Environment Agency

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Potential impacts of climate change on river water quality

Science Report – SC070043/SR1

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SCHO0508BOCW-E-P

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Our work includes tackling flooding and pollution incidents, reducing industry's impacts on the environment, cleaning up rivers, coastal waters and contaminated land, and improving wildlife habitats.

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Author(s):
Professor Whitehead, P., Butterfield, D. and Dr Wade, A.

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Research Contractor:
Water Resource Associates
Fairmile
Wallingford
01235 850836

Environment Agency's Project Manager:
Lamorna Zambellas, Science Department

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Steve Killeen

Steve Killeen

Head of Science

Executive summary

Climate change is a key driver of policy for the Environment Agency and the water industry. It is important to understand the potential impacts of climate change on water quality, so that good advice on the likely impacts can be given and used by policy makers. Whilst it is now accepted that human-induced climate change is occurring, predicting the degree and probability of climate change impacts on water quality is difficult because of the wide range of natural variability in hydrology, chemistry and ecology. Also, there is considerable uncertainty associated with a cascade of issues. For example, there is uncertainty over the quantification of the driving global climate processes, over the procedures for downscaling from global to local riverine situations, over the processes controlling behaviour in freshwater systems and over the complex interactions between hydrology, chemistry and biology. It is impossible for any one person to comprehend all these complexities, so mathematical models are increasingly being used to assist with understanding and predicting these processes.

In this report, we generate local weather information from General Circulation Models (GCMs, also referred to as Global Climate Models) using downscaling procedures, and use these data to drive the INCA (Integrated Catchment) suite of water quality models. We assess the potential impacts of climate change on water quality in five rivers systems in the UK: the Lambourn, the Tamar, the Lugg, the Tweed and the Tame. These rivers represent differing geology and geographical locations around the UK and differ significantly in the extent to which they are impacted by agriculture or point sources of pollution. We use the models to simulate flow, total and soluble phosphorus, nitrate, ammonia, sediments, and ecology (macrophytes and epiphytes). We run the models for four UK Climate Impacts Programme (UKCIP) climate change scenarios and for three periods, the 2010s, the 2020s and the 2050s. In addition, we simulate a longer term scenario from the present to 2100 for one river, to investigate transient impacts of climate change.

The results are complex as might be expected, but we do find consistent patterns and it is possible to make some statements about the likely outcomes of climate change.

In the lowland southern River Lambourn, we predict declining concentrations of Soluble Reactive Phosphorus (SRP) for winter months and increasing concentrations during summer and autumn months, caused by lower flows and reduced dilution.

We predict that sediments in the Lambourn will increase throughout the year, but will be particularly high in autumn after dry summers. The build up of sediments over dry periods, followed by increased autumn flows seems the main mechanism here.

Results from the urbanized midlands River Tame show similar increased SRP levels in summer but higher increases in winter, due to diffuse urban runoff.

In the western and rural River Lugg (a tributary of the River Wye), we predict the SRP will decrease in winter but increase in summer months.

Our models show nitrate levels in the northern River Tweed increasing in the winter in upland headwaters, as organic nitrogen is released, and decreasing in summer months due to drought and increased denitrification in the river. The lower Tweed also shows increased nitrates in winter but the highest increases are in summer. This difference is due to changes in land use to agriculture and point source discharges in the lower reaches of the Tweed.

Nitrate levels in the south western River Tamar have a similar response to those in the Tweed, with higher nitrates in winter and lower summer nitrates in the upper reaches. In the lower reaches of the river, nitrates increase both in summer and in winter.

For all of the above water quality results, the rate of change increases over time and also with the severity of the emission scenario.

The dynamics of macrophytes and epiphytes interact significantly within rivers, according to the modelling. The study suggests that increased drought could create problems by increasing nutrient concentrations. With more sunshine and reduced flows, epiphytic growth will be stimulated at the expense of macrophyte growth.

Overall, water quality impacts are different depending on geographic location and water body location within a catchment. However, we emphasize that the changes predicted by the modelling project are not large. Our analysis of the model results, using the delta change method of downscaling, shows relatively minor changes in the probability distributions. These results suggest there is no immediate need to change the processes of water quality planning.

However, simulations using a different method, the statistical downscaling model of Wilby *et al.* (2002), do show larger changes in probability distributions. This has yet to be fully explored, as few simulations using the full transition scenarios from GCMs have been evaluated from a water quality point of view.

We also undertook a comprehensive literature review on the impacts of climate change on water quality. There has been little study in this area and almost no quantification. However, the following points clearly emerge:

- Water quality will be affected by changes in flow regime.
- Lower minimum flows imply less volume for dilution and hence higher concentrations downstream of point discharges.
- Enhanced growth of algal blooms in rivers and reservoirs could affect levels of dissolved oxygen and the costs of treating water for potable supply.
- Increased storm events, especially in summer, could cause more frequent incidence of combined sewer overflows, discharging highly polluted waters into receiving water bodies. The potential impacts on urban water quality will be largely driven by these changes in short duration rainfall intensity overwhelming drainage systems, as well as rising sea levels affecting combined sewerage outfalls.
- The most immediate reaction to climate change is expected to be an increase in river and lake water temperatures.
- More intense rainfall and flooding could result in increased suspended solids, sediment yields and associated contaminant metal fluxes.
- Nutrient loads are expected to increase.
- In shallow lakes, oxygen levels may decline and cyanobacteria blooms may become more extensive.
- In the UK, there has been relatively little research on toxins in streams, lakes and sediments, as the problems are thought to be limited. However, climate change may alter this perception.

Climate change studies, especially in relation to water quality and ecology, are at fairly early stages and the outcomes are subject to considerable uncertainty. However, understanding the processes and mechanisms controlling water quality and ecology, and how these combine and interact, is essential for sustaining potable water supplies and conserving river systems. We identify the following options for further research:

1 Apply downscaling to a larger set of GCM scenarios, to investigate transient effects of climate change on water quality across a wider set of catchments. Evaluate the different probability results compared to the delta change method. Analyse a wider range of scenarios, such as those determined by New *et al*. (2007) or the new UK Climate Impacts Programme (UKCIP08) scenarios.

2 Further develop and test the macrophyte-epiphyte model, to elucidate the processes that control their growth and evaluate the potential impacts of climate change.

3 Develop the phytoplankton model in INCA, to help assess the overall impacts of climate change on algal populations.

4 Evaluate the apparent relationship between nitrate and biodiversity found in microcosm experiments (James *et al*. 2005), and combine this with INCA to predict possible effects on biodiversity of climate change and measures to control nitrate under the Nitrate Directive.

5 Refine the QUASAR river model (Whitehead *et al*. 1997) for DO-BOD (Dissolved Oxygen-Biochemical Oxygen Demand) and algae (or incorporate DO-BOD equations within INCA), to explore the joint probability of low DO (dissolved oxygen) events and potential fish kills.

6 Further analyse the INCA model simulations to fully evaluate the uncertainty inherent in water quality predictions, making use of the Monte Carlo tool developed within Eurolimpacs.

7 Investigate how the INCA-type functionality could be run on the national scale to evaluate national policies in terms of impact, costs and benefits.

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Contents

1. Introduction, background and objectives

The United Nations COP (Conference of the Parties to the UN Framework Convention on Climate Change) meeting in Bali and the latest IPCC Report (IPCC, 2007) confirmed a broad consensus among scientists and policy makers that human-induced climate change is now occurring. There is still considerable debate over the magnitude of future temperature changes and how these will drive precipitation, evaporation and hydrology at river catchment scales. However, it is not too early for the water industry and the Environment Agency to start considering the implications of climate change. The large costs of new infrastructure, treatment facilities, flood defence, river conservation and water resource management mean that an early assessment of climate risks is needed for the purposes of planning and design. Also, the time scales for some new developments and measures to adapt to, or mitigate the effects of climate change require an early understanding of the likely impacts.

To address the issue of climate change with respect to river water quality, the Environment Agency initiated a project with the objective to 'understand the potential impacts of climate change for river water quality, to support evidence-based policy development.' The specific tasks for this project are as follows:

(i) To gather, organise and interpret scientific information on climate change and its potential impacts on water quality, including the drivers of water quality.

(ii) To undertake a sensitivity analysis to explore the nature, magnitude and likelihood of the potential risks and benefits posed by climate change on river water quality.

(iii) To estimate the relative scale of different sources of uncertainty in the above analysis.

The first task produced a review paper, providing a comprehensive assessment of the available scientific literature on the potential direct and indirect impacts (damaging and beneficial) of climate change on water quality.

The second task explored the nature, magnitude and likelihood of potential impacts on river water quality under a number of different climate change scenarios. This work used an off-the-shelf model to explore the effects of diffuse and point sources of pollution under present and projected conditions of water flow and temperature. Compared with present conditions, the Environment Agency needs to explore potential impacts over three time horizons:

• the next 5-10 years, to inform current regulatory decisions;
• up to the 2020s, to inform decisions under the Water Framework Directive;
• up to the 2050s, to influence strategic decision making.

For model inputs, we have applied climate change scenarios from the UK Climate Impacts Programme 02 (UKCIP02, Hulme *et al*. 2002) and UK Water Industry Research UKWIR (Conlan *et al*. 2006), as these are currently in use in water resource planning. This ensures greater consistency in impact assessments across the Environment Agency and the water industry.

It is important to generate time series using a weather generator, and to convert variables from the General Circulation Models (GCMs) into long-run daily time series for river flow using an appropriate hydrological model. Such a model must be able to represent adequately both high and low flow conditions and their frequency, so as to estimate return periods. This modelling approach can also be used to explore longer term changes, and the changing probabilities of rare events, as the climate alters over the next 100 years.

This report describes the work undertaken to address the above three tasks.

2. Literature review

The literature review fills a real gap because there has been no comprehensive review on climate change and river water quality to date. Even the IPCC (2007) report had a relatively limited section on river water quality. This is because the IPCC priority was primarily from a water resource perspective to determine the reliability of fresh water supplies for human consumption, agriculture and hydropower, rather than water quality. There have been reviews of climate change impacts on UK water resources (for example, Romanowicz *et al*. 2006, Kundzewicz et al, 2008, Limbrick et al, 2000) and freshwater ecosystems (for example, the Environment Agency PRINCE project, Conlan *et al*. 2007). Hence, the current review is limited to river water quality.

The EU funded project Eurolimpacs (www.eurolimpacs.ucl.ac.uk) has been a significant source of information for the literature review. Eurolimpacs is a €22 million, 38 partner, 220 scientist project, with the specific task of investigating the impacts of climate change on freshwater systems across Europe, including rivers, lakes, wetlands and catchments. Figure 2.1 shows the structure of the work packages. These address laboratory research, field scale experiments, process modelling, restoration, potential adaptation measures and the development of management tools. Figures 2.2 and 2.3 show the range of catchments used in the modelling and the spread of partners within Eurolimpacs.

Our full literature review is presented in Appendix 1, in the form of a paper prepared for publication in the *Hydrological Sciences Journal*. Some key messages from the literature review are as follows:

- Water quality will be affected by changes in flow regime.
- Lower minimum flows imply less volume for dilution and hence higher concentrations downstream of point discharges.
- Enhanced growth of algal blooms in rivers, lakes and reservoirs could affect DO (dissolved oxygen) levels and water supply.
- Increased storm events, especially in summer, would lead to more frequent incidences of combined sewer overflows, discharging highly polluted waters into receiving water bodies.
- The most immediate reaction to climate change is expected to be an increase in river and lake water temperatures.
- More intense rainfall and flooding could result in increased suspended solids, sediment yields and associated contaminant metal fluxes.
- Nutrient loads are expected to increase.
- In shallow lakes, oxygen levels may decline and cyanobacteria blooms may become more extensive.
- In the UK, there has been relatively little research on toxins in streams, lakes and sediments, as the problems are thought to be limited. This may change.
- Potential impacts on urban water quality will be largely driven by changes in short duration rainfall intensity overwhelming drainage systems, as well as rising sea levels affecting combined sewerage outfalls.

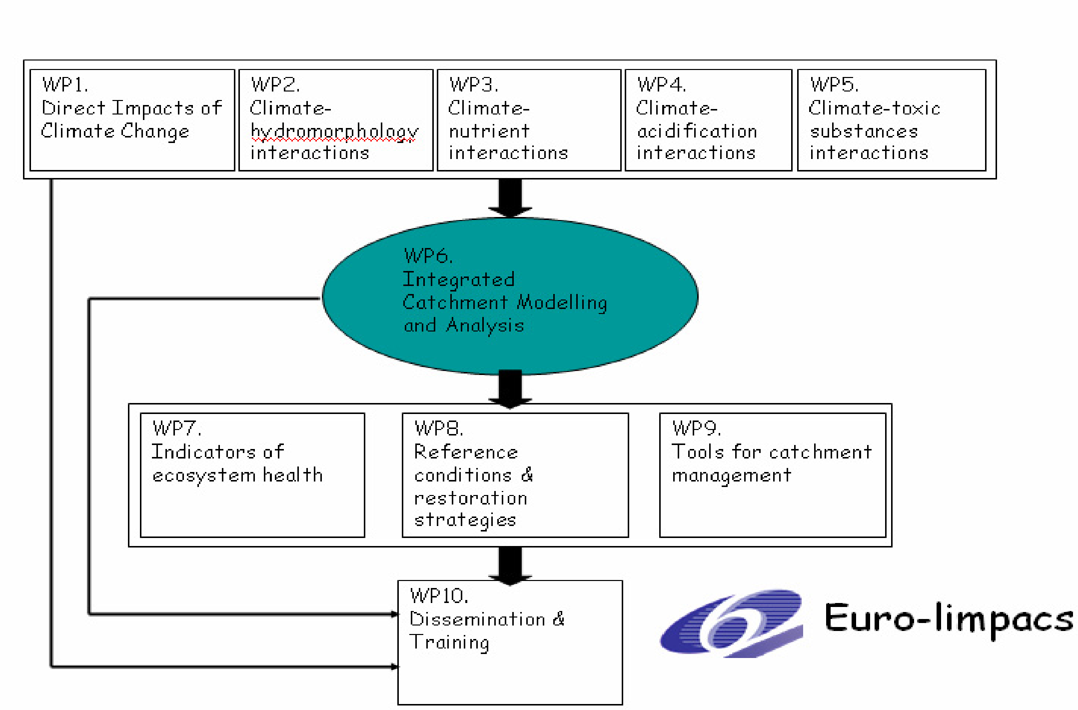

Figure 2.1 The Eurolimpacs work package structure

Figure 2.2 Catchments modelled in the Eurolimpacs project

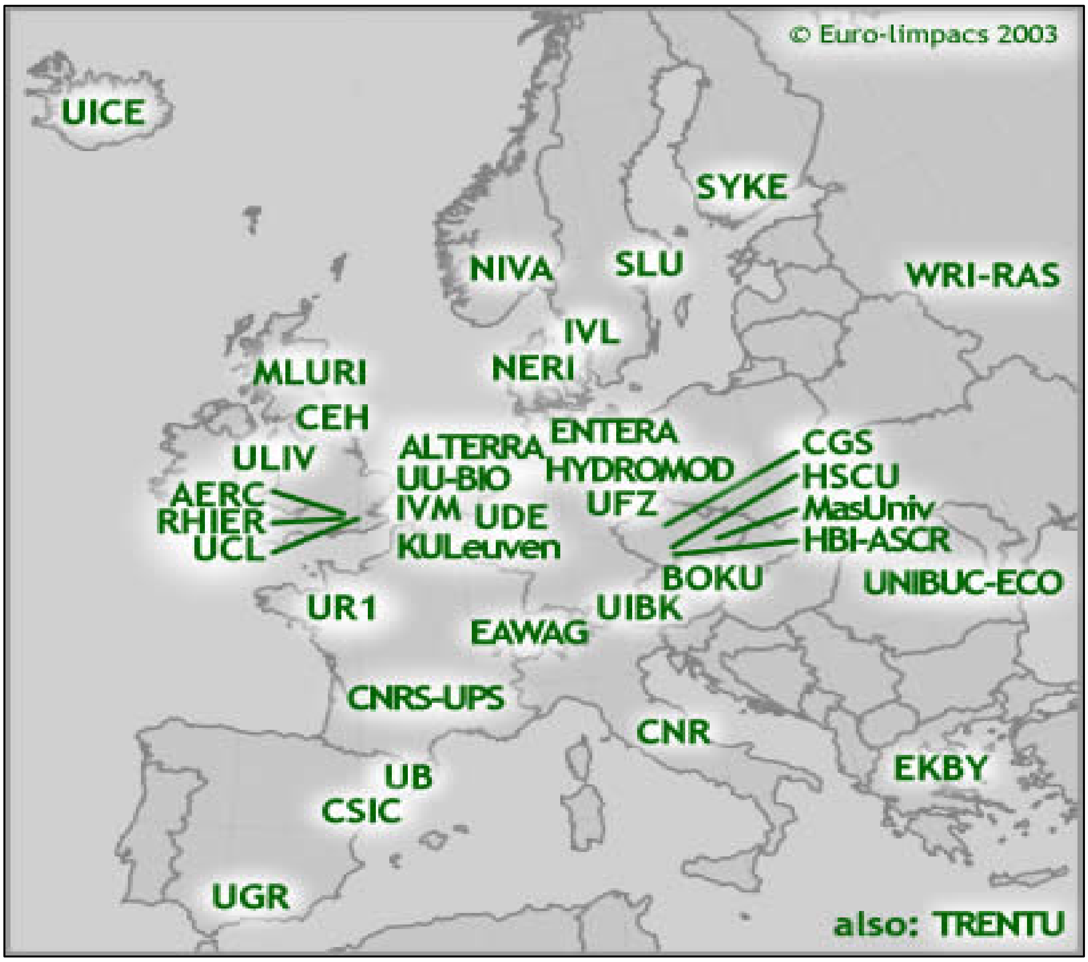

Figure 2.3 The Eurolimpacs project partners

KEY UCL = University College London, Environmental Change Research Centre (ECRC), London; NERI = National Environmental Research Institute, Department of Freshwater Ecology, Silkeborg; RHIER = Royal Holloway Institute for Environmental Research, Wetland Ecosystems Research Group, London; UDE = University of Duisburg-Essen, Centre for Microscale Ecosystem, Institute of Hydrobiology, Essen; AERC = University of Reading, Aquatic Environments Research Centre, Reading*;* ALTERRA = Alterra Green World Research, Team of Freshwater Ecology, Wageningen; CEH = Centre for Ecology and Hydrology, Wallingford, Edinburgh, Dorset, Windermere, Bangor; CSIC = Spanish Council for Scientific Research; IVL = Swedish Environment Research Institute, Gothenburg; NIVA = Norwegian Institute for Water Research, Oslo; SLU = Swedish University of Agricultural Sciences, Department of Environmental Assessment, Uppsala; SYKE = Finnish Environment Institute, Helsinki; UIBK = University of Innsbruck, Institute of Meteorology and Geophysics, Institute of Zoology and Limnology, Innsbruck; ULIV = University of Liverpool, School of Biological Sciences, Liverpool; BOKU = University of Natural Resources and Applied Life Sciences, Institute of Water Provision, Water Ecology and Waste Management, Department of Hydrobiology, Vienna; CNR = Consiglio Nazionale delle Ricerche; CNRS-UPS = Centre National de la Recherche Scientifique and University of Toulouse, Laboratoire Dynamique de la Biodiversité (LADYBIO), Toulouse; EAWAG = Swiss Federal Institute of Environmental Science and Technology, Departments of Water Resources, Drinking Water, Limnology and Surface Waters, Dubendorf*;* EKBY = Greek Biotope/Wetland Centre, Soil and Water Resources Department, Thessaloniki; ENTERA-Ingenieurgesellschaft für Planung und Informationstechnologie -, Hanover; HBI-ASCR = Czech Academy of Sciences, Hydrobiological Institute, České Budějovice; HSCU = Charles University, Hydrobiological station, Blatna; HYDROMOD

Scientific Consulting, Wedel; IVM = Institute for Environmental Studies, Amsterdam; KULeuven = University of Leuven, Department of Biology, Laboratory of Aquatic Ecology, Leuven; MasUniv = Masaryk University Brno, Faculty of Science, Department of Zoology & Ecology, Brno; UB = University of Barcelona, Department of Ecology, Barcelona; UFZ = Centre for Environmental Research Leipzig-Halle, Department of Conservation Biology and Natural Resources (CNBR), Leipzig; UGR = University of Granada, Department of Animal Biology, Granada; UICE = University of Iceland, Institute of Biology, Reykjavik; UNIBUC-ECO = University of Bucharest, Department of Systems Ecology and Sustainable Development, Bucharest; UR1 = University of Rennes, Research Unit 'Ecosystem Functioning and Biological Conservation', Rennes; UU-BIO = Utrecht University, Institute of Biology, Landscape Ecology Group, Utrecht; WRI-RAS = Russian Academy of Sciences, Water Problems Institute, Moscow; TRENTU = Trent University, Environmental and Resource Studies, Ontario; MLURI = Macaulay Land Use Research Institute, Aberdeen; CGS = Czech Geological Survey, Prague.

3. Modelling and sensitivity analysis

3.1 Methodology for the sensitivity analysis and catchment selection

Our approach to part 2 of the project has been to make use of existing models of water quality, and to subject them to a range of climate change scenarios. The model outputs are evaluated to determine monthly changes as well as changes in the statistical distributions of concentrations of substances. The Environment Agency's policy on water quality planning requires methods, such as SIMCAT (Simulated Catchment Model), which calculate changes in annual percentiles. Distributions of percentiles are therefore a key output of the project.

An earlier project on water quality (Conlan *et al*. 2006) used the UKCIP02 scenarios (Hulme *et al*. 2002) to assess the potential impacts of climate change. However, it was largely restricted to a sensitivity analysis without the application to calibrated catchment models. Conlan *et al.* (2006) used the 'delta change' method, where historic temperature and precipitation series are altered on a monthly basis according to the changes projected by the UKCIP02 scenarios for specific time horizons (for example, the 2050s). One limitation of this approach is that long term, transient effects of climate change are not considered. This can be important when changes in the temporal sequencing of hydrological events dictate water quality behaviour.

In this project we have used the delta change method to evaluate the impacts of four UKCIP02 scenarios and to provide information on potential changes in terms of the range of percentiles, as well as monthly statistics, for the 2010s, 2020s and 2050s. In addition, we have evaluated some long term scenarios reflecting transient effects, to explore an alternative approach to the delta change method.

Our modelled response variables include: nitrate, ammonia, total phosphorus, soluble reactive phosphate (orthophosphate or SRP) and sediments. Ecology is represented by including the biomass of macrophytes and epiphytes in chalk streams. Dissolved oxygen (DO) and biochemical oxygen demand (BOD) are also key variables of interest to water quality managers. DO and BOD were considered in detail by the Conlan *et al*. (2006) report. Also, a full dynamic modelling study for DO and BOD on the River Thames is described by Cox and Whitehead (2008). It should be emphasised that a dynamic model looks at a long sequence of time, and takes account of how the statistical distributions of the variables are combined. This allows the correct estimation of the probabilities of impacts, which is essential in deciding how to respond to risks.

In this study, we selected catchments that would allow us to evaluate a range of hydrology, geology and land use conditions. We also considered different types of reach, to represent upstream components, middle sections of rivers and downstream reaches that might have impacts on estuaries or coastal systems.

Although only three catchments were required to meet the Environment Agency specification for the project, the sensitivity analysis was applied to five catchments: the Rivers Tamar, Lugg, Lambourn, Tame and Tweed. These represent a wide spread geographically and are quite different in character. The Integrated Catchment Models (INCA) developed by Whitehead *et al*. (1998a,b), Wade *et al*. (2002a) and Jarritt and

Lawrence (2007), have been used as the main modelling tools in the project, drawing from previously established data sets and calibrated parameter values for all five catchments.

3.2 The INCA suite of models for river water quality

The INCA (Integrated Catchment) model is a dynamic computer model that predicts water quantity and quality in rivers and catchments (Whitehead *et al*. 1998a,b, Wade *et al*. 2002a). INCA is the product of several Natural Environment Research Council (NERC), Environment Agency and European Union (EU) funded projects over the past ten years. As shown in Figure 3.1, the primary aim of INCA is to represent the catchment topography and the complex interactions and connections operating at a range of scales. INCA is process based, so it can address the problems of scaling up from a small sub catchment to a large catchment.

The overall philosophy of the INCA model is to represent the factors and processes controlling flow and water quality dynamics in both the land and in-stream components of river catchments, whilst minimising data requirements and model structural complexity (Whitehead *et al*. 1998a, b). The model produces daily estimates of discharge, stream water quality concentrations and fluxes over a period of many years, and describes how these are correlated (important for estimating probabilities), at any point along a river's main channel. The model is semi-distributed, so that spatial variations in land use and management can be taken into account. The hydrological connectivity of different land use patches is not modelled in the same way as in a fully-distributed approach, however. Rather, the hydrological and nutrient fluxes from different land use classes and sub-catchment boundaries are modelled simultaneously and this information is fed sequentially into a multi-reach river model, as shown in Figure 3.2.

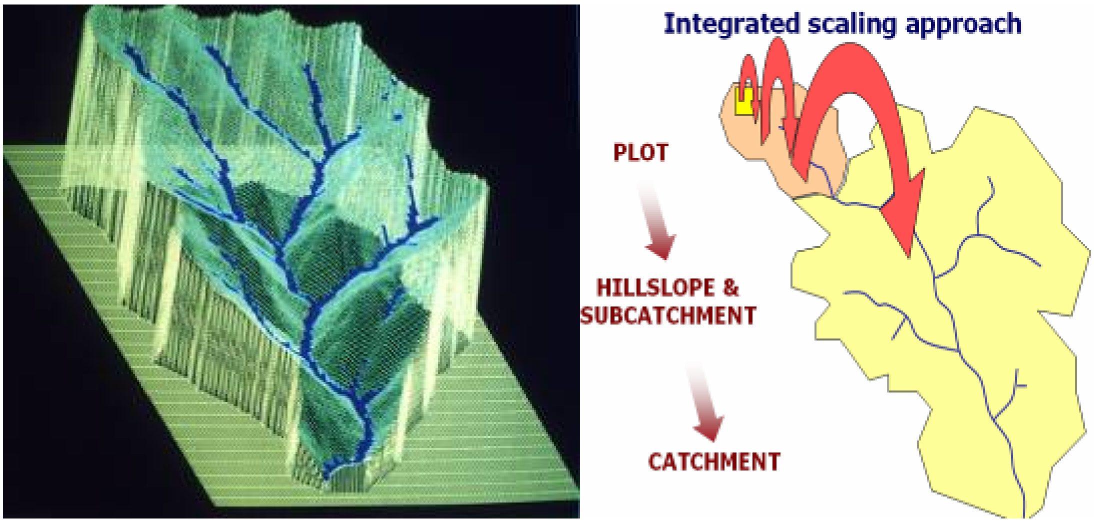

Figure 3.1 How the INCA model represents catchment topography (left), linkages, connectivity and scaling issues

The INCA model was originally tested on 10 catchments in the UK and 21 catchments across the EU (Table 3.1). This testing has been considerably extended in the current Eurolimpacs project. The major applications of INCA have been published in special volumes of *Hydrology and Earth System Sciences* (volume 6 (3), 2002) and *Science of the Total Environment* (volume 365, (1-3), 2006).

The INCA models have been designed to investigate the fate and distribution of water and pollutants in the aquatic and terrestrial environment. The original version of INCA has been enhanced over the course of Eurolimpacs to create new versions for nitrogen, phosphorus, sediments, ecology, mercury and a range of other metals. The models simulate flow pathways and track fluxes of pollutants in terrestrial and aquatic ecosystems. INCA is easy to use and fast, with a range of output graphics. The model system allows the user to specify the semi-distributed nature of a river basin or catchment, to alter reach lengths, rate coefficients, land use, velocity-flow relationships and to vary pollutant loads from atmospheric deposition, diffuse inputs or point sources.

A thorough review of the underlying factors and processes controlling nutrient transport and storage in river catchments was undertaken during the EU INCA project, using both historic and newly collected data. The basic equations of the INCA model were originally developed for the UK environment and these proved to be an adequate basis for the initial model applications (Whitehead *et al*. 1998b, Wade *et al*. 2002). However, to cover the wide variety of catchment types and pollution issues across the EU, and to incorporate the latest understanding of processes, parts of the INCA model were refined in relation to (a) the hydrology, (b) the representation of land management and (c) the factors controlling the biological processes of nutrient transformation. Specifically, these refinements relate to the addition of soil water and ground water retention volumes, more detailed vegetation growth periods and fertiliser application mechanisms, and additional soil moisture and temperature controls (Wade *et al.* 2002a). Reformulation of the equations and numerical integration ensured that a correct mass-balance of the statistical distributions was maintained by the model.

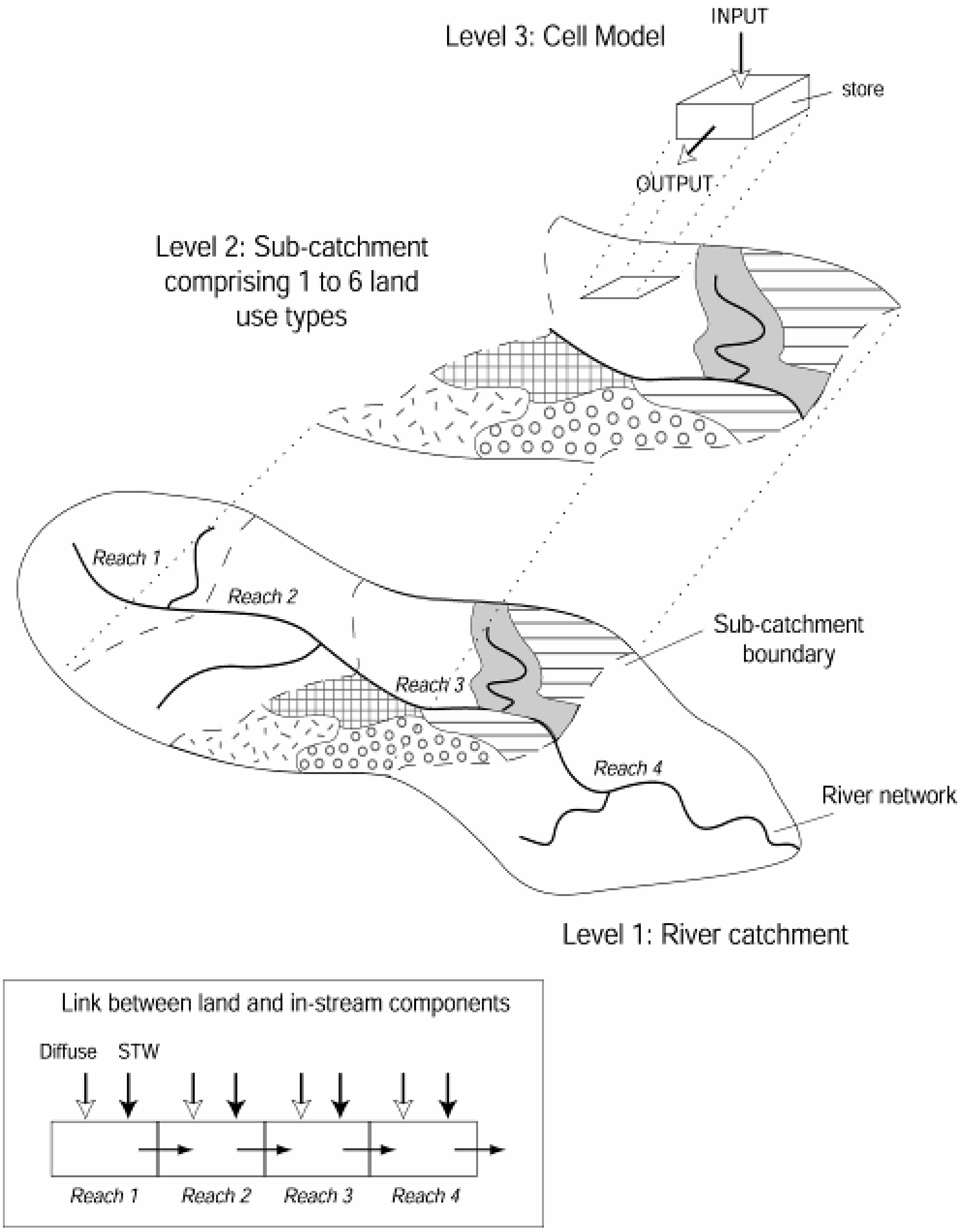

Figure 3.2 The integration of the landscape delivery and in-stream components of INCA. At level 1 the catchment is decomposed into sub-catchments. At level 2, the sub-catchments are sub-divided into six different land use types. At level 3, the soil chemical transformations and stores are simulated using the cell model. The diagram shows the link between the land-phase delivery and in-stream components at level 1: the diffuse inputs from the land-phase are added to the effluent point-source inputs such as Sewage Treatment Works (STW) and routed downstream.

