear and Newson, 2003) and modification of habitat availability and physical flushing out of sensitive populations, such as macroinvertebrates and fish (e.g., Harvey, 1987; Poff, 2002). Any systematic increase in the severity or return period of extreme flood events could have significant implications for stream and river evolution. Recent studies suggested that changing the magnitude and periodicity of flooding may influence species success, as desynchronisation of suitable flows at critical lifestages may influence recruitment (e.g., Fausch *et al*., 2001).

Increased flooding may also disrupt ecological processes such as riparian plant succession and availability, or low-energy, backwater habitats for aquatic species. If climate change induces more frequent disturbance, there may be a consequent reduction in diversity and selection for tolerant and/or mobile opportunistic species (including invasive species) evolved to benefit from such changes (e.g., Townsend and Hildrew, 1994).

The magnitude, frequency and duration of flooding are critical to many wetlands and washlands. The productivity of these areas is often governed by the flood return period, with flood waters providing organic matter and promoting nutrient cycling and ecosystem production (Nilsson *et al*., 1999). The flood-pulse concept for rivers (Junk *et al*., 1989) identifies the stimulation of the nutrient cycle and consequent promotion of macrophytes, phytoplankton and zooplankton growth, with consequent advantages for fisheries

(Bayley, 1995). The central importance of floods and wetland interaction to river ecology is now recognised (see, e.g., English Nature, 2004).

Flooding impacts as a result of climate change may be more severe than the simple transposition of additional rainfall into channel flow. Palaeohydrological studies suggest that climate shift can alter the run-off–erosion relationship that governs channel form. Shifts from wetter to dryer climates can reduce vegetative cover and thereby increase flooding and channel erosion for the same unit rainfall (e.g., Schumm, 1968). Historical reconstructions of flood histories for the upper Mississippi tributaries over 7000 years have demonstrated that small shifts in temperature (1-2°C) and precipitation (10-20%) caused sudden changes in flood magnitude and frequency (Knox, 1993) and the most extreme floods tend to be associated with periods of rapid climate change. Implications for river systems may be exacerbated with future climate change where a combination of drought followed by extreme high flow (as suggested by some climate change scenarios) could substantially increase the potential for channel erosion, as dried soils after rewetting tend to be more susceptible to erosion (National Rivers Authority, 1995).

At present in many parts of the UK river systems are heavily managed to prevent flooding. This has often resulted in highly constrained river channels, which in the short term may mean that many of the implications for wetlands and washlands will not be obvious. However, there is growing recognition that the integration of floodplains in flood management policy is beneficial, as considered, for example, *in Making Space for Water* (Defra, 2004). Adoption of more ecologically sustainable flood management practices should improve floodplain functionality, which would, as a consequence, be influenced by the climate-induced changes to flooding potential, and the potential impacts noted above.

The pathways that affect the hydrological regime determine the influent geochemical water-quality signature and the sediment load to the channel system. In-channel dilution capacity affects the contaminant and nutrient concentration. Interactions with velocity affect siltation rates, which consequently may amend the physical habitat. Floodplain inundation from out-of-bank high-flow events results in sediment mobilisation and has the potential to amend the geochemical and sediment load in the channel. Changes in historic climate (winter cyclonic Lamb weather type since 1861) have been shown to account for a significant proportion of the variation in sediment yields (Wilby *et al.*, 1997). The major negative or positive consequences of suspended sediment for feeding ecology in many riverine ecosystems, and also the consequences of suspended sediment for benthic distribution and habitat quality, are yet to be fully explored.

Implications of changes on fisheries

In river and stream systems, fish are profoundly influenced by all aspects of the hydrological water quality and ecological cycles, and their integrated effects. These include physical habitat availability, velocity, temperature regime, sedimentation pattern (Euliss and Mushet, 1999; Wood and Armitage, 1999), wetted perimeter, pollutant dilution, natural solute concentrations (Williams and Melack, 1997), biotic changes involving macrophytes composition (Holmes, 1999), primary production and organic retention (Mulholland *et al.*, 1997).

Temperature exerts a fundamental influence over the development and growth of fish, including egg and juvenile survival and recruitment of salmonids (e.g., Weatherley and Ormerod, 1990; Crisp, 1993) and coarse fish (e.g., Diamond, 1985; Mann, 1991). It is also an important factor in determining competitive strength of first-year classes (Mills and Mann, 1985). Studies of the 1995 drought, which may be replicated on a more frequent basis with climate change, suggest that coarse fish recruitment and growth may improve (Axford, 1998), provided that sufficient habitat remains (wetted area, marginal vegetation) and water quality (particularly dissolved oxygen) is maintained. The result is a strong year class that is propagated over the following generation (Mann, 1991;).

Implications for salmonids show some variability between species. High water temperature has been identified as responsible for the loss of the 1976 year class of young salmon in an unregulated stream in Wales (Cowx *et al*., 1984), although the trout year class survived. This suggests that other factors were at least as important, as salmon have been shown to have a higher thermal tolerance than trout (Elliot *et al*., 1997). An indirect effect of increased water temperatures was demonstrated for the River Wye in 1976, where late-season decomposition of excessive plant growth (encouraged by low flows and high temperatures) led to deoxygenation of the water and mass mortality of adult salmon (Booker *et al*., 1977).

Temperature also exerts control on the timing of salmonid migration and spawning, including the trigger for upstream and downstream migrations, egg emergence, etc. (e.g., Pavlov, 1994). Changes in the timing of migration and spawning could have significant implications for subsequent recruitment, with the possibility of desynchronising lifestage patterns with other assemblages (including macroinvertebrate emergence). Recent research for the Environment Agency suggests that salmon freshwater growth rates could improve in future years under the low emissions climate scenario, although maintenance and survival thresholds could be breached under the medium–high scenario with consequent negative impacts on salmon growth in the latter half of the 21st century (Davidson and Hazlewood, 2005). These effects would be more pronounced for trout than for salmon, as they are more temperature sensitive. If linked to increased drought and/or flooding, these temperature-mediated effects could be even more pronounced (see below).

Competitive interaction of fish species is also influenced by the temperature regime, either specifically or in combination with the possible indirect effects described elsewhere in this report (e.g., water quality). Certain cyprinids (e.g., carp) have a higher tolerance to temperature and low dissolved oxygen than do other species (e.g., dace and trout), which could influence distribution in rivers with a changed temperature and water-quality regime (Everard, 1996).

Significant modification to variation in the flow regime that potentially may be induced by climate change (high, average and low flows) may fundamentally influence migratory salmonids. Upstream migration is triggered by increased river flow into the estuary (freshets), which promotes the fish to move into the lower river (Atlantic Salmon Trust and Scottish Office, 1995). In low flow years the upstream limit to migration can be curtailed as insufficient depth of water is available in headwaters and spawning gravels become exposed or degraded by siltation (Axford, 1998). Downstream migration of smolts is also triggered by flow requirements. Juvenile and adult survival in headwaters and middle or lower river reaches also requires a minimum flow, linked to the provision of sufficient wetted area. Little work has been carried out on the effects of low flow and drought on salmonid recruitment, although Elliot *et al*. (1997) demonstrated evidence of density dependent population regulation. In low-flow years there may be increased movement of juvenile salmonids into the main stem of rivers from the tributaries, and increased competition (potentially between trout and salmon, which usually have different niches) that potentially leads to density-dependent mortality (Hendry, personal communication).

The influence of high flows on fish is less well described. An increase in winter flows may improve the availability of habitat for spawning salmonids, although evidence from the wet winter of 1995/1996 and other wet years suggests that unusually high flows in spring and summer can result in reduced recruitment of salmonid populations (Cascade Consulting and APEM, 2004). The mechanisms for this effect are not clear, but flushing of eggs and juveniles may be factors. Coarse fish larvae are also known to be very sensitive to current velocities and can be displaced by increased rates (Mann, 1991). Increases in future storminess and the potential for more extreme flows and velocities could therefore have implications for the recruitment and maintenance of fish and other aquatic populations. Further research in this area is required.

Finally, there is some evidence to suggest that climate change may influence Scottish ecosystems differentially, because of the moderation of climate to produce warmer, wetter conditions (Gilvear *et al*., 2002; Soulsby *et al*., 2002). Modifications of the hydrological regimes of the majority of rivers are expected, particularly those with a significant snowmelt component. The potential for a change in the acidification regime of sensitive watercourses remains and could be exacerbated, particularly in areas affected by increasing nitrogen deposition, with headwaters and low nutrient and/or little buffered standing waters also at risk.

Table 5.6 summarises the likely implications of climate change on rivers and streams. The analysis is based on the findings of the literature review and the expert judgement of the project team.

Table 5.6 Potential climate change impacts on rivers and streams.

Hydrological Domain	*Potential Climate Change Impacts*
Rivers and streams	
Predominantly baseflow-fed rivers	Variations in hydrology, altered groundwater recharge, thermal regime, residence times, interactions with abstraction and reduced lowland water quality (e.g., eutrophication) may cause substantial ecological changes. Effects may include: • greater winter recharge resulting in some support for summer river flows • increased frequency and magnitude of summer drought may reduce flows and wetted areas • increased average water temperature from increased air temperature and increased residence time, tempered by low variation in baseflow temperature • reduced instream dissolved oxygen saturation from increased water temperature and reduced flows during the summer period • changes in nutrient concentration of inflow from increased soil microbial rate associated with increased air temperature and changes to run-off regime from variation in rainfall pattern; in eutrophication-sensitive systems • reduced seasonal (summer) velocities from reduction of baseflow contribution to summer low flows and increased frequency and magnitude of drought • increased seasonal (summer) sediment accretion from reduced flows and velocities from reduction of baseflow contribution to summer low flows and increased frequency and magnitude of drought All of the above may contribute to an altered ecosystem, with specific impacts including the potential for increased growth of macrophytes, a change to macroinvertebrate communities to reflect a lower flow assemblage and modification of the fish community to reflect reducing water quality and increased temperatures
Headwaters (upland catchment)	Upland headwaters are by their nature often intermittent and subject to network shrinkage and their biota will reflect this (Wright and Berrie, 1987). Macroinvertebrate species abundance and richness can be severely reduced in drought situations, although this can be reversed rapidly with the onset of higher flows (Morrison, 1990; Wood and Petts, 1994). Variations in hydrology, thermal regime and interactions with water quality (e.g., acidification in sensitive catchments) may cause substantial ecological change. Potential impacts on upland catchment headwaters are most significantly associated with: • increase in water temperature of low temperature instream habitats • increased periodicity and extent of reduced wetted area from reduction of summer low flows and increased frequency and magnitude of drought • changes in nutrient concentration of inflow from increased soil microbial rate associated with increased air temperature and changes to run-off regime from variation in rainfall pattern; in low nutrient systems • changes in contaminant concentration of inflow from changes to run-off regime in low buffering-capacity systems Specific ecological effects may include direct loss of habitats through reduced or lost flows in summer and alteration of habitat structure through increased winter flooding. More complex interactions may result in the altered success of salmon recruitment, although the balance between possible increases in spawning success (higher winter rainfall, less freezing, more redds and fewer egg mortalities) may be tempered by a reduced ability to migrate (reduced migratory flows, reduced water quality, thermal effects inducing phenological impacts). Assemblages in headwaters affected by significantly reduced summer flows may shift to represent different ecosystem types, for example, with exposed river sediment macroinvertebrates succeeding true aquatic species, which could significantly reduce their range.
Headwaters (lowland catchment)	Variations in hydrology, catchment evapotranspiration, thermal regime, lowland water quality (e.g., eutrophication) and changes in catchment use and vegetation will cause substantial ecological changes. Potential impacts on lowland catchment headwaters are most significantly associated with: • changes in periodicity and extent of reduced wetted area from reduction of summer low flows and increased frequency and magnitude of drought • changes in nutrient concentration (increased soil microbial rate associated with changes to run-off regime) in nutrient-sensitive systems Ecological effects may include loss or modification to habitat availability to reflect the increasingly ephemeral nature of headwaters, with consequences for their aquatic assemblages. In lowland headwaters, associated land use is likely to have a more pronounced influence on ecological evolution, with a reduced wetted area and decreased dilution of contaminants capable of shifting habitat availability. Hydrological extremes may, however, be less severe than in upland systems, although this would have to be investigated.
Middle reaches of rivers	Middle reaches of rivers will be more robust in response to climate change than will headwaters. They will, however, be subject to variations in their hydrology, thermal regime and residence time, with consequent risks of reduced oxygen concentrations and reduced water quality (e.g., eutrophication). These factors may cause substantial ecological changes. Interactions with river regulation and water abstraction are also more likely. These systems are potentially sensitive to invasion by exotic species. Potential impacts on middle river reaches are most significantly associated with: • increased changes in periodicity and extent of reduced wetted area from reduction of summer low flows and increased frequency and magnitude of drought • reduced instream dissolved oxygen saturation from increased water temperature and reduced flows during the summer period • changes in nutrient concentration of inflow (e.g., increased soil microbial rate and changes to run-off regime) in eutrophication-sensitive systems • increased seasonal (summer) sediment accretion from reduced flows and velocities during increased frequency and magnitude of drought • potential increase in high flows and velocities from increased extreme rainfall events with implications for consequent habitat-changing geomorphological processes Reductions in flows and increased water temperature has the potential to increase the risk of toxic phytoplankton (cyanophyte) blooms in the middle and lower reaches of rivers, as witnessed as a result of the 1995/1996 drought on the Warwickshire Avon in 1995 and the Rivers Thames and Ouse in 1996 (Everard, 1996). Middle reaches will also show a transition from assemblages representative of faster flowing ecosystems to those of slow flowing and ponded character, including fish, macrophyte and macroinvertebrate communities. These attributes may not be as apparent in regulated rivers with flow or navigational constraints imposed.
Lower reaches of rivers	Many of the issues relevant to middle rivers will be similar to those for lower river reaches. However, the nature of lower rivers makes some of the potential effects more likely, including lower flows and velocities, increased residence time and reduced dilution of contaminants and nutrients. Ecological effects may be pronounced, including phytoplankton blooms during drought conditions, as a result of low river flushing and dilution, combined with high nutrient levels and temperatures (Everard, 1996). There may also be exceptional growth of other benthic algae such as blanket weed, seen in many places in 1995, when velocities and entrainment were reduced (Gordon *et al*., 1992). Reduced summer flows may also inhibit seaward migration and upstream spawning runs of migratory fish. In south-west England upstream salmon migration was delayed or failed because of high water temperatures and/or reduced dissolved oxygen concentrations, which resulted in missed physiological opportunity and possibly reduced recruitment (Solomon and Sambrook, 2004). There will be an increasing propensity for saline intrusion into the river systems as the freshwater–seawater interface migrates upstream in response to lower summer river baseflows and sea level rise. This may profoundly affect the freshwaters influenced by the change in salinity, as their ecology will evolve to one tolerant of low salinity, which tends to reflect communities with lower species diversity. In coastal and estuarine areas this may significantly impact existing designated nature conservation sites. Changes to the high-flow regime are less likely to have a significant impact on in-channel lower river ecology, as many lowland systems are modified to assimilate the effects of flooding (inundation of wetlands is considered separately). However, there may be implications for more extreme high flows and changed saline ingress in some systems, where the combination of effects may lead to modification of the aquatic assemblages.

5.6.4 Standing open waters and canals

Impacts can be direct or result from a number of indirect interactions. For example, changes in air temperature affect water temperature with direct consequences for lake processes and physiological rates. Together with wind speed (George, 2000b; George *et al*., 2000; Talling, 2004) temperature influences the occurrence of thermal stratification and establishes the boundaries of any existing stratification regime. The major importance of thermal processes in driving the development of stratification, with all its associated biological consequences, are described, for example, by DeStasio *et al*. (1996), King *et al*. (1999), Benson *et al*. (2000), George (2000a), Gerten and Adrian (2001), Straile *et al*. (2003b) and Winder and Schindler (2004).

Increased water temperature and fewer cloud days (with greater insolation) also has the potential to increase phytoplankton biomass and alter species composition (Findlay *et al*., 2001; Hart, 2004). Onset and offset of certain discrete conditions, such as ice cover, water temperature and insolation have been shown to be sensitive to climate change. As a consequence, so are the timings of the spring and summer–autumn phytoplankton blooms and the clear water phase (Carvalho *et al*., 2004). Desynchronisation of the phenology of plankton succession can occur when zooplankton emergence is modified by change to water temperature (e.g., Anneville *et al*., 2004; Wagner et al., 2004; Winder and Schindler, 2004). This leads to the breakdown of the relationship between phytoplankton and zooplankton maxima, which may have significant implications for lake functioning and the ecosystem, including fish populations.

Altered stratification characteristics can have negative consequences for hypolimnic oxidation status, oxidation–reduction processes, and nutrient and contaminant availability. The interaction between altered nutrient availability and temperature in driving algal blooms is well documented (Fujimoto *et al*., 1997; McQueen and Lean, 1987; Soranno, 1997). Evidence suggests that climate change will lead to the earlier onset of stratification in many lakes with potential water quality and ecological consequences (Romo *et al*., 1996; Gulati and van Donk, 2002; Moss *et al*., 2003). Drought in lakes has been shown to alter macroinvertebrate assemblages with recovery variable between groups (Gerard, 2001). Flushing and sediment accumulation processes also influence the bed–water column geochemical water-quality interactions.

The physical habitat of standing water margins and vegetated zones is strongly influenced by standing water level (seasonality, magnitude, variation and periodicity of variation) dependent on habitat availability controlling factors that include substrate and topography. The pathways that affect standing water level are complex, dependent on the site-specific influence of run-off, throughflow and baseflow to the inflow, as modified by evapotranspiration and outflow controls. Standing water level is a component in the control of marginal wetted area and its exposure regime, with feedback through the wave environment. Any significant and prolonged reduction in water level would have an influence on macrophyte dynamics (Hough *et al*., 1991), although increased temperatures could enhance the macrophytes standing crop if habitat availability was not constrained (Atkinson *et al*., 2004). Where standing waters dry out completely, faunal switches to communities typical of temporary water bodies would be expected (Jeffries, 1994).

The inflow regime to standing open waters and canals determines the influent geochemical water-quality signature and the sediment load to the system. Alteration of these processes and pathways and the relative contribution of water to the system from each affect the contaminant and nutrient concentration and availability. They may also influence changes in DOC loading with consequences for bacterial and algal production (James, 1991). Biological and physical processes in the waterbody also influence nutrient and oxygen dynamics. Drought-induced acidification in base-poor catchments (Arnott *et al*., 2001; Faulkenham *et al*., 2003) and altered patterns of allochthonous nutrient delivery (Eckert *et al*., 2003) with consequences for macrophytes (Tracy *et al*., 2003) have been demonstrated in the UK.

Light availability and water temperature are often the limiting factors for photosynthetic potential, especially in waterbodies without nutrient availability constraints. Although these are controlled by solar radiation and air temperature, they can also be modified by light penetration, controlled by turbidity, nutrient cycling and a range of biotic processes. The consequences of turbidity for lake clarity and of sedimentation for lake or reservoir benthos are well established (Kelly, 1992; Gibson and Guillot, 1997; Hoddell and Schelske, 1998).

Table 5.7 summarises the likely implications of climate change on standing open waters and canals. The analysis is based on the findings of the literature review and the expert judgement of the project team.

5.6.5 Wetlands and washlands

Wetland ecosystems are highly dependent on soil hydrology and the interactions with either groundwater or surface water. Changes in the seasonality, magnitude, frequency and duration of the supporting hydrological regime will influence the habitat availability and evolution of wetlands and washlands (Gowing and Spoor, 1996). The maintenance of fluctuating water levels plays a key role in maintaining biological diversity of the ecosystem, allowing competition between different vegetative groups within the wetland ecosystem (e.g., Sommer *et al*., 2004). Set against this is the relative resilience of wetland systems, which have evolved to withstand a certain level of inundation and desiccation (Palmer and Newbould, 1983).

Changes in vegetation character and cover often result from drought conditions, particularly when compounded by abstraction and drainage (Greening and Gerritsen, 1987; Harding, 1993; Fojt, 1994; Johnson *et al*., 2004). Reduced availability of water can also result in changes in amphibian distribution and breeding (Babbitt and Tanner, 2000) and impact on invertebrate assemblages (Bataille and Baldassarre, 1993; Brock *et al*., 2003) although in some cases rapid recovery is possibly because of dormant, drought-resistant stages evolved specifically to tolerate natural wetland desiccation.

Table 5.7 Potential climate change impacts on standing open waters and canals.

Hydrological Domain	***Potential Climate Change Impacts***
Standing open waters and canals	
Eutrophic lakes For example, lochs, meres and lakes throughout the UK often on calcareous-, limestone- or sandstone-rich base geologies	Climate change will potentially influence the thermal regime, water quality, turbidity, albedo and residence times of eutrophic lakes. These effects may cause accelerated eutrophication, which results in further ecological change (e.g., Ventela *et al.*, 2004). Altered oxygen dynamics are highly likely. Cyanobacterial blooms will be increasingly likely. These systems are also potentially sensitive to invasion by exotic species. Potential impacts on eutrophic standing waters are most significantly associated with: • decreased summer inflows and increased incidence of drought, with consequent reduction in summer water level fluctuations and suppressed minimum levels • increased peak winter inflows with short-term increases in peak water levels • increased water temperature, with additional potential impacts on thermal stratification and redox and/or oxygen status • reduction in dissolved oxygen saturation from increased water temperature • changes in nutrient concentration from changes to in flow quality and variation in rainfall patterns • increased turbidity from increased algal productivity, which reduces light penetration • on large lakes with significant fetch, there may a change in the induced wave height from changes in extreme wind pattern, which may be masked by variation in standing water level. The consequence of these effects is likely to be a change in the ecological composition of eutrophic lakes, including marginal, water column and benthic communities. Marginal and benthic communities include macroinvertebrates and macrophytes (plus fish spawning areas), which may modify their ranges and composition in response to changes in wetted area and water quality. Water column assemblages, including plankton and fish, would evolve in response to more highly eutrophic conditions. Significant shifts in water quality would influence all levels of biota in the lake system, and would also influence any downstream watercourses.
Mesotrophic lakes For example, lakes in Scotland and northern England, often on slightly base-rich rock – Bassenthwaite, Windermere, Lake of Menteith.	Mesotrophic lakes are likely to be influenced by the same trends as eutrophic lakes, although the degree of nutrification and ecological succession will be at an earlier stage of evolution. Abiotic factors that influence the lake systems will include changed thermal regimes and water quality, lake hydrology, including residence times and water levels, and interactions with catchment land use. Some may begin to develop blooms of nuisance algae, which potentially increases their eutrophication potential. These systems are potentially sensitive to invasion by exotic species. Potential impacts on mesotrophic standing waters are most significantly associated with: • increased fluctuation and changed periodicity of lake level from summer lows to winter peak inflows • increased water temperature, with additional potential impacts on thermal stratification and redox and/or oxygen status • reduction in dissolved oxygen saturation from increased water temperature • changes in nutrient concentration from changes to inflow quality variation in rainfall pattern Ecological change is likely to reflect the changed hydrological condition (greater fluctuation in lake level) and impacts on marginal biota together with general evolution in response to physicochemical water-quality changes. Significant shifts in water quality would influence all levels of biota in the lake system, and would also influence any downstream watercourses.
Broads and meres	Potential impacts of climate change on broads and meres are as those for mesotrophic lakes. The human-made nature of these ecosystems has created artificial and hybrid habitats that are sensitive to variation in water quality and eutrophication risk. The shallow nature of the systems will be reflected in changes to the thermal regime, which may be compounded by inflows of high nutrient concentration waters responding to the influence of climate change within the catchments. Ecological response will follow that for eutrophic and/or mesotrophic lakes, depending on the current nutrient status.
Oligotrophic lakes For example, on base-poor rock – Loch Lomond, Wastwater, Coniston, Buttermere, Llyn Ogwen, or upland tarns in Lake District and peaty lochs in northern Scotland,	Effects will be depth dependent, and will also reflect changes in headwater quality. Climate change may influence thermal regimes, potentially alter stratification patterns, and modify the hydrology, DOC and other aspects of water quality. These are likely to combine to cause substantial ecological change. Potential impacts on oligotrophic lakes are most significantly associated with: • increased water temperature of low temperature habitats • reduction in dissolved oxygen saturation with increased water temperature • increased variability in standing water level from seasonal changes in rainfall-induced inflows • changes in nutrient concentration from changes to inflow quality and regime and variation in rainfall patterns within low nutrient systems • changes in contaminant concentrations of inflow in low –buffering-capacity systems • change to the freeze–thaw cycling which may be currently present, from increases in winter temperature Changes to thermal stratification and direct and/or indirect consequences may occur, but the ecosystem consequences are dependent on the current presence or absence of stratification and likelihood of shift to stratification.
Dystrophic lakes	Dystrophic lakes are relatively rare in the UK. There are a number of pools and small lochs on blanket bog in Scotland and a few pools on acid substrate in southern England. Little is known of their potential response to climate change, but clearly they will respond to any climate-induced hydrological change. Their relatively small size would make them susceptible to changes in hydrological inflows.
Canals	Ecological data are relatively scarce and much would depend on the exact nature of any given canal network. including its feeder system. Management of the canal network to maintain water levels for navigation, together with the movement of boat traffic, generally present the overriding controls on ecosystem function. However, any alterations to thermal regimes and physicochemical water quality are also likely to influence the ecology. These systems are known to be sensitive to invasion by exotic species, including Signal crayfish and Zander. Future operation of the canal system in response to climate change, including the use of any changes to feeder reservoir resources (important as this would dictate how much water could be utilised to maintain flows) would be required to assess the likely implications for canal ecology. Potential changes to leisure activities and boat traffic would need to be included in the assessment.
Ditches	Modification to water tables and less summer rainfall will potentially reduce the ecological dynamics of many ditches, with a greater proportion becoming ephemeral. This may be exacerbated by altered water quality (influenced by future land use and/or agricultural change), oxygen concentrations and vegetation growth, which may cause substantial ecological change at least to those ditches that are important to nature conservation. These systems are likely to be increasingly sensitive to invasion by exotic species. Review of the current knowledge of ditch ecological function and trends in ditch maintenance would be required to give context to the ecological richness and sensitivity of such sites.
Ponds	Ponds include a highly diverse group of habitats about which it is difficult to generalise. Changes will reflect physical character, location, existing chemistry and thermal regime. Potential effects include modification to hydrological regime, leading to water level changes and, where lower, possible encroachment of marginal vegetation. Water column temperature effects will be the same as for lakes, although the shallow nature and lack of buffering capacity in ponds may make them more susceptible to extreme events and severe ecological consequence (algal blooms, deoxygenation, fish kills, etc.). Ephemeral or temporary pools, which were once very common, are now rare because of changes in land use and of agricultural drainage. These pools tend to develop a significant representation of rare species (Pond Action, 1994) and are resilient to periods of desiccation through their ability to retreat into moist areas. Although wetter environments may promote the formation of a higher number of these ecosystems, the increased drying in summer is likely to place temporary pools under greater stress in the future (particularly in the south and east).
Reservoirs	Potential impacts of climate change on reservoirs are similar to those for the lake typologies, but any changes will be further compounded by altered storage and potable water abstraction patterns. Management of reservoir water levels for water resource storage presents the overriding control on ecosystem function, notably the variation in standing water level, exposure of marginal areas and the thermal stratification regime. Since reservoirs are generally managed for drinking water supply, the management regime will usually include prevention of thermocline formation and minimisation of phytoplankton production. The draw-down regime will also be more severe than in a natural system, and often results in relatively denuded marginal habitats. These effects will all be exacerbated during drought conditions when reservoirs tend to be under greatest stress and operate at >50% draw-down. Reductions in rainfall and river flows in summer could result in greater stress within reservoir systems. This might include lower water levels and potentially increased incidence of phytoplankton blooms (Ferguson *et al.*, 1996), which results from higher water temperatures, increased insolation and reduced water column mixing. Although empirical evidence has demonstrated the increased occurrence of phytoplankton blooms in the warmer summers experienced since 1995 (Renton, personal communication), other research indicates that in the longer term, climate change may not increase total cyanophyte production (Howard and Easthorpe, 2002), although the growing season may extend. Further work in this area is required. Potential impacts of climate change on reservoirs would require site-specific investigation to reflect the wide range of operational circumstances that may prevail.

The importance of inundation dynamics, hydroperiod, hydrological regime, evaporation and soil moisture content to all forms of wetlands means that both thermal and hydrological aspects of climate change will have major impacts on vegetation composition (Schneider, 1994; Toner and Keddy, 1997; Siebel and Bouwma, 1998; Krebs *et al*., 1999; Lenssen *et al*., 1999; Bledstoe and Shear, 2000; Brose, 2001). Impacts on all other groups of wetland organisms will follow (de Szalay and Resh, 2000; Saunders *et al*., 2002; Sommer *et al*., 2004), including key groups in wetland conservation such as birds (Bethke and Nudds, 1995; Ausden *et al*., 2001; Rehfisch *et al*., 2003; Wilson *et al.*, 2004).

The primary source of water to the habitat, from river inundation or groundwater, has a significant influence, as does the relative frequency, magnitude and periodicity of standing water depth, soil moisture and influent geochemical water quality. Biological and physical processes in the wetland and/or washland are significant factors in the cycling and availability of nutrients, and also influence nutrient, contaminant and oxygen dynamics. Altered patterns of nutrient inputs can have potential consequences for plant communities (Serrano *et al*., 1999) and micro-organisms (Freeman *et al*., 1994). These processes include sediment mobilisation interactions.

The effects of any saline intrusions in coastal locations may also be crucial, particularly for freshwater groups (Moss, 1994; Floder and Burns, 2004), but also for a range of wetland plants (Keogh *et al*., 1999; Mauchamp and Mesleard, 2001). Clearly, any increase in saline intrusion within river systems and within freshwater wetlands at their downstream or coastal limit could have a significant impact on their ecological integrity.

Table 5.8 summarises the likely implications of climate change on the wetlands and washlands. The analysis is based on the findings of the literature review and the expert judgement of the project team.

5.6.6 Other potential anthropogenic influences on freshwater ecosystems

Although the focus of the research is to define the potential impacts of climate change on freshwater ecosystems, there is a recognised need to consider other potentially synergistic or confounding anthropogenic activities when seeking to define adaptation strategies. Any future adaptation strategies that are defined should include consideration of the potential additive or mitigating effects that the range of activities given in *Table 5.9* could have on possible solutions.

In most catchments some, if not all, of these activities are likely to have consequences for the ecology of the aquatic ecosystems that may be subject to study. Each will have signature impacts, but in combination their effects may be difficult to differentiate. In considering the potential for climate change impacts, the potential for cumulative impact should not be discounted. In some cases other activities may have the same or greater effect on ecosystem integrity over the longer term.

Table 5.8 Potential climate change impacts on wetlands and washlands.

Hydrological Domain	*Potential Climate Change Impacts*
Wetlands and washlands	
Fens	Climate change impacts would reflect altered patterns of inundation, water residence, evaporation, water quality and thermal regimes. Loss and fragmentation of fen habitat would be a likely consequence of increased drought, particularly in the south-east. Generalisation of potential impacts is difficult given the strong influence of site-specific topography and land management on fen integrity and function.
Reedbeds	Since *Phragmites* is largely euryhaline, eurytopic, eurythermal and even tolerant of changes in moisture regime, this may be a robust habitat that could even increase in some locations in future climates as lake shrinkage occurs. Implications for river reedbeds are less clear, given the lack of data on extent, topography and hydrological regimes. However, considerable drying would also change ecological quality in reedswamps – for example reed vigour and density. Specific investigation would be required to evaluate likely direction of change for freshwater reedbed.
Lowland raised bog	Lowland raised bogs will be affected by alteration in thermal regime, evaporation and changes in groundwater level. Loss and fragmentation of this habitat would be a likely consequence of increased drought, although wetter winters may prove beneficial. Generalisation of the potential impacts is difficult given the strong influence of site-specific topography and land management on lowland raised bog integrity and function. Site-specific investigation would be required to identify the balance between winter recharge and summer hydrological response. The sensitivity of this habitat to the hydrological conditions makes it very sensitive to climate change.
Improved grassland on floodplain	Changes will reflect altered patterns of inundation, evaporation, water quality and thermal regimes. Occurrence and persistence will be highly dependent on how the grassland is managed, although changes to floodplain inundation in wetter winter with drier summer futures may influence the hydrological regime required to keep the habitat functioning adequately. Site-specific investigation would be required to assess future trends in floodplain inundation, together with management requirements to maintain the habitat, before judgement could be made on the continued viability of such habitats across the UK. Thos habitats in the south-east would be under greatest stress and threat in the future.
Floodplain grazing marsh	As for improved grassland on floodplain. Wetter winters may improve inundation patterns for periods of the year, but drier summers are more likely to have an affect on grazing marsh management. Many may no longer be sustainable under drier futures.
Wet woodland	Impacts of climate changes would reflect altered patterns of inundation and evaporation on the habitat. Given the long periods for evolution of these habitats, changes here would be slow. Future land use and flood management policy may influence the success of wet woodlands, as well as the changing climate, with great interest in this habitat type by catchment managers at present.

Table 5.9 Examples of activities that may influence future adaptation strategies.

Activity	*Rivers and Streams*	*Standing Waters and Canals*	*Wetlands and Washlands*
Land management	Crop type and cropping pattern (land cover), fertiliser application, pesticide application, soil structure change, land drainage	Crop type and cropping pattern (land cover), fertiliser application, pesticide application, soil structure change, land drainage	Crop type, cropping pattern, grassland re-seeding (land cover), fertiliser application, pesticide application, soil structure change, land drainage
Conservation management	River restoration	Riparian zone management	Water level management through sluices and penstocks
Abstractions	Licenses held by a water company, by industry, for spray irrigation	Licenses held by a water company, by industry	n/a
Discharges	Point source discharges direct to surface water of treated sewage effluent and from industry	Point source discharges direct to surface water of treated sewage effluent and from industry	n/a
Flood management	Modification of bank structure, modification to flow regime, amendment to connectivity between river and floodplain	n/a	Amendment to connectivity between river and floodplain
Navigation	Channel modification, sediment resuspension	Canals and lake systems, bank damage, sediment resuspension	n/a
Human exploitation	Angling including stocking	Angling including stocking	Livestock grazing
Diffuse pollution	Agricultural run-off, farm pollution incidents	Agricultural run-off, farm pollution incidents	Agricultural run-off, farm pollution incidents

5.7 Sensitivity and potential indicators

With general ecological factors affecting the sensitivity and resilience of ecosystems still unclear (Ives and Cardinale, 2004), there is much debate about those that might render ecosystems sensitive to climate change. Candidates have included ecosystems characterised by (Chu *et al*., 2003; Hilborn *et al*., 2003; Peterson, 2003; Julliard *et al*., 2004; Thomas *et al*., 2004):

- species with restricted habitats or other niche specialism;
- species with limited range size;
- species with northerly or montane distributions;
- species in lowland areas of low rainfall;
- species with narrow geographical ranges;
- species in populations that are already fluctuating markedly;
- additional stresses that will interact with future climates.

There is, at present, a scarcity of analyses or data that translate these effects into freshwaters. Moreover, any other methods to identify the sensitivity or resilience of different freshwater ecosystems to future climatic change are unclear. This contrasts with other risks for which either sensitivity is clearly identifiable (e.g., acidification) or the magnitude of pressure can be quantified from risk factors (e.g., agricultural pollution).

Most likely from the analysis presented here, all will be sensitive, but to different combinations of effect. For example, upland running water systems might respond to altered thermal regimes, increased discharge (events) and interaction with acidification or catchment nutrient yields. Upland lakes might respond to altered thermal conditions. Lowland rivers might be affected by interactions between discharge and polluting discharge, droughts of increased severity and frequency, and interaction with changing catchment land use. Lowland wetlands will be at risk from altered flood frequency and consequent effects on soil hydrology.

Sensitivity will also be apparent at different levels of organisation. For example:

- *Populations* will be sensitive to altered vital rates (survival, mortality, dispersal) that reflect the proximate effects of temperature or the ultimate effects of the hydrograph. Distribution patterns will change depending on new regional conditions relative to species' tolerances, speeds of dispersal and effective colonisation. Organisms with populations concentrated in the south-east, the north-west and at restricted extreme altitudes are likely to be most affected. Catastrophic changes in distribution are likely in organisms sensitive to:
  - saline inundation in currently freshwater environments;
  - increased temperature in low temperature habitats of restricted distribution (e.g., cold water stenotherms at high altitude);
  - major increase in drought frequency;
  - increased flood-risk in flood-sensitive habitats.
- Different *species* will be sensitive to change through linkages between life cycle (e.g., seasonal production, breeding phenology), tolerance ranges and climatically mediated conditions.
- Organism *communities* will change through the effects of species loss, species gain and any rules that govern re-assembly. Such species additions might be at least as important to biological response as any other driver (Humphries *et al*., 2004).
- *Ecosystems* will alter in function relative to the effects of major species changes, the effects of functional redundancy, interactions across taxa, the effects of species substitution and over-riding effects on process rates. At present, biotic interactions are probably beyond the scope of modelling, though empirical evidence suggests they could be important (Schmitz *et al*., 2003).

Since the blend of processes that affect ecosystems and the exact mix of responses are currently beyond straightforward prediction, the best potential indicators of future climatic effects are likely to be:

- Aggregate indicators, for example at the community or ecosystem level, based on changes predicted by known response to floods, droughts, changes in discharge, changes in inundation or changes in thermal regimes. Candidates are available (e.g.,

diatoms, macrophytes, fish and invertebrates), but additional distributional analysis is required of existing data.
- Specific, hypothesis-driven responses in the distribution or population of selected organisms, for example among cold water stenotherms or other species of obligate distribution linked to climate.

An optimum ecological monitoring programme would involve a hierarchy linked across key habitats and key regions. Selection should be informed by existing datasets where there are sufficiently rich data to assess the existing spatiotemporal pattern linked to climate.

5.8 Summary

All of the freshwater ecosystems considered are sensitive to climatic change, either directly (e.g., through temperature and rainfall-mediated effects) or indirectly (through hydrological, water quality or competing ecological impacts).

It is difficult to rank the habitats in terms of sensitivity as their resilience to the multi-factoral stresses on each will not be the same and may combine in differing ways. However, it is clear that habitats that are already at the extremes of hydrological connectivity, such as headwaters, ditches and ephemeral ponds, would be the most likely to be affected by changing climatic conditions. The most resilient may be the middle and lower rivers and large lakes that have some buffering from climatic influence. Water bodies that have a significant management input, such as reservoirs and canals, will be influenced, but their ecology in many cases will be largely controlled by factors other than climate (depending on how the resource is used). In many cases there is insufficient knowledge to identify how the habitats may respond.

From this review a qualitative assessment of relative sensitivity to climate change would suggest the following (in order of most to least sensitive):

- upland headwaters
- lowland headwaters
- lowland raised bog
- ephemeral ponds
- groundwater-fed rivers
- ditches
- all lakes
- fens
- floodplain grazing marsh and grassland
- lowland rivers
- middle rivers
- wet woodlands
- reedbeds
- reservoirs and canals (operational management key consideration).

A summary of the potential impacts of climate change on the range of freshwater ecosystems has been provided to illustrate their relative sensitivity to relevant potential pathways, for a future climate change scenario representative of warmer dryer summers and wetter, warmer winters up to the 2050s. Summaries are provided for each of the three hydrological domains: rivers and streams (*Table 5.10a*), standing open water and canals (*Table 5.10b*) and wetlands and washlands (*Table 5.10c*). Each summary matrix presents the potential for impacts (could be either positive or negative) to provide a measure of the range of sensitive aquatic ecosystem typologies and key pathways.

Note that the tables do not show the integrating effects of potential ecological effects from within-system competition, immigration, emigration potential, disease, etc., although this has been considered as part of the relative ranking of sensitivity. The physicochemical (abiotic) and ecological (biotic) interactions will be tested using the range of modelling approaches defined in Section 7.

Table 5.10a Summary of potential changes in rivers and streams.

	Potential Changes in Sources and Pathways								
	Water temperature	Wetted area	Velocity	Contaminant concentration	Nutrient concentration	Downstream sediment accretion	Channel morphology	Freeze–thaw	Oxygen status
Predominantly baseflow-fed rivers	Low	High	High		High	High			High
Headwaters (upland catchments)	High	High	High	High	High			Low	Low
Headwaters (lowland catchments)	Low	High	High	Low	High			Low	Low
Middle reaches of river	High	High			High	High	High		High
Lower reaches of river	High	High			High	High	High		High

Table 5.10b Summary of potential changes in standing open water and canals.