Table 3.1 Summary of sites, data and policy issues studied in various INCA projects. Acid = acidification, Eutr = eutrophication, N Sat = nitrogen saturation (the point at which nitrogen starts to leach from catchments), CC = climate change

Country	Sites / river system	Area Kms^2	Predominant land use	Major issue
UK	Leith Hill	0.93	Forest and grassland	Acid/CC/N Sat
	Ant	49.3	Arable	Eutr
	Kennet	1033	Arable	Eutr/CC
	Tweed	4390	Improved pasture/arable	Eutr
	Ouse	8380	arable	Eutr
	Itchen	507	Improved pasture/arable	Eutr
	Test	1343	Improved pasture/arable	Eutr
	Tamar	916	Arable	Eutr
	Hafren/Hore at Plynlimon	6.8	Forest/grassland	Eutr,N Sat,Acid
Finland	Simojoki	3160	Coniferous forest/ Wetland	Acid
Germany	Lehstenbach	4.19	Coniferous forest	N Sat and Acid
	Steinkreuz	0.55	Deciduous forest	N Sat and Acid
France	Kerbernez	0.35	Arable	Eutr
	Stang Cau	0.86	Arable	Eutr
	Pouliou	0.75	Arable	Eutr
	Kervidy	4.9	Arable	Eutr
	Stimoes	12	Arable	Eutr
	Ponti-Veuzit	59	Arable	Eutr
	Garonne	56000	Arable, natural, forest	Eutr, CC
Netherlands	Buunderkamp	0.04	Oak forest	N Sat and Acid
	Leuvenum	0.04	Douglas Fir forest	N Sat and Acid
	Speuld	0.16	Douglas Fir forest	N Sat and Acid
	Kootwijk	0.16	Douglas Fir forest	N Sat and Acid
	Oldebroekse heide	0.005	Heathland	Eutr
	Edese bos	10	Heathland	N Sat and Eutr
Norway	Bjerkreim	619	Coniferous forest	N Sat and Acid
	Dalelv	3.2	Arctic tundra	N Sat and Acid
Spain	Fuirosos	16.2	Forest and arable	Eutr and Acid
Denmark	Vestskoven	Variable	Coniferous and deciduous forest	N Sat
	(18 plots)			
Romania	Mures	32,000	Forest, arable	Metals, nutrients
	Nealjov	3,465	Forest, arable	Nutrients

A more complete description of INCA, the equations and the application to a range of catchments is given in Appendix 2.

3.3 GCM downscaling and scenario data generation

In this report, we have used two approaches to evaluate the impacts of climate change on water quality. These were the delta change method as applied by UKWIR (Romanowicz *et al*. 2006), and the statistical downscaling scheme developed by Wilby *et al*. (Conlan *et al*. 2006). The UKWIR methodology was used to be consistent with water quality assessments ahead of the water companies planning round for 2009. The long term transient effects of climate change were also explored using downscaling, to give insights into potential water quality impacts arising from changes in the temporal sequencing of daily weather.

UKWIR delta change method

The UKWIR study (Romanowicz *et al*. 2006) developed a methodology for downscaling GCM information, translating the GCM outputs into key variables for river systems. These key variables were local precipitation, temperature, potential evapotranspiration (PET) and, via the water resource model CATCHMOD, into river flows. The procedure was applied to over 70 catchments across the UK, so it has provided a valuable resource for the current study. Information from the UKWIR project covers all the scenarios and time periods required for the water quality modeling tasks. The IPCC Special Report on Emission Scenarios (SRES) and corresponding UKCIP02 emission scenarios are shown in Table 3.2. These scenarios are described in detail by Hulme *et al.* (2002). The Eurolimpacs Project used the A2 and B2 scenarios for their investigations.

Table 3.2 UKCIP02 scenario nomenclature and matching SRES emission scenarios

UKCIP02 scenario	SRES emissions scenario
Low Emissions	B1
Medium-Low emissions	B2
Medium-High Emissions	A2
High Emissions	A1F1

The results of the UKWIR project are available in spreadsheets containing delta change factors for each catchment. In the case of precipitation and PET, the delta change is expressed as a percentage change, whereas for temperature the changes are in degrees centigrade, relative to the 1961-1990 baseline. Figures 3.3 and 3.4 show the precipitation and temperature changes for the River Kennet catchment incorporating the River Lambourn (Thames Region) and indicate significant changes in summer temperatures and precipitation.

These delta change values were used by Romanowicz et al (2006) to determine river flows and groundwater recharge for the catchments, using the CATCHMOD model (Wilby *et al*. 1994).

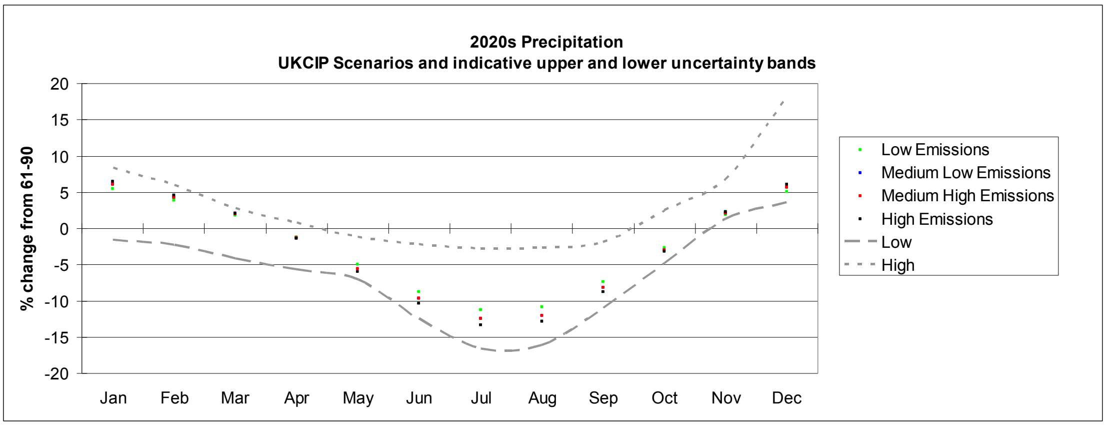

Figure 3.3 Precipitation changes for the Kennet Area (Thames Region) for the 2020s under a range of emission scenarios, together with the low and high uncertainty bands.

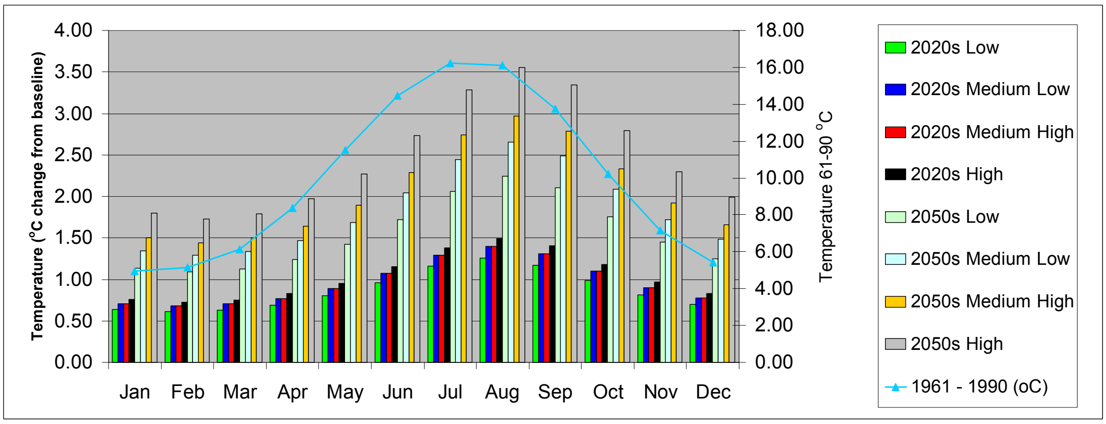

Figure 3.4 Temperature changes for the Kennet Area (Thames Region) for a range of emission scenarios, over the 2020s and the 2050s. Blue line shows actual temperature variations for the baseline period.

In the current project, we have taken the delta change information from the UKWIR project and used it to modify the input data for the INCA water quality models. Each scenario of interest (Table 3.2) has been run for the three time periods: the 2010s, 2020s and 2050s.

SDSM-Statistical downscaling method

In this study, GCM outputs were also downscaled to one UK river catchment – the Kennet and Lambourn - using the Statistical Downscaling Model (SDSM) (Wilby *et al*. 2002). This involves calibrating observed rainfall and PET series for Kennet catchment against gridded data from the National Centre for Environmental Prediction (NCEP). The UKSDSM archive holds 29 daily predictors originating from NCEP and four GCMs, for nine degree grid squares covering the British Isles, for the period 1961-2100. We only used predictors from grid-boxes centred on south east England (SEE) for the A2 emissions scenario (Medium-High Emissions). We chose A2 because this scenario has been widely used in previous impact studies.

Statistical downscaling of future climate change scenarios for the Rivers Kennet and Lambourn involved two main steps. First, we established empirical relationships between the variables of interest (daily temperature, precipitation and PET) across the catchment, and large-scale weather patterns obtained from the NCEP re-analysis for the current climate. These relationships were used in the second step to downscale ensembles of the target variables for the future climate, using data supplied by three GCMs (CGCM2, CSIRO, HadCM3) driven by A2 emissions scenario for the period 1961-2100. The procedures for downscaling GCM output are explained in detail by Wilby *et al*. (2006), alongside an analysis of impacts on evaporation, soil moisture, flow and groundwater in the Kennet catchment.

Figure 3.5 shows the daily data generated for the Kennet Catchment using this method for the period 1961-2100. It is difficult to discern trends except that the soil moisture deficit increased noticeably toward the end of the run, reflecting the drier conditions generated by the GCM. However, the river flows do show more frequent droughts in the second half of the 21st century (Figure 3.6). Note the variability generated by different GCMs in this figure.

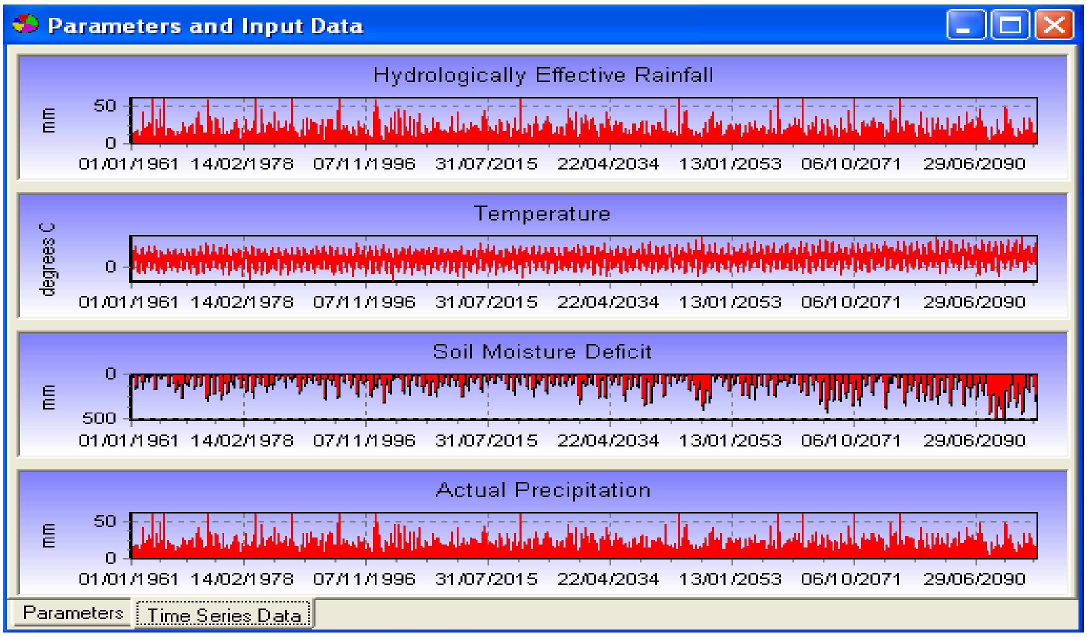

Figure 3.5 Daily data generated for the River Kennet by the HadCM3 under A2 emissions scenario for 1961-2100

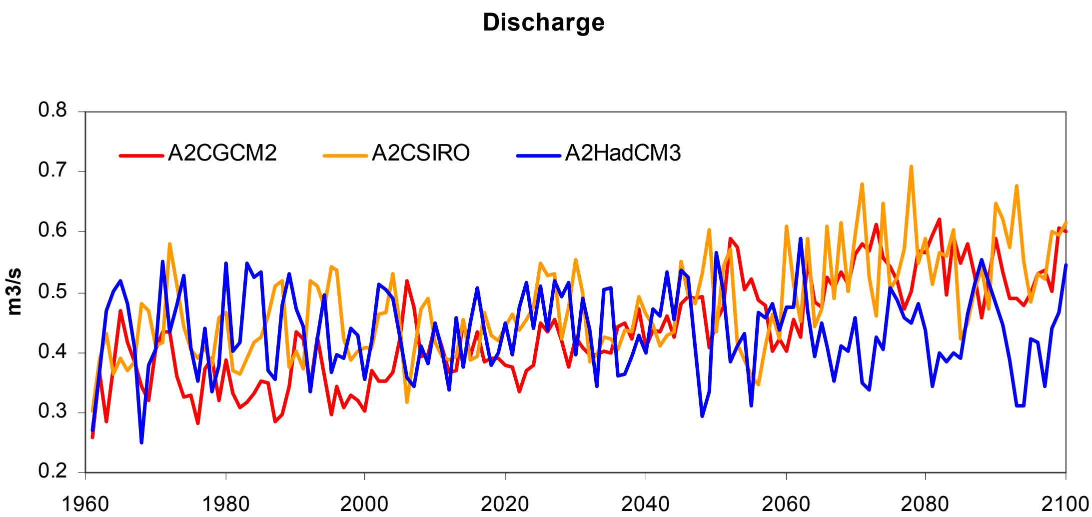

Figure 3.6 River flow simulated in the River Kennet for three different GCM outputs

3.4 Water quality model simulation results

Full descriptions of the INCA model applications to the five catchments selected for this study are given in Jarvie *et al*. (2002) for the Tweed, Wade *et al*. (2007) for the Lugg, Wilby *et al.* (2006) and Whitehead *et al.* (2002, 2006) for the Kennet, Wade *et al.* (2002d) and Jarritt and Lawrence (2007) for the Lambourn and Uncles *et al.* (2002) for the Tamar.

Rather than explain each model application in detail, we present selected model calibration results, to demonstrate the ability of the models to simulate a range of variables. Figures 3.7 to 3.21 show the model simulations for the Lambourn (Soluble Reactive Phosphorus or SRP and sediment), the Kennet (nitrate and ammonia), the Lugg (SRP and sediment), the Tamar (nitrate and ammonia), the Tame (phosphorus and sediment) and the Tweed (nitrate and ammonia). All are based on data obtained from the Environment Agency (flow and quality data), the UK Met Office, Agricultural Development and Advisory Service (ADAS), the Centre for Ecology and Hydrology (CEH) Land Cover maps, nitrogen deposition maps, agricultural census data, sewage treatment works (STW) discharges, river mapping, digital terrain maps and river velocity information. The model applications incorporate a significant quantity of historical data and encapsulate the hydrological, chemical and biological dynamics and processes operating in these diverse catchments. The various versions of INCA have been calibrated and validated in each case and provide a sound basis for climate change impact studies.

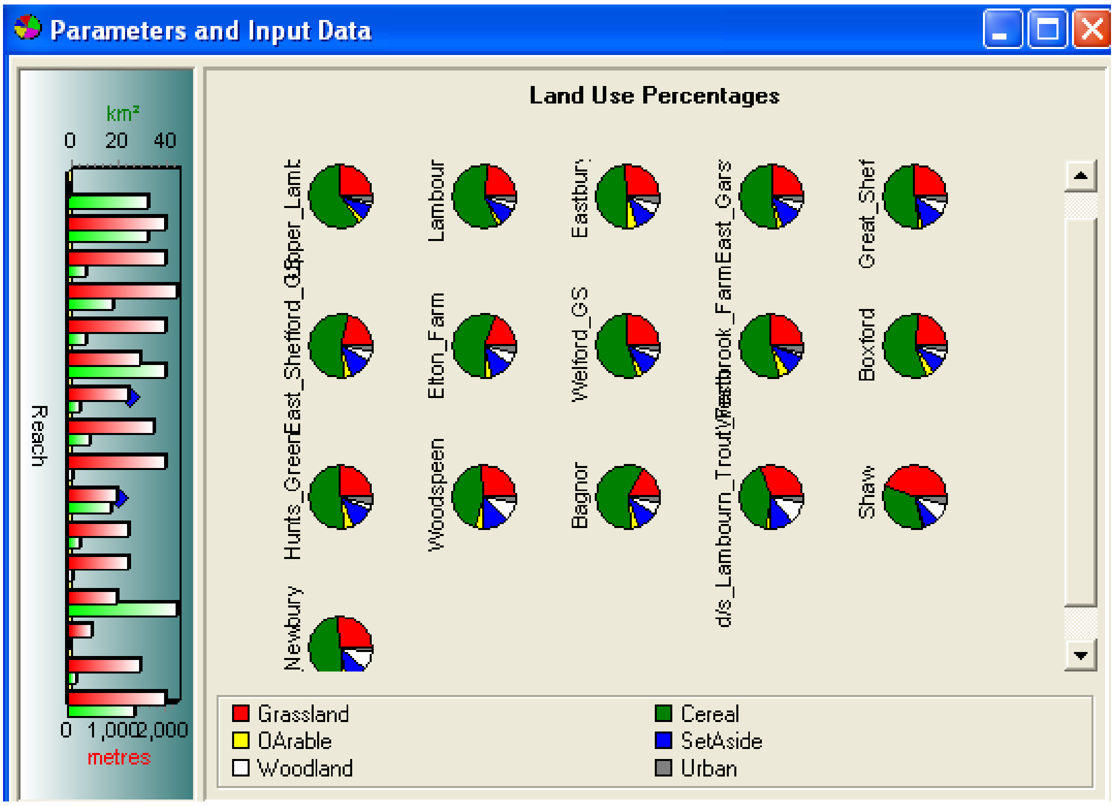

Figure 3.7 Land Use in the Lambourn

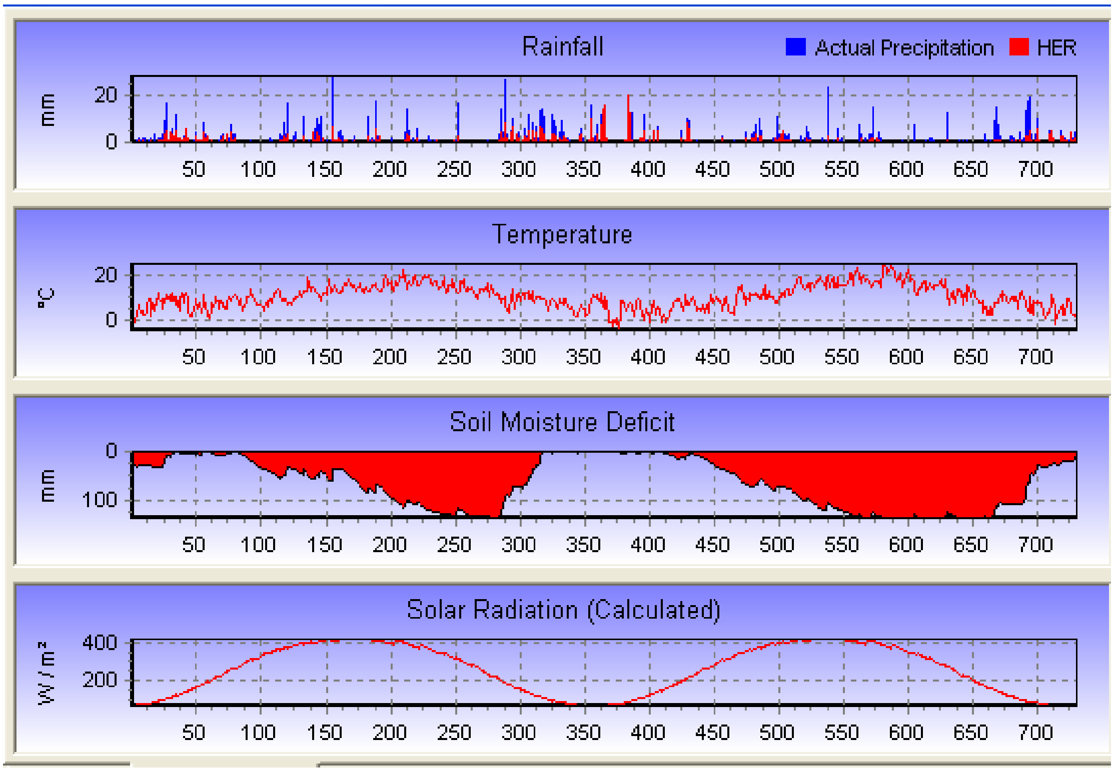

Figure 3.8 The hydrological input time series for 2002-2003 for the Lambourn, x axis is days since 1/1/2002

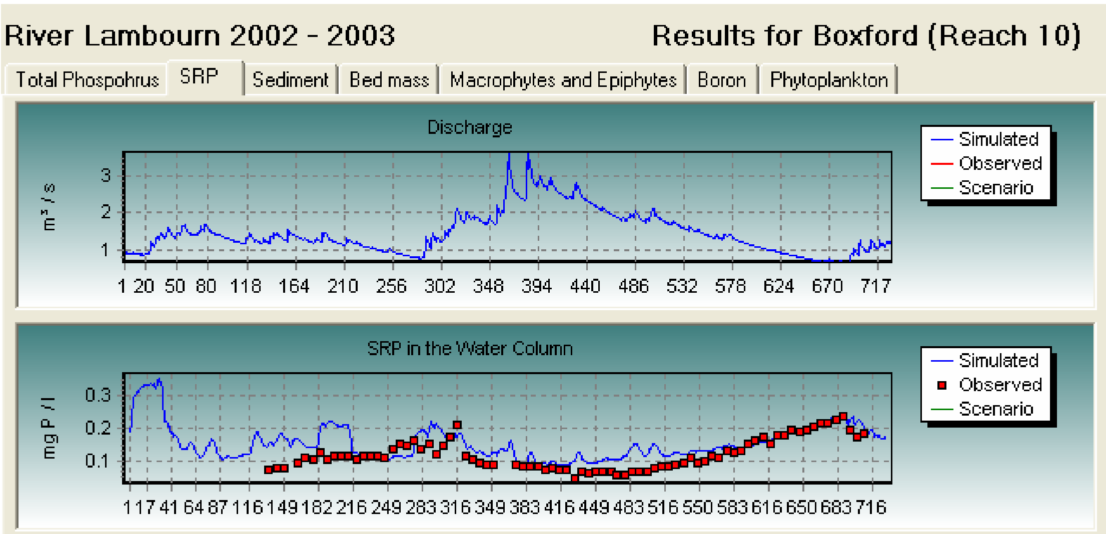

Figure 3.9 River Lambourn observed and simulated phosphorus 2002-2003

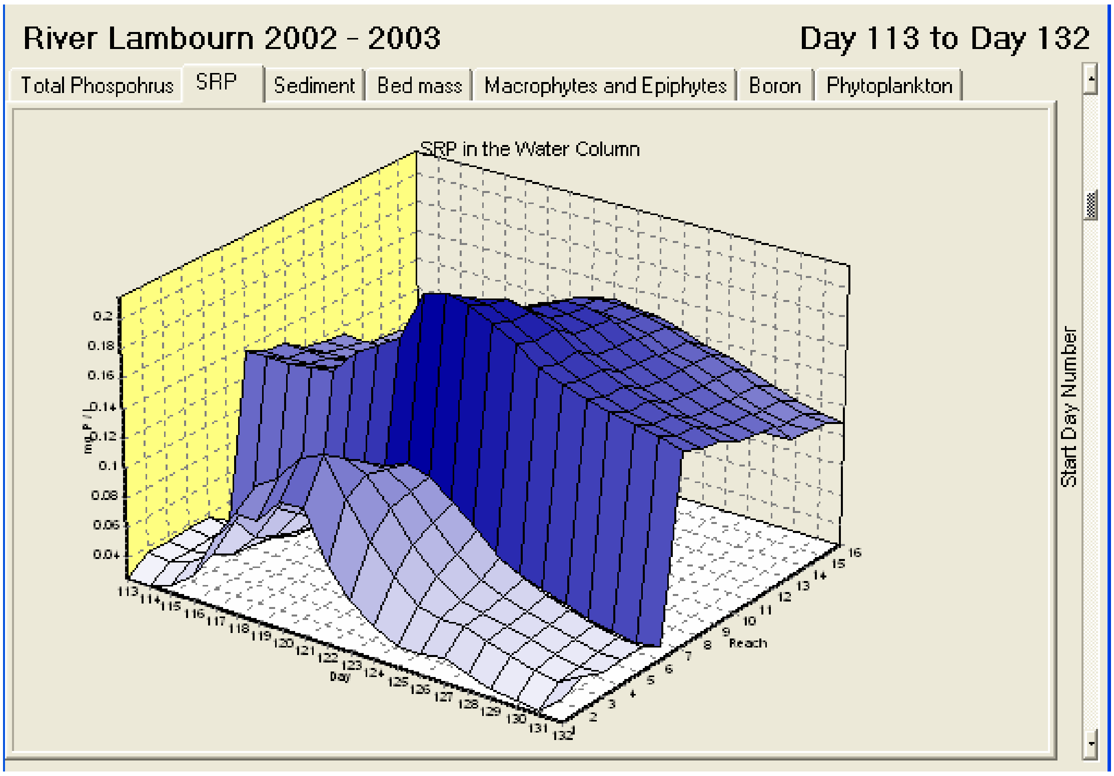

Figure 3.10 Phosphorus variations along the Lambourn during a storm event in 2002

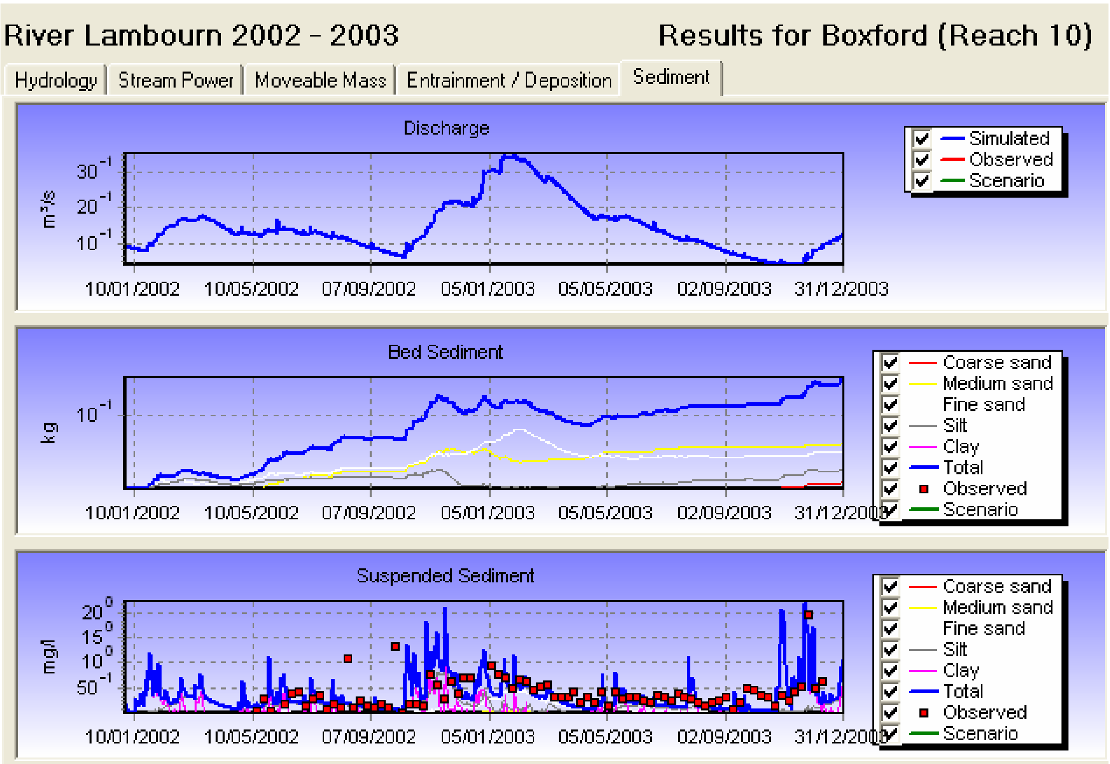

Figure 3.11 River Lambourn observed and simulated suspended sediment and simulated bed sediment 2002-2003

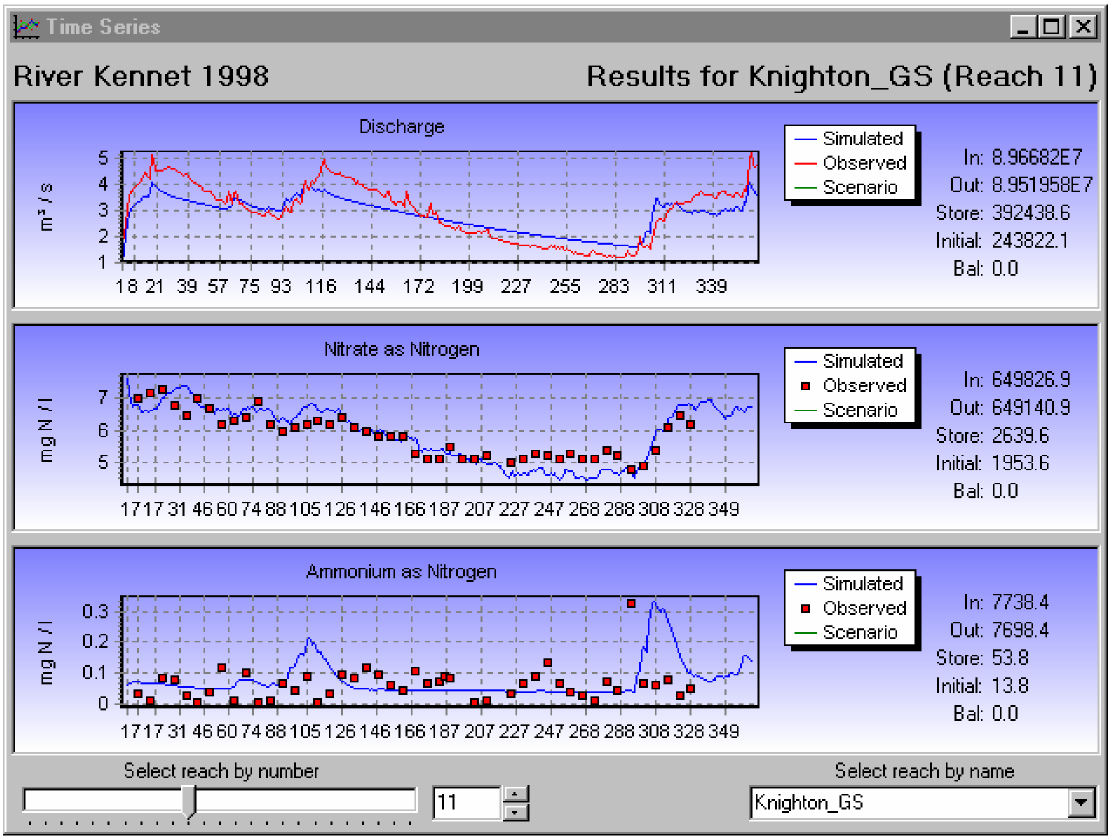

Figure 3.12 Simulated and observed flow, nitrate as N and ammonia as N for the upper Kennet, 1998