	Potential Changes in Sources and Pathways									
	Water temperature	Standing water level	Wetted area and exposure	Wave environment	Contaminant concentration	Nutrient concentration	Thermal stratification	Redox and oxygen status	Light penetration	Freeze–thaw
Dystrophic lakes	High	High	High		High	High				High
Oligotrophic lakes	High	Low	Low			High		High		
Mesotrophic lakes	High	Low	Low			High		High		
Eutrophic lakes	High	Low	Low			High		High	High	
Canals	Low				Low	Low				
Ditches	High	High	High		High	High				
Ponds	Low				Low	Low				
Reservoirs	High				Low	Low				

Table 5.10c Summary of potential changes in wetlands and washlands.

	Potential Changes in Sources and Pathways							
	Water temperature	Water table	Standing water depth	Soil moisture	Oxidation and reduction	Sediment mobilisation	Contaminant concentration	Nutrient concentration
Fens		High	High	High			Low	Low
Reedbed			Low					
Lowland raised bog	High	High	Low	High				
Improved grassland on floodplains			Low	Low				
Floodplain grazing marsh			High	High			Low	
Wet woodland			High					

Potential impacts		
High	Low	Negligible

6 Modelling framework

6.1 Introduction

This section introduces the modelling approaches that could be used to explore the possible future impacts of climate change on freshwater ecosystems, habitats and dependent species. It begins (Section 6.2) with an overview of the system that is to be modelled in terms of abiotic factors (6.2.1) and biotic (6.2.2) factors. Section 6.3 reviews the four main approaches that have been used to model freshwater ecosystems, including those that have considered climate change explicitly. On the basis of this, Section 6.4 identifies a set of issues that will need to be considered before evaluating the most appropriate modelling case studies.

6.2 Abiotic and biotic influences on freshwater ecosystems

The aim of this section is to develop the understanding of the abiotic and biotic influences upon aquatic ecology in a systems framework. This is valuable as it demonstrates the multiple feedbacks and influencing factors that will impinge upon aquatic ecology and so allows the possible options for the modelling activity (see Section 7) to be compared. The system conceptualisations in Section 4 provide a general framework in which to understand the interactions between variables in the drivers, sources, pathways and receptors of freshwater ecosystem sensitivity. In this section, we develop further the generic frameworks for modelling the interactions between abiotic and biotic processes in relation to climate change. With the same drivers and sources and/or pressures, the modelling linkages to ecological response are effected through the development of first and second order (including direct and indirect) pathways and receptors. This follows the drivers–sources–pathways–receptors (DSPR) framework, in which the ecological response is a third-order receptor.

6.2.1 A framework for considering abiotic processes

A framework for considering abiotic processes is provided in *Figure 6.1*, with the terms used defined in *Table 6.1*. It shows the direct effects of and interactions between abiotic components of the freshwater ecosystem in relation to climate change. The 'ecology' is left as a black box, but is unpacked in Section 6.2.2.

Figure 6.1 allows a number of important observations to be made in relation to developing a suitable modelling framework and considering suitable systems and/or habitats to model. Firstly, it emphasises that climate change impacts upon freshwater ecosystems will be both direct and indirect, the latter mediated by different pathways and receptors that influence the freshwater ecosystem. Thus, as we reflect upon the most appropriate systems and/or habitats to model, it is important to recognise that any delimitation of any single ecosystem may well remove important indirect influences of climate upon that ecosystem. This can be expressed differently – modelling of any one aquatic ecosystem may require analysis of how that ecosystem is connected to other parts of the environment if the full range of climate change impacts on that ecosystem is to be understood.

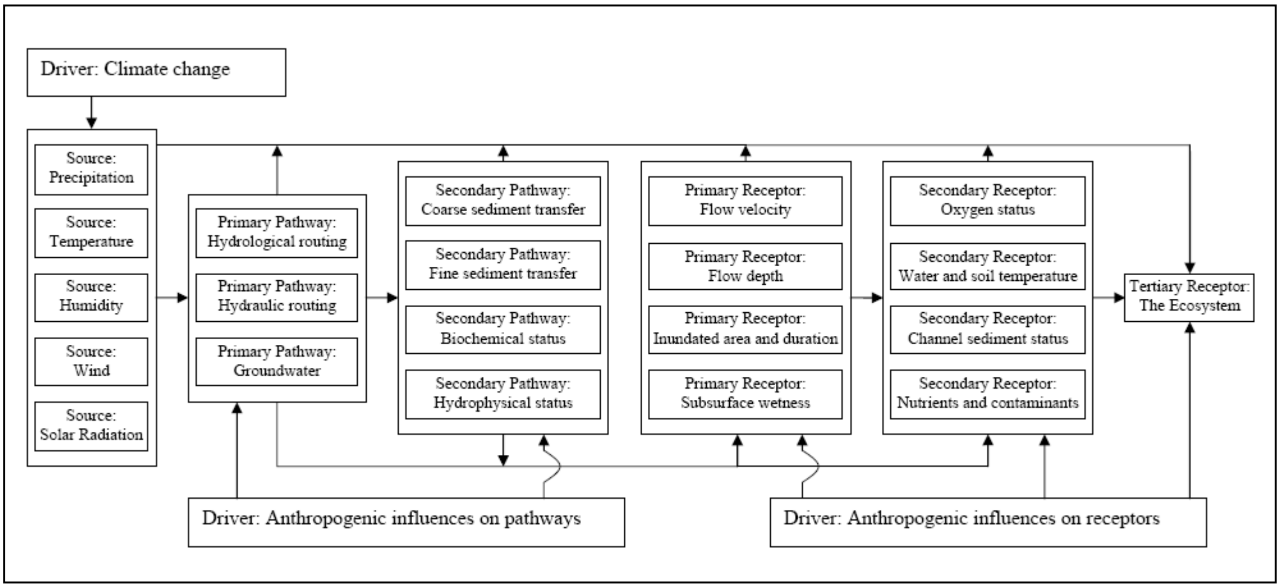

Figure 6.1 Modelling framework based on the driver–source–pathway–receptor model.

Secondly, it makes an important distinction between the climate change driver and anthropogenic impacts drivers upon the ecosystem. The impacts of climate change on freshwater ecosystems will be strongly mediated by these anthropogenic impacts. However, as the focus of this project is to understand how climate change will affect freshwater ecosystems, it will be necessary to assume that this is in the context of existing anthropogenic activity and not anthropogenic responses to the direct or indirect effects of climate change. Mitigation activities based on land use or catchment management could be simulated but, as the range and interaction of such activities is potentially very large, this will require significant additional modelling effort and so is beyond the feasibility of the current project.

Thirdly, *Figure 6.1* demonstrates that the sources, pathways and receptors will impact upon a freshwater system and habitat receptor. This will involve coupling a series of forecast physicochemical environments to freshwater systems and habitats. The nature of this coupling will need special consideration – the physicochemical environments can be modelled using continuous simulation (CS) methods. These will provide information with a particular spatial and temporal resolution. However, there are issues to address both in coupling these types of models to climate change and to freshwater systems and habitats. For instance, there are major challenges in downscaling climate predictions in space (from kilometres to less than 100 m) and time (from monthly or daily to sub-hourly) to generate input data for CS models that have the right spatiotemporal resolution. Similarly, there may be a wide range of timescales of ecosystem responses to an abiotic factor and some species will be able to respond to changes in abiotic factors through migration, which can also occur over a range of timescales.

Fourthly, and related to this issue, the exact nature of the DSPR system will depend upon the range of freshwater ecosystems being considered. Thus, each of the boxes provides a high-level generalisation that needs to be populated in relation to each system and habitat deemed to be of importance. In turn, this will help to identify which processes need to be modelled and the feasibility with which they can be modelled.

Table 6.1 Identification of terms used in *Figure 6.1*.

Class	*Terms*	*Definition*
Driver	Climate change	Human-induced and natural climate variability
Driver	Anthropogenic impacts on pathways	Human-induced changes to pathways (e.g., land management, conveyance management for flood protection, flood storage, abstraction of flow, navigation, effluent discharge)
Driver	Anthropogenic impacts on receptors	Human-related impacts on receptors (e.g., river channel engineering, dredging, flood management, drainage, restocking of fish, vegetation management, exploitation of aquatic resources)
Source	Precipitation	Primary control on aquatic ecosystems as the main source of water
Source	Temperature	Primary control on aquatic ecosystems through the impact on growth and secondary control through the impact on the hydrological cycle
Source	Humidity	Primary control on aquatic ecosystems through evapotranspiration
Source	Wind	Primary control on aquatic ecosystems through evapotranspiration
Source	Solar radiation	Primary control on aquatic ecosystems through growth and evapotranspiration
Pathway	Hydrological routing	Routing of water over the landscape, above ground and below ground, but where the water surface geometry has a negligible effect on the routing process
Pathway	Hydraulic routing	Routing of water over the landscape where the geometry of the water surface influences the routing process
Pathway	Groundwater	Specific pathway associated with routing of water through long-term sinks and stores
Pathway	Coarse sediment transfer	Transfers of sediment in the sand–gravel size range and coarser where the prime sediment movement mechanism involves particle stepping (important in ecological terms as a result of the linkage between coarse sediment delivery and certain instream habitats such as spawning gravels)
Pathway	Fine sediment transfer	Transfers of sediment in the silt–sand size range where material is moved in suspension (important in ecological terms as a result of the linkage between fine sediment and aquatic habitat)
Pathway	Biochemical status	Nutrient and contaminant interactions and transformations (e.g., denitrification)
Pathway	Hydrophysical status	Evapotranspiration and other processes that control physical aspects of the hydrological cycle at the land surface
Receptor	Flow velocity	Instream, wetland and floodplain flow speed, including a measure of the magnitude–frequency properties of those velocities
Receptor	Flow depth	Instream, wetland and floodplain flow depth, including a measure of the magnitude–frequency properties of those depths
Receptor	Inundated area and duration	Instream, wetland and floodplain area of inundation, including a measure of the magnitude–frequency properties of that inundation
Receptor	Subsurface wetness	Soil moisture status of the freshwater environment under consideration
Receptor	Oxygen status	The oxygen status of the freshwater environment under consideration
Receptor	Water and soil temperature	The temperature of the living environments within which things grow
Receptor	Channel sediment status	The sediment size distribution of the freshwater ecosystem under consideration
Receptor	Nutrients and contaminants	The nutrient status and contaminant status of the freshwater ecosystem under consideration
Receptor	The ecosystem	Indicator habitats and potentially species

6.2.2 A framework for considering biotic processes

Section 6.2.1 provided no consideration of biotic processes in terms of:

- the way in which abiotic processes impact upon biota;
- the role of interactions within the biota in conditioning biotic response to abiotic factors;
- the effects of the biotic system upon abiotic processes (e.g., the effects of macrophytes upon dissolved oxygen levels).

The review in Section 5 goes some way to describing the complexities and difficulties of associating abiotic and biotic processes, not least because many of the biotic processes are inter-linked and may influence the abiotic processes through feedback loops. The majority of the science to date has concerned the lumping of biotic processes within the development of empirical relationships that concern the outcomes of integrated abiotic and biotic processes on a given habitat, community or species.

Given the complexity of the biotic community interactions for the range of hydrological regimes identified in Section 4, it is likely that the linkage between dynamic abiotic change caused by climate change (which can be modelled deterministically) and the biotic implications (which are largely integrated through empirical relationships) will be a key focus of the proposed approach. This potential approach will be explored in Section 7.

6.3 Approaches to modelling future climate impacts on freshwater ecosystems

6.3.1 Introductory perspectives

We have identified three basic approaches to modelling future climate impacts on freshwater ecosystems:

- Space-for-time substitution;
- data-driven modelling: examination of trends in long-term ecological and ecosystem datasets;
- conceptually-driven mathematical modelling.

It is emphasised that data-driven and conceptually driven mathematical models are end members of a continuum. In this almost all (perhaps with the exception of models of a neural network type) data-driven models have some sort of conceptual underpinning and all conceptually driven mathematical models require boundary condition, parameter and validation data.

Gertsev and Gertseva (2004) provide a brief summary of issues in modelling in ecology. Firstly, they note that all models in ecology are homomorphic in that they are not symmetrical – a symmetrical or isomorphic ecological model is one in which the model's contents can be traced fully into the ecosystem under consideration *and* the ecosystem's contents can be traced fully into the model. As most ecosystems are too complex to be treated in full in a model, such models are asymmetric or homomorphic – the model's contents should be traceable fully in the ecosystem under consideration, but not vice versa. This is a statement that all models will involve some form of 'closure' or constraint, and is a natural part of making complex reality amenable to scientific study. For instance, the model may simplify reality by ignoring certain processes or process interactions (e.g., in deterministic models), assume that current statistical generalisations hold into the future (e.g., in data-driven models), assume that an existing climatic–ecosystem equilibrium will occur in a different spatial location in response to the development of climate at that location (e.g., in climate analogues), etc. The fact that all models are homomorphic means that they can only provide a set of plausible future scenarios (i.e., they will not be definitive) and that almost all models will be readily criticised in terms of the nature and number of assumptions associated with their formulation.

Secondly, Gertsev and Gertseva (2004) note that models can be described as stationary or time-dependent. In a stationary model, a particular time slice is forecast by applying future conditions to an ecosystem. There is no account for the effects of the trajectory followed from the present to the future upon the final state of the future that is realised. In a time-dependent approach, which is largely restricted to CS approaches, path dependence may be explicitly represented.

Thirdly, Gertsev and Gertseva (2004) note that models can be classified into whether or not they are spatially distributed or lumped. This is a particularly important consideration as biotic response can be spatial (e.g., through migration or occupation of a particular part of the environment that has implications for other parts of the environment, such as the impact of phytoplankton upon light penetration).

In the following sections, these sorts of descriptions are important. For each modelling approach we now:

- define each method in detail;
- provide some case examples and illustrative references;
- summarise the associated advantages;
- summarise the associated weaknesses.

6.3.2 Space-for-time substitution

Definition

Space-for-time substitution has been used in studies of biodiversity for some time, with some successes when applied to palaeo-ecological studies of ecological succession since the most recent ice age. The method is predetermined on the assumption that there is insufficient time to discover changes in ecosystems through monitoring and evaluation (as effects can be over long timescales) and that the examination of similar ecosystems in different geographical location may exhibit the same or similar evolutionary consequences for a given climatic condition as would be experienced (in the future) in the location of interest (in our case England and Wales). The technique examines ecological data to allow comparison between physiographically matched but climatically contrasting locations. This requires understanding of climate and ecological data (but not necessarily their interactions) for a potentially vulnerable site. The Euro-limpacs suite of studies is reviewing this approach as a potential avenue of research, although outputs of their studies are currently unavailable to the us.

Examples and illustrative case-studies

An example of the use of space-for-time substitution is given in Rutherford and Ezzy (submitted). The study investigated the potential future outcome of stream modifications during restoration on the vegetation structure of the resultant streams. Over 80 streams were analysed and the vegetative structures and age of communities described. A conceptual framework for the analysis was established, as shown in Figure 6.2.

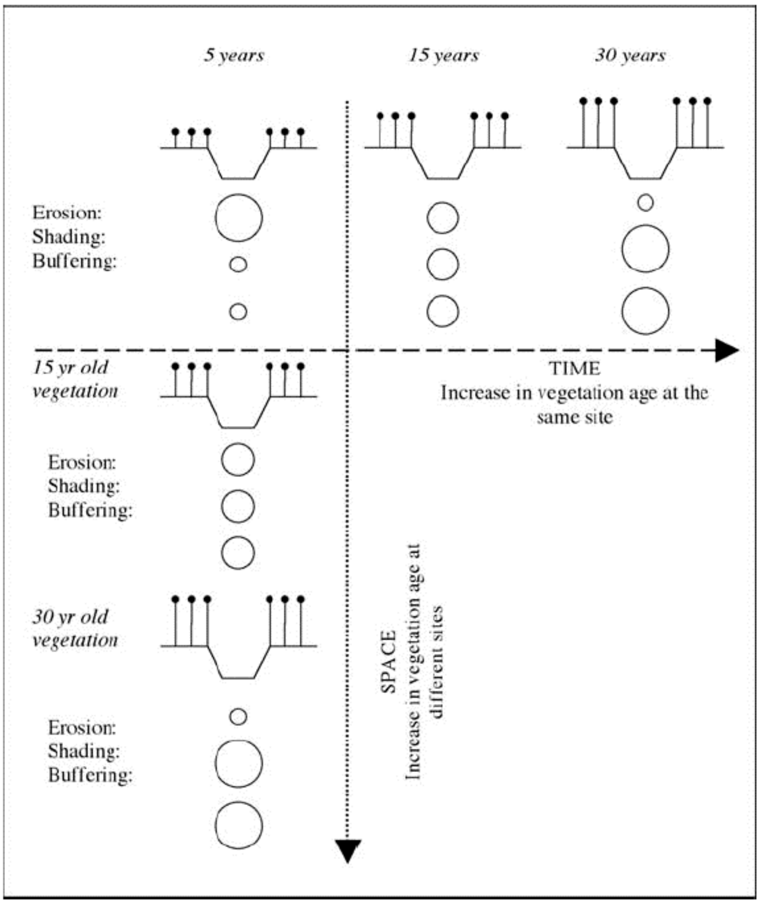

Figure 6.2 Conceptual model of space-for-time substitution at re-vegetation sites (the diameters of the circles indicate the hypothetical magnitude of the effect of vegetation on erosion, shading or buffering). (Taken from Rutherfors and Ezzy, submitted.)

The conclusion of the study was that there was potential in the method, but that vegetative (biotic) and a range of abiotic feature (hydrology, etc.) made it difficult to predict future variability with confidence. If this method were to be adopted, the issue of similarity (in all but climate) of potential spatial surrogates (e.g., from other countries at similar latitude in the northern hemisphere) would have to be carefully considered.

Advantages and disadvantages

The key advantage of this method would be its relative simplicity, provided that the data for each surrogate site were readily available at a suitable resolution. As these data are unlikely to be available, this also constitutes the key disadvantage.

Difficulties with the method include identification of the changes in climate at the host location, and then the requirement to match the 'future climate' data (at the host location) with somewhere with that existing climate signature and similar drivers of habitat availability (topography, geology, etc). It would also require good ecological datasets for the 'match' location. Although of some interest, it would be better suited to a consolidated monitoring programme with the express intention of using the data for such a purpose, in which case many of the potentially confounding factors could be designed out at the outset. A preliminary assessment suggests that this approach may be more valid for terrestrial analyses. A major limitation of space-for-time substitution for ecosystems is that the length of the day varies and is unlikely to represent the system being studied as well as the surrogate system.

6.3.3 Data-driven modelling: examination of trends in long-term ecological and ecosystem datasets

Definition

There is now a long record of data-driven approaches to the examination of the functioning of aquatic ecosystems. In the simplest of terms, these approaches seek to explain the variance in one or more biotic measures (which may be a binary representation of presence or absence, an ordinal representation of an ecosystem property such as abundance or a continuous or ratio measure) as a function of both biotic and abiotic variables. This is shown in Figure 6.3: once the statistical interactions between measured variables has been described in a statistical model, the model is driven by future climate change, under the assumption that the structure and rate of interaction will not change markedly from that found in the statistical model. There are a very large number of approaches to this type of statistical modelling. However, the nature of the water environment, with a high number of potentially relevant variables and variable interactions, means that many approaches require multivariate statistical analysis. This commonly involves selecting key driving variables or reducing the number of variables to components that represent most of the detail in the system. Most recent developments (see below) have explored the scope of analyses of the neural network type., In these the form of the statistical generalisation developed is 'learnt' from the available data, and so they require no user specification of the form of the model that is adopted, as happens in other statistical generalisations.

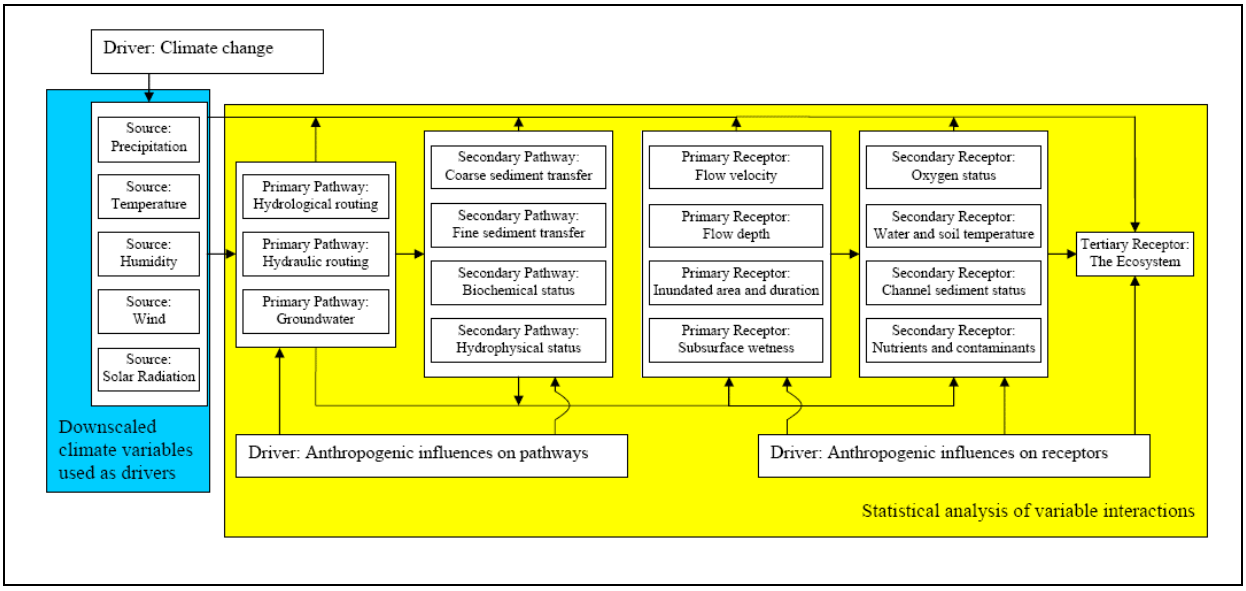

Figure 6.3 A conceptual illustration of how the abiotic environment would be represented in a data-driven analysis.

Examples and illustrative case studies

There are many examples of the use of this approach to model the freshwater environment (Van Tongeren *et al.*, 1992; Romo and Van Tongeren, 1995; Romo *et al.*, 1996). Lau and Lane (2002) used principal components analysis that included basic meteorological variables to explore the Barton Broad (Norfolk Broads) dataset on long-term phytoplankton and zooplankton dynamics. A simple extension of this work would take the statistically modelled data and drive it with 2050s and 2080s climate scenarios. It would provide explicit characterisation of phytoplankton and zooplankton dynamics, which can then be interpreted to understand broader ecological implications.

In recent years, one of the major developments has been the use of neural networks to forecast biotic variables in the future (e.g., Mastrorillo *et al.*, 1997; Paruelo and Tomasel, 1997; Recknagel *et al.*, 1997; Maier *et al.*, 1998, 2001; Lae *et al.*, 1999; Schleiter *et al.*, 1999; Lusk *et al.*, 2001). The basic principle of a neural network analysis is to use artificial intelligence methods to search among available data to develop forecasting relationships. This commonly involves some form of model training (i.e., backward propagation) and then forecasting (i.e., forward propagation). The method honours measured data explicitly and can be used in climate change studies by making sure that relevant climatic parameters are used as driving variables. Traditionally, a major criticism of neural networks has been their black box nature in that the internal model detail has no equivalent physical or biological basis. This is being addressed – Maier *et al.* (2001), for instance, describe a method that allows the information contained in a trained neural network to be expressed as a fuzzy rule base.

It is important to recognise that the distinction between data-driven and conceptual mathematical models is a continuous one. This is well illustrated by Bell *et al.* (2000), who used 30 years of data from Black Brows Beck (English Lake District) to construct a model of the population of migratory brown trout, *Salmo trutta* L., during normal rainfall conditions and under the influence of drought. The model is based upon a time-dependent approach suitable for proximate forecasting (i.e., over short periods into the future). he model is also based upon a Ricker stock-recruitment approach where the number of survivors (or recruits) *F*, at a later time, is determined as a function of the number of eggs (parent stock) at the start of each year-class, *S* (Bell *et al.*, 2000):

$$F = aSe^{-bs}$$

[6.1]

where *a* and *b* are empirically fitted parameters. In this case, the model has a strong conceptual underpinning in the form of Equation [6.1], but requires a very high-quality dataset to determine *a* and *b*. In this study, the model showed a strong dependence of survival rate upon the initial egg density. Subsequent analysis of higher resolution data showed that this density-dependence ends after a critical time period, the 'critical survival time', which in turn could be related to the initial egg density. The model did not include competition effects. A simple rainfall-dependent empirical scaling term was introduced to lower recruitment rates during drought years.

Advantages

Data-driven models have the key advantage that they can capture almost the full range of measured variability in an ecosystem and, with appropriate statistical generalisation, can be used as a forecasting tool. If the models are developed in the right way, then variable interactions will be accommodated as drivers are changed, but under the assumption that the nature of those interactions (e.g., process rates) does not change as a result of driver change.

Disadvantages

Perhaps the prime disadvantage of this modelling approach is that it is strongly dependent upon available data. These data must include all relevant variables. Many of the methods also make explicit assumptions about the nature of the data used to populate them (e.g., in some cases, the data must follow a Gaussian distribution) and effort may need to be put into data transformation before the model is developed. Indeed, there are situations for which technical development is required if application is to be successful (Momen *et al.*, 1996). This data dependence is illustrated in a number of senses. For instance, one of the major reasons for the success of the Bell *et al.* (2000) model of Brown Trout was the possibility of updating the model from known data. This emphasises that these types of approaches are only as good as the data available to populate them – they are restricted to those systems that have good data available from which to develop empirical generalisations.

Secondly, the majority of these types of models are suited to steady-state analysis and they can have particular problems when systems have path dependence. One of the best examples of this was early attempts to restore degraded shallow lakes through phosphorus control which overlooked that in the process of becoming more eutrophic, greater levels of surface floating algae led to macrophyte loss, which in turn sustained high algal populations, even when phosphorous loading was reduced. They are also not good at representing the spatial distribution and interaction of processes and, in most cases, can be described as ‘lumped’.

Thirdly, these models assume that the statistical generalisation of the system derived from the measured data is constant through time – both the structure of the system and the rates at which processes operate within the system are constant.

6.3.4 Conceptually driven mathematical modelling

Definition

This approach recognises that variable interactions will change as a result of climate change. Hence, it uses a conceptual analysis of a system to specify a set of variables and functional relationships between variables, which are solved in a mathematical model. It is the basis of the climate change forecasts that are to be downscaled for this project and it has been applied to many of the abiotic interactions identified in Section 4. *Figure 6.4* shows how conceptually driven mathematical modelling addresses abiotic variables.

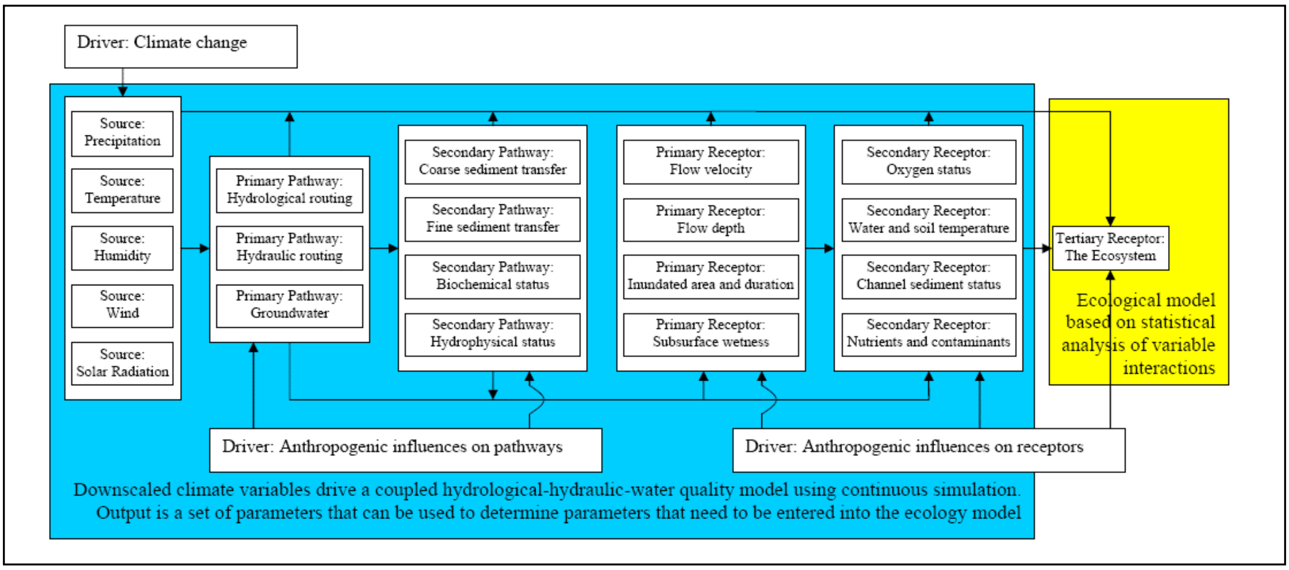

Figure 6.4 Conceptually driven mathematical modelling.

Examples and illustrative case-studies

i) Mass balance modelling of trophic interactions

One of the simplest forms of conceptually based mathematical modelling uses a mass balance analysis of trophic interactions. For example, Brando *et al.* (2004) used a mass balance approach based upon simulation of the functional relationships between ecosystem components to study the flow of nutrients within a coastal lagoon ecosystem. This involved a set of system-specific measurements to determine flows of nutrients between system components, with each trophic level essentially described as a black box. The model was developed to simulate trophic level impacts of manipulation of one part of the ecosystem, which was used to explore both changes in biomass in different components of the system and changes in flows of matter between different components of the system. This approach was highly data dependent. There was no introduction of rate equations to describe temperature-dependent effects and so, while this approach provides a good understanding of trophic interactions (*Figure 6.5*), it needs to be developed to extend it to climate change. A similar approach was used by Gamito and Erzini (2005) to explore the impacts of increased fish biomass upon trophic levels in a lagoon ecosystem.

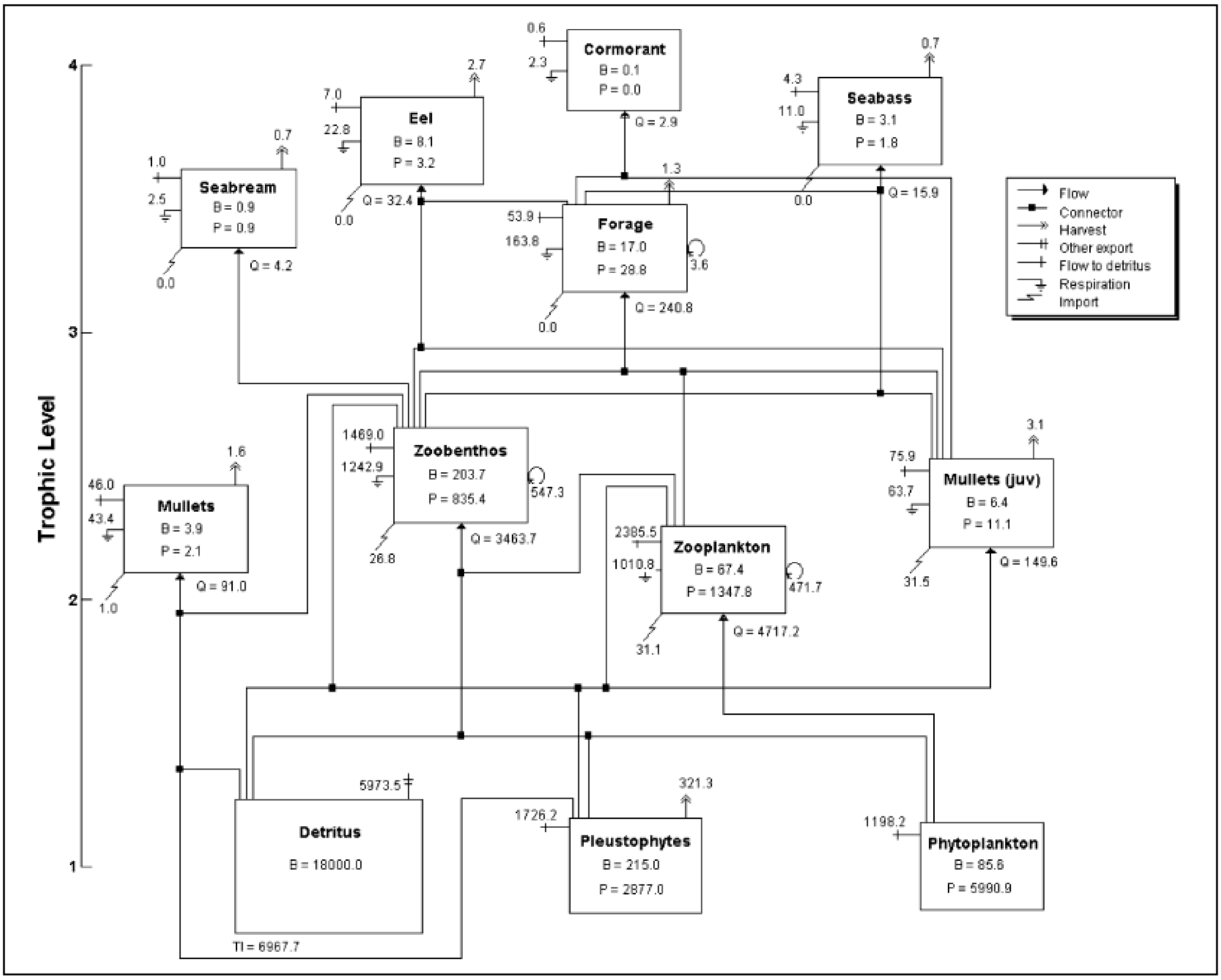

Figure 6.5 Flow rates for biomass and production (t/km^2/year) for Orbetello lagoon estimated from a mass balance analysis of trophic interactions (taken from Brando *et al.*, 2004).

The introduction of abiotic models is increasingly common for models of interactions between nutrients (e.g., phosphorus), primary producers and associated grazers. Both Malmaeus and Håkanson (2004) and Schauser *et al.* (2004) developed models for lake eutrophication, but based upon very different treatments of phosphorus cycling (*Figure 6.*). These kinds of models commonly represent climatic parameters explicitly through rate equations, as many of the associated process rates are water-temperature dependent (e.g., Moisan *et al.*, 2002). Indeed, climate change is likely to impact both water volumes (through evaporation and precipitation effects) and water temperatures (including stratification and reduced mixing) so resulting in quite major changes in phosphorus behaviour. Similar findings have emerged for dissolved oxygen (e.g., Park *et al.*, 2003) and, in a series of very important papers, it has been possible to couple climate change to dissolved oxygen and temperature characteristics and ultimately to fish populations (Fang *et al.*, 1999, 2004a, 2004b, 2004c).

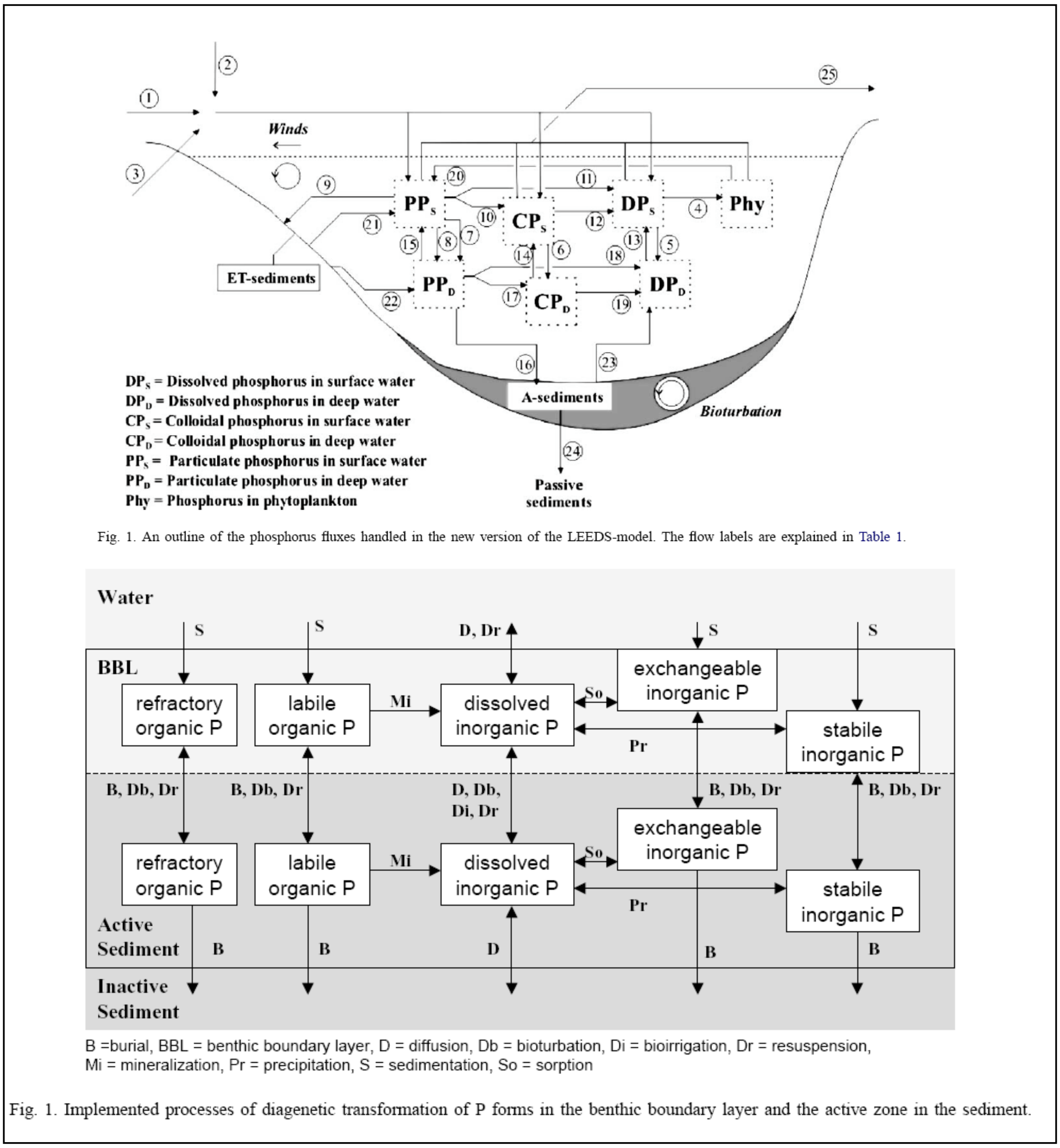

Figure 6.6 Illustrations of two different treatments of phosphorus cycling in lake ecosystems to model the eutrophication process (top taken from Malmaeus and Håkanson, 2004); bottom taken from Schauser *et al.*, 2004).

The differences in representation of the phosphorus cycle implicit in *Figure 6.* emphasise the uncertainty inherent in the associated science base, as well as the complexity associated with modelling only one small part, albeit an important one, of the lake ecosystem. More complex lake models using approaches of the mass balance type have been developed (e.g., Jimenez-Montealegre *et al.*, 2002; Güneralp and Barlas, 2003). For instance, Güneralp and Barlas (2003) used a mass balance type model to explore system dynamics, with four components: hydrology; nutrients and other indicators; chlorophyll-*a*, zooplankton, macrophytes; and fish and crayfish (*Figure 6.*). They offer much potential, but are heavily dependent upon parameters, some of which require calibration, others of which can be derived from experimentation and/or existing

literature. They also sacrifice representation of some parts of the phosphorus cycle (and other nutrient cycles) to model more of the ecology. This kind of issue emphasises that for many freshwater aquatic ecosystems we do not yet know what is 'sufficient physics', 'sufficient chemistry' or 'sufficient biology' for modelling the system effectively. In the classification of Gertsev and Gertseva (2004), we do not know what level of homomorphism is acceptable.