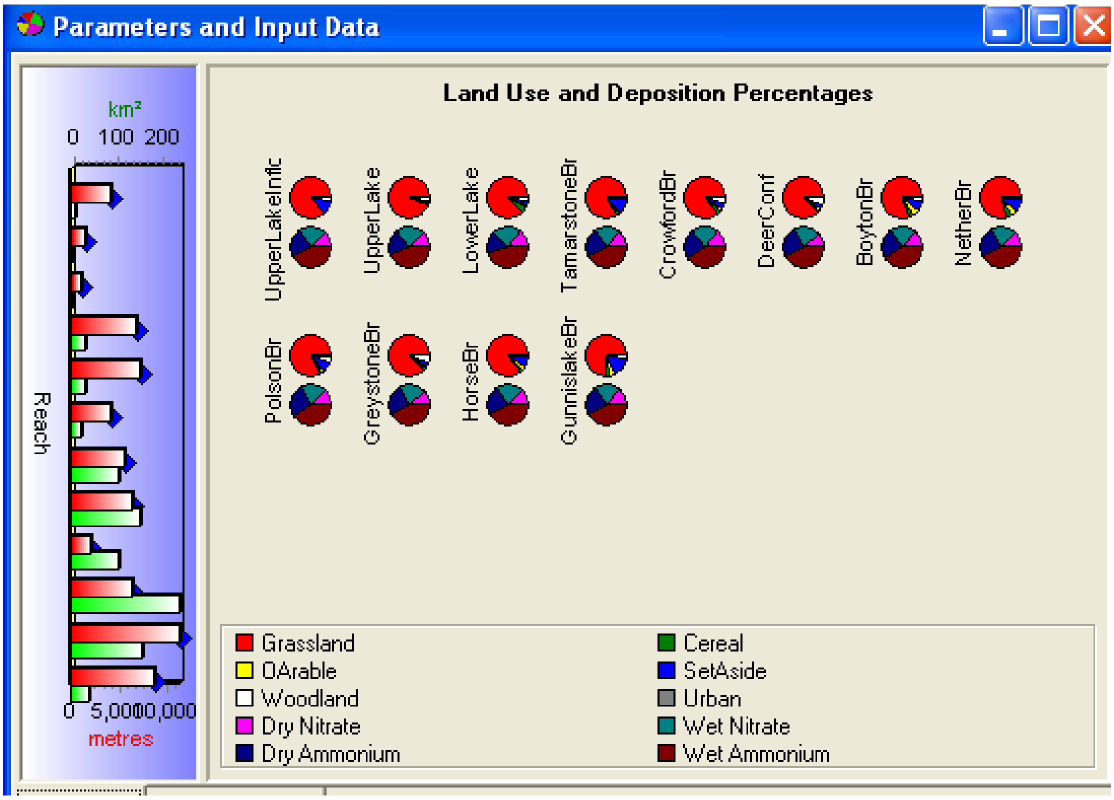

Figure 3.13 Land Use and N deposition for the River Tamar

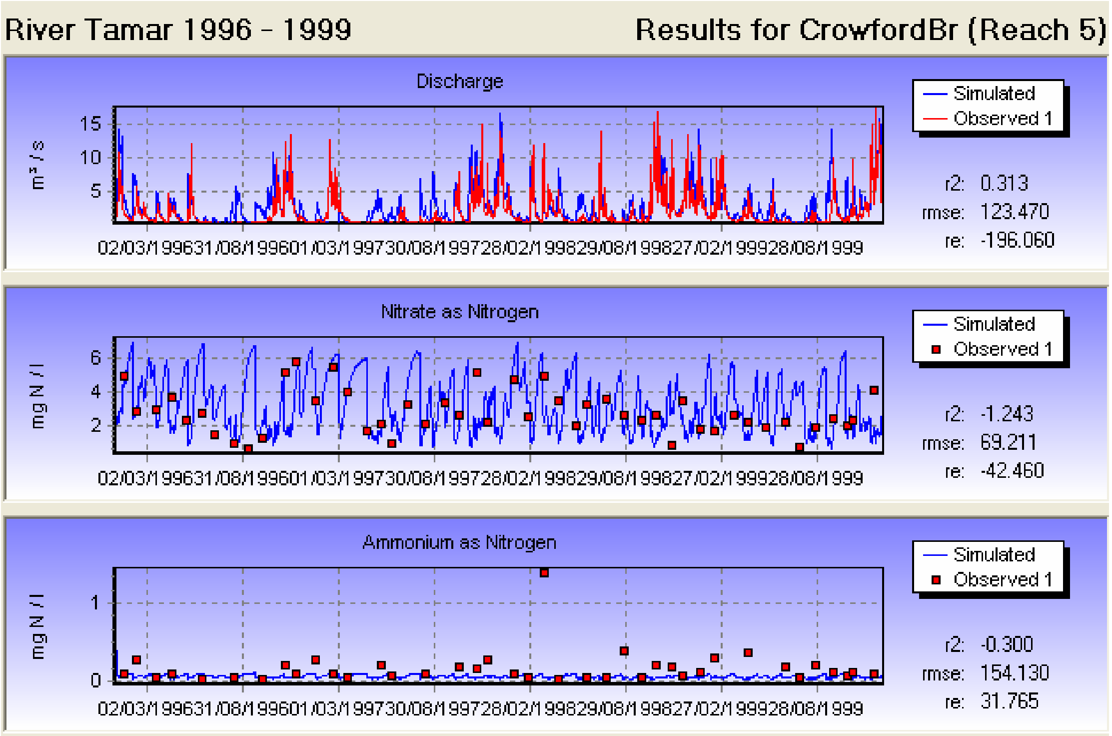

Figure 3.14 River Tamar simulated and observed flow, nitrate and ammonia 1996-1999

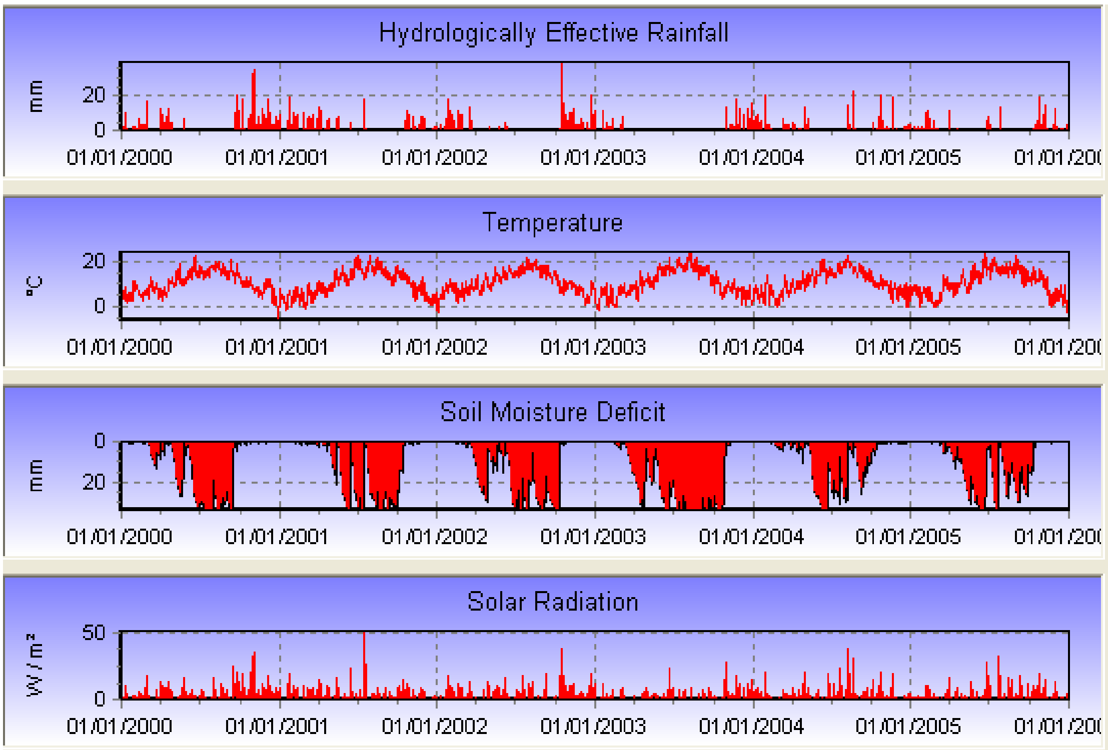

Figure 3.15 Hydrological data inputs for the River Tame 2000-2005

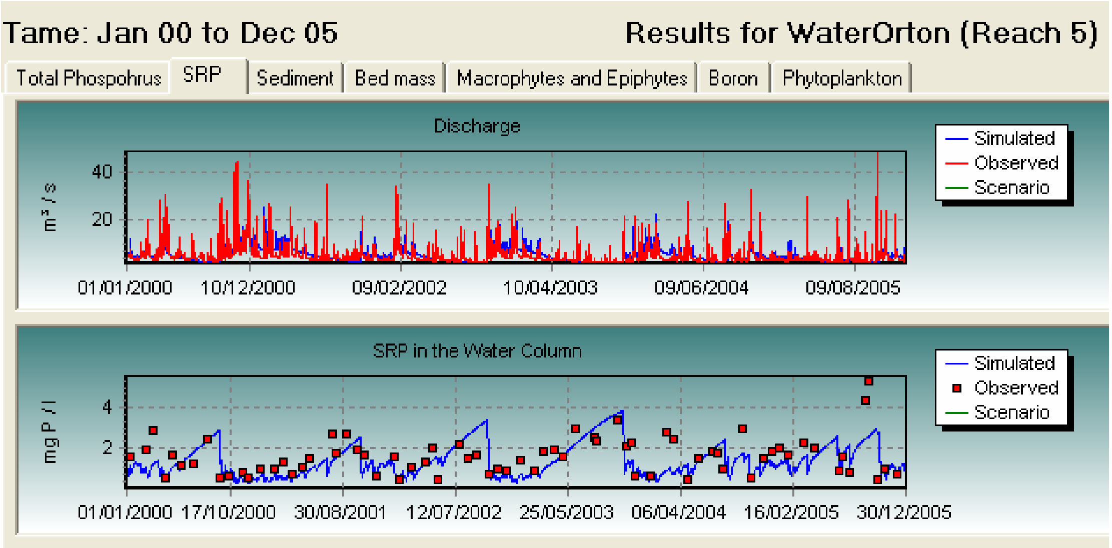

Figure 3.16 Flow and SRP model simulation and observed data for Water Orton on the Tame

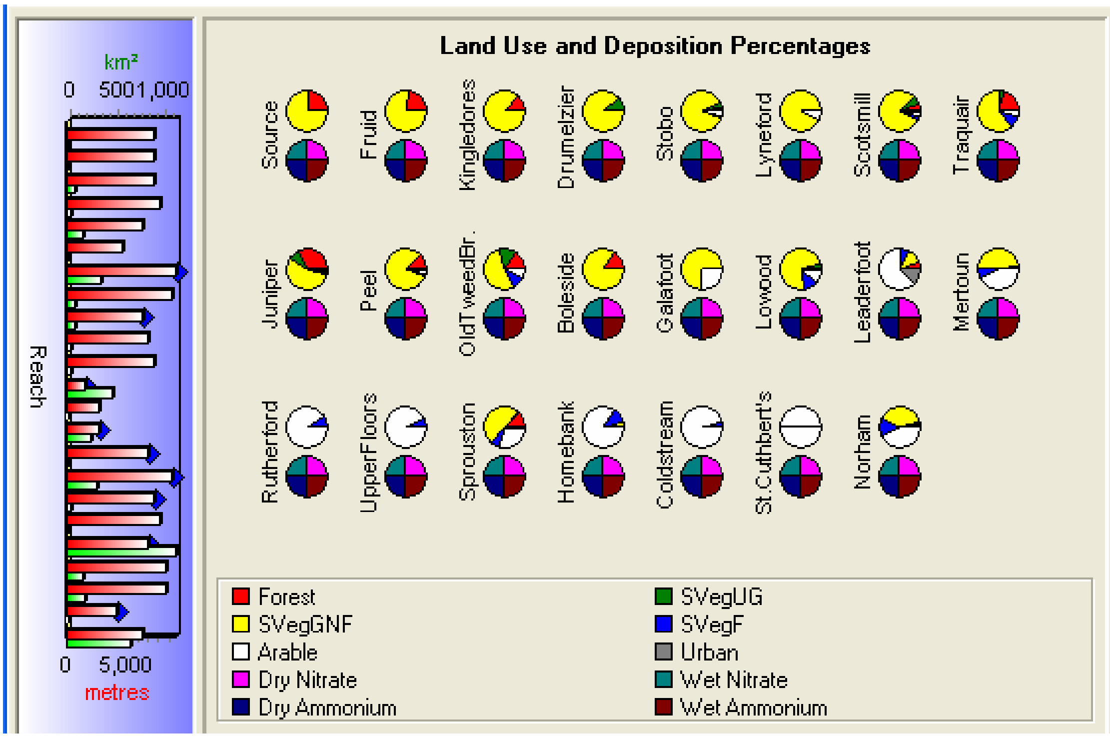

Figure 3.17 Land use and N deposition for the River Tweed (Jarvie et al, 2002)

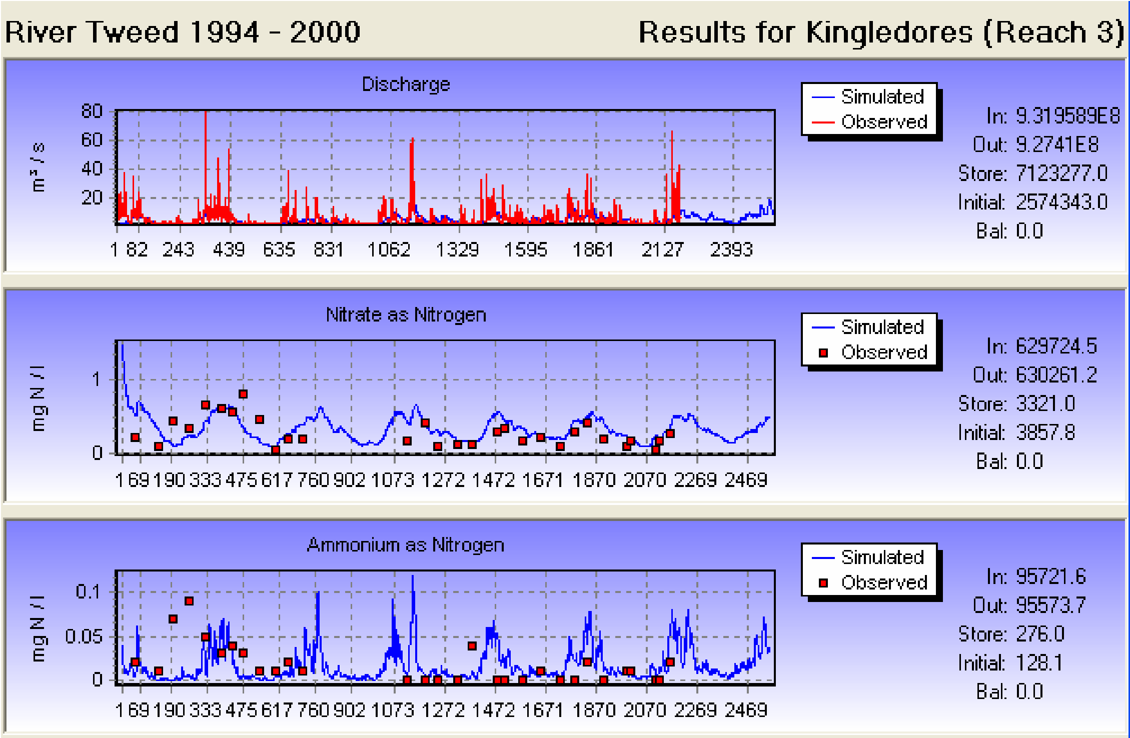

Figure 3.18 Upper River Tweed simulated and observed flow, nitrate and ammonia for 1994-2000

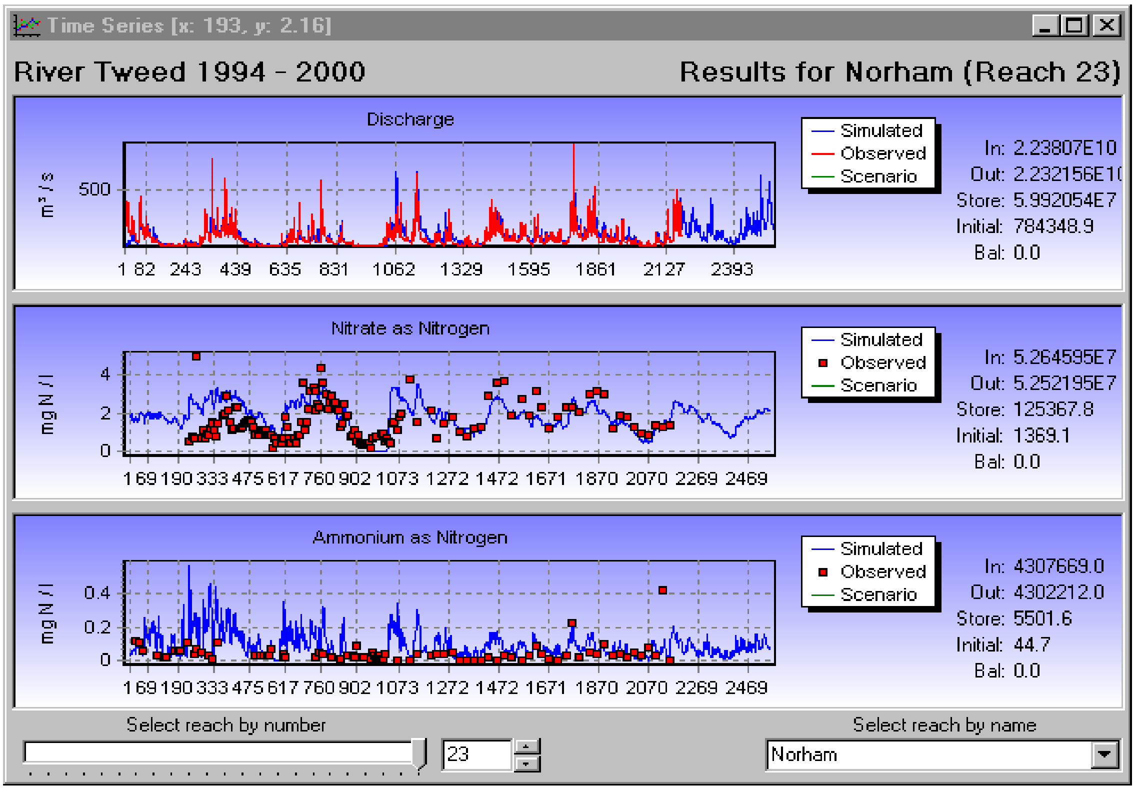

Figure 3.19 Lower River Tweed simulated and observed flow, nitrate and ammonia for 1994-2000

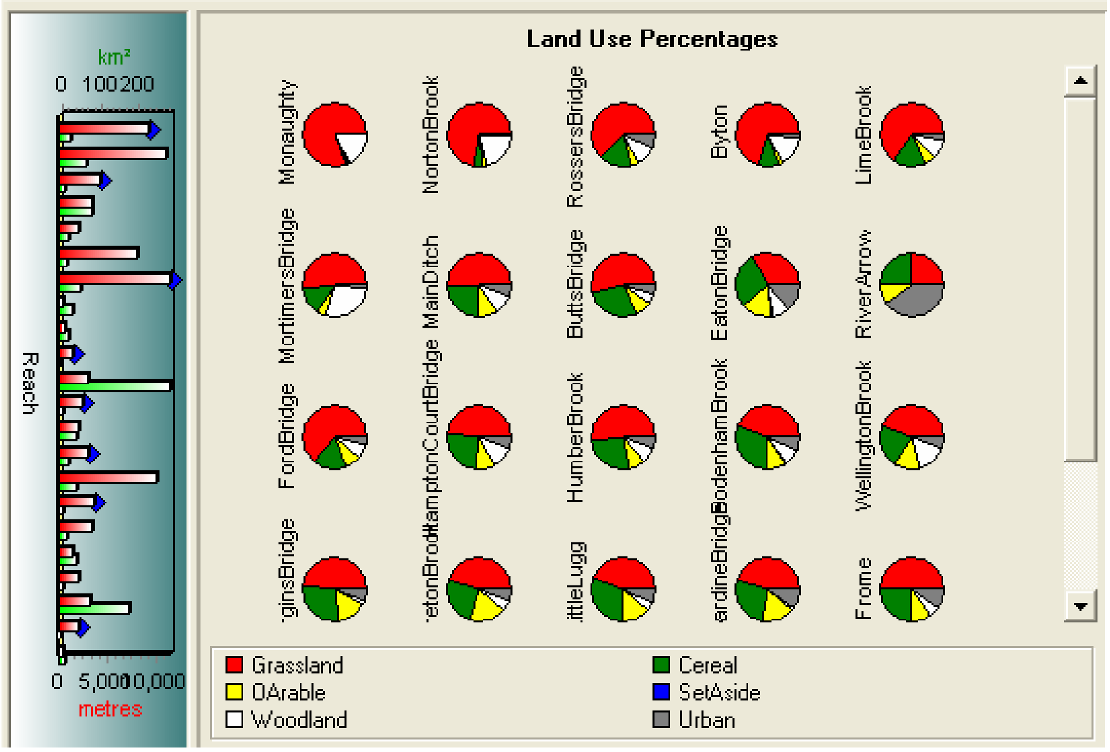

Figure 3.20 Reach structure and land use for the River Lugg

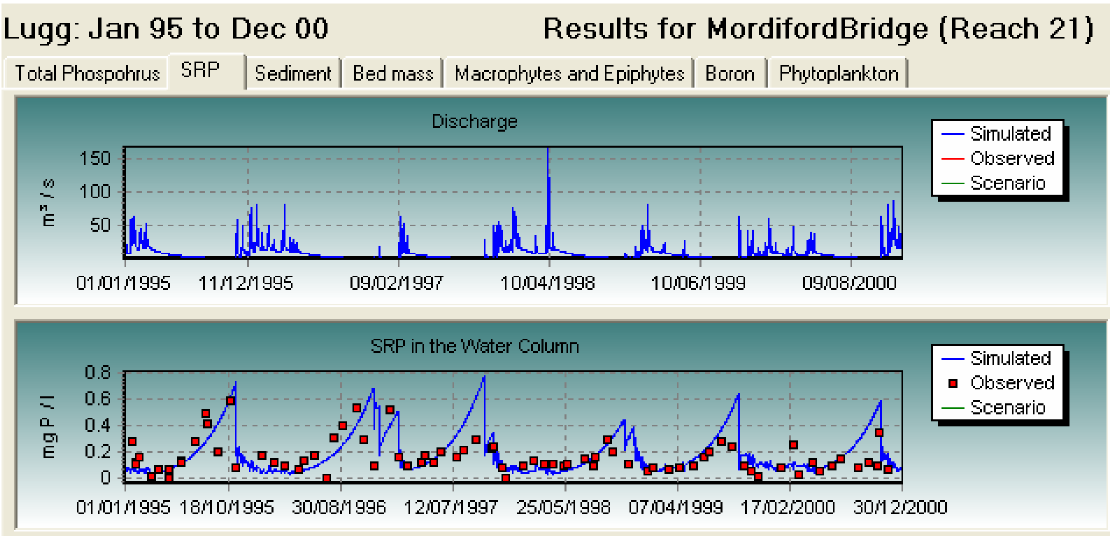

Figure 3.21 Simulated and observed phosphorus for the River Lugg 1995-2000

3.5 Delta change results

The INCA model simulations were performed for all four scenarios listed in Table 3.2, for the 2010s, 2020s and 2050s. This involved generating 12 input data files for all versions of INCA used for the catchments. Given the scope of the project, we only used well established applications of the models. These were INCA-P for the Lambourn, the Lugg and the Tame, INCA-N for the Tamar and the Tweed and INCA-SED for the Lambourn. INCA-P does incorporate sediment, so sediment was considered where possible. In addition, simulations have been executed for upper and lower reaches, thereby generating 144 sets of results. The information extracted from the simulations takes two forms: changes in monthly water quality and statistical distributions. Given the volume of model output, we have only presented summary results below.

River Lambourn phosphorus results

The impacts of climate change on the Lambourn under the four scenarios and three time periods are shown in Figures 3.22 to 3.26. The first three, Figures 3.22 to 3.24, show monthly percentage changes for the three time periods. The patterns of change are broadly similar but the magnitude of the effect increases with time. The effects are more significant in the 2050s than for earlier dates.

This is further demonstrated in Figure 3.25, which compares the three time periods under the high emissions scenario.

The pattern of change through the year is the same in all the scenarios. Concentrations of soluble reactive phosphate (SRP) decline during the winter and spring months when flows are generally increasing due to wetter winters, and increase during the summer and autumn months when flows are decreasing and therefore there is less dilution of discharges and agricultural runoff. Even so, the peak increases of four per cent of SRP are not particularly large. This result is very similar to the UKWIR study results (Conlan *et al*. 2006), which also show generally relatively little change in SRP under climate change scenarios.

The small amount of change could be an artefact of the delta change method, which does not build in the transient build up effect of climate change. This is explored in a later section of this report (section XX).

There is little change in the statistical distributions of phosphorus concentrations for the various climate changes scenarios. Figures 3.26 and 3.27 show percentile distributions for different time horizons and different emission scenarios. In both cases, the changes in distribution are minimal. This may be because the changes that are occurring have a symmetrical pattern over the year, as shown in Figures 3.22 to 3.27, so the decreases are balanced by increases and there is little change in the overall distribution. Even so, the changes are small compared with the range of statistical distributions covered by European water quality standards, which apply to all the Member States and their climates, and to year-on-year variations within Member States and across Europe.

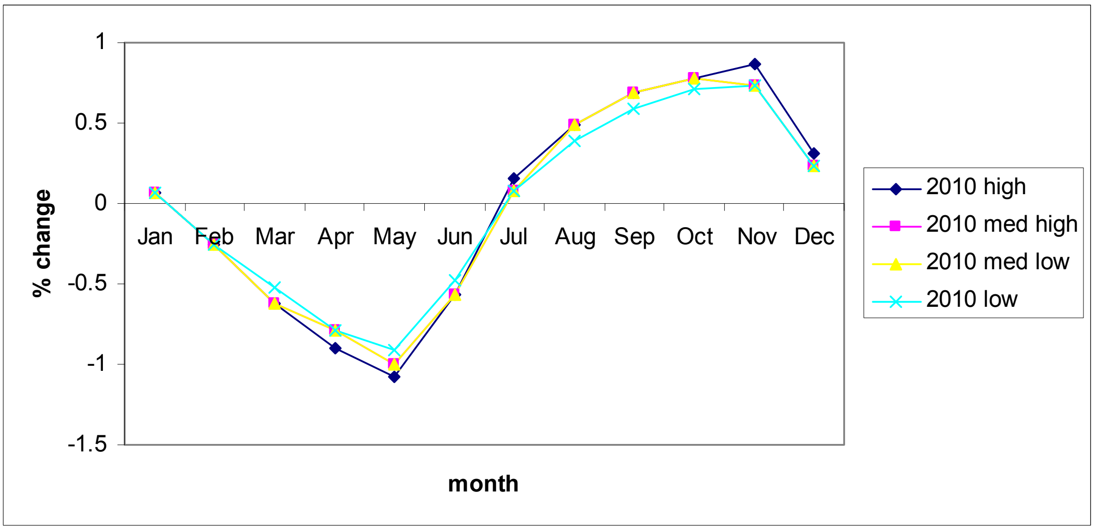

Figure 3.22 River Lambourn monthly percentage changes in phosphorus for the 2010s

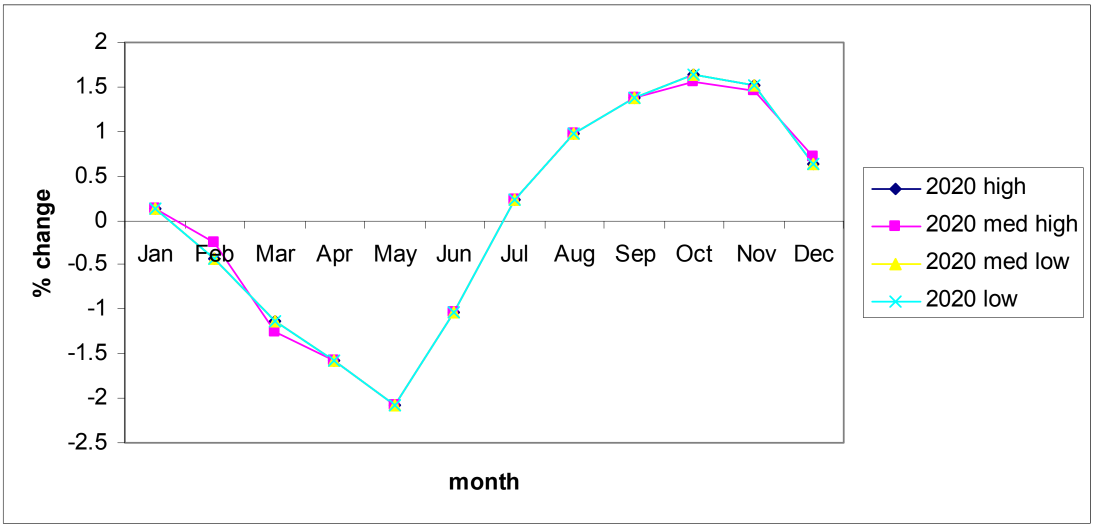

Figure 3.23 River Lambourn monthly percentage changes in phosphorus for the 2020s

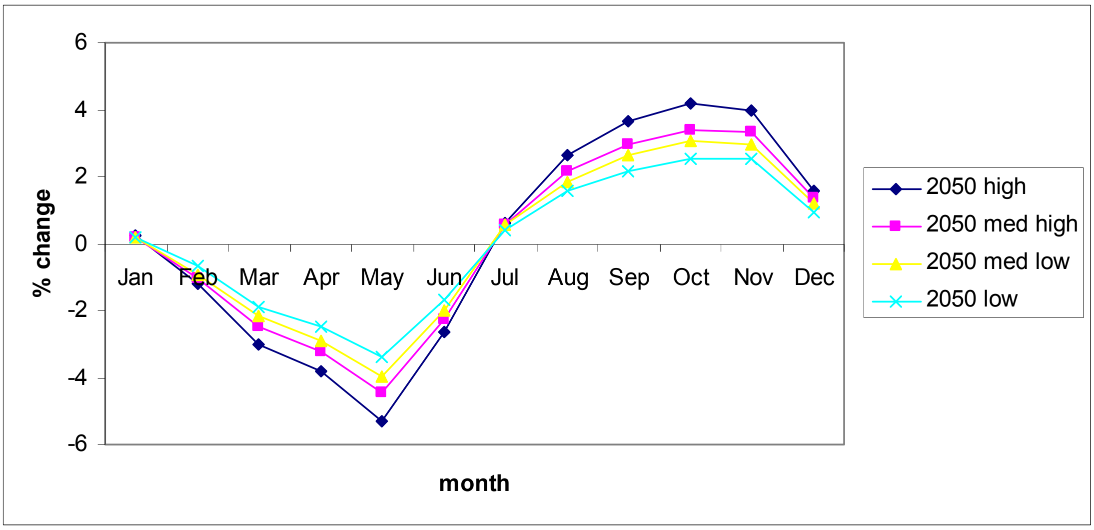

Figure 3.24 River Lambourn monthly percentage changes in phosphorus for the 2050s

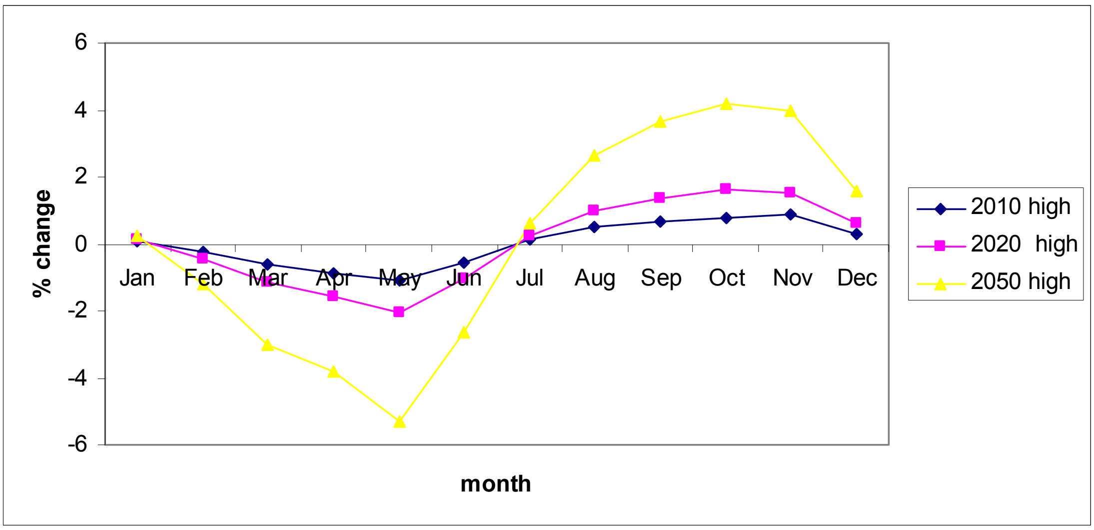

Figure 3.25 River Lambourn monthly percentage changes in phosphorus for all the high emission scenarios

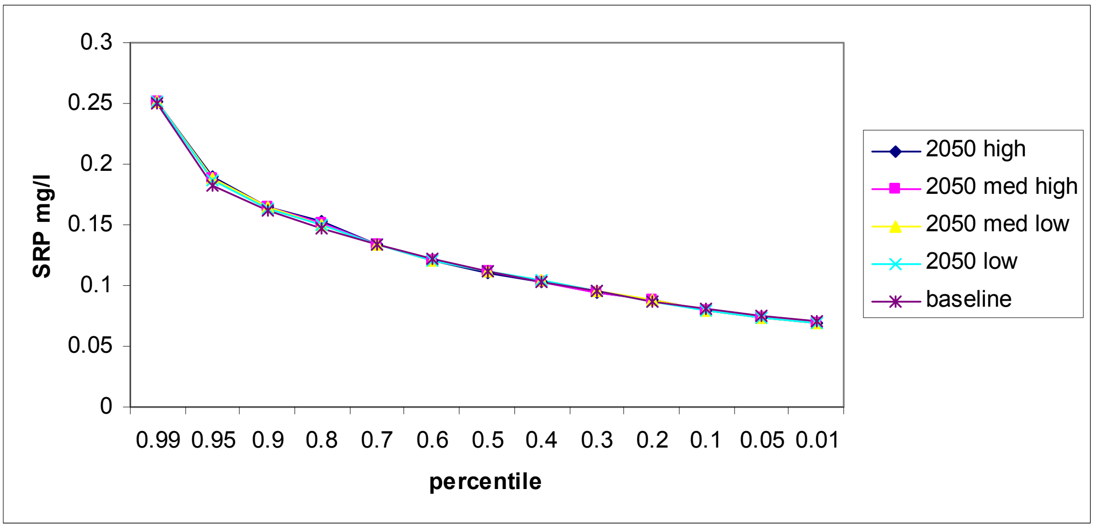

Figure 3.26 River Lambourn phosphorus concentrations as percentiles for the 2050s

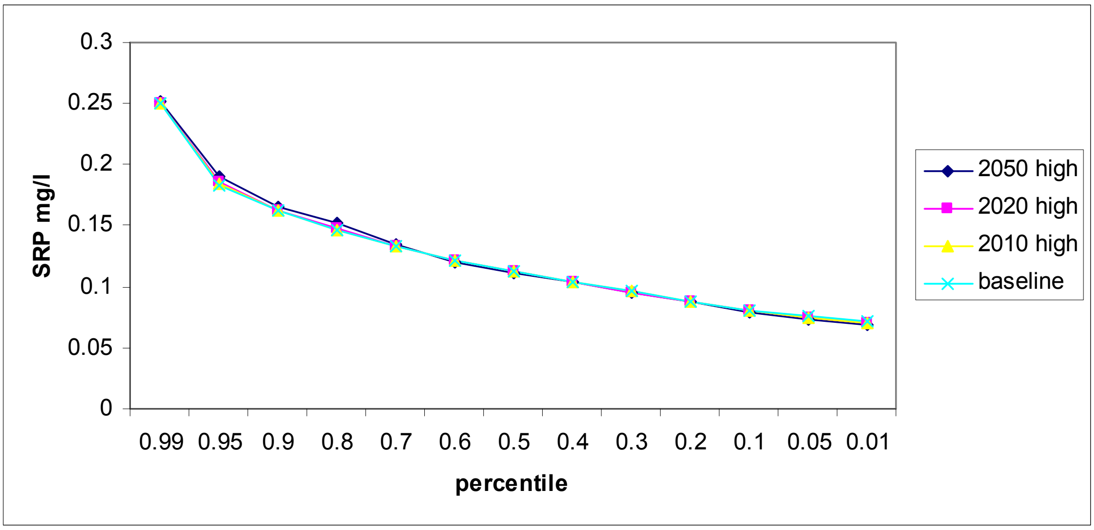

Figure 3.27 River Lambourn phosphorus concentrations as percentiles for all the high scenarios

River Tame results

The results for the River Tame are very different to those for the Lambourn, because the Tame drains the heavily urbanized area of Birmingham and its summer flows are sustained by the effluent discharges. For example, the Minworth sewage treatment works in the upper Tame has a discharge of approximately 4 m^3/s compared with an average summer river discharge of approximately 10 m^3/s. Also, the geology in the Tame is sandstone, mudstones and silt clay soils, significantly different from the chalk bedrock of the Lambourn. The Tame is more affected by direct runoff than the Lambourn, whether it is urban in the upper reaches or agricultural in the lower reaches.