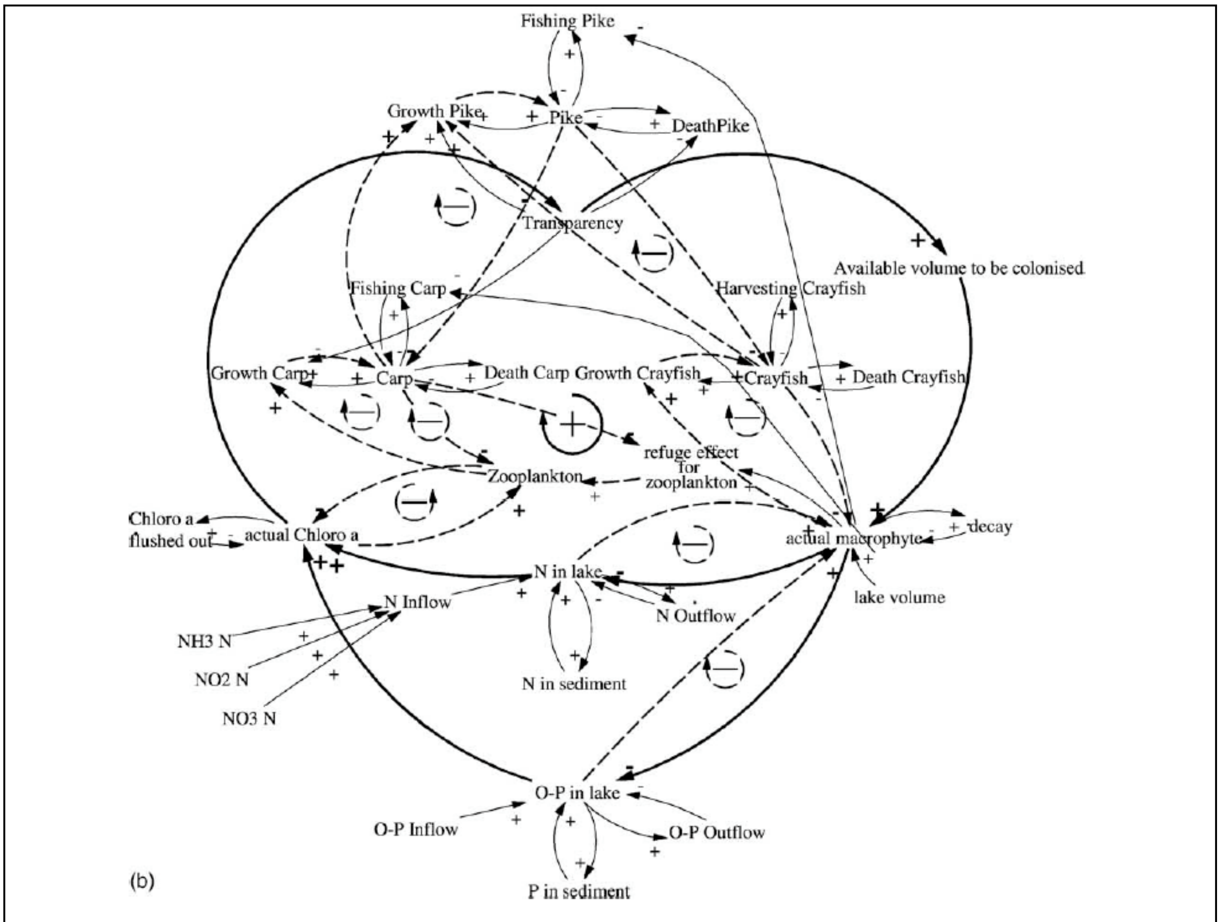

Fig. 1. (a) Model structure showing the sectors, sub-sectors and sections. (b) Broad causal-loop diagram of the *lake ecosystem* sector. (c) Fish-fishing effectiveness graph. (d) Broad causal-loop diagram of tomato cultivation from the *economic activities* sector. (e) Diagram of the *social structure* sector.

Figure 6.7 A more complex model of the mass balance type that includes a wider range of biotic processes, but with a simplified abiotic process (taken from Güneralp and Barlas, 2003).

ii) Spatially explicit mass balance models

The above treatments are all lumped. As such, they do not recognise the existence of considerable spatial gradients in abiotic and biotic processes in lake and river systems, largely driven by gradients in the photic depth (the depth of light penetration). Thus, the simplest of the spatially explicit mass balance models divides the water body into a series of layers in the vertical, to capture changes in the photic depth as well as the effects of vertical mixing processes. PROTECH (Phytoplankton RespOnses To Environmental Change) is one of the best examples of this (Elliott *et al.*, 2000b; Reynolds *et al.*, 2001; Elliott and Thackeray, 2004). It operates on the simple principle that chlorophyll-*a* concentration is a positive function of phytoplankton growth rate and a negative function of settling, grazing and dilution. It calculates the mixed layer as a function of wind shear and thermal fluxes. The model captured the timing and magnitude of spring and summer algal blooms in the test lake ecosystems. It was specifically developed for climate change studies – it is thermally and wind-stress driven and so it may be used with downscaled climate data.

In some cases, modelling a lake ecosystem requires a two-dimensional (2D) approach to capture horizontal process gradients. Thus, Beran and Kargi (2005) used a coupled 2D hydraulic and water-quality model to predict variations of chemical oxygen demand, ammoniacal nitrogen, phosphate phosphorus, dissolved oxygen, bacteria and algae concentrations with time for stabilisation ponds. The biological component of the model was based upon a rate equation for algal growth and decay, using the principle that the growth rate is constrained by the most limiting variable, but that the limiting variable may evolve as the system does. As with similar models, the prime impact of temperature is upon the rate equation that controls the rate at which certain biological and chemical processes operate. The model contains 39 parameters, 11 of which were calibrated and the rest of which were estimated from the literature. Drago *et al.* (2001; see also Ji *et al.*, 2002; Robson and Hamilton, 2004; Romero *et al.*, 2004) extended the hydrodynamic treatment to three dimensions, an approach suited to situations in which there are strong horizontal and vertical gradients in hydrodynamic variables such as velocity or turbulence. This can be especially important, even in lake ecosystems, where wind stress impacts upon the vertical advection and diffusion of parameters relevant to water quality. The model, as with other models of its type, models non-conservative processes using rate-type equations. The model includes low-level trophic representation and weak interaction through phytoplankton and zooplankton.

The interesting observation regarding many of the above is that they rarely explore more than primary producers and herbivores, but there are exceptions (e.g., Ji *et al.*, 2002; Jimenez-Montealegre *et al.*, 2002; Güneralp and Barlas, 2003). However, there is a growing sophistication in modelling activity with a much broader treatment of the biotic environment evident in some model applications, including consideration of higher trophic levels. A good example of this is Gertseva *et al.* (2004), who used a functional feeding group approach to simulate aquatic macroinvertebrate dynamics (*Figure 6.8*). Four functional groups of macroinvertebrates were identified – (1) shredders, (2) scrapers, (3) collectors and (4) predators. Each functional feeding group was linked to a module that addressed abiotic factors and a module that addressed biotic interactions. Water temperature, water flow, dissolved oxygen, pH and nutrients were identified as the most ecologically significant abiotic influences upon aquatic macroinvertebrate communities. The model was applied to small headwater streams and showed a strong sensitivity to temperature, which impacted both upon organism recruitment and emergence. This approach has much potential. However, as with many of the more complex models, the demands upon both abiotic and biotic data to initialise and parameterise the model prohibits the application of the model to specially designed data-acquisition programmes. Data collected for purposes other than this kind of modelling is generally not of the right kind.

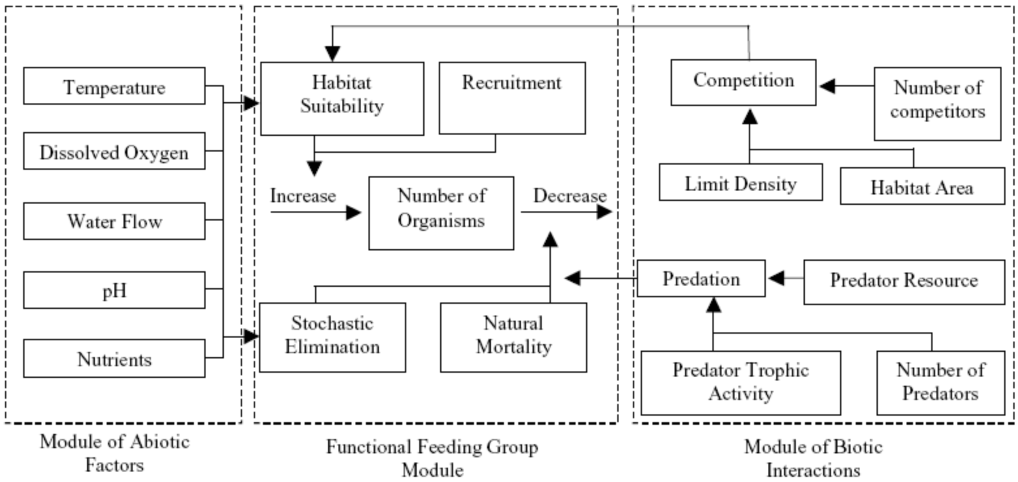

Fig. 1. Simplified diagram of the dynamics of number of organisms in a functional feeding group within a stream macroinvertebrate community.

Figure 6.8 The functional feeding group approach to modelling the dynamics of aquatic macroinvertebrate communities (taken from Gertseva *et al.*, 2004).

iii) Mathematical modelling of fish habitat suitability

The above cases generally modelled primary production. In a few cases, they also considered organisms higher in the food chain, but rarely fish (but note Güneralp and Barlas, 2003). However, the impacts of abiotic factors upon fish communities have been considered extensively through mathematical modelling of changes in habitat suitability. In practice, this does not translate into an explicit biotic response unless habitat suitability is the prime limiting factor for the aspect of the biota considered and there is no possibility of migration to existing environmental refugia or to areas of the environment that become habitat suitable as a result of the change in the abiotic factor. However, it does inform policy-makers as to where and when habitat suitability might change, which translates into an ecological impact in locations where habitat is a limiting variable.

Habitat suitability has been investigated for a wide number of freshwater organisms, not just fish. Typically, key physical parameters, such as flow velocity and depth, wetted perimeter, substrate, temperature and pH (e.g., Elso and Giller, 2001; Maddock *et al.*, 2001) are combined into some form of habitat score, such as a weighted usable area (WUA). The best-known example of this is PHABSIM (e.g., Milhous *et al.*, 1984), which is widely used to assess habitat suitability even though it has a hydraulic treatment that is not correct and severely distorts the habitat suitability results obtained (Lane and Ferguson, 2005). Leclerc *et al.* (1995) note that this will not produce reliable results for less than 10 m^2, so limiting its suitability in smaller streams or where spatial variation in habitat is high. Thus, although it is easy to drive PHABSIM with abiotic drivers (e.g., changes in flow), the results tend to be severe over-estimations of the amount of available habitat (Crowder and Diplas, 2002). Further, in freshwater systems a strong interaction between habitat composition and/or arrangement and habitat processes has been identified (Ward *et al.*, 2001), which creates both spatial variation in habitat (diversity) and changes in habitat and its structure through time (dynamics). Ward *et al.* (2001) argue that the failure to recognise fully the diversity and dynamism of natural river corridors has hindered conceptual advances in river ecology and will also result in a poorly developed understanding of how changes in abiotic factors, such as climate change, impact upon habitat suitability. It follows that an awareness of the spatial assemblage of different habitat types, and how they change through time, is a crucial input to conceptually based mathematical models of habitat.

This has been recognised in conceptually based mathematical modelling. As with forecasts of phytoplankton behaviour in strongly three-dimensional (3D) lake environments, hydraulic models in two- and three-dimensions have been used to determine habitat suitability. Leclerc *et al.* (1995, 1996) used a finite-element solution of the depth-averaged flow equations, including a wetting and drying treatment, with a habitat suitability index (HSI), to explore habitat changes on the Moisie River, Quebec (Leclerc *et al.*, 1995) and the Ashuapmushuan River, Quebec (Leclerc *et al.*, 1996). The HSI was derived from observed vertically averaged velocity, depth and substrate characteristics for Atlantic salmon fry and parr, using multivariate statistical techniques. The HSIs were driven by the hydraulic model predictions and were used to determine how the percentage usable area varied as a function of discharge. The advantage of this approach is that it recognises that as some habitat becomes less suitable as flow depth and/or velocity locally rises, so other habitats become more suitable (e.g., if they were originally dry, but become wet). This showed that there was a rapid increase in the percentage usable area for both parr and fry habitat up to 50 m^3s^{-1}. Above this, it remained relatively constant for fry, but rose more slowly for parr. Most importantly, they noted that the spatial scale of model predictions was similar to that of known ecological function – salmonids defend territories that rarely exceed 4 m^2. In a similar study, Tiffan *et al.* (2002) used a depth-averaged model to simulate flow depths and velocities at 36 steady state discharges. The biological model was based upon multivariate logistic regression in which the probability of the presence of subyearling fall Chinook salmon was predicted from physical habitat parameters predicted by the model. The results showed that estimates of rearing habitat decreased as flows increased and that estimates of the area that fish could become stranded in initially rose, but then fell. However, they noted that even the 16 m^2 resolution adopted was too coarse to characterise adequately the habitat needs of the subyearling fish. The biological model in the Tiffan *et al.* study was based upon a habitat probabilistic model and Guay *et al.* (2000) used a similar approach, but compared it with HSI approaches. They found that the habitat probability index (HPI produced better results than the HSI. They also noted that this may be because of the multivariate nature of the HPI approach in which predictions from the hydraulic models are considered simultaneously rather than independently, so dealing with a common criticism of approaches like PHABSIM.

Crowder and Diplas (2000) extended these approaches to the determination of energy gradients and ‘velocity shelters’ or refugia in gravel-bed streams. They recognised that point predictions of hydraulic variables may not always provide a sufficient representation of habitat, as spatial variation in these variables is also a crucial fish (e.g., Hayes and Jowett, 1994) and macroinvertebrate (e.g., Lancaster and Hildrew, 1993) requirement. Crowder and Diplas undertook a higher resolution modelling study, which included mesoscale topographic features (e.g., boulders), that generated predictions for input into spatial habitat metrics. These seek to capture the physical habitat difference between two locations with the same velocity but different surrounding velocities in terms of velocity and kinetic energy gradients. For instance, by scaling the spatial change in kinetic energy between two points by the kinetic energy at the point with the smaller velocity, they derived a metric that represents the kinetic energy spent by an organism to move from the point of lower velocity to the point of higher velocity. As these are based upon gradients, the determined metrics depend upon the spatial scale over which calculations are made. This needs to be evaluated in relation to the spatial resolution of the mesh used in the model and in relation to the behavioural aspects of the organism being considered. The research showed that the presence of boulders resulted in a substantially more complex spatial metric that provides an increased habitat range for fish. Linking this to observed fish behaviour (Crowder and Diplas, 2002) confirmed that boulders enhanced the potential availability of the right habitat.

Booker *et al.* (2004) coupled a 3D computational fluid dynamics (CFD) model to a bioenergetics model to predict physical habitat availability for juvenile salmon while drift feeding. The bioenergetic model included physiological and behavioural relationships that resulted in energy gain. The 3D CFD was important as it recognised the influence of locally significant gradients in flow velocity and depth in both the vertical and horizontal, which were in turn found to have a major impact upon possible foraging area. However, it was also noted that this kind of approach can be problematic when it comes to model validation as it assumes that the prime control on habitat selection is energetics rather than other factors. In addition to this being a validation issue, it emphasises that this kind of modelling approach predicts physical habitat availability in relation to model parameters and not the actual presence or absence of fish.

The above examples illustrate how habitat suitability modelling can now be undertaken for one (instream) part of the freshwater ecosystem. It has proved possible to link habitat suitability to abiotic changes in factors such as stream temperature and land use (e.g., Jessup, 1998). This kind of approach can also be applied to a range of other freshwater environments. However, this kind of modelling approach is only as good as the habitat data used to drive the models. In terms of instream fish habitat, it is strongly dependent upon characterisation of the mesoscale and microscale bed topography that determines habitat availability. This is not necessarily a major issue as habitat suitability curves are essentially statistical generalisations of habitat suitability within certain spatial units. Research being conducted in the University of Durham showed that there are ways to reproduce, using geostatistical methods, the range of topographic variability within a spatial unit of a river, and that this is sufficient to reproduce existing habitat suitability curves. This does open up the possibility of modelling the effects of changes in abiotic factors, such as river discharge, upon habitat suitability with significantly less field data collection.

iv) Spatially and temporally explicit models of fish behaviour

The above examples are entirely focused upon habitat suitability. They take no account of other factors that might impinge upon the actual characteristics of the ecosystem that are realised. The mathematical modelling required to do this is much more complex and still in the early stages of development. One of the best examples is that of Gaff *et al.* (2000), who used a simple cellular model to explore the spatial density of fish populations through time in freshwater marshes. It includes a predator–prey relationship between larger and smaller fish. Fish spread across the marsh during flood conditions. As the marsh dries, fish either retreat into refugia or, if they become isolated, die as the cell dries out. The model was used to assess the impacts of flow regulation. It could be driven climatically through evapotranspiration representation. It could also be used to explore climatic effects through changes in available flow.

Roy *et al.* (2004) modelled the growth rate potential of fish in response to temporal fluctuations in both prey availability and temperature structure of the water column at both long (seasonal) and short (daily) time scales. The model provides a cellular representation (*Figure 6.9*) of a column of water, within which specified abiotic factors are used to drive foraging and bioenergetics models that determine species and size-specific fish production potential. However, environmental change can be incorporated through applying environmental conditions to a suitable model of lake hydrodynamics and then using this to drive the fish population model.

Van Winkle *et al.* (1998) were able to link the hydraulic component of the physical habitat simulation system, PHABSIM, to an individual-based population model for brown trout, incorporating stochastic elements that represented movement, spawning time and mortality. Similarly, Thorpe *et al*. (1998) developed a life-history model for *Salmo salar* (Atlantic salmon) that identified the times at which developmental conversions are initiated or inhibited, and the relationship between physiological states and the thresholds for such conversions. The model concentrates on two major changes in the life cycle:

- the smolt stage when the fish migrate from fresh water to the sea;
- sexual maturation before the salmon return to fresh water.

Bell *et al.* (2000) note that the chief purpose of the model of Thorpe *et al*. (1998) was to identify the times at which the two major changes occurred, using physiological interactions between the genotype and the environment. It does not predict the number of survivors, but the timing of two crucial events that could be incorporated in a more general survival model, such as that which Bell *et al.* (2000) developed.

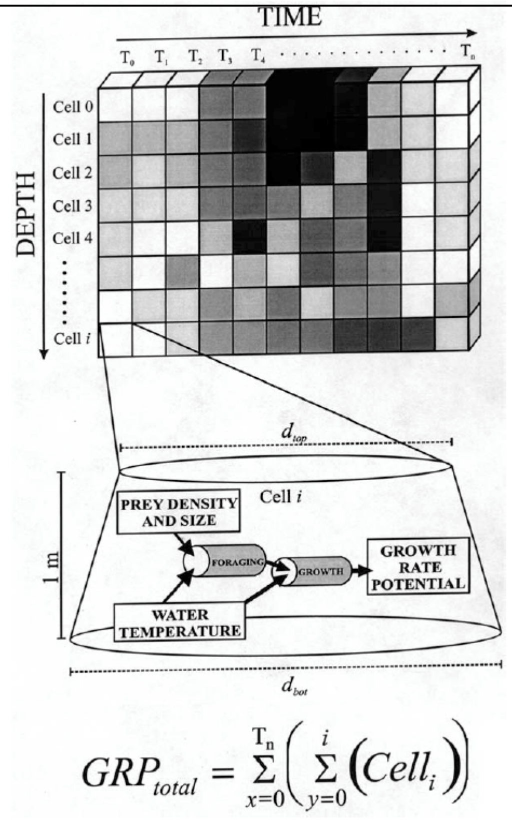

Fig. 1. Conceptualisation of the temporally explicit modelling framework, dividing the sampled environment into volumetric cells indexed by time and depth. Each cell ($Cell_i$) is characterised by a particular suite of environmental conditions which are entered into the foraging, and bioenergetics sub-models determining species, and size-specific fish production potential. Each cell has the shape of a cone and is re-sampled every minute (T_n) and provides a temporal assessment of potential production within that cell. Production potential integrated through each depth cell (G all $Cell_i$s) and summed over all times ($\sum T_n$) demonstrates and quantifies the production potential of the sampled environment over a specified time interval (adapted from Brandt and Kirsch, 1993).