The model results for the Tame reflect this differing hydrology and diffuse pollution response. Figures 3.28 to 3.31 show the scenarios for the 2010s, the 2020s and the 2050s, respectively. They all show the same pattern of increased phosphorus concentrations in the summer, due to the reduced flows and decreased dilution. However, much higher increases are seen in winter. This appears counterintuitive but it reflects the fact that increased winter rainfall will flush increased nutrients from the urban area (for example via storm water drainage) and from the agricultural areas (for example via nutrient runoff) in the lower reaches of the Tame. As in the case of the Lambourn, the probability distributions are very similar for the different scenarios, but there is a slight shift from the baseline situation (2000-2005), as shown in Figures 3.32 and 3.33, reflecting the more dynamic or surface dominates river.

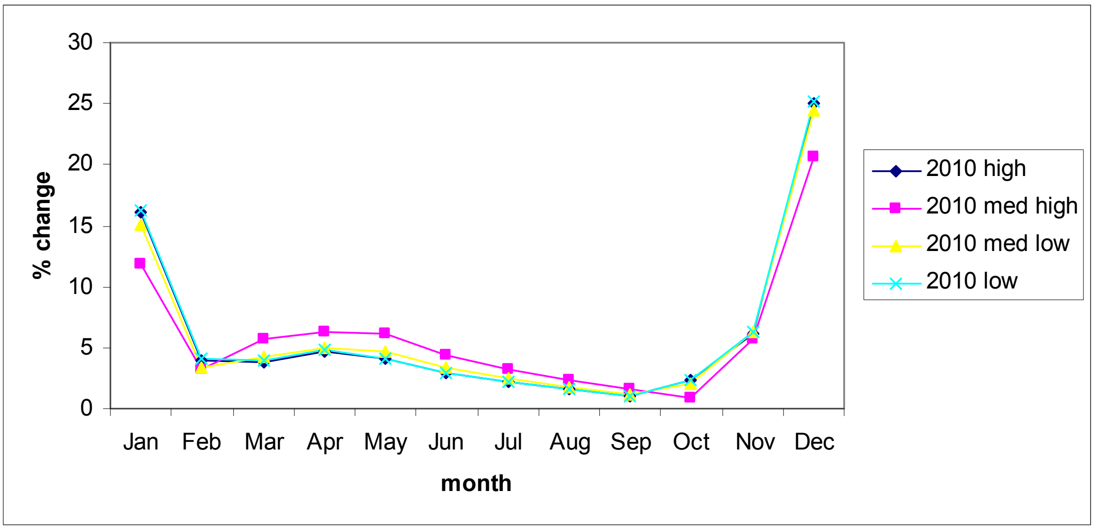

Figure 3.28 River Tame monthly percentage changes in phosphorus for the 2010s

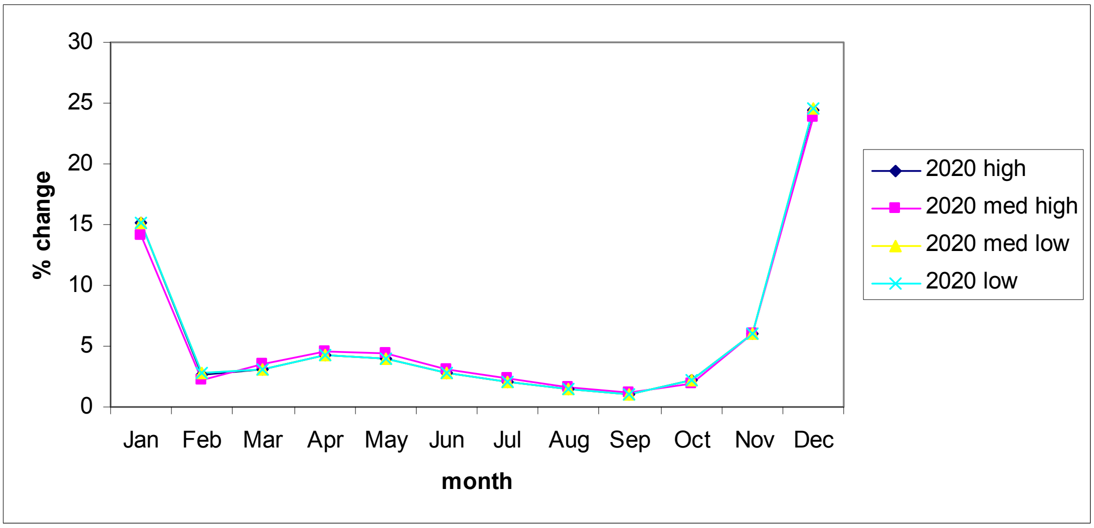

Figure 3.29 River Tame monthly percentage changes in phosphorus for the 2020s

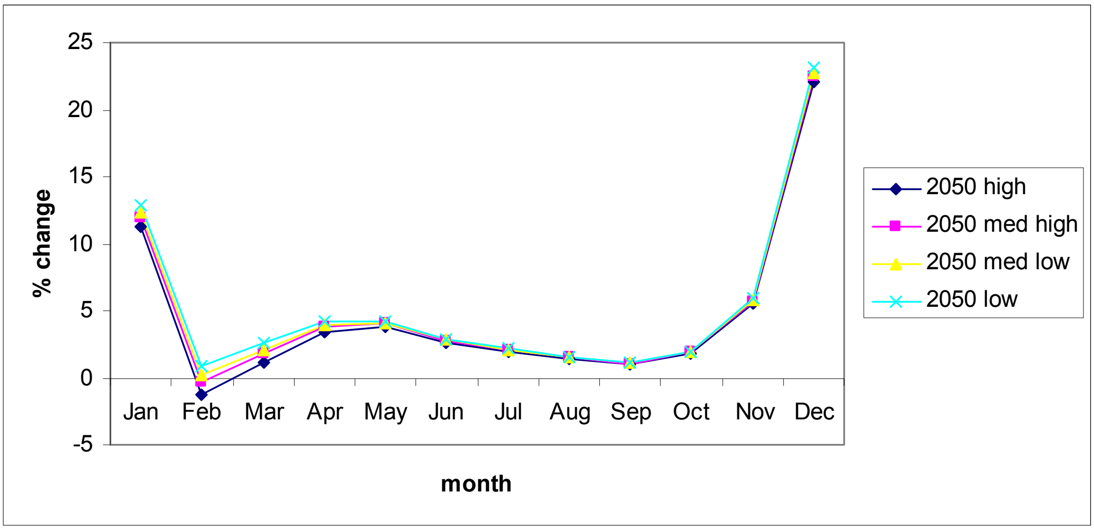

Figure 3.30 River Tame monthly percentage changes in phosphorus for the 2050s

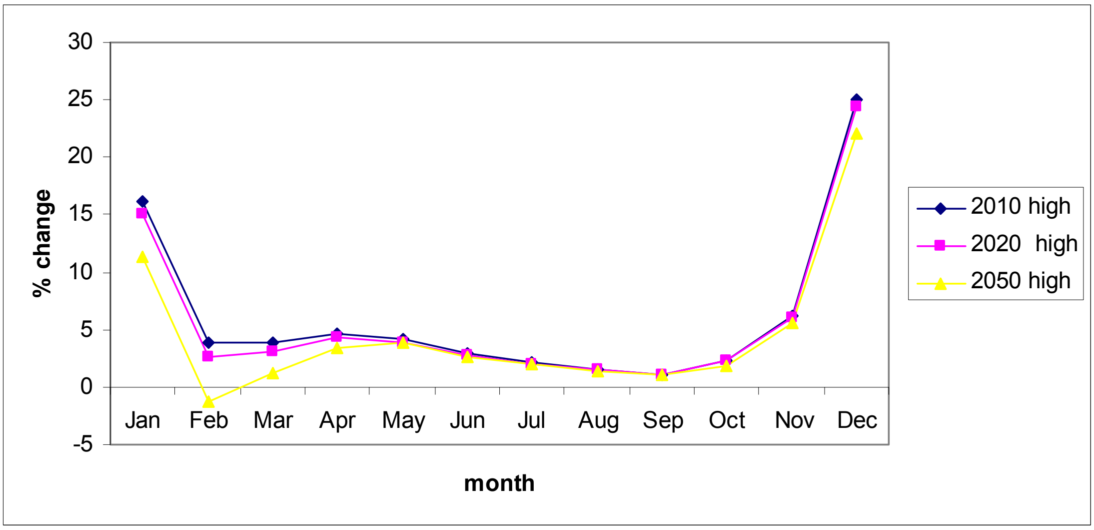

Figure 3.31 River Tame monthly percentage changes in phosphorus for all the high scenarios

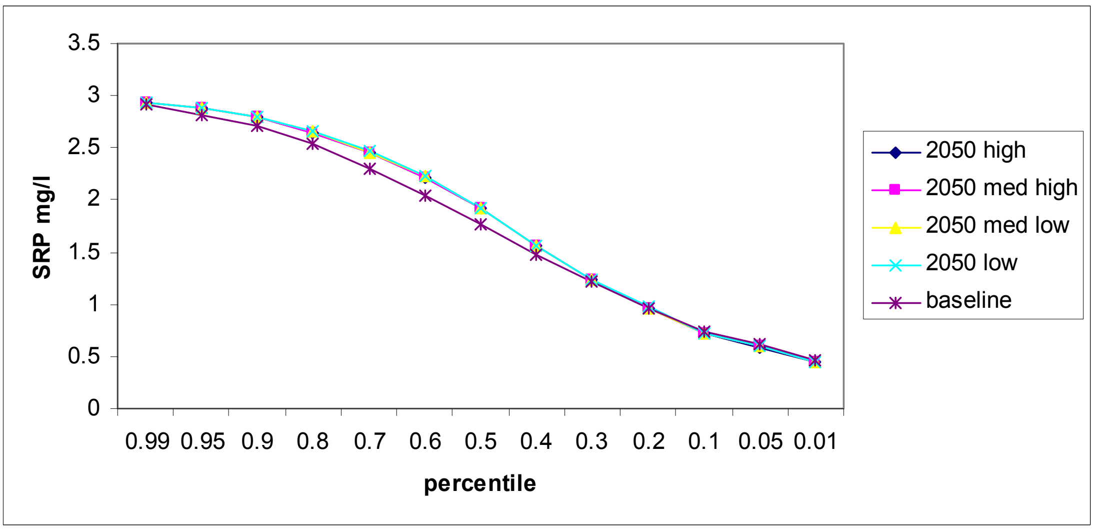

Figure 3.32 River Tame phosphorus concentrations as percentiles for all the high scenarios

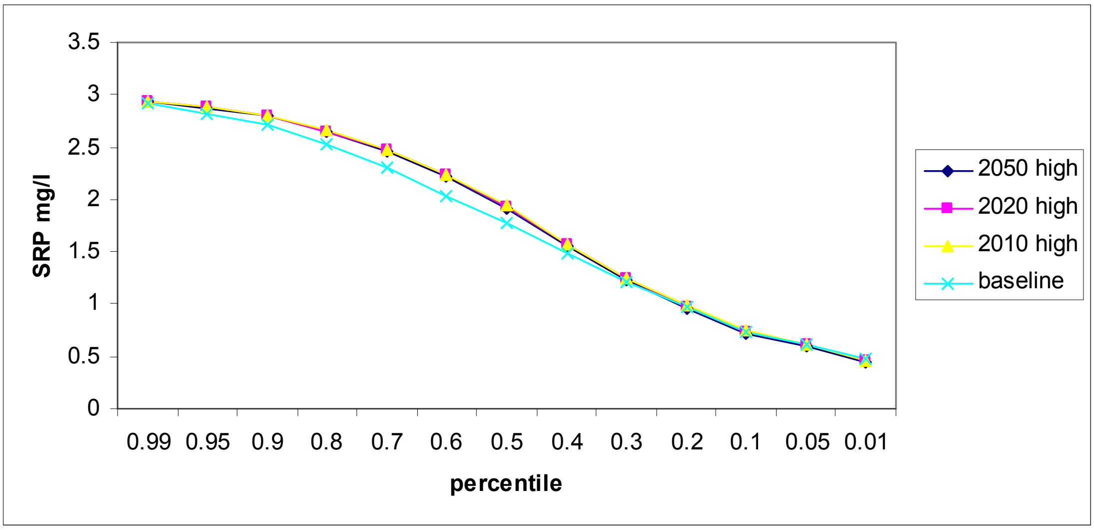

Figure 3.33 River Tame phosphorus concentrations as percentiles for all the high scenarios

River Lugg results

The River Lugg phosphorus results for the upper reaches of the river (Figures 3.34 to 3.38) are very similar to those for the Lambourn. The models produce decreased phosphorus concentrations in the winter and spring, and increased concentrations in the late summer. This reflects the change in flow conditions due to climate change, with reduced summer flows producing less dilution. There is also not much change in the distributions of phosphorus concentrations, in the upper reaches of this rural catchment (not shown).

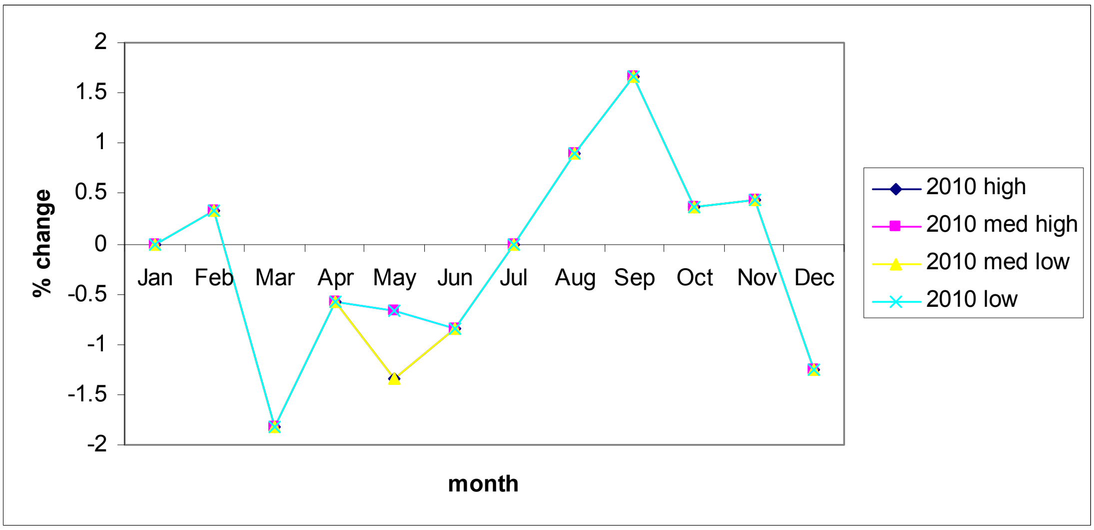

Figure 3.34 River Lugg monthly percentage changes in phosphorus for the 2010s

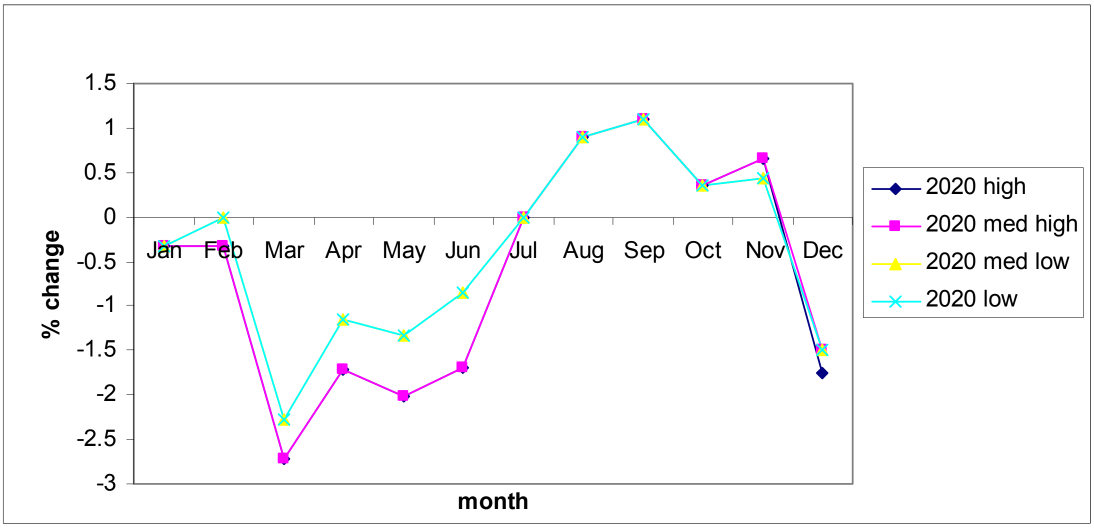

Figure 3.35 River Lugg monthly percentage changes in phosphorus for the 2020s

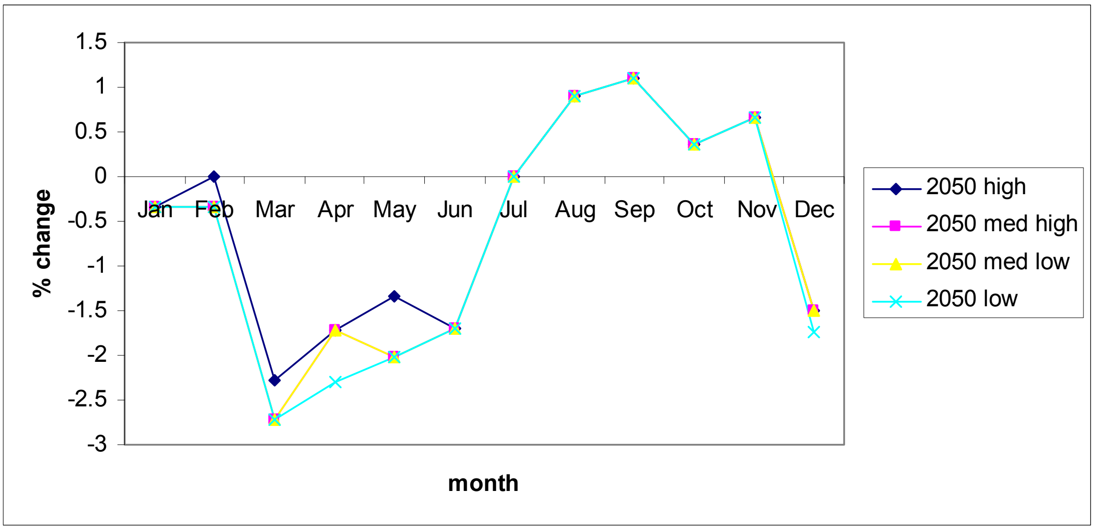

Figure 3.36 River Lugg monthly percentage changes in phosphorus for the 2050s

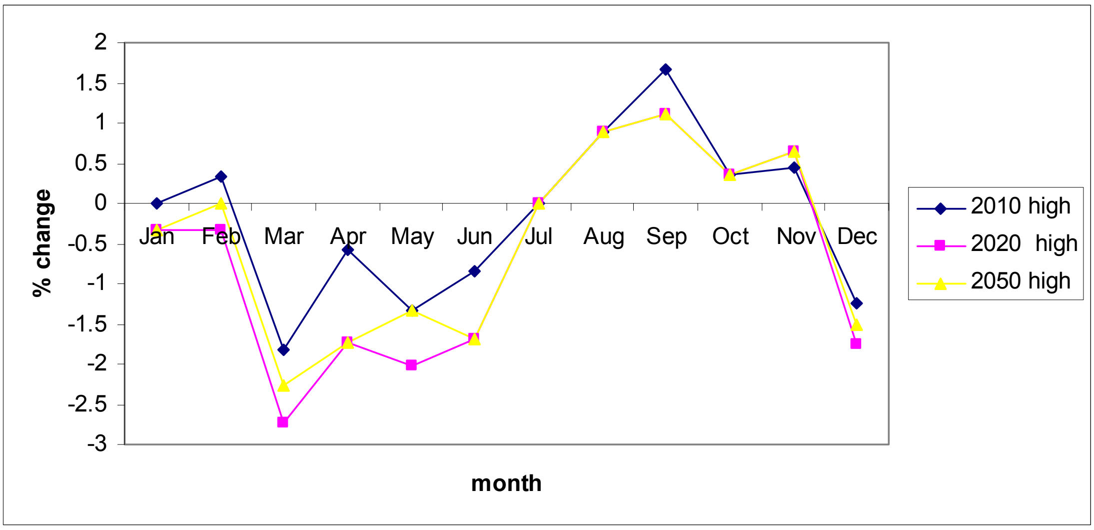

Figure 3.37 River Lugg monthly percentage changes in phosphorus for the 2010s

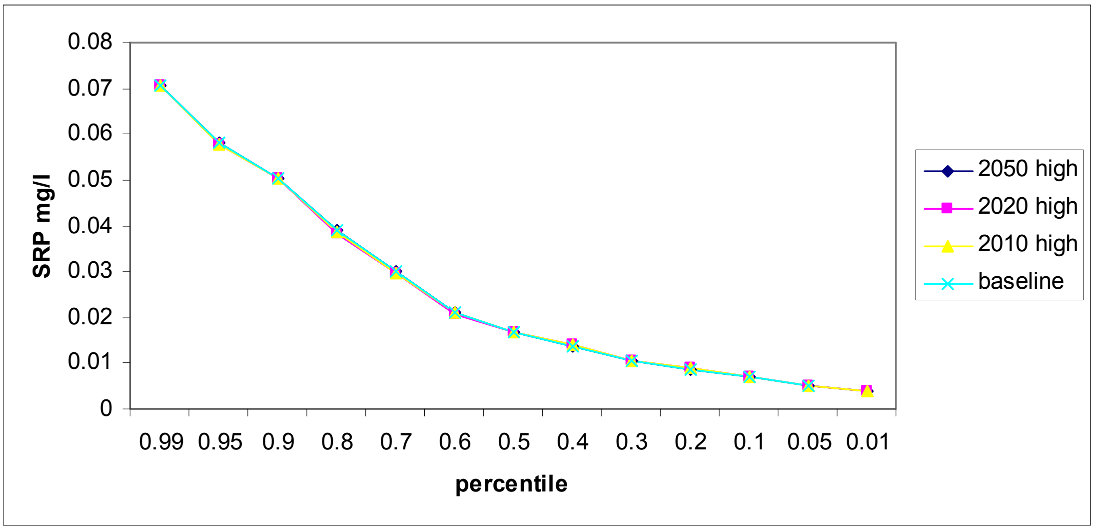

Figure 3.38 River Lugg phosphorus concentrations as percentiles for the high emission scenarios

River Tweed results

The results for nitrate (as N) in the upper Tweed are shown in Figures 3.39 and 3.40. Here the nitrate concentrations are seen to be slightly higher in the winter months under the climate change scenarios, perhaps reflecting the higher flushing of nitrogen load from the upland peaty soils. However, in the summer months nitrates fall significantly. This probably reflects enhanced denitrification processes in the river. Denitrification is highly dependent on temperature and residence time. With warmer conditions and slower velocities in the river, there is a greater rate of nitrate loss via denitrification. This generates lower nitrate concentrations in the river. The downside of this is that denitrification results in the release of nitrous oxide, which is a significant greenhouse gas. INCA could be used to calculate the flux of nitrous oxide under these situations, although this would need to be looked at in the context of other changes affecting nitrate, in terms of the percentage change in the total nitrous oxide from denitrification.

Interestingly, the nitrate results lower down the river are quite different, as shown in Figures 3.41 to 3.43. Here, the nitrate concentrations are significantly higher in the summer months. This is most probably because there is significant runoff from agriculture in the lower reaches of the Tweed, where upland moorland and grassland are replaced by arable farming. There are also some significant discharges from sewage treatment works. Lower flows in summer reduce the dilution of point and diffuse sources of nitrate, and concentrations rise. By the 2050s, the simulation suggests that denitrification is overcoming this effect and having a larger effect than the lack of dilution (Figure 3.43). It is known that large lowland rivers such as the Thames can lose up to 70 per cent of the nitrate through denitrification processes in summer low flow conditions and this situation could be happening in the Tweed from the 2050s (Whitehead and Williams 1982). Note, however that the percentile distributions do not change significantly, as shown in Figure 3.44.

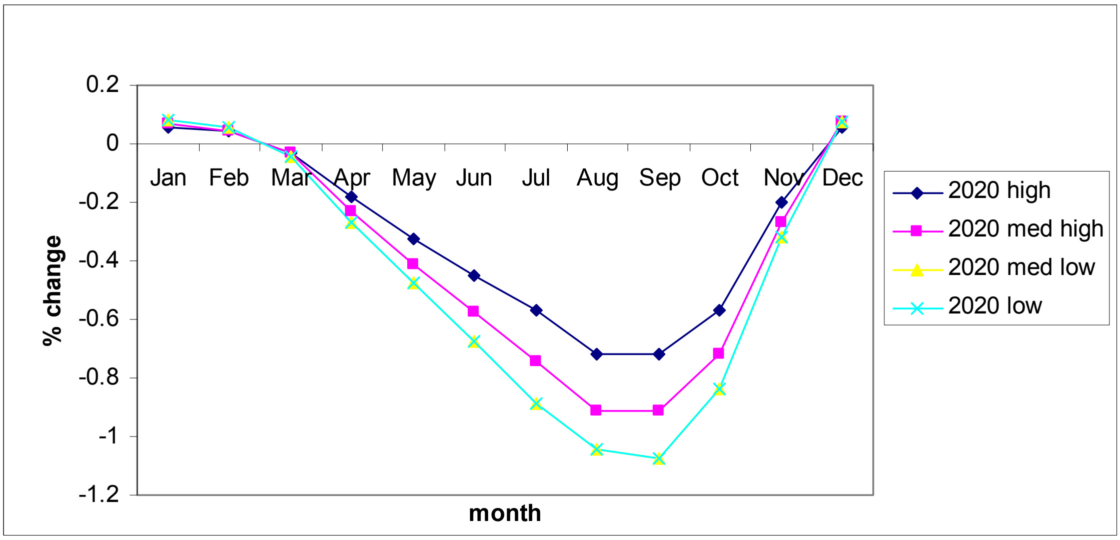

Figure 3.39 River Tweed (upper reaches) monthly percentage changes in nitrate (as N) for the 2020s

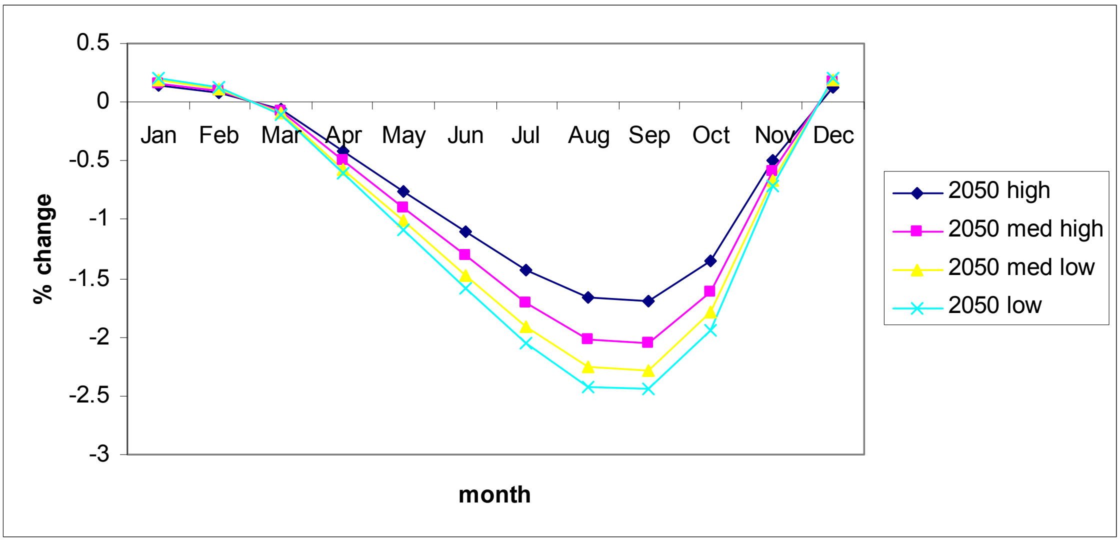

Figure 3.40 River Tweed (upper reaches) monthly percentage changes in nitrate (as N) for the 2050s

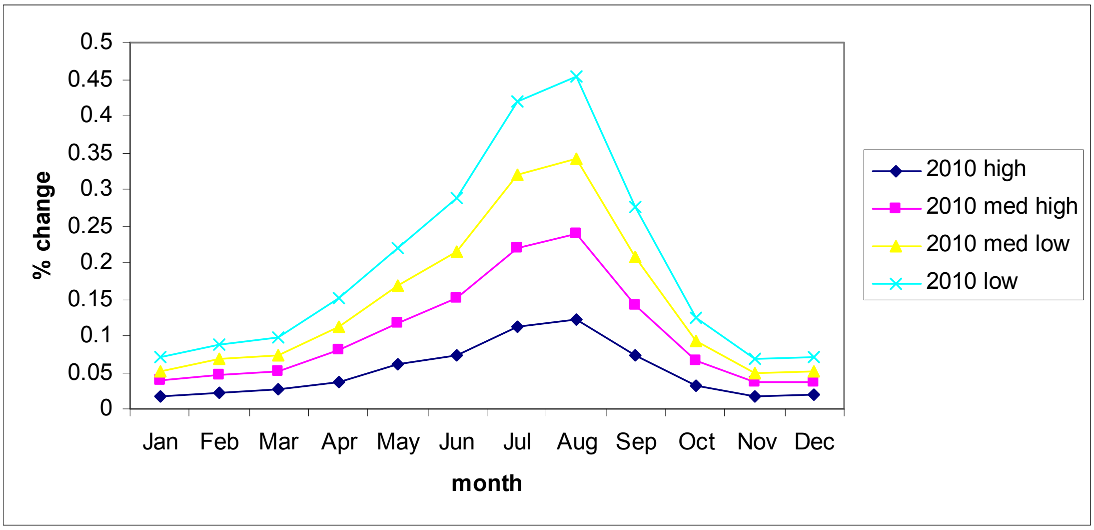

Figure 3.41 River Tweed (lower reaches) monthly percentage changes in nitrate (as N) for the 2010s

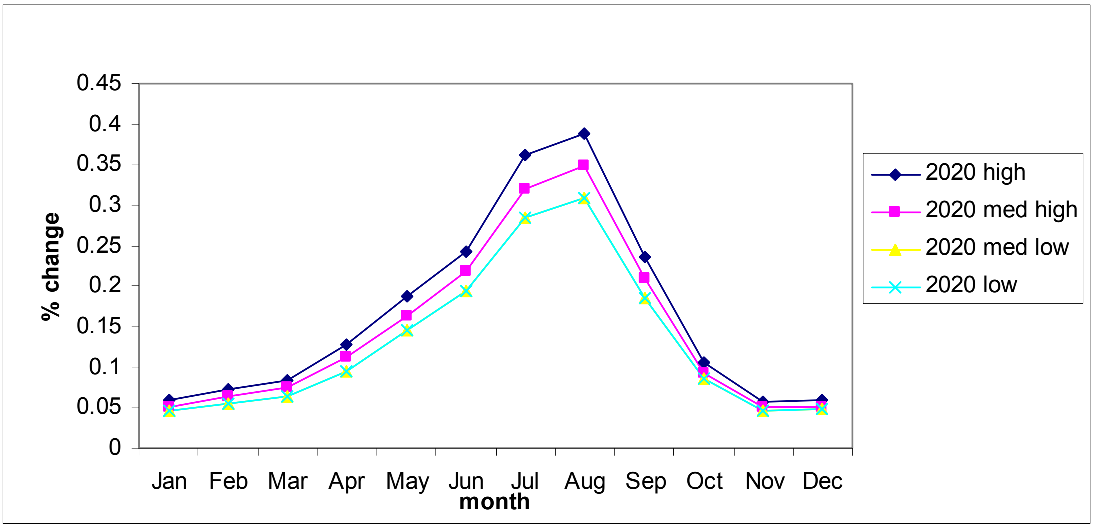

Figure 3.42 River Tweed (lower reaches) monthly percentage changes in nitrate (as N) for the 2010s

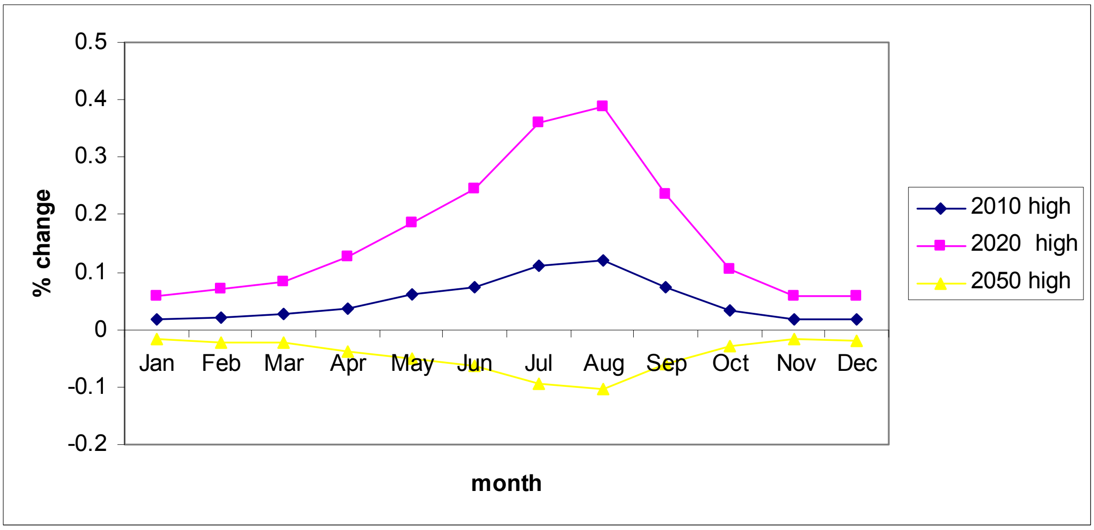

Figure 3.43 River Tweed (lower reaches) monthly percentage changes in nitrate (as N) for the high scenarios

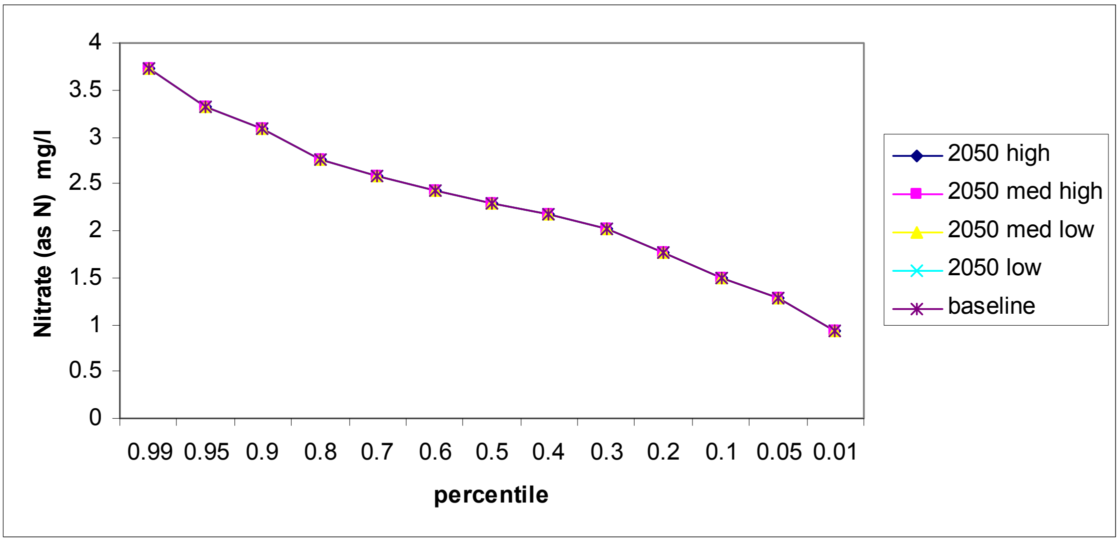

Figure 3.44 River Tweed nitrate (as N) concentrations as percentiles for all the high scenarios

River Tamar results

The results for the River Tamar are similar to those for the River Tweed. Upper reaches of the river behave as a natural stream, whereas lower down the river system, the impacts from sewage treatment works and diffuse sources from agriculture have a large effect. Figures 3.45 to 3.47 show a strong seasonal pattern of change in the upper Tamar, with slightly increased nitrate concentrations in the winter months, as in the case of the upper Tweed, but significantly lower concentrations in summer months. The reduction in summer is again, probably linked to the denitrification processes operating in the upper reaches of the river. Lower down the river system, the pattern is reversed with significant increases in summer reflecting the lack of dilution, as shown in Figures 3.48 and 3.49. As in the case of the Tweed, the statistical distributions of nitrate concentrations do not change significantly (Figure 3.50).