Figure 6.9 The cellular representation of time and depth in the water column model of Roy *et al.* (2004), which was used to drive foraging and bioenergetics models of fish response.

Advantages

Conceptually driven mathematical models have a number of important advantages. Firstly, if they are run in a time-dependent mode, they are the only modelling type that, in principle, can represent path dependence in an ecosystem for which there is no *a priori* empirical evidence of the nature of that path dependence. Secondly, the approach explicitly recognises feedbacks and interactions between components included in the model. Thirdly, the approach may allow modelling of coupled ecosystems. One of the fundamental properties of the freshwater environment is strong feedback – a change in water balance in an upstream part of the catchment will have impacts upon the riparian corridor downstream. This type of modelling approach may be able to simulate these types of interactions, and hence the impacts of climate change upon more than one part of the ecosystem simultaneously.

Disadvantages

The main disadvantage of this modelling approach is that it is a significant undertaking. Firstly, the range of processes that has to be modelled is extensive. In many cases, the exact nature of these processes and the way they interact are only now been revealed through field and experimental studies. For instance, phosphorus is a key nutrient for primary productivity and can have a major effect upon the ecology of shallow lake ecosystems. However, it is only now that we are really gaining a proper understanding of the nature of phosphorus cycling in river and lake ecosystems, and this research is revealing that system behaviour is sensitively dependent upon conditions that are difficult to predict. The extensive nature of the possible process representation will restrict candidate environments for modelling using this approach to those for which the science base behind these processes is well-developed and the processes themselves can be represented mathematically.

Secondly, developing models that are effective requires a careful balance between the inclusion of processes that are possibly relevant and processes that are definitely relevant. The natural tendency in many modelling exercises is to include as many processes as possible in the hope that this will make the model more representative of the system under consideration. However, this can lead to serious problems of parameter specification and may sometimes result in model equifinality (where different combinations of model parameters can be used to give observed system behaviour). The alternative is to include only those processes required to give a sufficient representation of the system, possibly with some simplification. However, this immediately opens the model up to criticism, in terms both of those processes that have been included versus those that have been excluded and of the nature of the simplifications to processes that have been introduced. Thus, these kinds of models are easily criticised when opened up to scrutiny, especially by those who are not aware of the limits to creating a fully – specified, conceptually driven mathematical model.

Thirdly, the data demands to implement these models (specification of boundary conditions, estimation of parameter values, acquisition of data for validation) can be significant. For instance, it is possible to model dissolved oxygen dynamics in river systems, and these can be an important component of ecosystem behaviour. However, dissolved oxygen levels are commonly sensitively dependent upon re-aeration processes, and these can require quite detailed knowledge of local river morphology and instream flow structures, as re-aeration can dominate the dissolved oxygen signal.

Fourthly, the models may contain a large number of adjustable parameters (e.g., *Table 6.2*). Indeed, the more complex the model, the larger the number of processes that must be parameterised. There are a number of options for parameterisation:

i) derivation from field or laboratory experimentation;
ii) determination from field measurement or sampling (e.g., the chlorophyll-*a* content per unit weight of organic matter);
iii) estimation through changing model parameter values to give the correct fit to measured data.

Most modelling approaches involve all three of these methods (e.g., Romero *et al.*, 2004; Beran and Kargi, 2005). The main issue with both (i) and (ii) is the extent to which derived values transfer from field or laboratory measurements (which are specific, by definition) to generic parameter values that are constant in space and time. In a model with a small number of parameters (commonly less than five), (iii) is a useful option, provided suitable validation data are available (see above). However, as the number of parameters increases, so different combinations of parameters may give the same result, and the model displays equifinality. This problem may be reduced if only a subset of the parameters associated with a model has a major impact upon model predictions. This is why sensitivity analysis is a crucial component of conceptually driven mathematical modelling. It also requires careful design of modelling methodologies to identify a number of sets of model parameters that give plausible (sometimes called behavioural, e.g., Beven, 1979) simulations. The model's predictions can then be described based on parameters that give the full range of model predictions (e.g., mean, standard deviation, 5^{th} and 95^{th} percentiles).

Table 6.2 Illustration of the dependence of conceptually-driven mathematical models upon adjustable parameters.

Authors	*Model Description*	*Number of Adjustable Parameters and Comments*
Beran and Kargi (2005)	Coupled abiotic–biotic hydrodynamic model of algal growth in a wastewater stabilisation pond	39, 11 of which required calibration
Gamito and Erzini (2005)	Only considered biotic processes in a mass balance type model of trophic interactions for a reservoir in Portugal	50
Jimenez-Montealegre *et al.* (2002)	Mass balance model of nitrogen transformation in fish ponds with a fish, a phytoplankton and a water-sediment module	29
Malmaeus and Håkanson (2004)	Mass balance model of lake eutrophication with a particular focus on phosphorus	17
Robson and Hamilton (2004)	Coupled hydrodynamic-ecological model to explore post-flood algal blooms	150
Romero *et al.* (2004)	Algal model based upon a coupled nutrient cycling and hydrodynamic treatment	40, seven of which required calibration. Number of parameters involved precluded uncertainty analysis

Fifthly, and following from the above, these models require propagation of boundary conditions and parametric uncertainties through the modelled variables, which means that predictions of ecologically relevant variables may be swamped with uncertainty. This creates two problems. Firstly, it can result in excessive computational demands if the analysis is to be undertaken correctly, especially if the number of adjustable parameters is large, and hence the number of plausible parameter combinations is very large. Secondly, it means that relatively small and/or subtle changes in the state of the system may not be clear from model predictions.

The final challenge for this type of approach is the severe difficulties encountered in modelling aspects of the biota. The case examples provided above demonstrate that it can be done. However, a number of difficult issues arise. For instance, the timescales of biotic and abiotic responses are commonly different, such that coupling a biotic and an abiotic model can be numerically difficult. As some components of ecosystems will move, sometimes over long distances, in response to both abiotic and biotic drivers, it is necessary to be able to couple this kind of behaviour to such models. Similarly, elements of the biotic system have life-cycle dependent behavioural traits that require them to be modelled as individual members of a species (i.e., as Lagrangian entities, with interactions between them, such as fish) rather than as functional groups (i.e., as Eulerian entities, such as certain types of phytoplankton). Fully Lagrangian treatments of ecosystems are not computationally feasible as a result of population sizes lower in the food chain. Fully Eulerian approaches do not necessarily represent the full range of species-specific and individual-specific behaviours. Some innovative examples of modelling have sought to couple Lagrangian and Eulerian approaches (e.g., Thorpe *et al.*, 1998; Roy *et al.*, 2004). The key point that emerges from this discussion is that conceptually based mathematical models of biotic systems are feasible. As the case examples section noted, these are especially well-developed for lower trophic levels. For higher order trophic levels, this is still, very much, an emerging research area, for which the state of our ecological understanding may still be insufficient for whole ecosystem modelling and for which most modelling activities remain focused upon one part of ecosystem dynamics.

6.4 Key issues to consider in assessing target systems and habitats for modelling

The aim of this section is to synthesise the material addressed in Section 6.3 to summarise the key issues in considering which ecosystems will be modelled in the next stage of the PRINCE project.

6.4.1 Integrated versus ecosystem-specific modelling

Choosing which ecosystem to model requires us to recognise the complex interaction between abiotic and biotic variables in terms of how a freshwater ecosystem responds to climate change. Ecologically relevant variables (e.g., flow inundation extent) are strongly controlled both by sources and pathways, and to understand climate change requires consideration of the integrated effects of these pathways. This implies that to gain a proper understanding of the system the analysis should be holistic, as far as is possible. Care must be shown when a particular component of the aquatic environment is isolated and studied when other components of the terrestrial–freshwater system will also change and have a bearing upon the environment being considered. This is why space–time substitution, climatic analogues and data-driven approaches are potentially important. It is clear from the review of deterministic models above that, while they can be sophisticated in their process representation in certain areas, there are very major uncertainties in capturing the full biotic complexity of the system being considered.

6.4.2 Regional variations in UK climate predictions

UKCIP02 scenarios suggest very different climatic responses in different regions. Thus, the response of a given aquatic environment to climate change will be regionally dependent. This means that identifying representative environments may be difficult – the same category may respond very differently to ‘climate change’. Thus care must be shown in extrapolating the application of climate change scenarios to any one case study environment in any one region to other regions in which climate change could be different. This was found in the recent UKWIR climate change and water quality project

(CL/06) in which, using a conceptually driven mathematical mode, different water-quality parameters responded in different ways to climate forcing, according to their geographical location. Note that this may mean we need to run climate scenarios for different regions on a case-study environment from one particular region.

6.4.3 Process representation and modelling framework

Prior to identification of the chosen environments for modelling, it will be vital to consider what can be modelled in terms of our current process understanding and modelling capabilities. This is true of all of the approaches described above. For instance, and as shown above, we know that the ecological response of a lake ecosystem is strongly predicated upon phosphorus and nitrogen cycling; but there remains considerable scientific uncertainty as to the behaviour of these nutrient cycles, let alone a clear consensus on how best to model them. Thus, an important part of the PRINCE project will be to:

(1) identify what the right modelling approach will be (e.g., physically based versus statistical–empirical) for each of the systems identified in Section 3;
(2) flag those systems for which our process understanding or technical capability is insufficiently developed to model at this point in time (i.e., additional research is required).

6.4.4 Data demands

The review of possible modelling approaches presented here emphasises one fundamental issue that will severely limit the choice of both modelling approaches and case examples that we can consider – data demands. All of the modelling approaches described require a data-based understanding of the system being studied. How these data are used varies between modelling approaches. In the space–time substitution and climate analogue approaches, data are required to provide an understanding of current climate–ecology linkages, to identify how climate change may cause the geographical distribution of species to change. In data-driven modelling approaches, data are required on all relevant components of the system to be modelled to derive generalised statistical relationships for forecasting. In conceptually driven mathematical modelling, data are required as boundary conditions for the model, to specify process rates and, in most cases, to parameterise and to validate the models. It follows that data demands will also depend upon the type of system being considered. Hence, data demands must be factored into the choice of both the system to be studied and the modelling approach to be adopted.

Sites with potentially suitable datasets (long term datasets) in the UK (e.g., those held by the Environment Agency for river hydrochemistry) are generally sourced for specific purposes (e.g., compliance assessment). They are therefore frequently at inappropriate time intervals or insufficiently representative to enable their use as input data to climate change studies. Difficulties are specific to individual databases, but include incomplete sets of parameters, lack of event-based data (e.g., rarely coinciding with drought or flood periods) and insufficient representation of diurnal, seasonal or inter-annual variation.

A handful of long-term hydro-ecology research programmes collect data that may be suitable for input to the proposed modelling framework, such as:

- Environmental Change Network
  (15 river sites and seven lakes in England and Wales – water chemistry, surface water discharge, macroinvertebrates, aquatic macrophytes, zooplankton and phytoplankton). Fieldwork started in 1994.
- Lowland Catchment Research (LOCAR)
  (three river catchments in England – hydrology and ecology). Fieldwork runs from 2002 to 2006.
- Catchment Hydrology and Sensitive Management (CHASM)
  (Eden, Upper Severn – with two upland mini-catchments).

6.4.5 Uncertainty

One issue merits special mention at this point – uncertainty. All three modelling approaches have uncertainties, but these vary in nature between each approach. For instance, in the data-driven approaches, uncertainties largely relate to possible future system behaviours that cannot be captured by the statistical generalisations contained within the model. In conceptually driven mathematical modelling, uncertainties can arise from parameters or from poorly specified boundary conditions. This kind of uncertainty is an important reminder that predicting climate change impacts on freshwater ecosystems will not lead to definitive statements of what the water environment, habitats and species will look like at future time periods. Rather, and following the principle behind future climate scenarios, it should lead to the identification of plausible scenarios given our current understanding of the biotic and abiotic environment. Thus, each case-study application will contain a statement of the possible uncertainties associated with the application, as well as the scientific work required to address them.

7 Discussion

7.1 Introduction

This Project Record supports the view that climate change is already having observable effects on ecosystems across the globe. In the UK there is relatively little published research on the implications of climate change for freshwaters. The purpose of the research is, therefore, firstly to identify potential climate change impacts on the studied ecosystems, and secondly to derive, if appropriate, the 'proof of concept' for each of the suggested modelling approaches. If the approaches are suitable they could be used subsequently on other ecosystem types or at different geographical locations to give a more thorough assessment of potential climate-mediated changes (spatial differences in ecosystem types and climate scenario outputs may be considered in the future).

The review sought to identify the current scientific knowledge base from which to identify the potential vulnerability of freshwater ecosystems. A logical framework has been developed, based on the general hydrological functioning of aquatic systems, which has identified rivers and streams, standing open waters and canals, and wetlands and washlands as the functioning groups. There are a range of supported habitats, communities and species with each hydrological grouping.

Climate change will have a fundamental effect on a range of meteorological parameters (e.g., rainfall, humidity, wind speed, etc.) that in turn influence the abiotic (hydromorphological and physicochemical) regime of each supported habitat. The review sought to identify, in a broad sense, the interaction between these parameters to arrive at an understanding of which ecosystems, habitats and/or species are most vulnerable to climate change.

The sensitivity of freshwater ecosystems to climate change depends on two major components:

- i) the magnitude, frequency (return period), timing (seasonality), variability (averages and extremes) and direction of the predicted climate changes;
- ii) the sensitivity and resilience of the ecosystem, habitat and/or species to those changes and its interaction (including synchronicity) with the other various ecological components.

In a general sense, those ecosystems at the extreme range of their supported environmental envelope will be most affected. For example, these may include changes in distribution for organisms sensitive to:

- increased temperature in low-temperature habitats of restricted distribution (e.g., cold-water stenotherms at high altitude);
- major increase in drought frequency (e.g., salmonids that require spawning and juvenile habitat in upland tributaries; species within lowland raised bog in the south-east);
- increased flood risk in flood-sensitive habitats (e.g., floodplain grazing marsh, reedbed).

However, other ecosystems are either functionally more important (e.g., support a wider range of habitat types) or have major nature conservation significance must also be considered. The magnitude of change may not be as great, but the significance may be higher. These issues were considered in selecting the first habitat types to be modelled.

7.2 Vulnerable ecosystems

From the assessment undertaken, a qualified analysis of the relative vulnerability of the habitats supported by freshwaters has been undertaken. In order of reducing vulnerability these are:

- upland headwaters
- lowland headwaters
- lowland raised bog
- ephemeral ponds
- groundwater-fed rivers
- ditches
- all lakes
- fens
- floodplain grazing marsh and grassland
- lowland rivers
- middle rivers
- wet woodlands
- reedbeds
- reservoirs and canals (operational management is the key consideration).

It is recognised that not all of these habitats can be covered in the first tranche of studies, but we suggest that development be considered of those proposals in which sufficient interest is generated by the wider stakeholder community.

7.3 Proposed ecosystem modelling approaches

A number of ecosystem modelling approaches are potentially available, either by application of existing methods or by developing a novel methodology (based on emerging catchment modelling techniques). The evidence gathered through the Project Record suggests that at, this stage in the project, with the potential for future expansion of the project portfolio, it would be useful to test different approaches to seek the best blend of predictive outcome. In effect, to arrive at a “proof of concept” for one or a number of potential modelling approaches.

As the review has progressed it has become increasingly evident that two methodologies merit further attention:

- an empirical data-driven approach – integrating biotic and abiotic factors;
- a deterministic modelling approach – predicting changes in abiotic parameters to use as inputs to an assessment of changes to selected habitats, communities or species.

In response to direction given by the PRINCE steering group, the Cascade Consulting project team have developed three broad hypotheses to test through research during the remaining phases of this PRINCE programme. Research effort will initially focus on addressing hypotheses 1 and 2 below in detail, moving on to addressing hypothesis 3 at a high level:

(1) The signals of present and future climate change are sufficient to be detectable against the background variability of the (i) hydrology, (ii) hydrochemistry, (iii) sediment dynamics and (iv) thermal regime of a given river system.
(2) The resulting changes in hydrology, hydrochemistry, sediment dynamics and thermal regime are sufficient to cause measurable biological change that is detectable against the background variability of aquatic community composition.
(3) The effects of climate change on aquatic community composition are greater than those caused by variation in land use.

The proposed research framework to address hypotheses 1 (*Table 7.1*) and 2 (*Table 7.2*) is described below. Further discussion with the Environment Agency project management team is required to develop a suitable research framework to address hypothesis 3.

Table 7.1 Research framework to test hypothesis 1 (abiotic factors).

Location	*Habitats Investigated*	*Abiotic Factors Investigated*	*Suitable Model*	*Model Runs*	*Interpretation*
Upper Wharfe, Yorkshire	Headwaters (upland catchment)	Discharge Wetted area Velocity Water temperature	CAS-Hydro coupled to hydrodynamic modelling code	Sensitivity testing Present climate 2050s climate 2080s climate	Seasonality Extremes Averages Duration Periodicity
Middle Wharfe, Yorkshire	Middle reaches	Changes in discharge	FEH	Present climate 2050s climate 2080s climate	Seasonality Frequency Duration Periodicity
Upper Wharfe, Yorkshire	Headwaters (uplands catchment)	Coarse sediment turnover	S Lane	Present climate 2050s climate 2080s climate	Frequency

Table 7.2 Research Framework to Test Hypothesis 2 (Freshwater Ecosystem/ Biodiversity Consequences).

Location	*Habitats Investigated*	*Abiotic Factors Investigated*	*Suitable Model*	*Model Runs*	*Interpretation*	*Assemblages Investigated*	*Biotic Data*
Yorkshire rivers	Headwaters (upland catchment) Middle reaches of river	Discharge	CAS-Hydro	1990-1998 2050s climate 2080s climate	Q_5, Q_{10}, Q_{90}, Q_{95}	In-channel benthic macro-invertebrates	Environment Agency annual macro-invertebrate survey across 91 sites 1990-1998
		Sediment risk	Empirical		Ranking/ banding		
		Water temperature	CAS-Hydro		Appropriate statistics		
Upper Tywi, mid Wales	Headwaters (upland catchment)	Discharge	FEH	1984-2004 2050s climate 2080s climate	Appropriate statistics	In-channel benthic macro-invertebrates	Cardiff University macroinvertebrate species dataset
		Water temperature	Empirical				

Notes: Interrogation of Environment Agency routine monitoring water quality dataset for Yorkshire rivers (1990-1998) to be investigated further. Prediction of future change in hydrochemistry not currently feasible.

Hydrochemistry data available for 1984-2004 upper Tywi dataset, prediction of future change in hydrochemistry not currently feasible.

Implementing the approaches described, the modelling would provide predictions of the ecological consequences of climate change for, at least:

- headwaters (upland catchments);
- middle reaches of river;
- improved grasslands on floodplains;
- floodplain grazing marsh.

Macroinvertebrate and trout communities that inhabit the headwaters and middle river reaches would be analysed to identify potential effects from the known consequences of changes to physicochemical attributes of the freshwater ecosystems (e.g., flow, temperature, wetted area, etc.).

7.3.1 Empirical data-driven approaches

Previous research on acidification provides a template for the need to link large-scale climatological effects with impacts on hydro-ecology. Studies that examined the effects of acid deposition showed how deterministic hydrochemical models (e.g., MAGIC – the Model of Acidification of Groundwaters in Catchments) could be linked to a range of empirical, statistically derived models to predict invertebrate assemblage composition, macro-algae, salmonid density and even the distribution of river birds (Ormerod *et al.*, 1988; Ormerod and Tyler, 1989; Weatherley and Ormerod, 1989). This approach was subsequently used to address policy issues, for example to compare outcomes from different future emission and deposition scenarios with those engendered by catchment mitigation (Ormerod *et al.*, 1990). It has since continued to influence Defra in international negotiations over acid deposition impacts (http://www.nbu.ac.uk/negtap/)

and other issues that reflect diffuse pollutants (e.g., Dalton and Brand-Hardy, 2003). Statistical modelling of this type has been at the heart of one of the longest running tools of all used by the Environment Agency, RIVPACS, and equally is helping to develop a more modern counterpart built on machine-learning. It is applicable to a wide range of organisms – but will be demonstrated here using invertebrates in view of their contribution to biodiversity, their indicator value and data availability.

The approach circumvents the need to parameterise the individual, multivariate and highly complex species response to climatic variation – for example, that would result through individual based models or population viability analysis. These approaches are not only extremely data hungry, but also cannot be readily scaled up to wider community responses. Whole assemblage response models, while 'black-box' in character, can be highly effectively tested using a range of experimental or survey-based methods (e.g., Rundle *et al*., 1995).

In this particular case, two empirical modelling approaches will be developed to link data on invertebrate assemblage composition to hydrological input data for:

- the Wharfe river system, representing river middle reaches, using data from the Environment Agency;
- upland mid-Wales rivers, based on one of the longest running datasets on upland rivers anywhere in the UK at Llyn Brianne.

Data available in both cases already reveal climate and/or hydrological response as major drivers of invertebrate variation (Bradley and Ormerod, 2001, and unpublished data). In this case, the steps involved would be:

- parameterise year-to-year variation among invertebrates using ordination (e.g., canonical correspondence analysis, hierarchical classification, …);
- develop transfer functions with hydrometeorological drivers (e.g., discriminant analysis, …);
- drive the resulting models using new input hydrometeorological data (e.g., on future scenarios).

7.3.2 Conceptually driven mathematical modelling

Adaptation of the CAS-Hydro (fully distributed catchment-scale hydrology) model for the River Wharfe

The research area of Upper Wharfedale in the Yorkshire Dales National Park is recommended to evaluate the conceptually driven mathematical modelling approach. The catchment area is large, for this kind of study, at 72 km^2. This will involve development of the UKWIR-derived Cas-Hydro hydrological model, which currently includes coupling downscaled climate scenarios to:

- a hillslope rainfall run-off model, and
- the routing of the generated flow through the drainage network to floodplain zones,

with new process representations for:

- partitioning flow between river and floodplain;
- modelling the spatial patterns of floodplain inundation;
- modelling the spatial patterns of instream trout habitat using habitat suitability curves.

This is a substantial undertaking, but important as it will illustrate the advantages and disadvantages of a conceptually driven mathematical modelling approach as well as yield predictions of how climate change might impact upon instream habitat for a case study of an upland gravel-bed river. The Wharfe has been chosen because the system is particularly well-constrained and has all the data necessary for this kind of modelling approach.

Reid (2004) provides a review of the Upper Wharfe. In recent decades, the catchment has experienced large volumes of coarse sediment transport and generation, severe bank erosion, increased incidence of downstream flooding and water-quality deterioration. Previous work has synthesised the hydrological, aerial photography, laser altimetry and channel survey data necessary for the proposed modelling. The Upper Wharfe is of significant ecological importance – parts of the upper floodplain included in the proposed catchment area are designated SSSI and the river itself is designated as an SSSI. The river includes short reaches of limestone pavement, with associated mosses, some higher plants typical of lowland rivers and bankside reeds. There is also herb-rich grassland, a range of sedges (including nationally rare examples) and important birdlife (e.g., dipper and kingfisher).

7.3.3 Further considerations

Given the complexity of the ecosystems that are being modelled and the potentially different approaches, there is merit in considering f the various approaches being applied to a single catchment, but focusing on different attributes. There are a number of advantages of such an approach:

- the baseline catchment data are the same and only need to be collected and collated for one system;
- one suite of climate change scenarios and downscaled data would be needed;
- outputs of the deterministic model would inform the empirical assessment;
- modelling outputs for each type of modelling approach could be analysed to assess their relative value for future climate research;
- coupling of the model outputs in this way should enable our understanding of ecosystem dynamics to be developed.

Given the budget and timescale constraints, this is seen as the most suitable way forward at this time. A notable disadvantage of the approach is that it does not explicitly consider climate variability across England and Wales. However, the studies will enable the potential for climate change to be defined for some of the recognised vulnerable ecosystems, meeting the criteria set out in the project Terms of Reference.

A final consideration is the problem of identifying the potential future pattern of re-distribution of aquatic habitats and species in response to climate change. If it is agreed that there will be a general shift in ecosystem distribution, it would be useful to understand the mechanism by which it will be achieved, and the potential constraints to migration. Aquatic ecosystems are often constrained by their hydrological boundaries (catchments), and the processes by which inter-catchment transfer of habitats and species may occur remains unclear. If the initial studies suggest that shifts in ecosystem components are likely (e.g., species ranges constraining northward, introductions from abroad), it would be useful to promote a study to better understand the likely mechanisms of catchment immigration and emigration, and the assemblage and/or population propagation for non-migratory species. Desk-based research could be considered that focuses on White clawed–Signal crayfish interactions and/or Zander as analogues of introduction–extinction and mechanisms of increasing or decreasing distribution.

7.4 Stakeholder hypotheses and high-level objectives

Subject to a meeting of the Project Steering Group, at which the draft Project Record was discussed, the Environment Agency project managers requested the submission of potential testable hypotheses from the Project Steering Group. A large number of hypotheses were submitted, which could be loosely collated into the following:

- investigation of abiotic factors;
- investigation of ecosystem and biodiversity consequences;
- investigation of potential adaptation strategies.

The full range of hypotheses and 'high-level objectives' that were submitted is given in Appendix 3. A number of the hypotheses can be tested by the proposed modelling studies, with the remainder potentially suitable for later stages of the PRINCE research programme.

7.5 Suggested approaches for vulnerable ecosystems not in the proposals for further study at this stage

For those hydrological regimes for which relatively little is known of the abiotic and/or ecosystem interactions, or for those that we have not suggested research for in this initial tranche of study, a number of potential issues and research proposals are identified in *Table 7.3*.

Table 7.3 Proposals for extensions to the PRINCE programme.

Domain	*Climate Issues*	*Ecosystem Interest*	*Potential Locations*	*Approach*
Lowland headwaters	Reduced/lost summer flows, reduction in aquatic habitat availability	Ecological communities at the edge of their range, including macroinvertebrates and juvenile fish	Catchment in south-east England	Similar to that advocated for the upland headwaters. Will depend on outcome of present PRINCE modelling studies
Lowland raised bog	Reduced groundwater flows and loss of hydrological connectivity. Drying out of bogs through increased temperatures and evapotranspiration	Rare and reducing habitat with recognised conservation status	Habitat in the south-east of England	Site(s) required with good hydrological and ecological records, including surveys during previous droughts. Changes to hydrological conditions assessed, in consideration of other activities (e.g., abstractions), and identification of ecosystem responses. Modelling of future hydrology using downscaled rainfall data
Ponds	Reduced water levels in summer, with increased temperatures and potential water-quality problems	Increased marginal vegetation, macrophyte growth and changes to fish population and fish kills. Modification to pond ecology as hydrology changes	Sub-sample of inter-regional 1 ha ponds	Empirical modelling using existing pond hydrology data linked to key indicators, likely to include macroinvertebrates and fish. Lack of data may require monitoring programme. Analysis over drought period would be beneficial
Predominantly groundwater-fed rivers	Increased incidence of summer drought and wetter winters on river hydrology Influence of thermal regime and physico-chemical water quality on biota	Altered habitat availability and water quality on recruitment of salmonid and coarse fish	Rivers Itchen, Kennet or Frome	Similar modelling to that proposed for Wharfe. River must have suitable groundwater model at appropriate resolution. Could make use of extensive Itchen datasets
Ditches	More extreme flows, increased erosion, increased low flows and desiccation	Greater number of ephemeral ditches and loss of associated habitats	Ock catchment; est catchment such as RPA 8	Ecological surveys linked to fully distributed catchment modelling at high resolution, using downscaled rainfall data

Domain	*Climate Issues*	*Ecosystem Interest*	*Potential Locations*	*Approach*
Lakes	Reduced inflows leading to change in water level, wetted area and physico-chemical water quality Initial focus on oligotrophic and eutrophic lakes	Change in marginal vegetation Change in plankton and fish populations	Windermere Ullswater Broads	Lake inflow and level modelling using existing water company hydrological models to predict change in variability over time. Use of previous droughts as analogues of potential future climates. For Broads could use empirical data on hydrology and ecology to derive relationships to predict future changes in climate
Fens	Slight increase in winter flooding and summer drought altering hydrological regime	Change in fen community structure in response to hydrological regime change or water quality impacts	Site-specific investigation where data already exist. Sites currently unknown	Empirical modelling where data exist to link hydrological conditions to changing ecology. May require monitoring to provide baseline data
Floodplain grazing marsh and improved grassland on floodplain	Increased winter and reduced summer inundation Higher temperature and nutrient loads Change to sediment supply interactions	Change in hydrological regime leading to modification to habitats and supported communities	River system with extensive functioning floodplain system	Use existing databases and analyses to identify potential sites (e.g., Open University (Dr Gowing) or Cranfield University (Dr Joe Morris)). Changes to flooding potential, inundation periodicity, etc., to be derived from catchment hydrological model. Ecological effects to be derived from empirical relationships
Lower rivers	More extreme flows, increased temperatures, reduced water quality Sea level rise increasing saline ingress and moving estuarine boundary upstream	Increased macrophytes growth, dissolved oxygen problems, evolution of biota in response to higher salinities	River Thames Essex rivers	Use 2D hydrodynamic and water-quality model of existing system to predict change in saline ingress for range of climate and sea level rise scenarios

Domain	*Climate Issues*	*Ecosystem Interest*	*Potential Locations*	*Approach*
Wet woodland	Change to hydrological regime, linked to inundation patterns and potentially groundwater connectivity	Influence of changing hydrological response to climate on uptake of water by wet woodland, and also adaptation of woodland to new inundation patterns	Site-specific investigation – could potentially use Defra Ripon project as template	Consideration of studies by Cambridge University (FLOBAR (Prof. Keith Richards)) to identify sites for analysis. Probably requires linked surface and groundwater modelling to describe changes to hydrological regime. Complex unless empirical approach taken
Reedbeds	Increased habitat availability during droughts as marginal areas extend. May be countered by reduced upper limit on inundation	Increase or decrease in range in response to change in hydrological and/or water-quality conditions	Site-specific investigation	Site required with good hydrological and ecological datasets. Water-level modelling will be required to identify how habitat availability may vary over time and with future hydrological conditions
Canals	Stress in supply reservoirs leading to change in canal velocity, temperature and water quality	Change in aquatic populations, including plankton and fish	Extensive canal network, with significant variability in operating rules and boat traffic. Will require site-specific investigation	Use of British Waterways (BW) ecological datasets. Hydrology will require simulation using BW hydrological modelling suite. Prediction of impacts on ecology from use of extensive datasets (different canals, different flows) and prediction of future water resource allocations (including climate change scenarios)
Reservoirs	Change to inflow patterns, modifications to abstraction regimes, leading to more frequent draw-down. Water quality may change	Effects on marginal communities, plankton and fisheries	Operational reservoir with suitable hydrological and ecological datasets	Hydrological modelling of inflow characteristics and water-level fluctuations over time. New inflow records from climate scenario treatment to allow future inflows to be predicted. Water level and outflows can be identified and impacts on biota estimated

Once the most sensitive ecosystems in the general typology have been investigated at a strategic level, it is recommended that the typology be revisited and greater resolution included for those ecosystems most at risk. This may, for example, include sub-dividing the rivers and lakes into a greater number of categories, most probably on the basis of the WFD typology. It is also likely that a description of the relative risk of the different freshwater systems will require a geographical context, given the north-east–south-west axis of predicted climate change effects under UKCIP02 scenarios. A spatially derived matrix of rivers, lakes, etc., within a national geographical framework could then identify which systems are most at risk. The outputs from this initial research would inform which types of predictive methodology would be most suitably applied to each.

Consideration should also be given to fundamental science on the response of micro-organisms, fungi and lower order organisms to climate change. The implications for survival and vectoring of pathogens have not been reviewed in detail and merit further discussion.

Glossary

Acidfication – a process whereby air pollution (ammonia, sulphur dioxide and nitrogen oxides) are converted into acid substances

BAP Priority Habitats - habitats that represent distinctive and biologically important land use systems which have given rise to characteristic habitat mosaics

Carbon cycle – a biochemical cycle that describes the movement of carbon, in its many forms, between the biosphere, atmosphere, oceans and geosphere

Cyanobacteria – photosynthetic bacteria formerly called blue–green algae

Denitrification – a process by which bacteria and fungi decompose plant or animal waste, converting it into nitrogen gas

Diatoms – a microscopic algae with plate-like structures composed of silica

Dystrophic lake – lakes that are very acidic and poor in plant nutrients

EU Birds Directive – a major European initiative that aims to take measures to conserve all naturally occurring bird species across Europe

EU Habitat Directive – a major European initiative that aims to contribute towards protecting biodiversity - the variety of life - through the conservation of natural habitats and wild plants and animals

Eutrophication – the addition of mineral nutrients to an ecosystem, generally raising the net primary productivity

Evapotranspiration – the diffusion of water vapour into the atmosphere from a vegetated surface

Greenhouse gases - gases whose absorption of solar radiation is responsible for the greenhouse effect, including carbon dioxide, methane, ozone and the fluorocarbons

Groundwater – all subsurface water whether it is in its liquid, solid or gaseous state, provided it is not chemically combined with the minerals present

Hydrological cycle – central concept of hydrology that describes the continuous movement of all forms of water (vapour, liquid and solid) on, in and above the earth's surface

Macrophyte – multi-celled plants that grow in or near water and are generally beneficial to lakes and ponds because they produce oxygen and provide habitat for aquatic invertebrates and fish

Mesotrophic lake – characterised by having a narrow range of nutrients, the main indicative ones being inorganic nitrogen and total phosphorus

Nitrification – the capture of nitrogen from the air, a process carried out by bacteria that live symbiotically in association with leguminous plants

Oligotrophic lake – lakes that have low concentrations of nutrients and as a result contain few species and relatively small populations of aquatic plants and animals

Phytobenthos – microscopic plants that live in the surface layers of the water–sediment interface

Phytoplankton – microscopic plants that live in the water

Ramsar Site – a wetland area that has been designated as internationally important according to a set of criteria under the terms of the (Ramsar) Convention on Wetlands

Run-off – a process that occurs when the infiltration capacity of the soil surface is exceeded and can no longer absorb moisture at the rate at which it is being supplied

Throughflow – downslope flow within the soil

Turbidity – a measure of water cloudiness caused by suspended solids

Water Framework Directive – a European Directive that is an integrated approach to the protection, improvement and sustainable use of Europe's rivers, lakes, estuaries, coastal waters and groundwater

Wildlife and Countryside Act – the principle mechanism for the legislative protection of wildlife in Great Britain

Acronyms

ASPT	Average score per taxon
BAP	Biodiversity Action Plan
CCW	Countryside Council for Wales
CEH	Centre of Ecology and Hydrology
CFD	Computational fluid dynamics
CHESS	Climate, Hydrochemistry and Economics of Surface-water Systems
CROW	Countryside and Rights of Way (Act)
CS	Continuous simulation
cSAC	**Candidate Special Area of Conservation**
Defra	Department of Environment, Food and Rural Affairs
DETR	Department for Transport, Local Government and the Regions
DOC	Dissolved oxygen content
DSPR	Drivers–sources–pathways–receptors
EEA	European Environment Agency
EIONET	European Environment Information and Observation Network
EU	European Union
EUNIS	European Nature Information System
GCM	General circulation model
HAP	Habitat Action Plan
HPI	Habitat probability index
HSI	Habitat suitability index
IPPC	Intergovernmental Panel on Climate Change
LIFE	Low flow index score
MAFF	Ministry of Agriculture, Food and Fisheries
MPI	Max Planck Institute
NAO	North Atlantic Oscillation
NERC	National Environmental Research Council

PRINCE Preparing for Climate Change Impacts on Freshwater Ecosystems

PSG Project Steering Group

RIVPACS River Invertebrate Prediction and Classification System

SAC Special Areas for Conservation

SAP **Special Protection Area**

SLP Sea-Level Pressure

SSSI Sites of Special Scientific Interest

UCL University College London

UKCIP United Kingdom Climate Impact Programme

UKWIR United Kingdom Water Industry Research

WCA Wildlife and Countryside Act

WFD Water Framework Directive

WUA Weighted usable area

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Appendix 1 Terms of reference

Project Specification **SC030300 (X1-0454/3)**

Preparing For Climate Change Impacts On Freshwater Ecosystems (PRINCE)

1 OVERALL OBJECTIVE

Conservation-related work is often driven by UK and European legislation that does not explicitly consider the implications of climate change. There is a need to review information and understanding of the implications of climate change for freshwater ecosystems; to inform a wide range of policies; to communicate an improved understanding of climate change effects; and to apply this to predictions of the consequences (e.g., Pearson et al., 2002).

The MONARCH consortium has identified freshwater ecosystems as a priority research area (Harrison et al., 2001). However, it is generally recognised that the ‘climatic envelope’ approach is inadequate for aquatic ecosystems. Integrated catchment analysis and process modelling techniques being developed under the EU Framework VI Euro-limpacs project (www.eurolimpacs.ucl.ac.uk) offer an alternative strategy for investigating climate change impacts on freshwater ecosystems at key sites (e.g. Wade et al., 2002). This approach will also assess the impact of climate change on freshwater processes and also in particular the impact of extreme events that may be particularly significant for freshwater species.

This project will build on current research investigating factors affecting coarse fish and salmonid recruitment. There is potential synergy with Environment Agency-funded climate change projects addressing water resources, flood risk management, diffuse pollution, land use, habitat changes and broad-scale ecological modelling. Defra are also proposing to undertake a review and update of the DETR and MAFF publication 'Climate Change and UK Nature Conservation' by 2006. This will involve looking at the vulnerability to climate change of all BAP Priority habitats and freshwater species. PRINCE will inform this review

2 SPECIFIC OBJECTIVES

The project is divided into ten main tasks that will run sequentially. Important activities at the outset will be the scoping and quality assessment of existing data, together with a comprehensive literature review that will address a wide range of freshwaters and species. This overview will inform the development and testing of process-based models for trial UK freshwater species, habitats or locations to evaluate climate sensitivity. These models will improve understanding of how climate change and its interaction with other drivers (e.g., land-use change, nutrient loading, acid deposition, toxic pollution, etc) will change the structure and functioning of selected UK freshwater ecosystems. A modular approach will be adopted to facilitate the extension of the project to additional sites and functional groups identified by co-funding partner organisations. Funding already secured by the Environment Agency will enable the literature review, model development, selection and subsequent testing of the approach for 3 freshwater ecosystems. The findings will be extrapolated to indicate the impacts on key species located in these ecosystems and where possible to priority BAP habitats and species.

3 TARGET AUDIENCE

The project will lead to authoritative and comprehensive technical reviews of the implications of climate change on UK freshwater ecosystems; scoping of data needs and key scientific uncertainties for freshwater ecosystem processes, target species or functional groups; models allowing risk-based assessment of future climate change impacts at key sites; and policy guidance for catchment/ habitat/ species management in the face of climate change and other environmental drivers.

The target audience includes: EA; Defra; CEFAS; SEPA; SERAD; DARD; English Nature; RSPB; Countryside Commission for Wales; National Assembly of Wales; National Trust, SNIFFER; British Waterways; Angling Associations; Broads Authority; the Atlantic Salmon Trust; and the MONARCH consortium. Internal Environment Agency partners include: Fisheries Policy and Process; Conservation; Ecosystems and Human Health; Environmental Policy; Habitats Co-ordinators; Water Resources and the Flood Risk Management Research Consortium. Furthermore, by aligning the project modelling strategies with those of Euro-limpacs there will be value-added through expanding the network of (UK) sites, species and functional groups investigated within a European wide programme of research.

Appendix 2 Supplementary data used in the development of the freshwater ecosystem typology

Correspondence between classification systems

Reproduced from Jackson (2000).