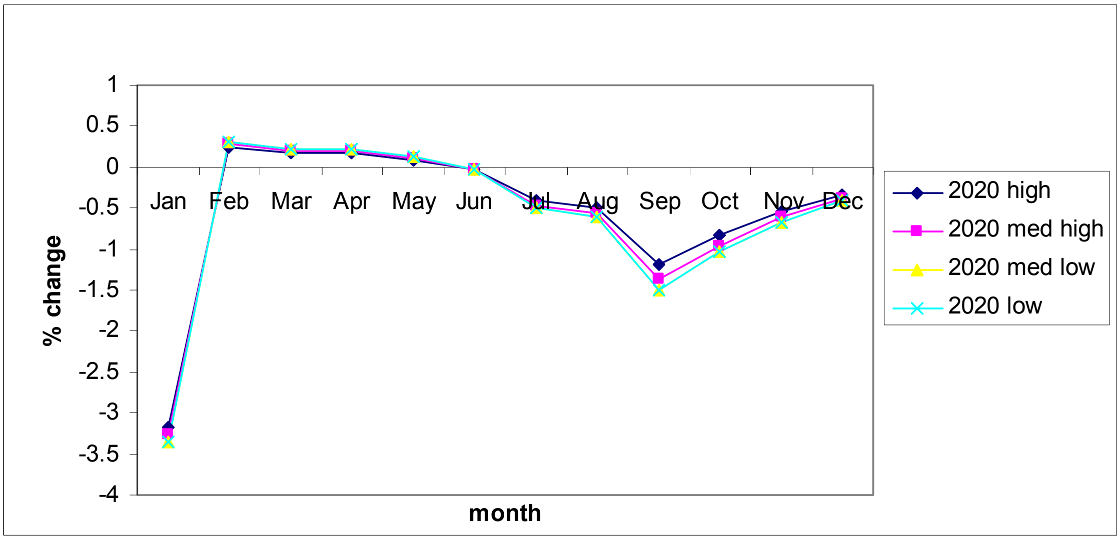

Figure 3.45 Upper River Tamar monthly percentage changes in nitrate (as N) for the 2020s

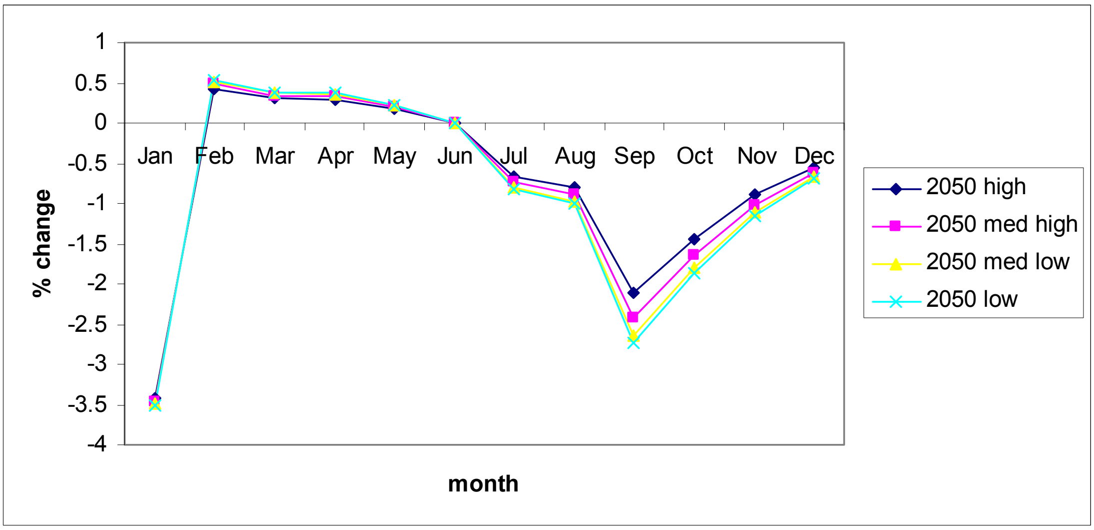

Figure 3.46 Upper River Tamar monthly percentage changes in nitrate (as N) for the 2050s

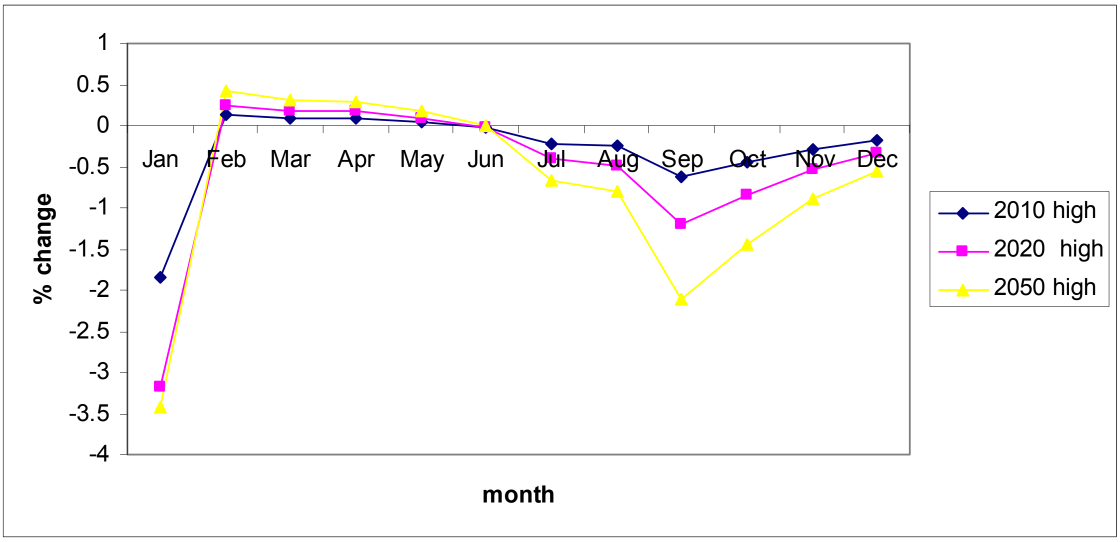

Figure 3.47 Upper River Tamar monthly percentage changes in nitrate (as N) for the high scenarios

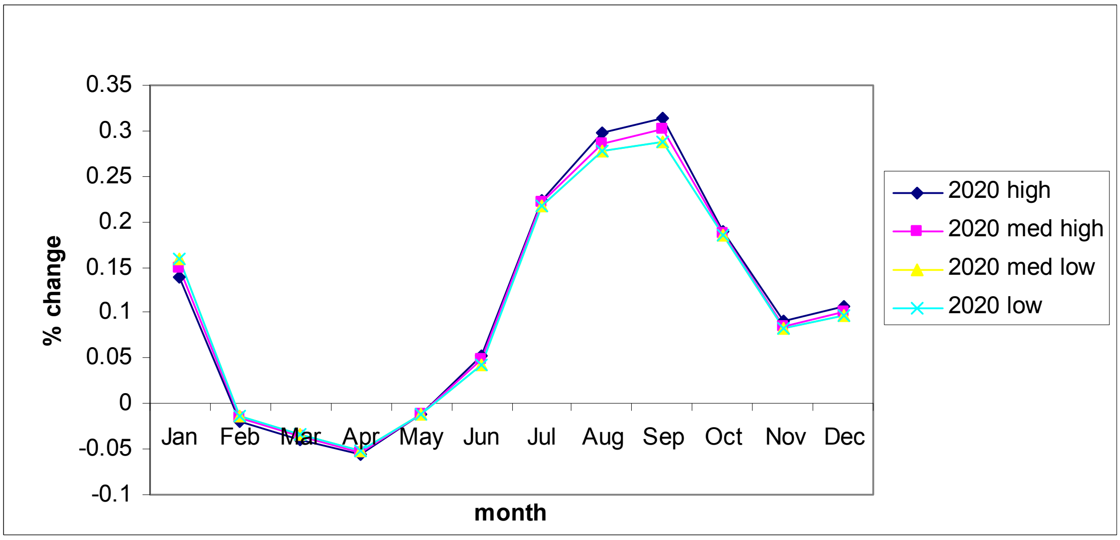

Figure 3.48 Lower River Tamar monthly percentage changes in nitrate (as N) for the 2020s

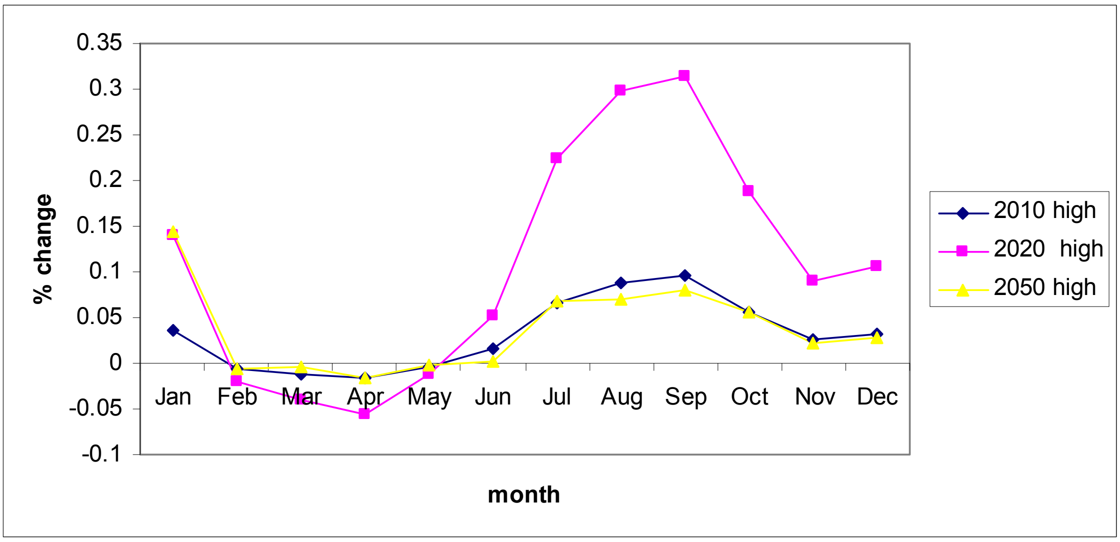

Figure 3.49 Lower River Tamar monthly percentage changes in nitrate (as N) for the high scenarios

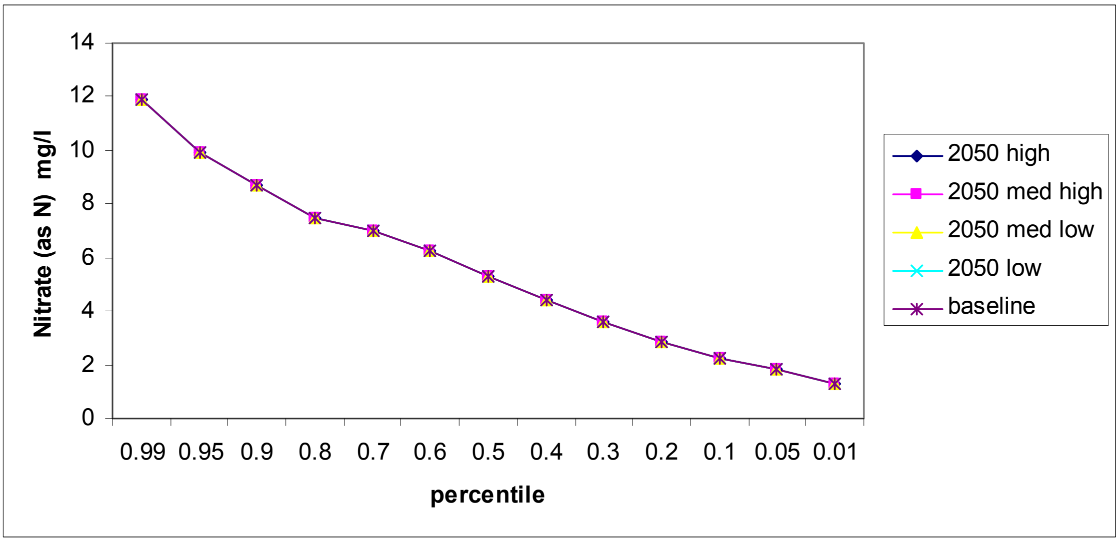

Figure 3.50 Tamar nitrate (as N) concentrations as percentiles for all the high scenarios

River Lambourn fine sediment simulation

The Lambourn sediment results show two periods of increase in the spring and the autumn for fine sediment delivery (Figures 3.51 to 3.53). These increases reflect the river flow changes and storm conditions, with increased storm events in early spring and at the end of the summer. Sediment release is also a function of antecedent conditions and the rate of change of flow. Hence, by the end of the summer, there has been sufficient time for the accumulation of sediments on the land surface and, with the onset of autumn rainfall, these sediment deposits are flushed into the river. Even so, the changes are not large and this is reflected in minimal change in the percentile curve (Figure 3.54). The percentile curve for the lowest reach of the Lambourn indicates a larger change in the sediment releases under the climate change scenarios for the 2050s (Figure 3.55). We attribute this to the cumulative effect of sediment release down the river system.

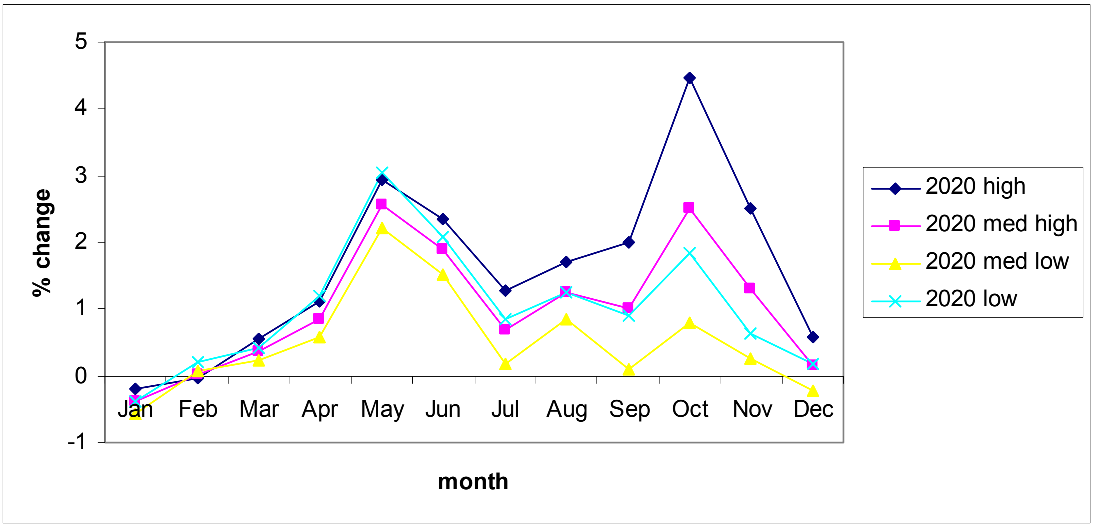

Figure 3.51 Percentage changes in monthly sediment concentrations in the upper Lambourn for the 2020s

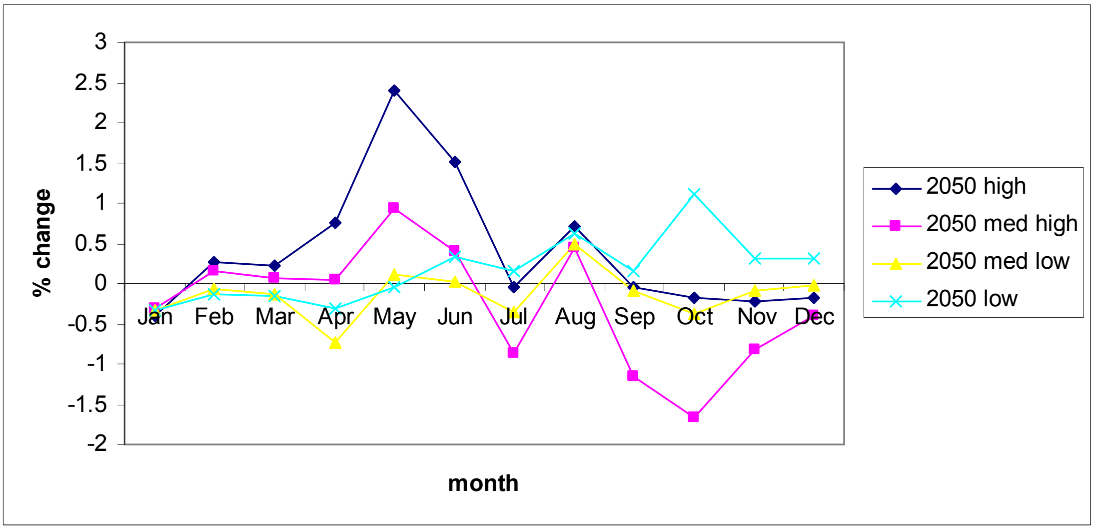

Figure 3.52 Percentage changes in monthly sediment concentrations in the upper Lambourn for the 2050s

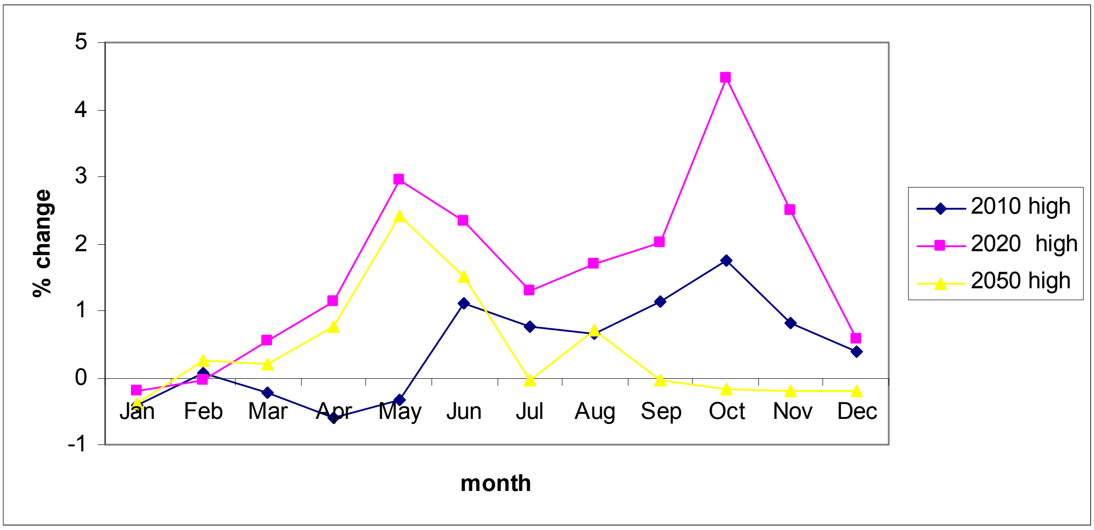

Figure 3.53 Percentage changes in monthly sediment concentrations in the upper Lambourn for the high Scenarios.

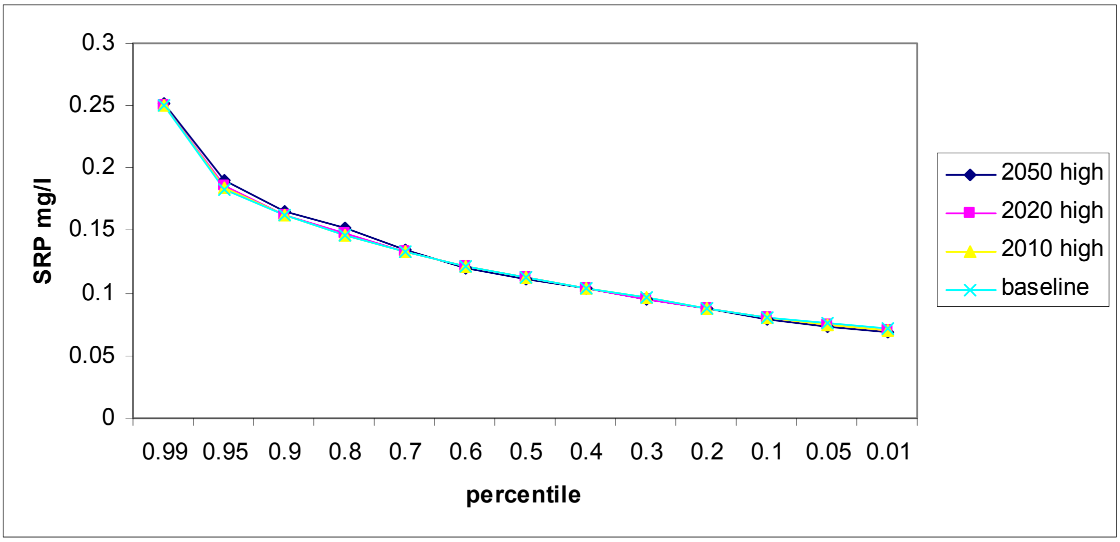

Figure 3.54 Sediment concentrations as percentiles in the upper reaches of the Lambourn for the high scenarios.

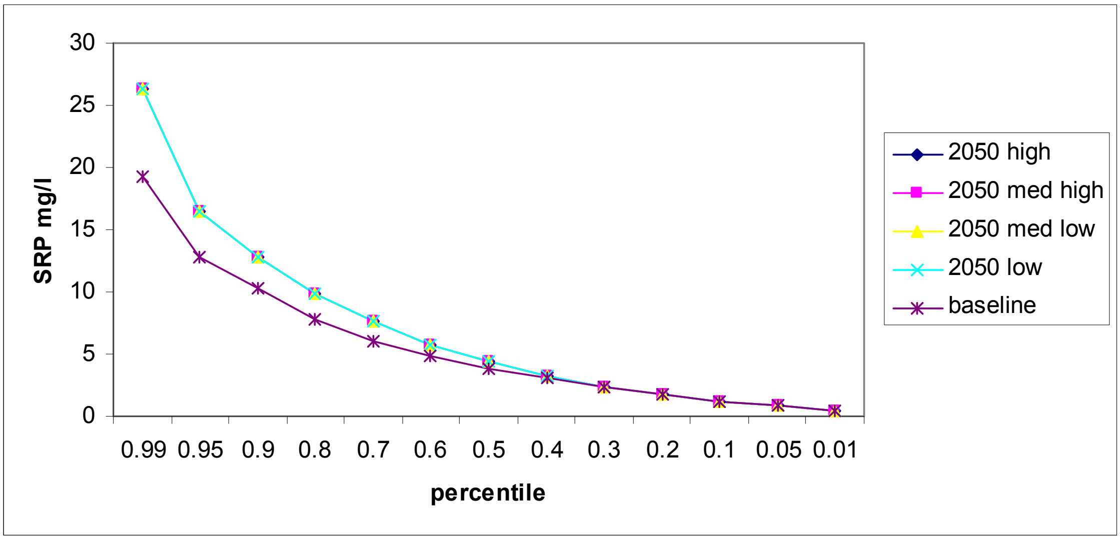

Figure 3.55 Sediment concentrations as percentiles in the lower reach of the Lambourn for the 2050s

3.6 Long term simulations

Long term scenario results in the Lambourn and the Kennet

We also ran the INCA-P model using the long term transient downscaled A2 scenario utilized in an earlier study of the River Kennet (Wilby *et al*. 2006, Whitehead *et al*. 2006). The input daily data are shown in Figure 3.56 for the period 1961- 2100. There appear to be significant increases in soil moisture deficit at certain times, reflecting enhanced summer drought conditions in the future.

As noted above, this can be important because it affects the recovery of the stream following a long dry summer and it also affects soil processes such as phosphorus absorption or release, nitrogen mineralisation, nitrification and denitrification.

Figure 3.57 shows the daily time series of flow, phosphorus, sediment, macrophyte and epiphytes generated by the model over the period 1961-2100 for the River Lambourn.

River flow declines marginally in response to drier conditions and phosphorus concentrations rise due to these declining flows and reduced dilution of effluents in the Lambourn.

What is remarkable is the stable nature of macrophyte and epiphyte dynamics simulated for the Lambourn. The biomass levels follow the patterns seen at present, although at certain times the annual patterns of growth and death change. For example, there appears to be increased epiphytic growth towards the end of the period, perhaps reflecting increased phosphorus concentrations in the stream.

The parameters used to set up this aspect of the model are those used by Wade *et al.* (2002d) and the stable populations obtained indicate that the parameters chosen are reasonable for the Lambourn. There is a wide range of parameter values for macrophyte and epiphyte growth in the literature (Wade *et al.* 2002d), so selecting appropriate rates is difficult and subject to considerable uncertainty.

Figure 3.58 shows the percentage changes in phosphorus in the River Lambourn for different time periods, according to the long term model. Increased concentrations are evident by the 2050s. This change is also reflected in the percentile distribution of the predicted data (Figure 3.59), where we find a significant increase in percentile values over time, relative to the baseline period. This result contradicts the delta change method results and the reason for this is not clear. It suggests that the changes in phosphorus concentration are more significant in the long term transient scenario. This may be due to the build up of altered soil moisture conditions, or the slow ramping up of temperature. This area clearly needs further work.

Figure 3.60 and 3.61 show monthly macrophyte and epiphyte biomass values for the three time periods. Although there are no major biomass increases, there is a shift in production to later months of the year, which probably reflects changes in flow patterns and phosphorus conditions.