Table A2.1 Correspondence between the biodiversity Broad Habitat Classification and the biodiversity Priority Habitat types

Broad Habitat Type		*Priority Habitats*
11	Fen, marsh and swamp	Purple moor grass and rush pastures
		Fens
		Reedbeds
12	Bogs	Lowland raised bog
		Blanket bog
13	Standing open water and canals	Mesotrophic lakes
		Eutrophic standing waters
		Aquifer-fed naturally fluctuating water bodies
14	Rivers and streams	Chalk rivers

Table A2.2 Correspondence between the biodiversity Broad Habitat Classification and the EC Habitat Directive Annex I types

Broad Habitat Type		*Annex I Code*	*Annex I Type*
13	Standing open water and canals	3110	Oligotrophic waters containing very few minerals of sandy plains: *Littorelletea uniflorarae*
		3130	Oligotrophic to mesotrophic standing waters with vegetation of the *Littorelletea uniflorae* and/or of the Isoeto-Nanojuncetea
		3140	Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp.
		3150	Natural eutrophic lakes with Magnopotamion or Hydrocharition-type vegetation
		3160	Natural dystrophic lakes and ponds
		3170	Mediterranean temporary ponds
14	Rivers and streams	3260	Water courses of plain-to-montane levels with *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation
12	Bogs	7110	Active raised bogs
		7120	Degraded raised bogs still capable of natural regeneration
		7130	Blanket bog (active only)
		7150	Depressions on peat substrates of the Rhynchosporion (vegetation on raised and blanket bogs)
11	Fen, marsh and swamp	6410	Molinia meadows on calcareous, peaty or clayey silt-laden soils (*Molinion caenuleae*)
		7140	Transition mires and quaking bogs
		7150	Depressions on peat substrates of the Rhynchosporion (vegetation on valley mires)
		7210	Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*
		7220	Petrifying springs with tufa formation (Cratoneurion)
		7230	Alkaline fens
		7240	Alpine pioneer formations of *Caricion bicoloris-atrofuscae*

Elements of the EUNIS Habitat classification system

A: Marine habitats
B: Coastal habitats
C: Inland surface water habitats
- C1 – Surface standing waters
  - C1.1 – Permanent oligotrophic lakes, ponds and pools
  - C1.2 – Permanent mesotrophic lakes, ponds and pools
  - C1.3 – Permanent eutrophic lakes, ponds and pools
  - C1.4 – Permanent dystrophic lakes, ponds and pools
  - C1.5 – Permanent inland saline and brackish lakes, ponds and pools
  - C1.6 – Temporary lakes, ponds and pools (wet phase)
- C2 – Surface running waters
  - C2.1 – Springs, spring brooks and geysers
  - C2.2 – Permanent non-tidal, fast, turbulent watercourses
  - C2.3 – Permanent non-tidal, slow, smooth-flowing watercourses
  - C2.4 – Tidal rivers, upstream from the estuary
  - C2.5 – Temporary running waters (wet phase)
  - C2.6 – Films of water flowing over rocky watercourse margins
- C3 – Littoral zone of inland surface waterbodies
  - C3.1 – Species-rich helophyte beds
  - C3.2 – Water-fringing reedbeds and tall helophytes other than canes
- C3.4 – Species-poor beds of low-growing water-fringing or amphibious vegetation
- C3.5 – Pioneer and ephemeral vegetation of periodically inundated shores
- C3.6 – Unvegetated or sparsely vegetated shores with soft or mobile sediments
- C3.7 – Unvegetated or sparsely vegetated shores with non-mobile substrates

D: Mire, bog and fen habitats
- D1 – Raised and blanket bogs
- D2 – Valley mires, poor fens and transition mires
- D3 – Aapa, palsa and polygon mires
- D4 – Base-rich fens
- D6 – Inland saline and brackish marshes and reedbeds

E: Grassland and tall forb habitats
F: Heathland, scrub and tundra habitats
G: Woodland and forest habitats and other wooded land
H: Inland unvegetated or sparsely vegetated habitats
I: Regularly or recently cultivated agricultural, horticultural and domestic habitats
J: Constructed, industrial and other artificial habitats
- J5 – Highly artificial man-made waters and associated structures
  - J5.3 – Highly artificial non-saline standing waters
  - J5.4 – Highly artificial non-saline running waters

Definitions of hydrological domains and habitats used in the PRINCE freshwater ecosystem typology

Rivers and streams

This hydrological domain covers rivers and streams from bank top to bank top or, where there are no distinctive banks or banks are never overtopped, it does not include the floodplain area affected by out-of-bank events. This includes the open channel (which may contain submerged, free-floating or floating-leaved vegetation) water fringe vegetation and exposed sediments and shingle banks. Distinction is made, in the first instance, between hydrological regime: predominantly baseflow-fed rivers; and predominantly run-off-/throughflow-fed rivers.

Predominantly baseflow-fed rivers

This includes chalk rivers (UK BAP Priority Habitat) and sandstone rivers.

All chalk rivers are fed from groundwater aquifers, producing clear waters and a generally stable flow and temperature regime. Chalk rivers have a characteristic plant community, often dominated in mid-channel by river water crowfoot *Ranunculus penicillatus* var. pseudofluitans and starworts *Callitriche obtusangula* and *C. platycarpa*, and along the edges by watercress *Rorippa nasturtium-aquaticum* and lesser water-parsnip *Berula erecta*. They have low banks that support a range of water-loving plants and a rich diversity of invertebrate life and important game fisheries.

Most chalk rivers have 'winterbourne' stretches in their headwaters. These often run dry, or partially dry, in late summer because of lack of rainfall recharging the aquifer. A characteristic range of invertebrates has adapted to these conditions, as has the brook water crowfoot *Ranunculus peltatus*.

Headwaters (upland catchments)

From the EUNIS classification C2.2 **Permanent non-tidal, fast, turbulent watercourses**: flowing water courses of the Palaearctic region, permanent or temporary, and their associated animal and microscopic algae pelagic and benthic communities. Specifically, this category includes acid oligotrophic vegetation of fast-flowing streams.

Headwaters (lowland catchments)

From the EUNIS classification C2.2 **Permanent non-tidal, fast, turbulent watercourses**: flowing water courses of the Palaearctic region, permanent or temporary, and their associated animal and microscopic algae pelagic and benthic communities. Specifically, this category includes lime-rich oligotrophic vegetation and mesotrophic vegetation of fast-flowing streams.

Middle reaches of river

From the EUNIS classification C2.3 **Permanent non-tidal, slow, smooth-flowing watercourses**: flowing water courses of the Palaearctic region, permanent or temporary, and their associated animal and microscopic algae pelagic and benthic communities. Distinction is made from lower reaches of river on the basis of higher stream power leading to differences in geomorphology, notably reduced sedimentation of fine deposits.

Lower reaches of river

From the EUNIS classification C2.3 **Permanent non-tidal, slow, smooth-flowing watercourses**: flowing water courses of the Palaearctic region, permanent or temporary, and their associated animal and microscopic algae pelagic and benthic communities. Distinction is made from lower reaches of river on the basis of lower stream power leading to differences in geomorphology, notably increased sedimentation of fine deposits. The classification terminates downstream at the tidal limit and saline influences are not included.

Standing open water and canals

This hydrological domain includes natural systems, such as lakes, meres and pools, as well as man-made waters, such as reservoirs, canals, ponds and gravel pits. It includes the open water zone (which may contain submerged, free-floating or floating-leaved vegetation) and water fringe vegetation. Ditches with open water for at least the majority of the year are also included in this habitat type.

Standing waters are usually classified according to their nutrient status and this can change naturally over time or as a result of pollution. There are three main types of standing waters, namely: oligotrophic (nutrient-poor), eutrophic (nutrient-rich) and mesotrophic (intermediate). These lake types exist along an environmental gradient and intermediate types occur. Other types of standing water include dystrophic (highly acidic, peat-stained water), marl lakes, brackish-water lakes, turloughs and other temporary water bodies. Lake depth and altitude may also be significant factors in the nature and extent of impacts resulting from climate change.

Dystrophic lakes

Dystrophic lakes are acidic, shallow bodies of water that contain much humus and/or other organic matter, and are peat-stained in appearance. They contain very low levels of nutrients, especially nitrogen and pH is less than 6. Dystrophic lakes are generally small and of low productivity. Species diversity is low, with few macrophytes, and cold-loving fish species.

Oligotrophic lakes

Oligotrophic lakes (i.e., of low nutrient status) are deep, clear lakes with little organic matter and a high dissolved-oxygen level. Usually the biomass of plants and animals is typically lower than in more nutrient-rich waters. Most natural lakes of glacial origin in Great Britain are classified as oligotrophic. Oligotrophic waters support a characteristic suite of plant and animal communities, most adapted to low nutrient status and vulnerable to increases in nutrient levels.

Mesotrophic lakes (UK BAP Priority Habitat)

Mesotrophic lakes (i.e., those in the middle of the trophic range) are relatively infrequent in the UK and largely confined to the margins of upland areas in the north and west. They are characterised by having a narrow range of nutrients, the main indicative ones being inorganic nitrogen (N) and total phosphorus (P). Typically, mesotrophic lakes have nutrient levels of 0.3-0.65 mgN/l and 0.01-0.03 mgP/l. While such levels simplify the complex interaction between plant nutrients and the hydrological and physical characteristics of individual lakes (for instance, virtually all available nutrients are 'locked up' in algae during the growing season), they serve to show the sensitivity of the trophic state to artificially increased levels of nitrogen and phosphorus.

Eutrophic lakes (UK BAP Priority Habitat)

Eutrophic standing waters are water bodies characterised by dense, long-term populations of algae in mid-summer, often making the water green. Their beds are covered by dark anaerobic mud, rich in organic matter. The water column typically contains at least 0.035 mg/l total phosphorus (which includes phosphorus bound up in plankton) and 0.5 mg/l or more total inorganic nitrogen (mainly in the form of dissolved nitrates). Many lowland water bodies in the UK are now heavily polluted, with nutrient concentrations far in excess of these levels, although there is some geographical variation in the extent of the enrichment. Eutrophic waters are most typical of hard water areas of the lowlands of southern and eastern Britain, but they also occur in the north and west, especially near the coast.

Canals

From the EUNIS classification J5.3 **Highly artificial non-saline standing waters**: artificial watercourses and basins carrying freshwater.

Ditches

From the EUNIS classification C1.6 **Temporary lakes, ponds and pools**: freshwater lakes, ponds, pools, or parts of such freshwater bodies, which become periodically dry, with their associated animal, green algal or lower algal pelagic and benthic communities.

Ponds

From the EUNIS classification J5.3 **Highly artificial non-saline standing waters**: artificial watercourses and basins carrying freshwater. Ponds can be separated from lakes in terms of surface area, with lakes having a surface area greater than 1 ha. A functional distinction can also be made, where ponds are sufficiently small and shallow for the whole waterbody to be influenced by marginal and edge effects.

Reservoirs

From the EUNIS classification J5.3 **Highly artificial non-saline standing waters**: artificial watercourses and basins carrying freshwater.

Wetlands and washlands

The transition between open water and land is often occupied by tall emergent vegetation called swamp or reedbed, or wet woodland called carr. In practice this vegetation often forms a continuum, but is included separately in the typology.

Similarly, the semi-natural wetland habitats adjacent to rivers and streams, such as unimproved floodplain grasslands, marshy grassland, wet heath, fens, bogs, flushes, swamps and wet woodland, although intimately linked with the river, are covered in the wetlands and washlands hydrological domain.

This hydrological domain is characterised by a variety of vegetation types that are found on minerotrophic (groundwater-fed), permanently, seasonally or periodically waterlogged peat, peaty soil or mineral soils. Fens are peatlands that receive water and nutrients from groundwater and surface run-off, as well as from rainfall. Marsh is a general term usually used to imply waterlogged soil; it is used more specifically here to refer to fen meadows and rush-pasture communities on mineral soils and shallow peats. Swamps are characterised by tall emergent vegetation. Reedbeds (i.e., swamps dominated by stands of common reed *Phragmites australis*) are also included in this domain.

This hydrological domain also includes neutral and improved grasslands on floodplains and grazing marshes. Lowland raised bogs and areas of carr (fen woodland dominated by species such as willow *Salix* spp., alder *Alnus glutinosa* or birch *Betula* spp.) are also included.

Fens (UK BAP Priority Habitat)

Fens are peatlands that receive water and nutrients from the soil, rock and ground water as well as from rainfall: they are minerotrophic (groundwater fed). Two types of fen can broadly be distinguished: topogenous and soligenous. Topogenous fens are those in which water movements in the peat or soil are generally vertical. They include basin fens and floodplain fen. Soligenous fens, in which water movements are predominantly lateral, include mires associated with springs, rills and flushes in the uplands, valley mires, springs and flushes in the lowlands, trackways and ladder fens in blanket bogs and laggs of raised bogs.

Reedbed (UK BAP Priority Habitat)

Wetlands dominated by stands of the common reed *Phragmites australis*, wherein the water table is at or above ground level for most of the year. They tend to incorporate areas of open water and ditches, and small areas of wet grassland and carr woodland may be associated with them. Reedbeds are among the most important habitats for birds in the UK.

Lowland raised bog (UK BAP Priority Habitat)

Peatland ecosystems that develop primarily, but not exclusively, in lowland areas, such as the head of estuaries, along river floodplains and in topographic depressions. In such locations drainage may be impeded by a high groundwater table, or by low permeability substrata, such as estuarine, glacial or lacustrine clays. The resultant waterlogging provides anaerobic conditions that slow down the decomposition of plant material, which in turn leads to an accumulation of peat. Continued accrual of peat elevates the bog surface above regional groundwater levels to form a gently curving dome from which the term ‘raised bog’ is derived.

Improved grasslands on floodplains

Periodically inundated pasture, managed by agricultural improvement for livestock grazing or hay and/or silage production. Improvement may include increasing the drainage capacity (typically using land drains), reseeding of selected grass species to provide a grazing sward and nutrient enhancement through manuring and/or artificial fertiliser application.

Floodplain grazing marsh (UK BAP Priority Habitat)

Periodically inundated pasture or meadow with ditches that maintain the water levels, containing standing freshwater. The ditches are especially rich in plants and invertebrates. Almost all areas are grazed and some are cut for hay or silage. Sites may contain seasonal water-filled hollows and permanent ponds with emergent swamp communities, but not extensive areas of tall fen species like reeds, although they may abut with fen and reed swamp communities.

Wet woodland (UK BAP Priority Habitat)

Wet woodland, or carr, occurs on poorly drained or seasonally wet soils, usually with alder, birch and willows as the predominant tree species, but sometimes including ash, oak, pine and beech on the drier riparian areas. It is found on floodplains, as successional habitat on fens, mires and bogs, along streams and hill-side flushes, and in peaty hollows.

Appendix 3 PRINCE PSG hypotheses and high level objectives

PRINCE steering group potential testable hypotheses

The Project Steering Group at the first project steering group meeting, after receipt of the draft Project Record, proposed a wide range of hypotheses that could form the basis of the research (listed below as received). Given the broad range of hypotheses, in both content and style, they have been collated by Cascade Consulting under three general headings:

- investigation of abiotic factors;
- investigation of ecosystem and/or biodiversity consequences;
- investigation of potential adaptation strategies.

The hypotheses are very broad and encompass a greater number of attributes than were:

i) defined in the tender specification;
ii) envisaged by the Cascade Consulting project team during the tendering process.

Contributions were received from Harriet Orr (HO) and Rob Wilby (RW), project managers for the Environment Agency; Keith Colqhoun (KC) representing UK Water Industry Research Ltd; Stewart Clarke (SC), English Nature; Graham Pierson (GP), Environment Agency; and Rhian Thomas (RT), Countryside Council for Wales. It is recognised by the Environment Agency project managers that some of these questions (marked #) are outwith the scope of the proposed modelling work.

Investigation of abiotic factors

i) Which physicochemical variables are most sensitive to climate change? (HO)
ii) Can we measure the relative sensitivity of physicochemical variables to climate and land use? (HO)
ii) Is it possible to measure the relative sensitivity of physicochemical variables to future climate change scenarios (combined with future land use scenarios)? (SC)#
iv) What impact will future climate change have upon key fluvial system processes – sediment transport, nutrient dynamics, …? (SC)#
v) What are the likely effects upon annual flow regimes? (GP)
vi) Under future climate change scenarios what is likely to be the magnitude, duration and periodicity of extreme events (floods, droughts)? (SC)#
vii) What is the true extent of natural climate variability on the target systems? (RW)
viii) What are the effects of climate change on mean and seasonal temperature patterns? (RT/GP)
ix) How will climate change affect sediment deposition and erosion, in different river reaches? (GP)

x) What effect will multi-season droughts have? In considering droughts the effects of lower rainfall in the autumn should also be considered in conjunction with summer droughts. (RT)
xi) What are the effects of sea level rise and saline intrusion on rivers.
xii) What are the main pollutants that could be released from sediments and/or soils due to climate change that we need to be concerned about?

Investigation of freshwater ecosystem and/or biodiversity consequences

xiii) What impact would these extreme events have upon river ecology (e.g., what drought return periods are likely to prevent community persistence)? (SC)#
xiv) How much would invert community be expected to fluctuate (e.g., use phases of NAO) (RW).
xv) Will climate change cause physicochemical changes, can the climate change signal be detected and if so how might these changes impact on aquatic ecosystems? (KC).
xvi) Will the river have increased probability of reaching lethal temperatures (for salmonids) and with what frequency (e.g., one in every 10 years?). (GP)
xvii) What effects will there be on the migration times of fish both to and from rivers – would the fish spend longer in estuaries? What are the implications here? (RT)
xviii) How will climate change affect turbidity, and what are the implications for freshwater species and/or habitats? (RT)

Investigation of potential adaptation strategies

xix) What controlling factors could be manipulated to mitigate climate change impacts on river ecology (e.g., drainage patterns, land use, instream habitat)? (SC)# adaptation
xx) Can current drainage patterns be manipulated to mitigate climate change impacts? (HO) adaptation
xxi) What is the relative significance of land use and climate change to the target biota and/or systems? (RW)
xxii) What land management strategies could be invoked to counter undesirable impacts on the biota and/or others? (RW) adaptation
xxiii) What are the potential adaptation options that could be developed to respond the impacts of climate change? (KC).
xxiv) Extreme rainfall would have flood management policy issues, and hence we would like to know what the implications of such works would have on conservation.
xxv) What can we do to mitigate the effects of climate change on freshwater species and/or habitats of conservation importance? How can we manage for these effects –through any of policy, site management, influencing other organisations to change their policies and/or practices, change our agri-environment scheme? This is one of the main questions for us.

High-level objectives for the project, developments in the future or ultimate end points example

To ensure that the R&D can be used to inform the full risk assessment, the PSG has defined a number of longer-term high-level objectives within which the research should fit. These are given as bullet points below, with notes to describe their status within this initial R&D programme.

The current proposed studies are tasked with defining the following:

- To identify freshwater habitats or species particularly sensitive to climate change.

Adaptation responses will be testable by the modelling, but are not part of this proposal:

- What adaptation responses do we adopt?
- To identify adaptation options for freshwater habitats and species.

Results of the impact assessment will be relevant as inputs to policy assessment, but this is not an objective in itself of the project:

- Will existing policy mechanisms be adequate (e.g., WFD, Habitats Directive, etc.)?

Other non-climate change activities that may influence freshwater ecology:

- to link climate change to a range of future land use scenarios;
- to identify anthropogenic factors that could exacerbate the impacts of future climate change (e.g., abstraction, nutrient enrichment, flood management);
- try to anticipate anthropogenic responses, such as increase in arable farming in the Dales in response to warmer drier summers, is a subsequent stage.

Other interactions and potential competitive dynamics that may need to be tested for fish populations:

- Fish may be directly affected by some of the things mentioned above, but other effects will doubtless be mediated through changes in other floral and faunal abundance and/or composition, which in turn will react directly to some of the abiotics.
- Subtle changes in the competitive balance between fish species in mixed fishery, such as the Wharfe, will also manifest under even quite small climate changes. For example, hot summers and low summer flows will be generally favourable for cyprinid recruitment and growth and survival of adults, and these may outperform salmonids in competing for food and space, even though these conditions may not in themselves be directly harmful to salmonids.

It is clear from the above that there are a wide range of aspirations within the PSG. The Cascade Consulting project team request that some level of integration of the original project objectives and these high-level objectives be undertaken to provide a clear course for the remaining part of the PRINCE ecological modelling studies. This integration could also usefully include rationalisation of the hypotheses to establish objectives for the present studies and promote aspirations for future studies.

We welcome views from our users, stakeholders and the public, including comments about the content and presentation of this report. If you are happy with our service, please tell us about it. It helps us to identify good practice and rewards our staff. If you are unhappy with our service, please let us know how we can improve it.