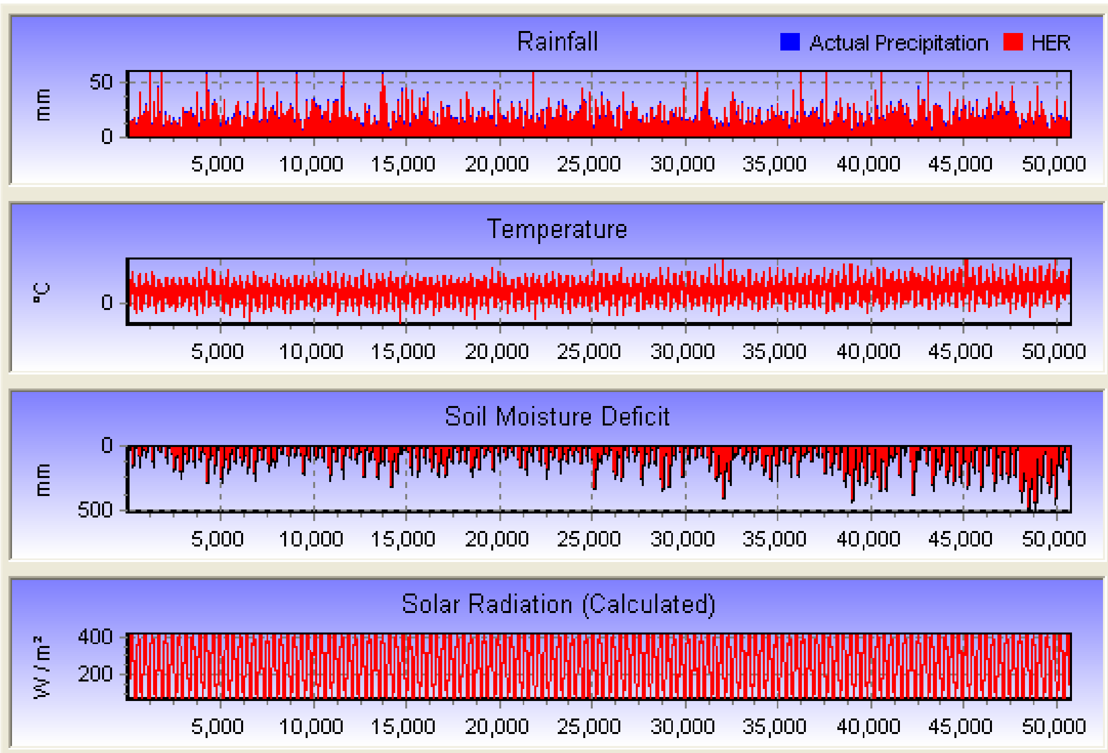

Figure 3.56 Daily rainfall, temperature, soil moisture deficit and solar radiation from 1961 in the River Kennet and Lambourn catchment, for the A2 scenario

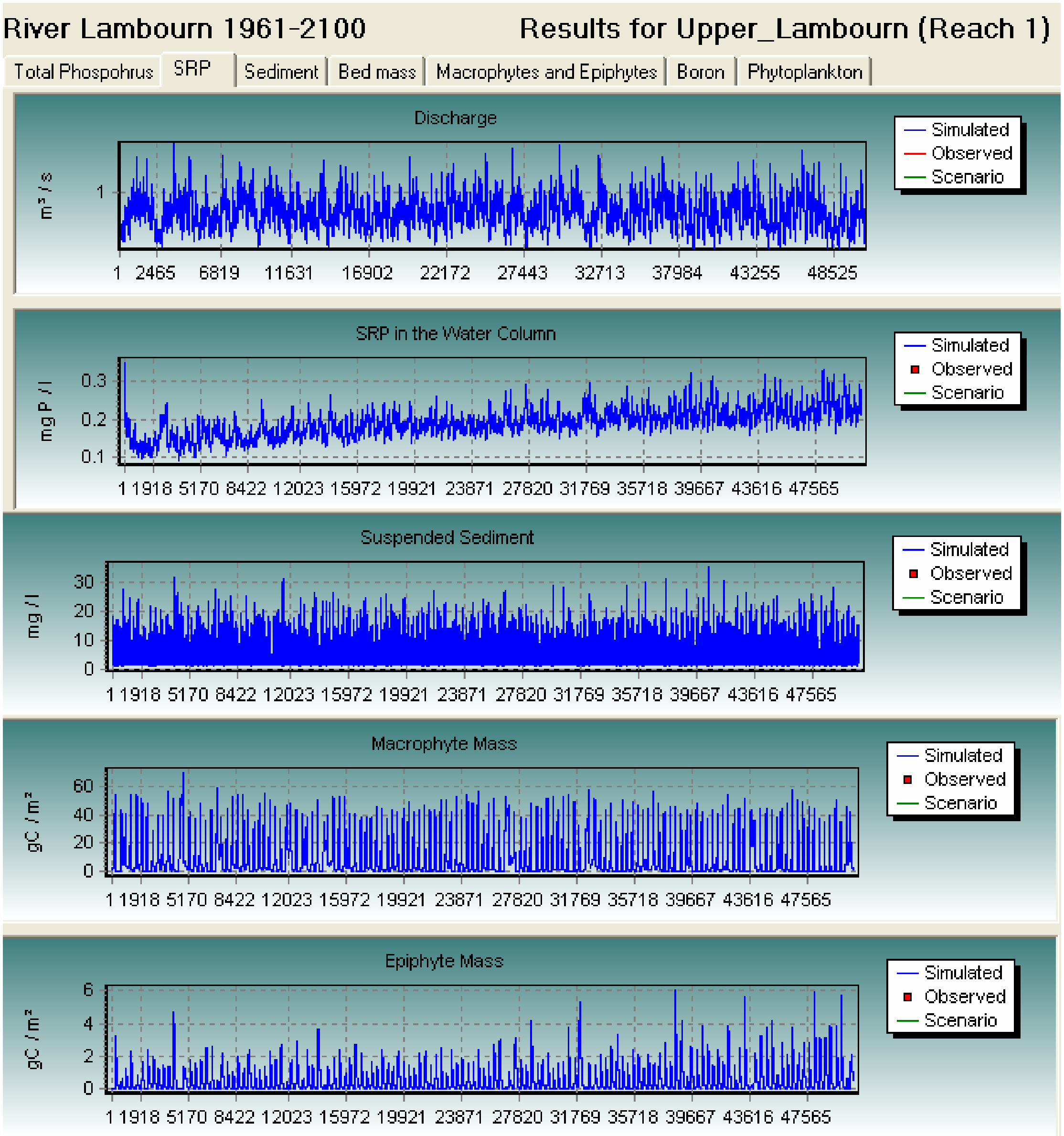

Figure 3.57 Simulated flow, phosphorus, sediment, macrophyte and epiphyte biomass in the Lambourn, for the A2 scenario 1961-2100, x axis shows day number from 1st January, 1961

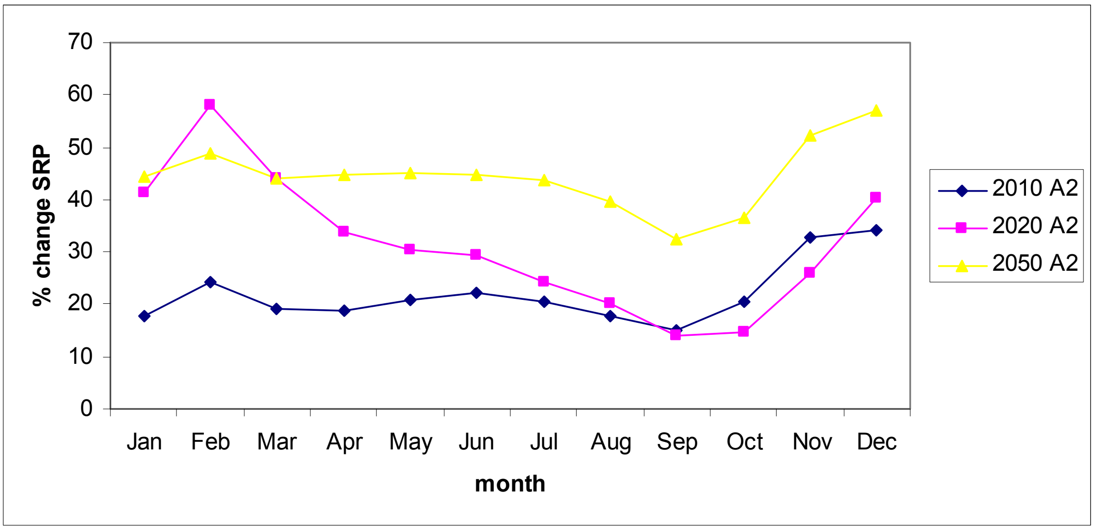

Figure 3.58 River Lambourn monthly percentage changes in phosphorus for the 2010s, 2020s and 2050s, using the long term GCM A2 scenario outputs

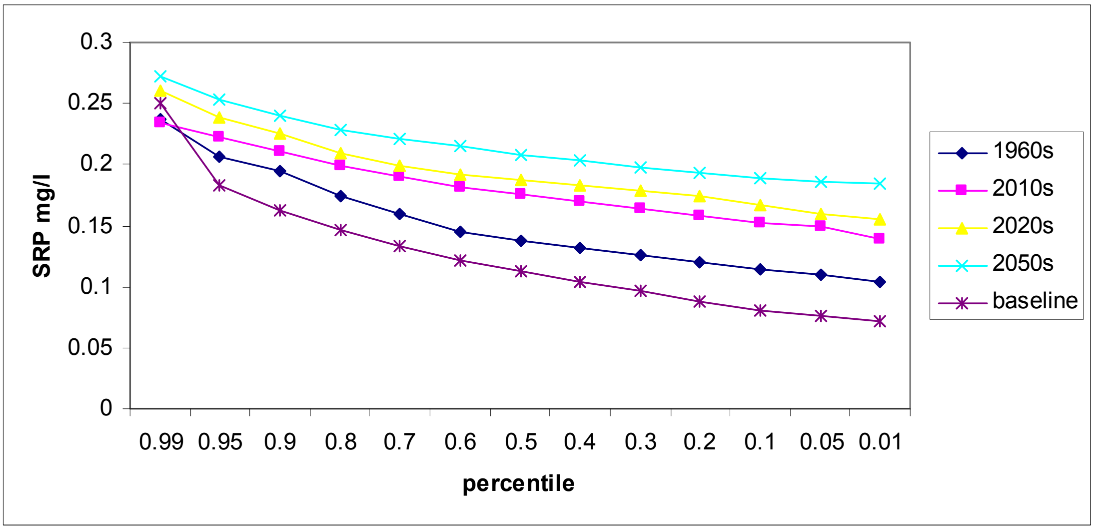

Figure 3.59 Percentile distributions for phosphorus in the Lambourn assuming the long term A2 scenario

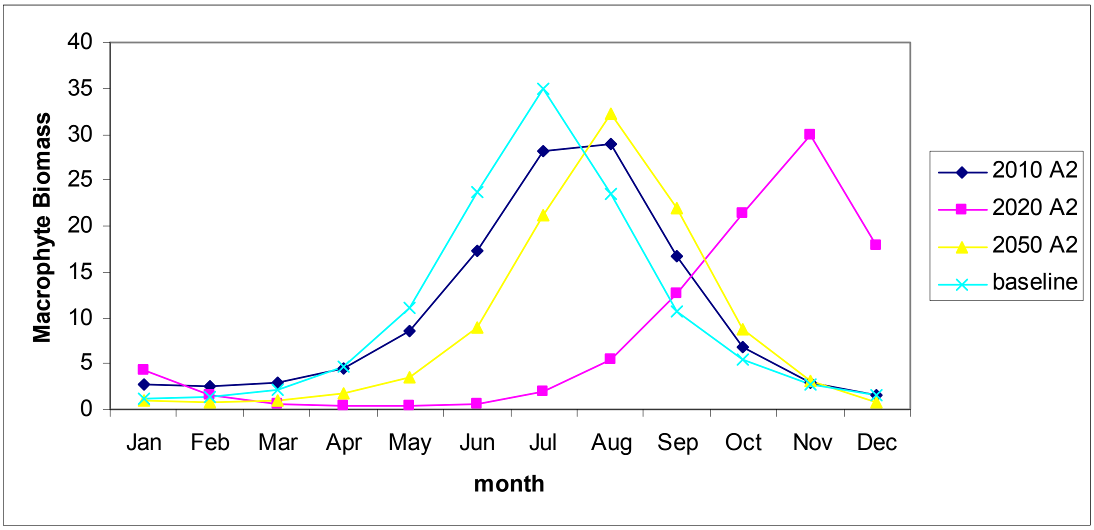

Figure 3.60 Lambourn macrophyte biomass (gC/m^2). Monthly statistics over a range of time periods for the A2 long term scenario.

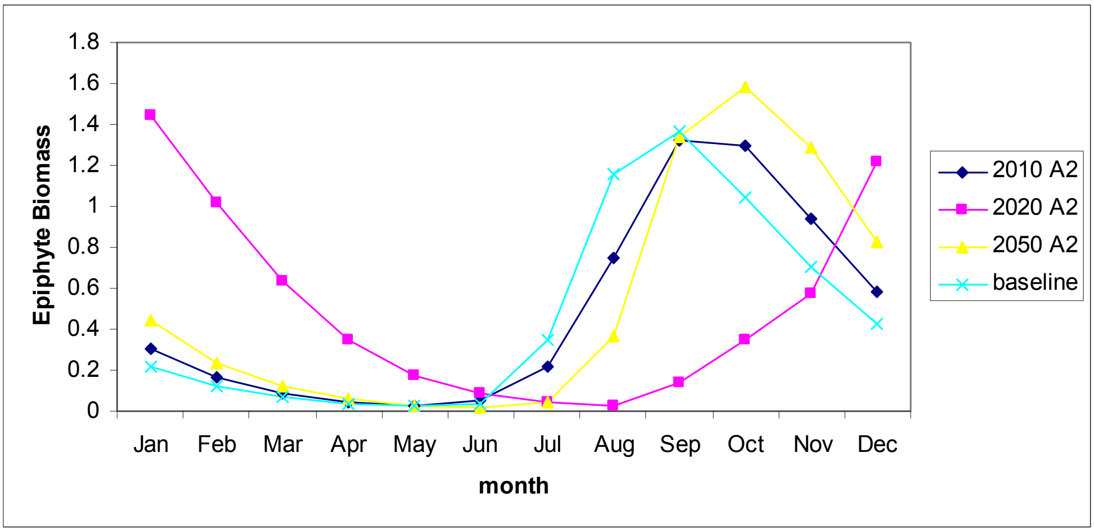

Figure 3.61 Lambourn epiphyte biomass (gC/m^2). Monthly statistics over a range of time periods for the A2 long term scenario.

4. Uncertainty considerations

All model predictions are uncertain to some degree, but quantifying and reducing this uncertainty are priorities for many climate change studies. The sources of uncertainty are extensive (Jakeman *et al*. 1993, McIntyre *et al*. 2005, New and Hulme 2000, Wilby 2005). As illustrated in Table 4.1, there are major sources of error at every stage of climate change studies (Jenkins and Lowe 2003, Kay *et al*. 2006, Prudholme *et al*. 2005, Wilby and Harris, 2006). Added to these are uncertainties associated with water quality modelling structures and parameters, and the uncertainty associated with water quality observations.

Table 4.1 Upper and lower percentage error ranges in GCM modelling and downscaling (Kay *et al*. 2006)

Source of error	Lower bound %	Upper bound %
Natural variability	-34	17
Emissions	-14	-9
GCM structure	-13	41
GCM initial conditions	-25	-5
Downscaling	-22	-8

Since catchment hydrochemical models can include many parameters, some of which are very imperfectly known, one might think that their overall uncertainty will be so large as to render them useless. But this is not necessarily the case. For example, the calculation of critical loads, deposition thresholds used in pollution control policy, involves models that combine 10 to 20 uncertain parameters. However, uncertainty in the calculated critical loads is typically less than the uncertainty in any of the input parameters (see Skeffington *et al*. 2007, for example).

This behaviour is to be expected where parameters are independent and subject mainly to random errors. A process of averaging occurs in the model's calculations. To some this is counterintuitive, but it suggests that complex behaviour patterns can give rise to surprisingly low variability in model outputs. This is an important consideration in mathematical models generally. When looking at the combined effect of many mechanisms, it is the way the impacts are correlated that is important in deciding whether the overall impact is important.

Various techniques for correctly estimating uncertainty in environmental models are currently being applied (Jackson *et al*. 2007) within the Eurolimpacs Project. One of these is Monte Carlo analysis. Monte Carlo analysis allows the correct combination of errors and the correct calculation of uncertainty in the estimates produced by the model (Whitehead and Young, 1979). It has been applied to all the INCA models to identify key parameters and place confidence limits on model predictions (Wade *et al.* 2002b), Cox and Whitehead 2004, Jarritt and Lawrence 2007, Wilby 2005). The technique is also the basis of the Environment Agency's effluent discharge and consent setting procedure (Warn and Brew, 1980), and this procedure has a long track record of providing a coherent and unbiased methodology for England and Wales.

The results from the present project give an idea of the kind of variability that can be expected from water quality climate change simulations. For example, Figure 4.1 shows the error bounds from the 2050s scenario for SRP on the River Lambourn based on a statistical analysis of the monthly statistics. The 95 per cent confidence bounds show a relatively narrow band of uncertainty during lower flow summer months when conditions are more stable. However, in winter months the uncertainty increases as flow condition become more variable and storms generate more runoff of water and nutrients. These uncertainty bounds are of comparable magnitude to the sources of variability in the climate change signal, shown in Table 4.1.

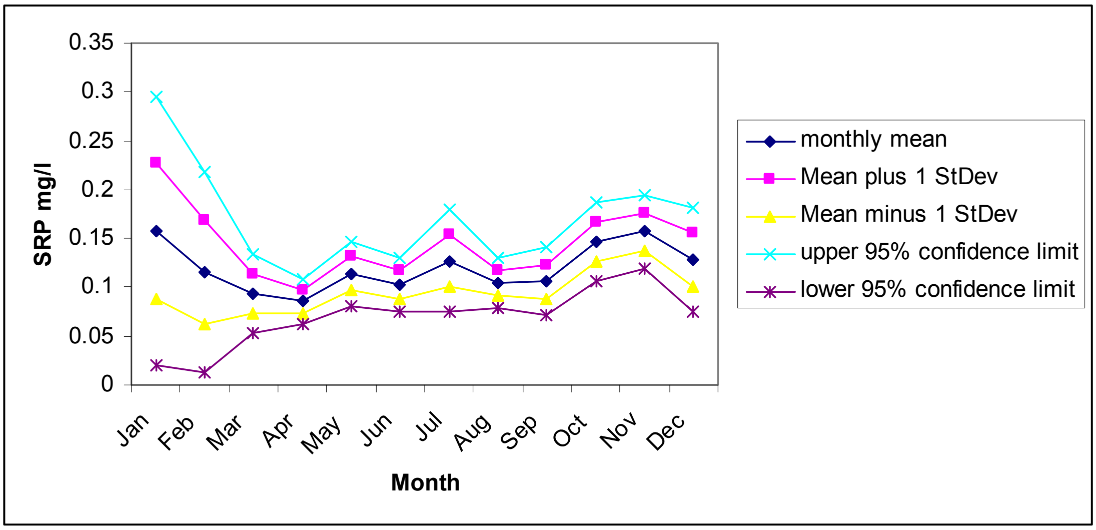

Figure 4.1 Phosphorus concentrations simulated for the 2050s in the Lambourn together with uncertainty bounds

5. Conclusions and future research

The results from this project are complex, as might be expected, but consistent patterns are obtained and it is possible to make some statements about the likely outcomes.

In the lowland southern River Lambourn, declining concentrations of Soluble Reactive Phosphorus (SRP) were predicted for winter months and increasing concentrations during summer and autumn months, caused by lower flows and hence reduced dilution. The rate of change increases over time and also increases with the severity of the emission scenario.

Sediments in the Lambourn are predicted to increase throughout the year but are particularly high in autumn after dry summers. The build up of sediments over dry periods followed by increased autumn flows seems the main mechanism here. Again, rate of change increases over time and also with the severity of the emission scenario.

Results from the urbanized midlands River Tame show similar increased SRP levels in summer but higher increases in winter due to diffuse urban runoff. The rate of change increases over time and also with the severity of the emission scenario.

In the western and rural River Lugg (a tributary of the River Wye) the SRP is predicted to decrease in winter but increase in summer months. The rate of change increases over time and also with the severity of the emission scenario.

Nitrate levels in the northern River Tweed increase in winter in upland headwaters as organic nitrogen is released, and decrease in summer months due to drought and increased denitrification in the river. The lower Tweed also shows increased nitrates in winter but the highest increases are in summer. This difference is due to the change in land use to agriculture and point source discharges in the lower reaches of the Tweed. The rate of change increases over time and also with the severity of the emission scenario.

Nitrate levels in the south western River Tamar have a similar response to those in the Tweed, with higher nitrates in winter and lower summer nitrates in the upper reaches. In the lower reaches of the river, nitrates increase both in summer and in winter. The rate of change increases over time and also with the severity of the emission scenario.

The macrophyte and epiphyte dynamics within rivers, according to the modelling, show significant interaction with each other. They suggest that increased drought could create problems by increasing nutrient concentrations and stimulating epiphytic growth at the expense of macrophyte growth.

Overall, water quality impacts are different depending on geographic location and water body location within a catchment. However, we emphasize that the changes predicted by this modeling project are not large and the statistical analysis of the modelling results using the delta change method of downscaling shows relatively minor changes in probability distributions. However, simulations using the statistical downscaling procedure do show larger changes in probability distributions.

We also undertook a comprehensive literature review on the effects of climate change on water quality. This has been prepared as a paper, appended to this report. Although there has been little study in the area, some clear points emerge.

Future Research

Climate change studies, especially in relation to water quality and ecology, are at fairly early stages and their outcomes are subject to considerable uncertainty. However, understanding the processes and mechanisms controlling water quality and ecology are essential for sustaining potable water supplies and conserving river systems. This project has identified a number of options for further research:

1 Apply statistical downscaling to a larger set of GCM scenarios, to further investigate transient effects of climate change on water quality across a wider set of catchments. Compare the results to those from the delta change method of downscaling. Also, analyse a wider range of scenarios, such as those determined by New *et al*. (2007) or the new UK Climate Impacts Programme 2008 scenarios.

2 Further develop and test the macrophyte-epiphyte model, to evaluate the overall impacts of the processes controlling behaviour, the thresholds of water quality that may control growth effects and the potential impacts of climate change.

3 Develop the phytoplankton model in INCA to help assess the overall impacts of climate change on algal populations.

4 Evaluate the apparent relationship between nitrate and biodiversity found in microcosm experiments (James *et al.* 2005). Combine this with INCA to predict possible effects on biodiversity of climate change and measures to control nitrate under the Nitrate Directive.

5 Refine the QUASAR river model (Whitehead *et al*. 1997) for DO-BOD and algae (or incorporate DO-BOD equations within INCA) to explore the joint probability of low DO events and potential fish kills.

6 Further analyse the INCA model simulations to fully evaluate the uncertainty inherent in water quality predictions, making use of the Monte Carlo tool developed within Eurolimpacs.

7 Investigate how the INCA-type functionality could be run on the national scale to evaluate national policies in terms of impact, costs and benefits.

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Abbreviations

The following abbreviations and acronyms are used frequently in this report:

DO-BOD	Dissolved oxygen - biological oxygen demand
GCM	Global Climate Model OR General Circulation Model
HER	Hydrologically effective rainfall
INCA	Integrated Catchment model
PET	Potential evapotranspiration
QUASAR	Quality Simulation Along Rivers (a water quality model)
SRES	Special Report on Emissions Scenarios
SRP	Soluble reactive phosphorus
UKCIP	UK Climate Impacts Programme
UKWIR	UK Water Industry Research

APPENDIX 1

A REVIEW OF THE POTENTIAL IMPACTS

OF CLIMATE CHANGE ON SURFACE WATER QUALITY

By

P.G.Whitehead[1], R. L. Wilby[2], R.W. Battarbee[3], M. Kernan[3], A. Wade[1]

[1]Aquatic Environments Research Centre, University of Reading, Reading RG6 6AB,

[2]Lancaster Environnent Centre, Lancaster University, Lancaster, LA1 4YQ

[3]Environmental Change Research Centre, University College London, Gower St., London WC1E 6BT

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Correspondence:

Professor Paul Whitehead

Email:p.g.whitehead@reading.ac.uk

ABSTRACT

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

KEY WORDS: Climate Change, Water Quality, Rivers, Catchments, Lakes, Estuaries, Ecology, Hydrochemistry

INTRODUCTION

The 2007 Conference of the Parties to the United Nations Framework Convention on Climate Change meeting in Bali and the latest IPCC Report (2007) confirmed the consensus amongst scientists and policy makers that human induced global climate change is now occurring. However, there is less certainty about the magnitude of future temperature changes and how these will drive precipitation, evaporation and hydrology at regional scales. Nonetheless, climate model scenarios provide the best available information for assessing future impacts of climate change on the water quality and ecology of surface water bodies (Kundzewicz et al., 2007).

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

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

This paper provides a review of water quality seen through the lens of anticipated impacts in the UK. The material is organised as follows. Firstly, we review potential impacts on *surface* water bodies such as rivers and lakes in terms of their hydrological regimes, hydromorphology, nutrient status, mobilisation of toxic substances and acidification potential. Second, we review long-term changes in the water quality for specific aspects of freshwater environments such as estuaries and urban areas. These sections are followed by a critique of recent water quality model developments, including the treatment of uncertainty. Finally, the indirect consequences to water quality of wider climate change policies affecting land and water management or emission reductions are considered.

HYDROLOGICY, WATER QUALITY AND THERMAL REGIMES

A review of surface water quality cannot be undertaken without considering changes in hydrological regimes. The UKCIP02 scenarios (Hulme et al., 2002) suggest that winter precipitation in the UK could increase by 10-20% for a low emissions scenario and by 15-35% for a high emissions scenario by the 2080s. The largest changes are predicted in the south and east of England and the smallest in north-west Scotland. In the summer, the pattern is reversed and almost the whole of the UK could become drier, with precipitation decreases of up to 35% under a low emissions scenario and 50% or more under a high emissions scenario. Marsh (2007) has shown that summer precipitation has already fallen to some extent (**Figure 1**). Furthermore, the frequency of extreme events is also predicted to increase with two-year (return period) winter precipitation event intensities estimated to become between 5% (low emissions) and 20% (high emissions) heavier by the 2080s. These changes in precipitation have been used to simulate changes in flow across the UK (Arnell, 2003, Limbrick et al, 2002). More recently, Romanowicz et al (2006) modelled changes in river flow for a range of catchments, under different climate model projections. They conclude that by the 2020s flows in winter could increase by between 4% and 9% and that summer flows could decrease on average by 11% but this could range between 1% to 32% depending on the catchment location, land-use, soils, geology, and model uncertainty.

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

Reduced dilution effects will also impact organic pollutant concentrations with increased biological oxygen demand (BOD) and hence lower dissolved oxygen (DO) concentrations in rivers. Cox and Whitehead (2008) show that DO in the Thames will be affected in the 2080s under a range of UKCIP scenarios by enhanced BOD and by direct effects of temperature which reduces the saturation concentration for DO. **Figure 3** shows the effects are not large and this would not be an issue for ecosystems under normal circumstances. However, in the Thames, occasional algal blooms in some summers are a feature of the river ecology the frequency and intensity of these may increase. When an algal bloom occurs there are large diurnal variations in DO and, on poor quality rivers, low oxygen levels can be exacerbated by pollution events during summer low flow conditions.

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

The most immediate reaction to climate change is expected to be in river and lake water temperatures (Hammond and Pryce, 2007; Hassan et al., 1998). River water temperatures are in close equilibrium with air temperature and as air temperatures rise so will river temperatures. There has already been a 1-3°C temperature rise over the past one hundred years in large European rivers such as the River Rhine and the River Danube (EEA Technical Report, 2007). Small streams have shown an increase in winter temperature maxima in Scotland (Langan et al, 2001), and there have been large increases in temperature reported for water courses in Switzerland at all altitudes (Hari et al., 2006). There have been two sudden shifts in river temperatures in 1988 and 2002 following changes in air temperature. Abrupt water temperature rises could have important implications for some aquatic organisms, if species are unable to adapt at the same pace. Furthermore, a recent study of long-term trends in UK surface waters revealed marked regional variations, with the greatest rates of change in the south and east (**Figure 4**).

Most chemical reactions and bacteriological processes run faster at higher temperatures. In addition, temperature controls the growth rates of phytoplankton, macrophytes and epiphytes making freshwater ecosystems sensitive to rising temperatures (Whitehead et al, 1984, Wade et al, 2002). Water temperatures also regulate the behaviour of aquatic organisms, such as fish migration, and the timing of emergence and abundance of insect populations at different life-cycle stages (e.g., Durance and Ormerod, 2007; Davidson and Hazelwood, 2005). This has implications in that meeting WFD objectives and reference conditions for the restoration and improvement of the ecology of streams could be more difficult with future climate change (Wilby et al., 2006).

HYDROMORPHOLOGY AND ECOLOGY

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

In many parts of Europe, hydromorphology is a key factor controlling ecosystems behaviour. Alterations to river forms through channel straightening, loss of connectivity with flood plains, weir and dam construction, and loss of riparian vegetation also impact on river ecology. Under the WFD there is a requirement to reverse some of these changes and restore the ecology of rivers and lakes towards their natural states. However, climate change may act against restoration, making it difficult, if not impossible, to return to the previous ecosystem status (Orr and Walsh, 2006) (**Figure 5**). Changes in climate could affect sediment transfer, channel morphology and inundation frequency, thereby altering ecosystems at both catchment and habitat scales (Verdonschot, 2000). The impact of low flows on biotic communities has been studied extensively in the River Lambourn (Wright et al., 1982). In this case, drought has a deleterious effect on aquatic ecology with *Ranunculus* being smothered by epiphytic algae (Wade et al, 2002). Drought also significantly damages macro-invertebrates although recovery can be fast (Ladle and Bass, 1982).

Extreme events could have significant impacts on upland rivers, releasing higher concentrations of sediments by erosion and resuspension, thereby creating new or disturbed habitats downstream. However, this can be beneficial to upland stream ecology with natural formations of pool and riffle sequences and a wider range of habitats such as meandering side channels, larger dead zone areas and deeper sediment zones to support aquatic life. An extensive study in German rivers shows that habitat restoration may be enhanced by the effects of a more variable flow regime (Hering et al, 2008).

The PRINCE project specifically addressed the potential impacts of climate change for selected UK freshwater ecosystems (Conlan et al., 2007). It was shown that changes in climate could influence aquatic ecosystems through episodic pulsed effects (i.e., changes in the frequency, duration and magnitude of extreme events) and by progressive change in ambient conditions. Many freshwater species are sensitive to the water temperature regime as they are cold-blooded and many have a limited range of thermal tolerance. Thus, changes in the temperature regime could have significant effects on the life cycle of a wide variety of aquatic organisms. Temperature effects could combine with changes in water velocity and DO to affect life cycles and inter-relationships of organisms such as invertebrates, amphibians, fishes and birds. In addition, there may be impacts on dispersal or migratory patterns across ecosystems, for example, between marine systems and freshwaters by long-distance migrants (Atlantic salmon, eel, shad), or across watersheds during inter-basin dispersal flights by invertebrates; and via the introduction, survival and population dynamics of exotic organisms.

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

In summary, climate change could affect: 1) the magnitude, frequency (return period), timing (seasonality), variability (averages and extremes) and direction of predicted changes of flow and water quality; and 2) the sensitivity and resilience of the ecosystem, habitat and/or species to those changes. Habitats that are already in vulnerable stream sections such as headwaters, ditches and ephemeral ponds, could be the most sensitive to changing climatic conditions.

NUTRIENTS AND EUTROPHICATION

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

Meanwhile, nutrients released from agriculture or from sewage treatment works could be less diluted due to the reduced flows in summer. Whitehead et al. (2006) simulated these combined effects on the River Kennet in terms of projected nitrate and ammonia concentrations. As shown in **Figure 6**, nitrate concentration increases over time as higher temperatures increase soil mineralisation. This is particularly significant under high flow conditions following a drought. Whilst this is a theoretical modelling exercise, similar responses have been observed in the field. For example, **Figure 7** shows nitrate concentrations in the Thames at Teddington at the termination of the 1976 drought (Whitehead, 1990). Nitrate-N concentrations rose from 4 mg/l to 18 mg/l as nitrates were flushed from the Thames catchment. Increased frequency of flushing events is expected under some climate change scenarios and this extra nitrogen could enhance eutrophication in receiving water bodies.

As part of Euro-limpacs a set of mesocosm (small artificial lake) experiments have been established to simulate climate change impacts (Moss et al, 2003, 2004). These controlled environments show that growing seasons are extended by increased temperatures as are growth rates of algae and zooplankton. Oxygen concentrations fall as temperature reduces saturation levels and increased nutrient levels enhance respiration. This could, in turn, lead to increased risks of fish deaths even for tolerant species. The ecology of the mesocosms changed significantly with exotic species out-competing native species. Future projections suggest that oxygen levels may decline and cyanobacteria blooms may become more extensive. The findings from these mesocosm experiments could have implications for lowland rivers, as well as for shallow lakes, where water levels are controlled by weirs and where there can be long residence times in summer. Van Doorslaer et al (2007) have also shown that zooplankton evolution can occur over relatively few generations, raising the possibility that ecosystems might maintain their current structure and functionality by adapting to temperatures increases. Whilst one or two species might achieve sufficient rates of change, it seems unlikely that whole ecosystems could evolve in parallel.

Lake ecosystems respond to changes in inflow volumes, water quality and water temperature as well as to changes in thermocline behaviour and residence times (George et al., 2007). Numerous studies have highlighted the links between the winter North Atlantic Oscillation (NAO) and coherent responses in lake water temperature, ice conditions, and spring plankton phenology across Europe (e.g., Blenckner et al., 2007). Higher wind speeds could reduce lake stability, and enhance mixing of nutrients (George et al, 2007). Conversely, higher temperatures lengthen the period of thermal stratification and deepen the thermocline (Hassan et al., 1998). Shallow lakes may be particularly susceptible to climate induced warming, changes in seasonal mean residence times (George et al., 2007) and nutrient loads (Carvalho and Kirika, 2003). Climate impact assessments typically show associated changes in ecosystem functioning (**Table 1**) such as earlier blooms (**Figure 8**) or increased concentrations of planktonic algae (e.g., Arheimer et al., 2005; Komatsu et al., 2007). A sensitivity study of phytoplankton in Loch Leven, Scotland showed larger responses to increases in phosphorous loads than water temperature (Elliott and May, 2008).

TOXIC SUBSTANCES

Although many of the most toxic substances introduced into the environment by human activity have been banned or restricted in use, many persist, especially in soils and sediments, and either remains in contact with food chains or can be re-mobilised and taken up by aquatic biota (Catalan et al., 2004; Vives et al., 2005). High levels of metals (such as mercury, Hg and lead, Pb) and persistent organic pollutants (PCBs) are present in the tissue of freshwater fish in arctic and alpine lakes (Grimalt et al., 2001; Vives et al., 2004a). This attests to the mobility and transport of these substances in the atmosphere (Carrera et al., 2002) and their concentration in cold regions (Fernandez and Grimalt, 2003). Biomagnification within aquatic systems with long food chains can elevate concentrations in fish to lethal levels for human consumption. The major concern with respect to climate change is the extent to which toxic substances will be remobilised and cause additional contamination and biological uptake in arctic and alpine freshwater systems as water temperatures rise. Storm events and flooding might also increase soil and sediment erosion and lead to the re-mobilisation of metals and persistent organic compounds (Grimalt et al., 2004a; b, Rose et al., 2004). In the case of Hg, changing hydrology in Boreal forest soils may lead to the enhanced production of methyl mercury (Meili et al, 2003, Munthe, 2008).

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

In the UK, research on toxics has focused on upland lakes exposed to air pollutants from long range transport. In particular detailed studies at Lochnagar in Scotland has shown high concentrations of both trace metals and trace organic compounds in sediments (Yang et al. 2002, Rose et al. 2001) and fish (Rosseland et al. 2007), and Rose et al. (2004) have argued that increased storminess in future might cause the remobilisation of trace metals from catchment soils.

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

ACIDIFICATION AND DOC IN THE UPLANDS

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

Droughts can further exacerbate acidification by creating lower water tables, aerobic conditions and enhanced the oxidation of sulphur to sulphate (Dillon et al, 1997; Wilby, 1994). Acid anions are exported during subsequent storm events along with heavy metals (Tipping et al, 2003). Peat catchments are particularly vulnerable to climate change as they have significant stores of sulphur which could be released following summer droughts (Aherne et al, 2006). Nitrogen is another source of acidification in upland catchments as nitric acid is a strong acid anion that can be flushed after droughts (Adamson et al., 1998; Curtis et al., 2005, Whitehead et al, 2006; Wilby et al, 2006). The Norwegian CLIMEX study (Wright and Jenkins, 2001; Wright et al., 1998) showed significant mineralization of nitrogen following increases in temperature and CO_2 and this switched a small catchment from being a nitrogen sink to a nitrogen source.

Dissolved organic carbon (DOC) concentrations have doubled across the UK since the 1980s (Freeman et al 2001; Worrall et al, 2004; 2003; Evans et al 2001; 2005; Monteith et al, 2000) (**Figure 9**). The reasons for this are not yet clear. Freeman et al (2001) proposed a climate related enzyme latch mechanism that can release carbon from wetland soils. However, there is also strong circumstantial evidence that reduced sulphur deposition has an influence on DOC trends (Monteith et al, 2007). There is no doubt that hydrological conditions affect DOC export, with lower DOC concentrations during drought and greater concentrations during high flows (Hughes et al, 1993;

Clarke et al, 2005). Water colour is correlated with DOC so as DOC increases so will water colour. Whilst colour *per se* is not a public health issue, the chlorination processes at water treatment plants generate by-products such as trihalomethanes which are carcinogens (Chow et al, 2003). DOC modelling by Futter et al (2007; 2008) suggests that warmer, wetter climates could lead to higher levels of surface water DOC (**Figure 10**), but there remain large uncertainties due to the complex dynamics and biochemical processes controlling soil carbon flux.

ESTUARIES

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

To date, attention has been paid to monitoring the effects of rising water temperatures and nutrient loads on existing challenges such as eutrophication and severity of hypoxia. For example, Preston (2004) reports a warming of the Chesapeake Bay estuary by 0.8-1.1°C since the mid-20th century. Historically, winters with more frequent wet/warm weather patterns have been followed by greater phytoplankton biomass occurring later in the spring, covering a larger area and extending farther seaward in the estuary (Miller and Harding, 2007). Higher rates of primary production have also been observed in the Hudson River estuary during dry summers when freshwater discharges are lower and residence times, stratification and depth of the photic zone increase (Howarth et al., 2000). Conversely, more frequent peak discharges can increase scour, and alter sediment patterns and hence the morphology of estuarine mouths (e.g., Fox et al., 2001). Furthermore, Strufy et al. (2004) report a downstream shift of the salinity gradient and concentrations of nutrients (but greater total nutrient loads) associated with a threefold increase in annual discharge of the upper Schelde estuary, Belgium/Netherlands.

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

URBAN AREAS

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

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

The assumption of stationarity of rainfall properties for hydraulic infrastructure design is no longer tenable so retrofitting or upgrading of drainage capacity may be required (Denault et al., 2006). Adaptation responses must accommodate potential changes in both the frequency of extreme precipitation events of given magnitude, and changes in the severity of events with given return periods. For example, under a doubled CO_2 climate change scenario the 1 in 100-year flood becomes a 1 in 10-year event for Canberra, Australia (Schreider et al., 2000). However, detailed hydrological modelling for urban areas continues to be confounded by a lack of high-resolution (space and time) climate change scenarios (Grum et al., 2006; Wilby, 2007). Indeed, many climate models treat built areas as vegetated surfaces, whilst realistic simulations of localised, high-intensity, summer precipitation events have yet to be realised (see Fowler and Ekström, 2008). Water quality impacts associated with rising air and water temperatures may be more confidently projected. A beneficial consequence of rising temperatures could be improved performance of water treatment works.

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

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

MODELLING FRAMEWORKS

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

Applications

The most widely-used models in the Eurolimpacs project are the INCA suite. The first INCA model (INCA-N) was used to investigate the response of rivers to changes in nitrogen inputs and catchment nitrogen metabolism (Whitehead *et al.*, 1998a; b; Wade *et al.*, 2002). Since then the INCA framework has been extended to phosphorus (INCA-P); particulates (INCA-SED); dissolved organic carbon (INCA-C) and mercury (INCA-Hg). Another dynamic model being used in Eurolimpacs is the Mike11-TRANS model. This has a suite of ecological sub-models linked to a hydrodynamic model, and has been used in conjunction with the rainfall-runoff model NAM to simulate nitrogen fluxes in lowland Danish catchments (Anderson *et al.*, 2006). Another development of existing models is the use of a landscape-based mixing model (PEARLS) coupled with the acidification model MAGIC (Cosby et al, 1986) to simulate the recovery from acidification of the Conwy in North Wales, a large heterogeneous river basin (Evans *et al.*, 2005).

INCA-N was used in one of the first attempts to model outcomes of different adaptation strategies to address the rising nitrate concentrations under climate change (Whitehead *et al.*, 2006). Precipitation scenarios were downscaled from three GCMs for the River Kennet in Southern England for the period 1961 to 2100. The scenarios yield different effects in detail, but all show a general increase in nitrate concentration due to enhanced microbial activity. **Figure 11** shows the results of applying different interventions on this overall trend under one precipitation scenario (downscaled from HadCM3).

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

Model chains and integration

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

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

Model uncertainty

All model predictions are uncertain to some degree, but characterising and reducing this uncertainty is one of the priorities for Eurolimpacs. The sources of uncertainty are manifold (see Jakeman et al, 1993; McIntyre et al, 2005; New and Hulme, 2000; Wilby, 2005). Some catchment hydrochemical models can include more than 100 parameters, many of which are imperfectly known. This suggests that the overall uncertainty will be so large as to render projections useless, but this is not necessarily the case. The calculation of critical loads, deposition thresholds used in pollution control policy, involves models which are combinations of 10-20 uncertain parameters. However, rather surprisingly, translating input uncertainties into uncertainty in the outputs is typically less than the summed uncertainty in the input parameters (e.g. Skeffington *et al.*, 2007). This counterintuitive result suggests that complex behaviour patterns can reduce to surprisingly low variability in model outputs. A number of techniques for estimating uncertainty in environmental models are currently being explored by Eurolimpacs participants. For example, a Monte Carlo-based tool is being used to perform sensitivity analysis, to identify key parameters, and hence place confidence limits about model predictions (Wade *et al.* 2002; Cox and Whitehead, 2004; Wilby, 2005). At the same time less formal approaches are being used, such as applying different models to the same problem and comparing the results.

CONCLUDING REMARKS

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

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

As noted above, climate change is expected to cause more hydrological extremes with enhanced drought and flooding episodes (EEA, 2006, 2007; Kundzewicz et al., 2007). Adaptation to these impacts will have to be substantial and could include new water transfer schemes to transport water to drier areas and/or new reservoirs to improve security of supply. Resulting changes in flow regimes will influence the chemistry, hydro-morphology and ecology of regulated water bodies. Agricultural activity will have to adapt to longer growing seasons combined with reduced water availability, with new crops suited to drier, warmer conditions. Biofuel crops are already in demand and their intensive production could exacerbate water supply problems. Increased recycling of water in the UK, Central and Southern Europe is also an option but this implies lower volumes and final discharges of effluents with higher concentrations. Adaptation measures to restore polluted ecosystems might involve managing water levels in wetlands, lakes and rivers, or the designation of vegetation corridors and buffer zones, all of which potentially affect water quality.

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

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

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

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APPENDIX A

Climate change and freshwaters – knowledge gaps

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

Direct effects of climate change on freshwaters

- What will the effects of lower flows be on pollutant concentrations in rivers?
- What are the potential impacts of changing climate on catchment chemical fluxes?
- How will changing discharge regime affect stream ecology?
- How do changing air temperature and precipitation impact on discharge patterns in glacierised and non-glacierised upland river basins?
- How have freshwater biological communities responded to natural climate variability in the past?
- What effects will changing temperature and wind patterns have on the structure of lake water columns?
- How will changes in ice-cover duration, stratification and mixing regime affect lake biota?
- How will climate change affect the hydrology of marginal wetlands?
- What effects will changing hydrology and biogeochemical processes have on plant communities, nutrient dynamics and productivity in marginal wetlands?
- How will climate change affect the quality and quantity of dissolved organic carbon release from soils?
- What was the amplitude of natural variability of dissolved organic carbon concentrations in surface waters, prior to any impacts from greenhouse gas-forced climate change?

Interactions between climate and hydromorphological/land-use change

- How do climate, hydrology, land-use and morphology interact in space and time?
- How do these interactions affect aquatic ecosystems at the catchment scale?

- What effect will changing hydrological conditions (both directly and through morphological change) have on stream aquatic communities at the habitat scale?
- How will climate change affect mountain stream restoration?
- How will climate change affect channel morphology and stability in meandering lowland streams?

Climate change and eutrophication

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

Climate change and acidification

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

Climate change and toxic substances

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

Modelling climate change effects on surface water

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

Indicators of ecosystem health

- What chemical parameters are best suited as indicators of climate change?
- How can functional indicators be identified to address climate change impacts on wetlands, rivers and lakes?
- How can biological indicators of climate change be identified and can these be used to assess the response of communities to change?

- How can the different indicator types be linked to provide a common framework for rivers, lakes and wetlands?
- How can existing assessment and prediction methods for European freshwater systems be expanded and modified to address climate change?

Reference conditions and restoration strategies

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

Policy and management

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

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

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

Country	Study Area	Area km^2	River	Lake	Wetland	Additional Key Issues
Austria	Piburger See	2	✓	✓		Eutrophication
Finland	Savijoki	15.4	✓		✓	Eutrophication, Sed.
	Simojoki	3160	✓	✓	✓	Acid., N sat.
	Tueronjoki	439	✓	✓	✓	N sat, Eutrophication
France	Garonne-Adour	56500	✓	✓	✓	Eutrophication
Norway	Bjerkreim	685	✓	✓	✓	Acid., N sat., C
	Tovdalselva	1855	✓	✓	✓	N sat, Eutrophication
Romania	Lower Danube Wetland	210			✓	Eutrophication
Spain	La Tordera	124	✓		✓	Eutrophication
Sweden	Gårdsjön	0.005		✓		Acid., N sat., C
	Svartberget	0.5	✓	✓		Acid., Hg
UK	Conwy	590	✓		✓	Acid., Eutrophication, C
	Kennet	1030	✓		✓	Eutrophication, Sed.
	Lambourn	263	✓		✓	Eutrophication
	Endrick / Falloch	781	✓	✓		Eutrophication
	Tamar	917	✓		✓	Eutrophication
	Wye	4140	✓	✓		Acid., Eutrophication, C

Table 2 Principal areas used in the Eurolimpacs project for catchment-scale modelling. Issues: Acid. = Acidification C = Carbon, N sat. = N saturation,. Sed = sediment.

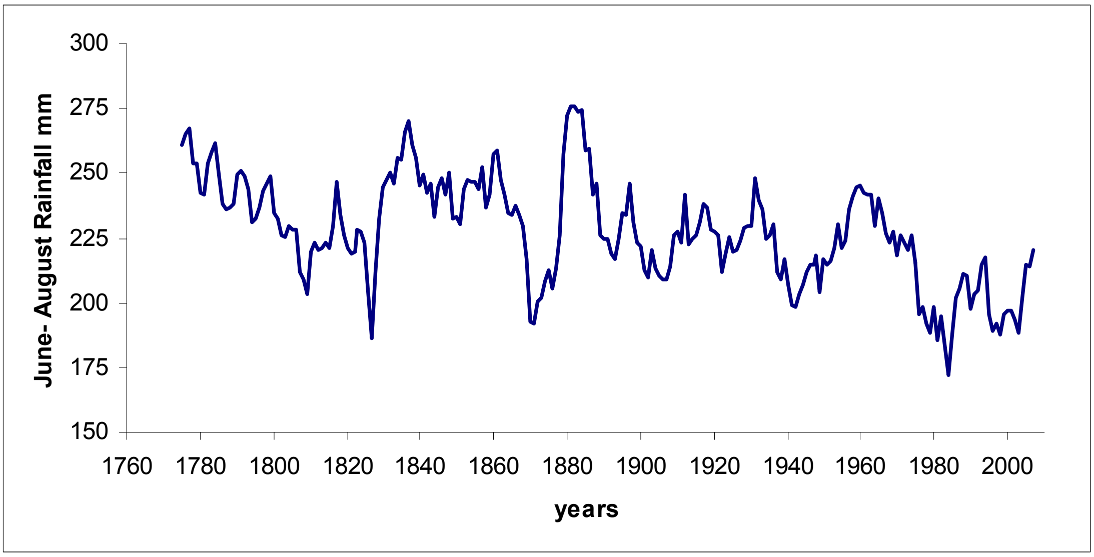

Figure 1 Summer (June-August) rainfall totals (mms) showing long term decline (Source: Marsh, 2007)

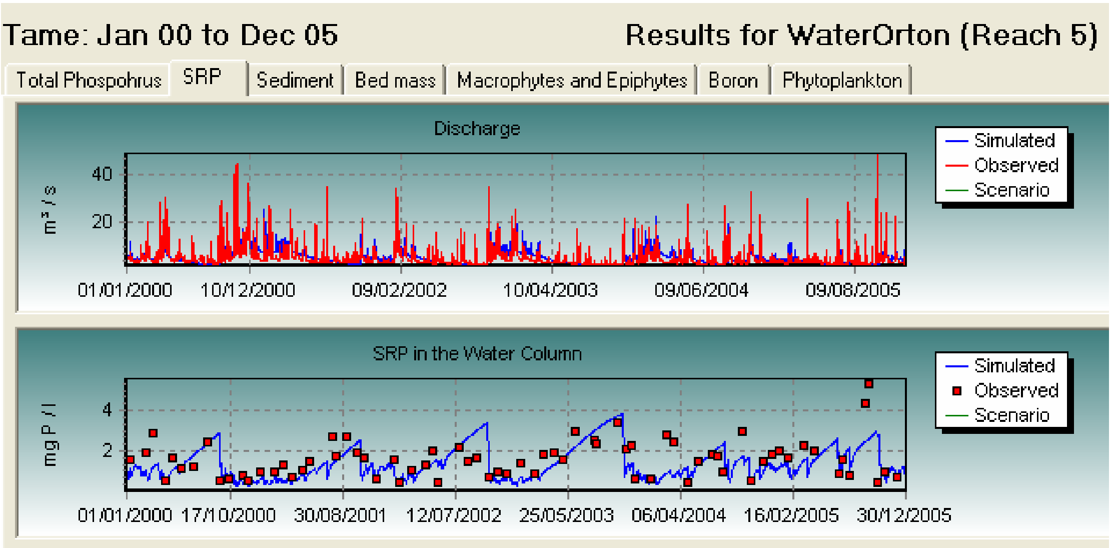

Figure 2 Flow and Phosphorus simulation and observed data for Water Orton on the River Tame in Birmingham.

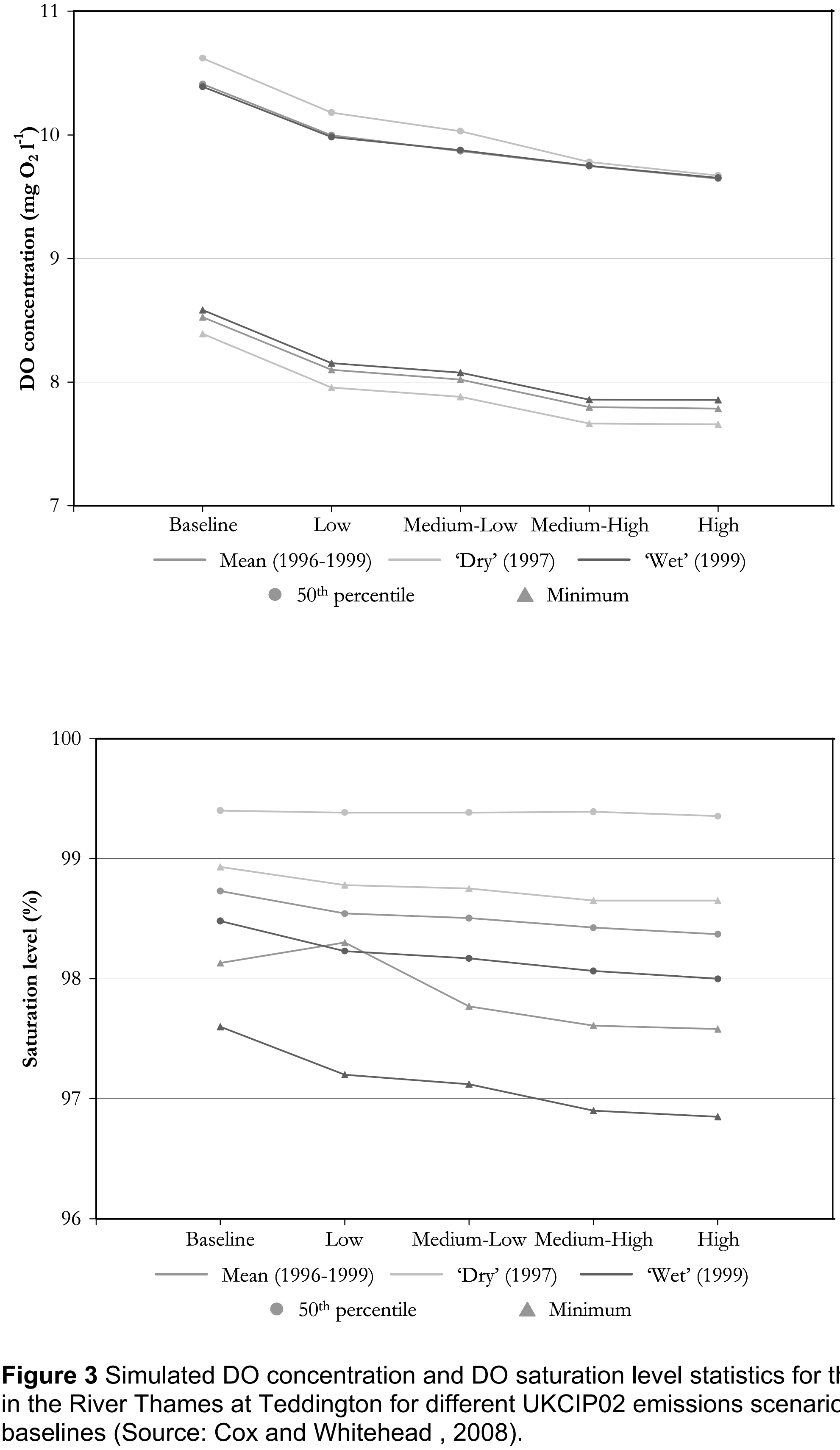

Figure 3 Simulated DO concentration and DO saturation level statistics for the 2080s in the River Thames at Teddington for different UKCIP02 emissions scenarios and flow baselines (Source: Cox and Whitehead , 2008).

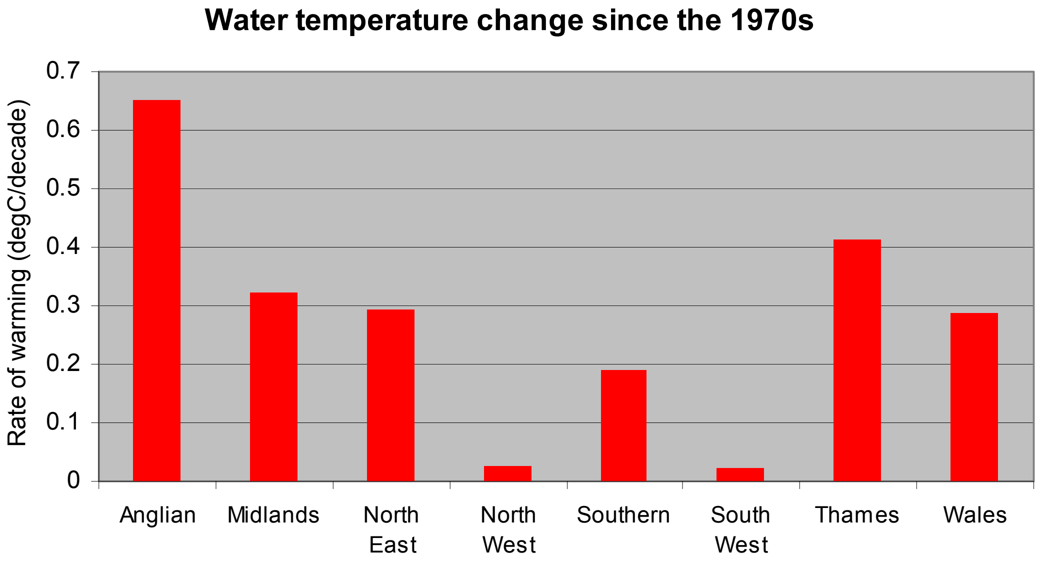

Figure 4 Regional variations in UK surface water temperatures since the 1970s. Data adapted from Hammond and Pryce (2007)

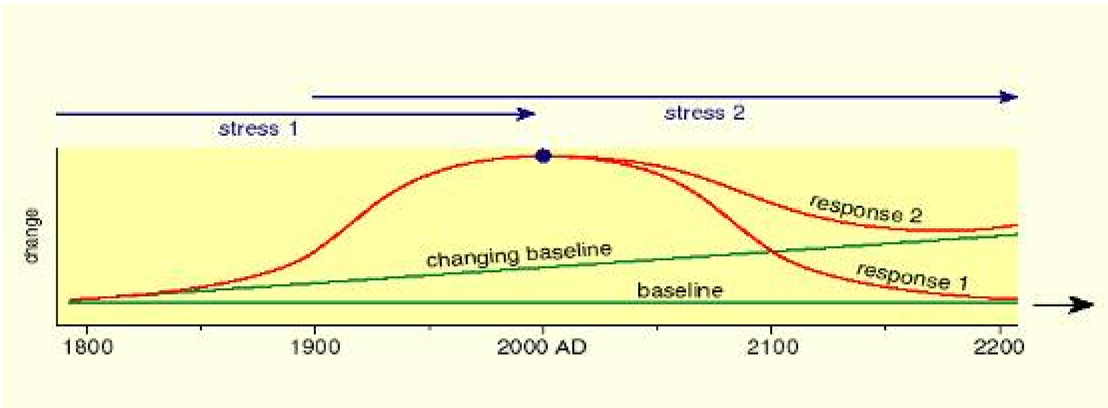

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Figure 5 Conceptual diagram illustrating ecosystem response to increasing and decreasing stresses in relation to climate change, expressed as a changing baseline (Source: Battarbee *et al.* 2005).

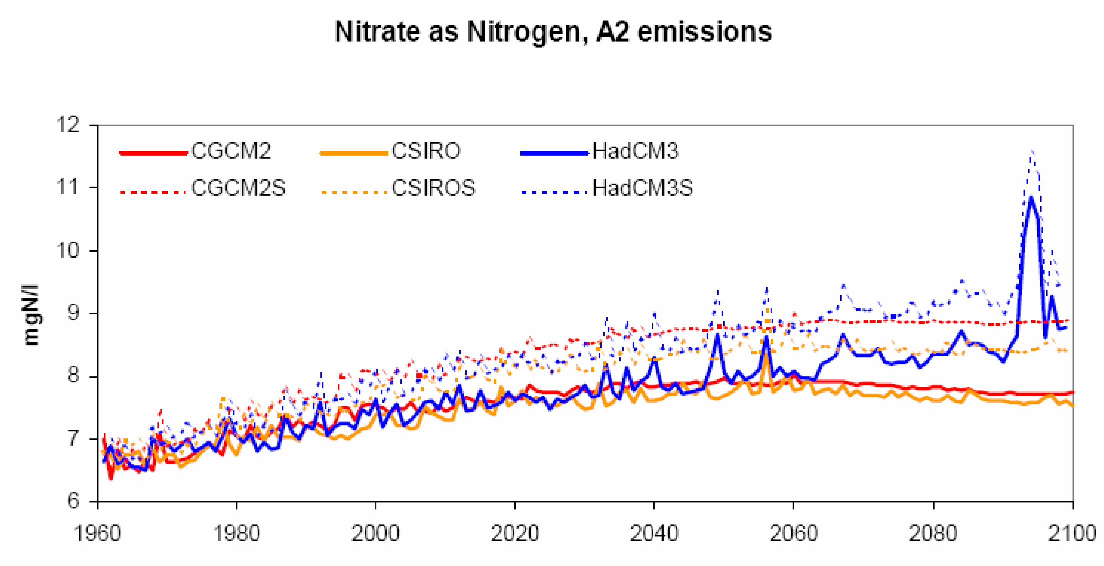

Figure 6 Nitrate-N concentrations (5th percentiles shown as dotted lines) over the 21st century under a range of GCMs for the Upper Kennet River System

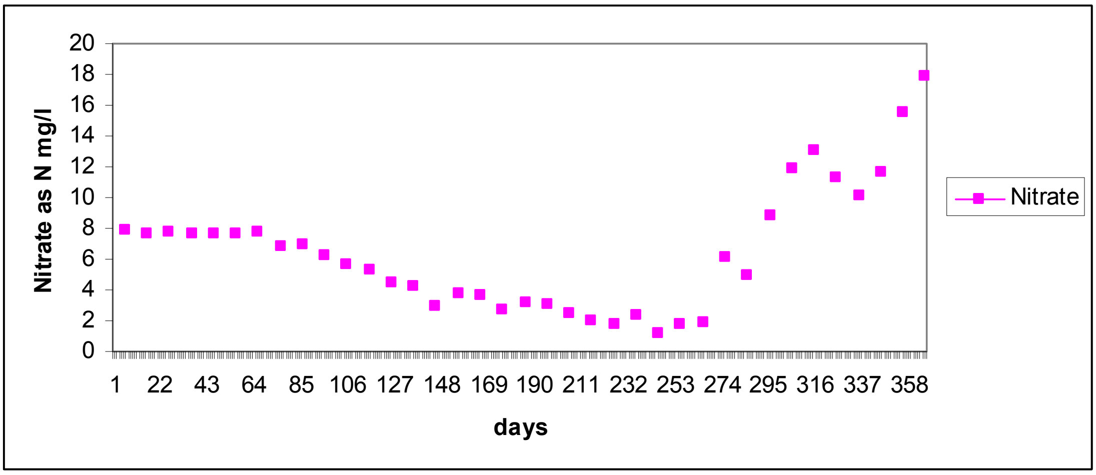

Figure 7 Nitrate- N at Swinford on the River Thames during 1976 showing increased concentrations when the drought ended and river flows recover on 3rd October (day 277) – Whitehead and Williams (1983)

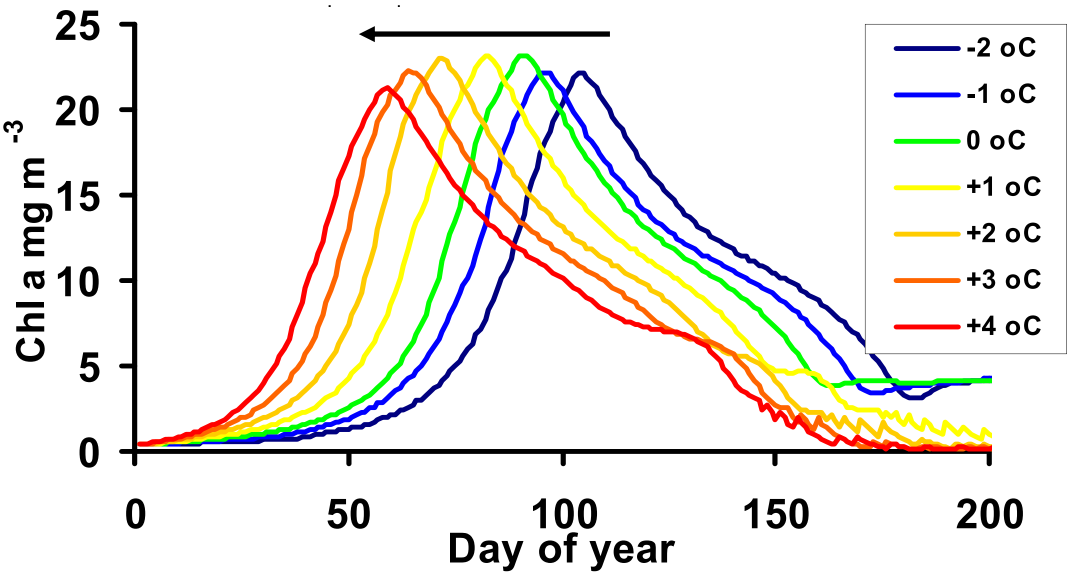

Figure 8 Effects of lake changing temperatures on algal blooms in Bassenthwaite Water

(source Elliot et al, 2007)

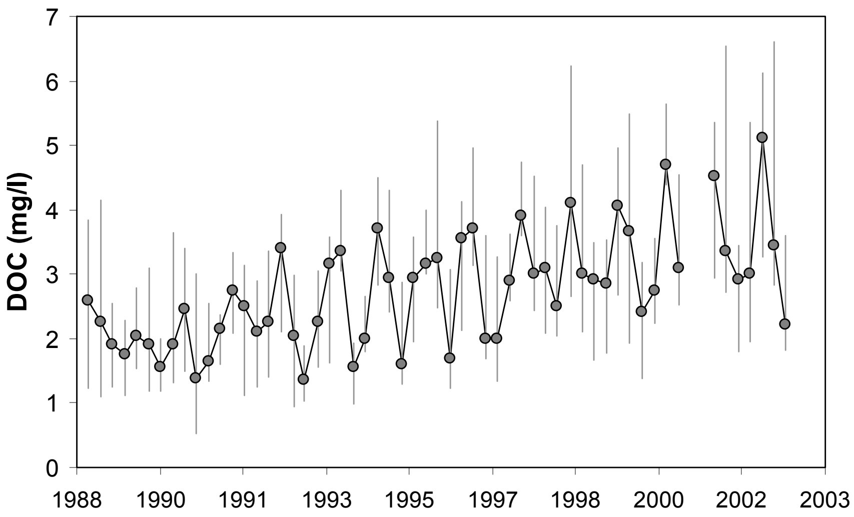

Figure 9 Trends in DOC across the UK from the Acid Waters Monitoring Network. (Source: CEH Acid Waters Monitoring Network)

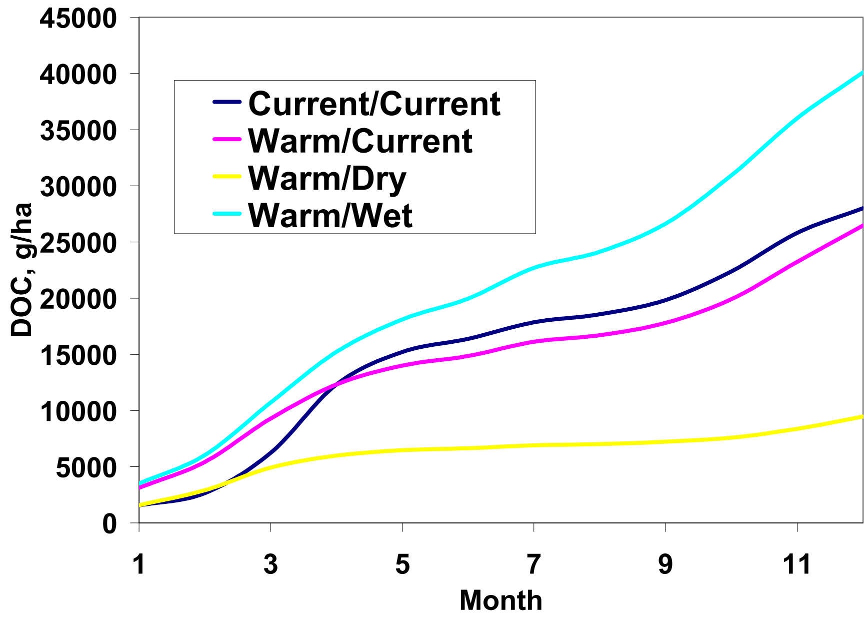

Figure 10 Simulated effects of climate change on DOC fluxes from upland catchments. (Source: Whitehead et al, 2006)

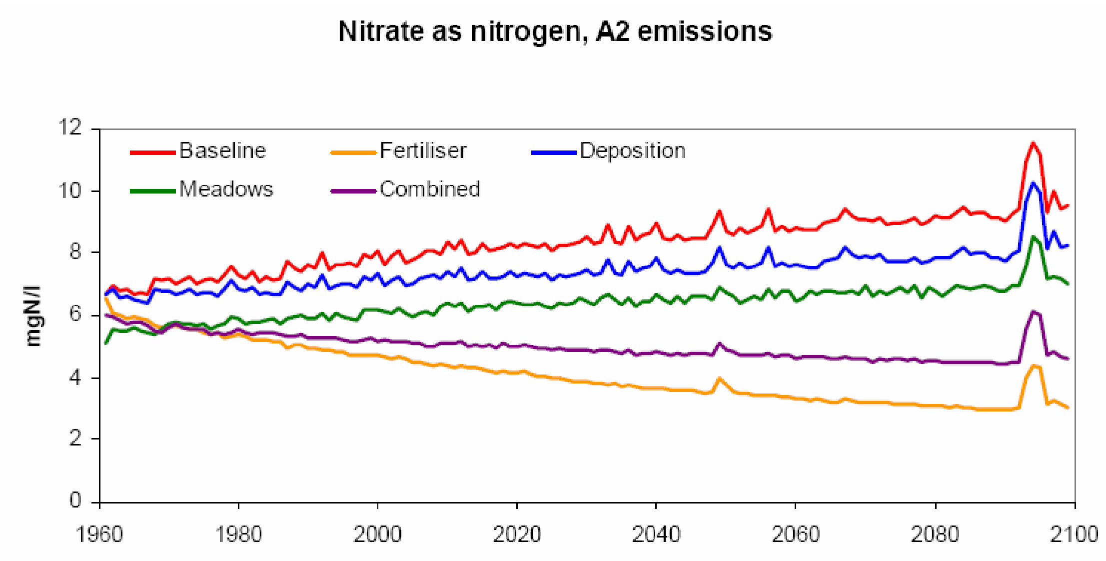

Figure 11 Modelled nitrate concentrations in the lower River Kennet, UK, 1960 – 2100, given various management treatments and downscaled climate scenarios from the HadCM3 GCM, medium-high emissions scenario. Source: Whitehead *et al.* (2007).

APPENDIX 2

The INCA models - description and model application

1. Introduction

The INCA (Integrated Catchment) models are the result of several NERC, Environment Agency and EU funded projects over the past 10 years (Whitehead *et al*. 1998a,b, Wade *et al*. 2002a,b,c). Together, they form a dynamic computer model that predicts water quantity and quality in rivers and catchments. The primary aim of INCA is to represent the catchment topography and the complex interactions and connections operating at a range of scales. INCA is process-based, so it can address the scaling up issue that often limited the potential of other water quality models. This appendix provides a brief description of the main components of INCA.

Key components of INCA

The INCA models have been designed to investigate the fate and distribution of water and pollutants in the aquatic and terrestrial environment. The models simulate flow pathways and track fluxes of pollutants such as nitrogen (N), phosphorus (P), sediments, carbon and metals in the land and in aquatic ecosystems. There are five components to the INCA modelling system:

1. A GIS interface, which defines sub-catchment boundaries and calculates the areas of different land use types in each sub-catchment.
2. A pollution input model, which calculates the total mass inputs from all sources to each sub-catchment, scaling wet and dry deposition and other inputs such as fertiliser applications according to land use.
3. A hydrological model, which simulates the flow of effective rainfall in the reactive and groundwater zones of the catchment and within the river itself. This component of the model drives the pollutant fluxes through the catchment.
4. The catchment process models, which simulate pollutant transformations in the soil and groundwater of the catchment.
5. The river pollution process model, which simulates dilution and in-river transformations and losses. Outputs from each sub-catchment (component 4 above) provide the mass flux into the corresponding river reach and input to the river quality process model, as shown in Figure A2.1.

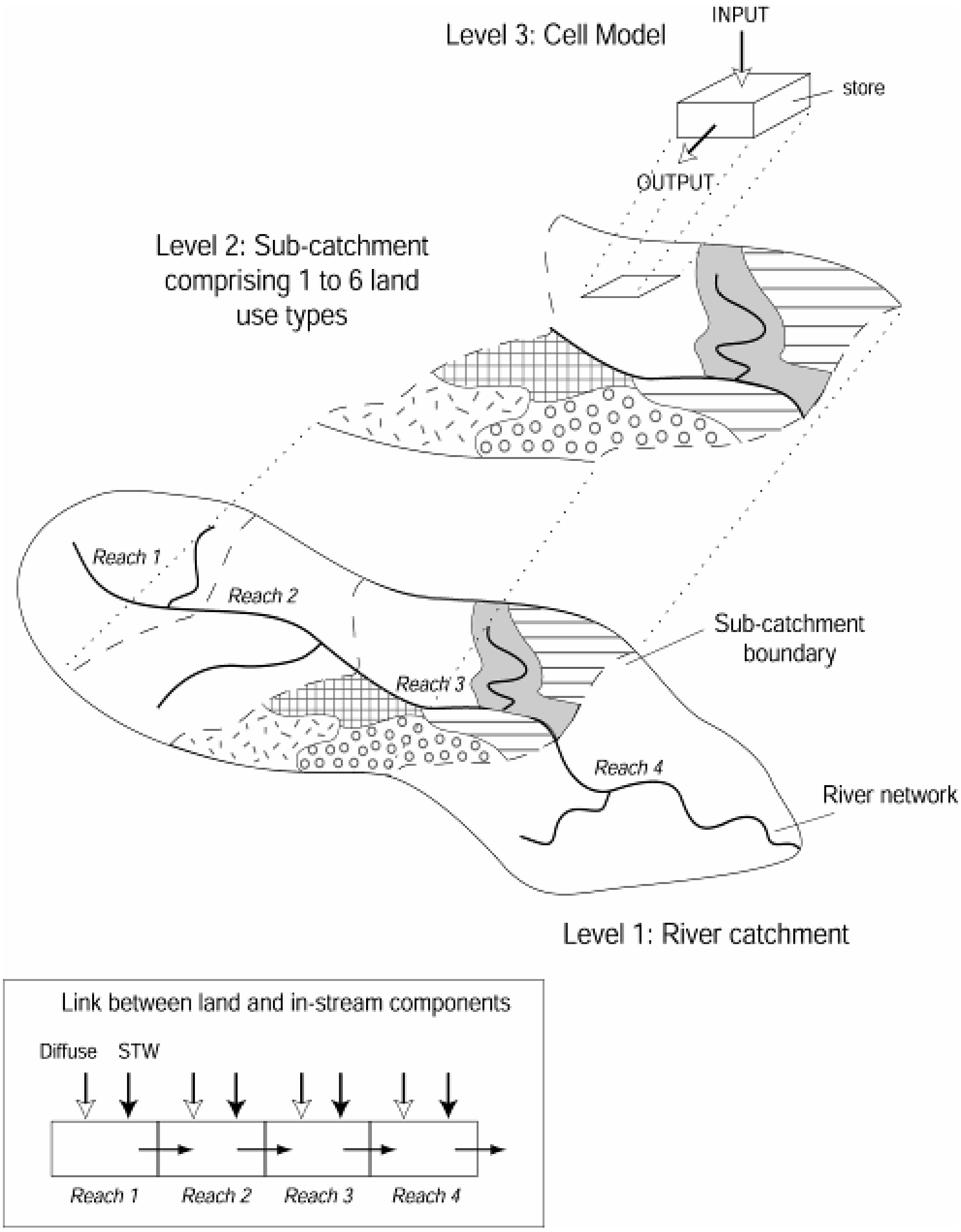

Figure A2.1 Land transfers of water and chemistry into streams

The INCA stand-alone software package consists of components three, four and five above. Components one and two are pre-processing operations, required to set up the parameter and data files for INCA.

INCA has been designed to be easy to use and fast, with excellent output graphics. The menu system allows the user to specify the semi-distributed nature of a river basin or catchment, to alter reach lengths, rate coefficients, land use, velocity-flow relationships and to vary input pollutant deposition loads.

INCA provides the following outputs:

- daily time series of flows and water quality outputs, such as metals, cyanide, nitrate-nitrogen and ammonium-nitrogen concentrations, at selected sites along the river;

- profiles of flow or water quality along the river at selected times;
- cumulative frequency distributions of flow and water quality at selected sites;
- table of statistics for all sites;
- daily and annual water quality loads for all land uses and all processes;
- three dimensional pictorial representations of flow and water quality;
- time series plots of the soil and groundwater responses;
- output time series for transfer to other analysis packages such as Excel;
- procedures for saving modified parameter sets;
- scenario simulation results, presented graphically or as output files.

2 The hydrological model

The hydrological model provides information on the flow moving through the soil zone, the groundwater zone and the river system. Figure A2.2 shows the hydrological model as a simple two-box system, with hydrologically effective rainfall moving through the soil system and then either recharging the groundwater system or leaching into the river. The groundwater flows are also routed to the river reaches after a suitable delay controlled by a residence time.

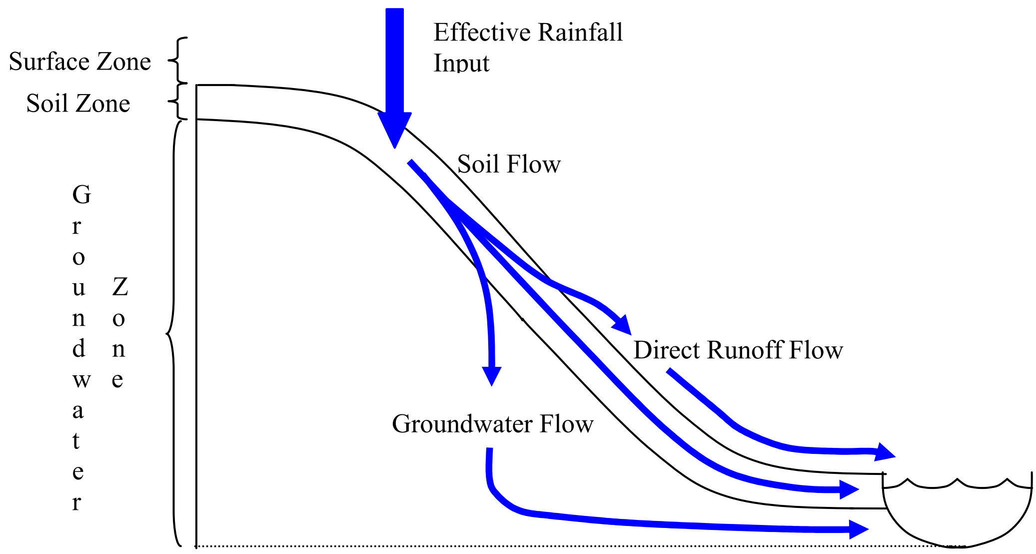

Figure A2.2 Structure of the cell model used to simulate the hydrological and N processes and transport mechanisms within the land component of INCA-N

The flow model for the two zones in the plant/soil system component of the INCA model is:

Soil zone
$$\frac{dx_1}{dt} = \frac{1}{T_1}(U_1 - x_1) \qquad (1)$$

Groundwater zone $$\frac{dx_2}{dt} = \frac{1}{T_1}(U_8 x_1 - x_2) \quad (2)$$

where x_1 and x_2 are output flows (m^3s^{-1}) for the two zones and U_1 is the input driving hydrologically effective rainfall (HER). T_1 and T_2 are time constants associated with the zones and U_8 is the baseflow index (the proportion of water being transferred to the lower groundwater zone). The HER data can be obtained using standard meteorological data collected locally or nationally. Outputs from the soil and groundwater compartments are released into the stream and then routed along the river system, as shown in Figure A2.3.

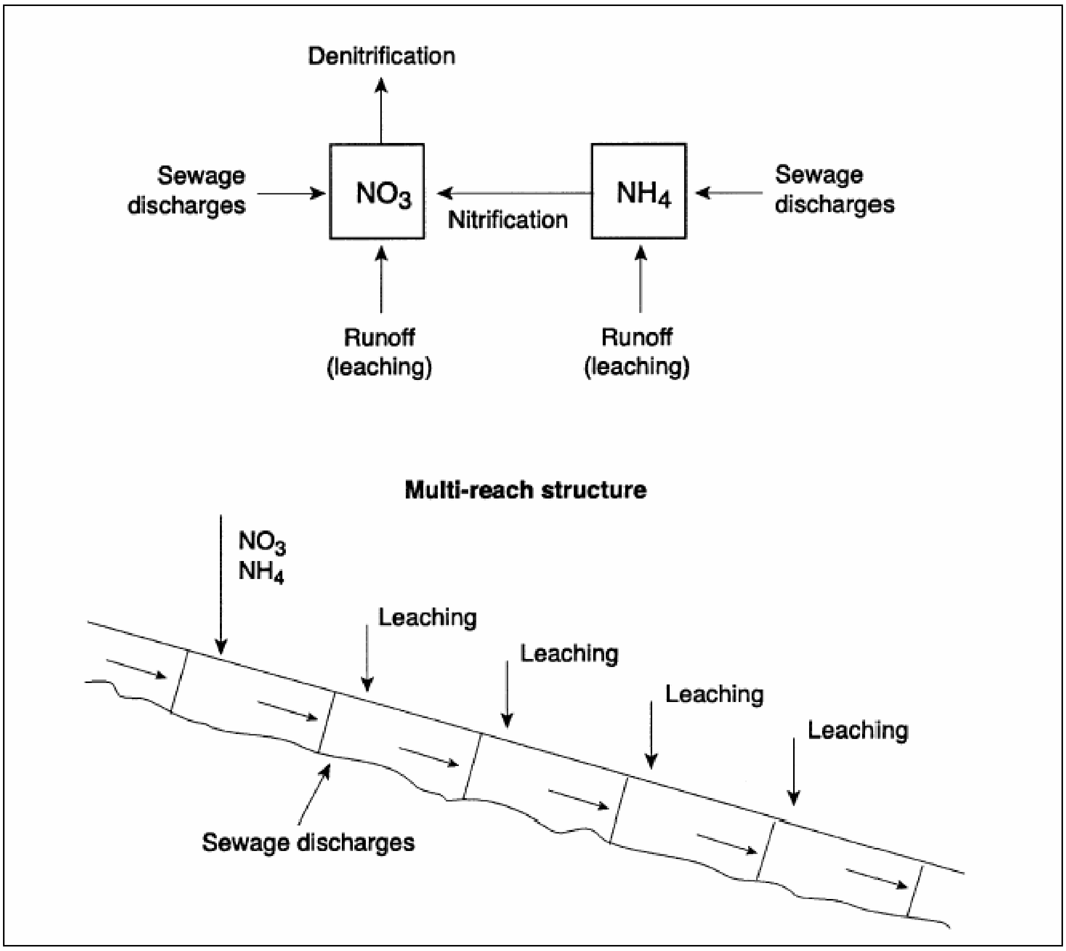

Figure A2.3 Instream processes and river reach structure

The river flow model is based on mass balance and uses a multi-reach description of the river system. Within each reach, flow variation is determined by a non-linear reservoir model. In hydrological flow routing terms the relationship between inflow, I, outflow, Q and storage, S, in each reach is represented by

$$\frac{dS(t)}{dt} = I(t) - Q(t) \quad (3)$$

where *S(t) = T(t)*Q(t)*, and *T* is a travel time parameter, which can be expressed as

$$T(t) = \frac{L}{v(T)} \tag{4}$$

L is the reach length and *v*, the mean flow velocity in the reach (ms^{-1}), is related to discharge, *Q* through

$$v(t) = aQ^b(t) \tag{5}$$

Where, a and b are constants to be estimated from tracer experiments or theoretical considerations.

Whilst this model is relatively simple, it is quite effective in simulating flows along rivers as shown in applications to the Bedford Ouse and a range of other river systems (Whitehead *et al*. 1998b). The equations are solved using a fourth order Runga Kutta method of solution with a Merson variable step length integration routine. This enables stable numerical integration of the equations and minimises numerical problems. The advantage of this scheme is that scientific effort can be directed to ensuring correct process formulation and interaction rather than numerical stability problems.

The hydrological model uses the hydrologically effective rainfall (HER) shown in Figure A2.4 to drive the model and generate the flows from the soils and groundwater system. The residence times in the model control the recession behaviour of the catchment and the area of the sub-catchments scale up the flows to give the full catchment flow. A typical model simulation for the River Twyi is given in Figure A2.5, and shows a very good fit. This is fairly typical of the hydrological model simulation and other results are given in the following sections and in the reference list below.

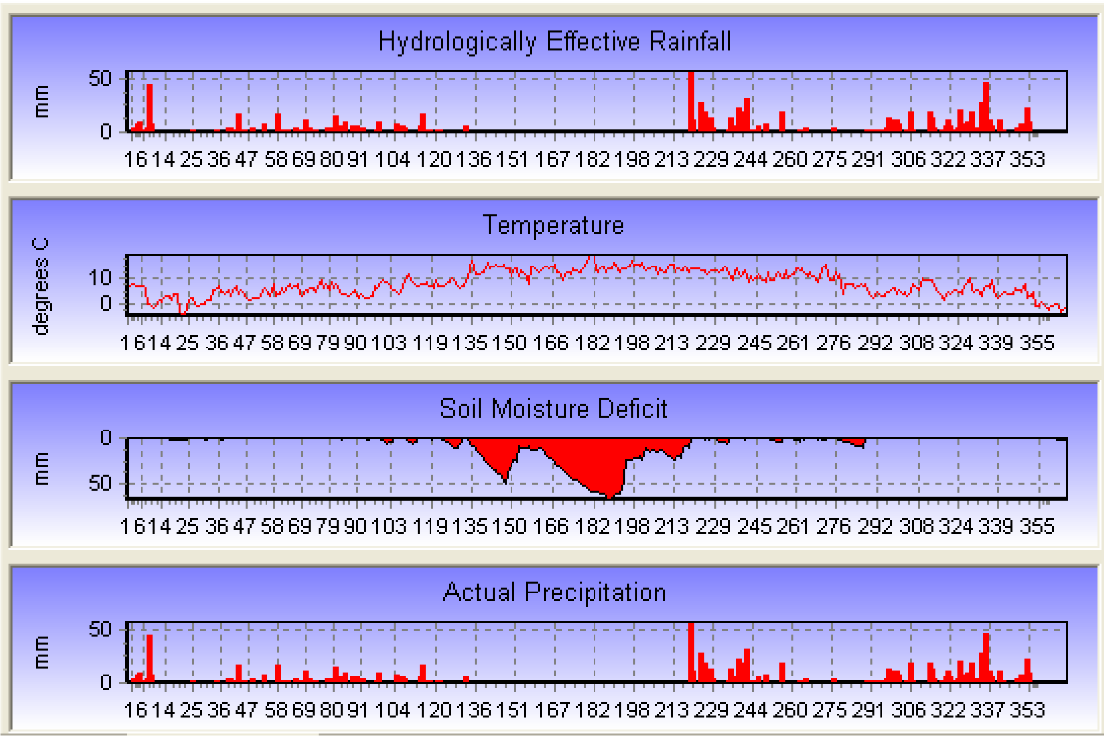

Figure A2.4 1992 input hydrological data for River Tywi in South Wales

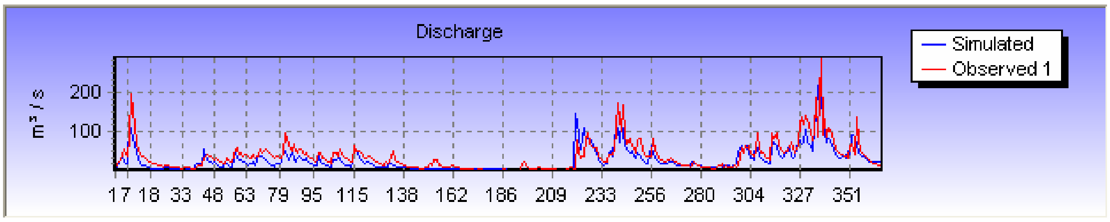

Figure A2.5 River Tywi observed and simulated flows at Ffinnant over 1992

3 INCA-N: the nitrogen and ammonium model

The hydrological model provides information on the flow moving through the soil zone, the groundwater zone and the river system. Simultaneously, whilst solving the flow equations, it is possible to solve the mass balance equations for nitrate-nitrogen (NO_3-N) and ammonium-nitrogen (NH_4-N) in both the soil and groundwater zones. The key processes that require modelling in the soil zone, as shown in Figure A2.6, are plant uptake for NH_4-N and NO_3-N, ammonia nitrification, denitrification of NO_3-N, ammonia mineralisation, ammonia immobilisation and N fixation. All of these processes will vary from land use to land use and a generalised set of equations is required, for which parameter sets can be derived for different land uses. The land phase model must also account for all the inputs affecting each land use, including dry and wet deposition of NH_4-N and NO_3-N, and fertiliser addition for both NH_4-N and NO_3-N (for example, as ammonium nitrate). Also, temperature and soil moisture control certain processes. For example, nitrification reaction kinetics are temperature dependent, and denitrification and mineralisation are both temperature and soil moisture-dependent.

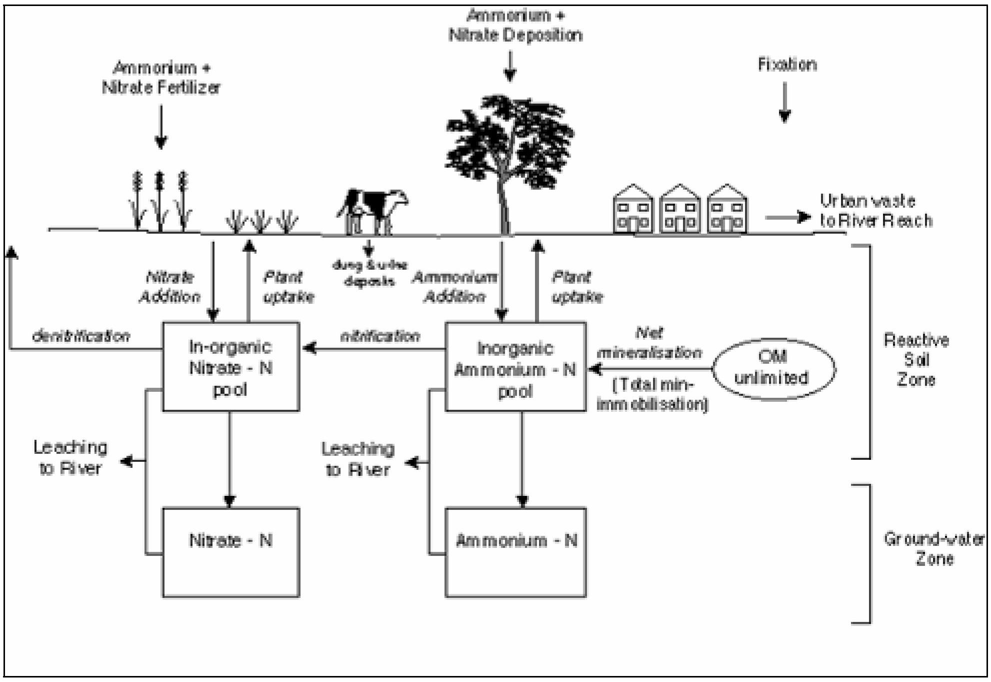

Figure A2.6 The key inputs, outputs and processes in the INCA nitrogen component

In the groundwater zone, the model assumes that no biochemical reactions occur and that there is mass balance for NH_4-N and NO_3-N. The equations used in INCA are as follows:

Nitrate-N

Soil zone $$\frac{dx_3}{dt} = \frac{1}{V_1}(U_3 - x_1x_3) - C_3U_7x_3 + C_6x_5 - C_1U_5x_3 + C_2 \qquad (6)$$

Groundwater $$\frac{dx_4}{dt} = \frac{1}{V_2}(x_3x_1U_8 - x_2x_4) \qquad (7)$$

Ammonium-N

Soil zone $$\frac{dx_5}{dt} = \frac{1}{V_1}(U_4 - x_1x_5) - C_{10}U_7x_5 - C_6x_5 + C_7U_6 + C_8x_5 \qquad (8)$$

Groundwater $$\frac{dx_6}{dt} = \frac{1}{V_2}(x_5x_1U_8 - x_2x_6) \qquad (9)$$

where x_3 and x_4 are the daily NO_3-N concentrations (mgl^{-1}) in the soil zone and groundwater zone respectively, and x_5 and x_6 are the daily NH_4-N concentrations (mgl^{-1}) in the soil zone and groundwater zone respectively.

U_8 is the baseflow index and C_3, C_6, C_1, C_2, C_{10}, C_7, C_8 are rate coefficients (per day) for plant uptake of nitrate, ammonia nitrification, nitrate denitrification, nitrate fixation, plant uptake of ammonia, ammonia mineralisation and ammonia immobilisation, respectively. U_3 and U_4 are the daily NO_3-N and NH_4-N loads entering the soil zone and constitute the additional dry and wet deposition and agricultural inputs (for example fertiliser addition). All rate coefficients are temperature dependent, using the equation:

$$C_n = C_n 1.047^{(\theta_s - 20)} \qquad (10)$$

where θ_s is soil temperature estimated from a seasonal relationship dependent on air temperature as follows:

$$Soil\ Temperature = Air\ Temperature + C_{16} \sin\left(\frac{3}{2}\pi \frac{day\ no.}{365}\right) \qquad (11)$$

where C_{16} is the maximum temperature difference between summer and winter conditions (^oC).

U_7 is a seasonal plant growth index, where:

$$U_7 = 0.66 + 0.34 \sin\left(2\pi \frac{[day\ no. - C_{11}]}{365}\right) \qquad (12)$$

where C_{11} is the day number associated with the start of the growing season and U_5 is a soil moisture threshold below which denitrification will not occur. Denitrification generally will only be significant when soil moisture levels are high. Similarly U_6 is a soil moisture control for mineralisation, which permits mineralisation when soil water content is above a threshold level.

Nitrogen process equation: river system

In the river, the key processes are denitrification of NO_3-N, nitrification of NH_4-N and mass balance. The reach mass balance needs to include the upstream NO_3-N and NH_4-N, together with inputs from both the soil zone and groundwater zone, as well as direct effluent discharges, as shown in Figure A2.5.

The equations for NO_3-N and NH_4-N in the river reaches are:

Nitrate
$$\frac{dx_8}{dt} = \frac{1}{V_3}(U_{10}U_9 - x_7 x_8) - C_{17}x_8 + C_{14}x_9 \quad (13)$$

Ammonium
$$\frac{dx_9}{dt} = \frac{1}{V_3}(U_{11}U_9 - x_7 x_9) - C_{14}U_3 x_9 \quad (14)$$

where U_9 is the upstream flow (m^3s^{-1}), U_{10} is the upstream NO_3-N (mgl^{-1}) and U_{11} is the upstream NH_4-N (mgl^{-1}). T_3 is the reach time constant (or residence time), which varies from day to day. x_7 is the estimated downstream flow rate (m^3s^{-1}) and x_8 and x_9 are the downstream (reach output) concentrations of nitrate and ammonia, respectively. C_{17} and C_{18} are temperature-dependent rate parameters for denitrification and nitrification respectively. Temperature effects are related to river water temperature σ as follows:

$$C=C1.047^{(\sigma-20)} \quad (15)$$

Whilst these equations are quite complex, the numerical solution is extremely fast so model runs typically take just a few seconds.

4 INCA-P, the phosphorus and ecology model

INCA-P is a dynamic, mass-balance model that attempts to track the temporal variations in the hydrological flowpaths and P transformations and stores, in both the land and in-stream components of the river system. INCA-P provides the following outputs:

- daily and annual land-use specific organic and inorganic P fluxes ($kgha^{-1}yr^{-1}$) for all transformation processes and stores within the land phase;
- daily time series of land-use specific flows, and organic and inorganic P concentrations in the soil and groundwaters, and in direct runoff;
- daily time series of flows, Total Phosphorus (TP), Soluble Reactive Phosphorus (SRP) ($mgPl^{-1}$), chlorophyll ‘a’ concentrations (μgchl ‘a’l^{-1}), and macrophyte and epiphyte biomass (gCm^{-2}) at selected sites along a river;
- profiles of flow and P concentrations along a river at selected sites;
- cumulative frequency distributions of flow and P concentrations at selected sites;

- detailed mass-balance checks.

Spatial data describing the major land use types are required. INCA-P also requires time series inputs (see Table A2.1), describing:

- The hydrology, namely the soil moisture deficit (mm), hydrologically effective rainfall (mmday^{-1}), air temperature (^{o}C) and actual precipitation (mmday^{-1}).

- Land management practices, namely estimates of growing season for different crop and vegetation types, and fertiliser application quantities and timings, which are estimated from DEFRA farm statistics and surveys. and local knowledge.

- Sewage effluent flow rates and SRP concentrations.

Table A2.1 Data requirements of INCA-P and examples of typical data sources.

Data	Description	Example source
TP and SRP streamwater concentrations	Spot samples taken at points within river system	Environment Agency routine monitoring, research studies
Land use	Classification at resolution of 1 km^2 grid	Institute of Terrestrial Ecology: Land Survey of Great Britain
Precipitation	Daily time series	Environment Agency, Meteorological Office.
Discharge	Daily time series	Environment Agency
Flow velocity	Occasional measurements for flow ratings	Environment Agency
MORECS rainfall, temperature and soil moisture	Daily time series (derived)	Meteorological Office
Base flow index	Derived for each flow gauge and extrapolated to other tributaries	Centre for Ecology and Hydrology
Fertiliser practice: application	Survey	DEFRA Farm Survey data, Fertiliser Manufacturers' Association
Fertiliser practice: timing	Survey	Local knowledge
Growing season	Survey	Local knowledge

The model has an interface designed to permit the inclusion of detailed time series data describing growing seasons, atmospheric deposition, fertiliser and effluent inputs from sewage treatment works if available, or alternatively to accept single lumped values. This allows the model to be applied to systems that are data rich or poor. To describe the spatial variations in rainfall, soil moisture deficit and air temperature within a catchment, multiple hydrological time series can be loaded if available.

There are four components to the INCA-P model:

1. A GIS interface that defines the sub-catchment boundaries, calculates the area of each land use type within each and decomposes the area of each sub-catchment into a maximum of six land use classes.

2. A land-phase hydrological model that calculates the flow of effective rainfall through soil water and groundwater stores, and as direct runoff. This component drives the water and P fluxes through the catchment.

3. The land-phase P model that simulates P transformations and stores in the soil and groundwater of the catchment. This is shown in Figure A2.7.

4. The in-stream P model that simulates the dilution and in-stream P transformations, and the corresponding algal, epiphytic and macrophytic growth response.

As in the case of INCA-N, the land phase component model was developed to simulate a generic 1 km^2 cell (Figure A2.1) and INCA-P is semi-distributed rather than fully-distributed. Twenty generalised equations define the P transformations and stores within a cell, and six user-defined parameter sets derived through calibration are used to simulate the differences between the six land-use types. The P fluxes from each of the transformations are determined by modifying the equation parameters through calibration against experimental or field data available in literature. The numerical method for solving the equations is based on the fourth order Runge Kutta technique. This allows a simultaneous solution of the model equations and ensures that no single process represented by the equations takes precedence over another.

To estimate the water and P outputs from each land-use type within each sub-catchment, the volume and load output from the cell model is multiplied by the land-use area, and the outputs from each land use are summed to provide a total sub-catchment volume and load. The resultant volume and load is then fed sequentially into a multi-reach river model, as shown in Figure A2.1.

The fertiliser, wastewater, slurry and livestock P inputs to the cell model vary with land use type, to simulate variations in land management practice. In addition, the effective rainfall, soil moisture and temperature can also vary between sub-catchments. Thus, it is possible to simulate the spatial variations in land management practice and hydrological inputs to some degree. This model structure entails the following necessary assumptions:

- The fertiliser, wastewater, slurry and livestock inputs are the same for a particular land use type, irrespective of the catchment location.

- The P process rates are the same irrespective of the location within the catchment. The process rates can still vary according to spatial variations in the soil moisture and temperature.

- The initial stores of water and P associated with each land use type are the same, irrespective of the location within the catchment.

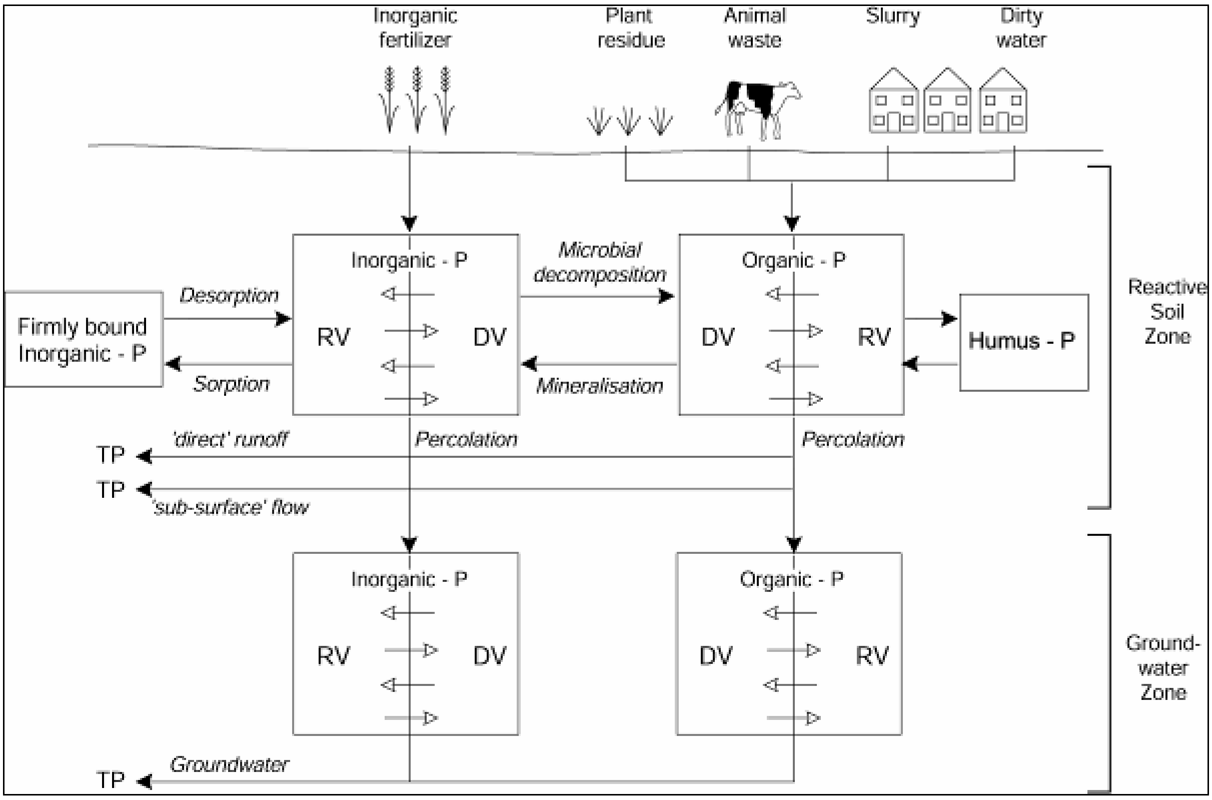

Figure A2.7 The land phase component of the INCA-P model, showing phosphorus inputs, processes and outputs in the direct runoff, soil and groundwater stores in each cell.

The instream processes built into INCA-P are illustrated in Figure A2.8. Here soluble P in the water column is linked to the total P by an equilibrium equation. There is a store of phosphorus in the sediments that can leach into the pore water pool in the sediments and this P can then be recycled back into the overlying water. Also, macrophytes and epiphytes are modelled in the reach, extracting P from the sediments and/or the water column. The epiphytes grow on the macrophytes and so will cause the death of the macrophytes as the epiphyte population levels build up. Thus, we have a highly interacting system that represents the chemical and biological processes occurring in the river.

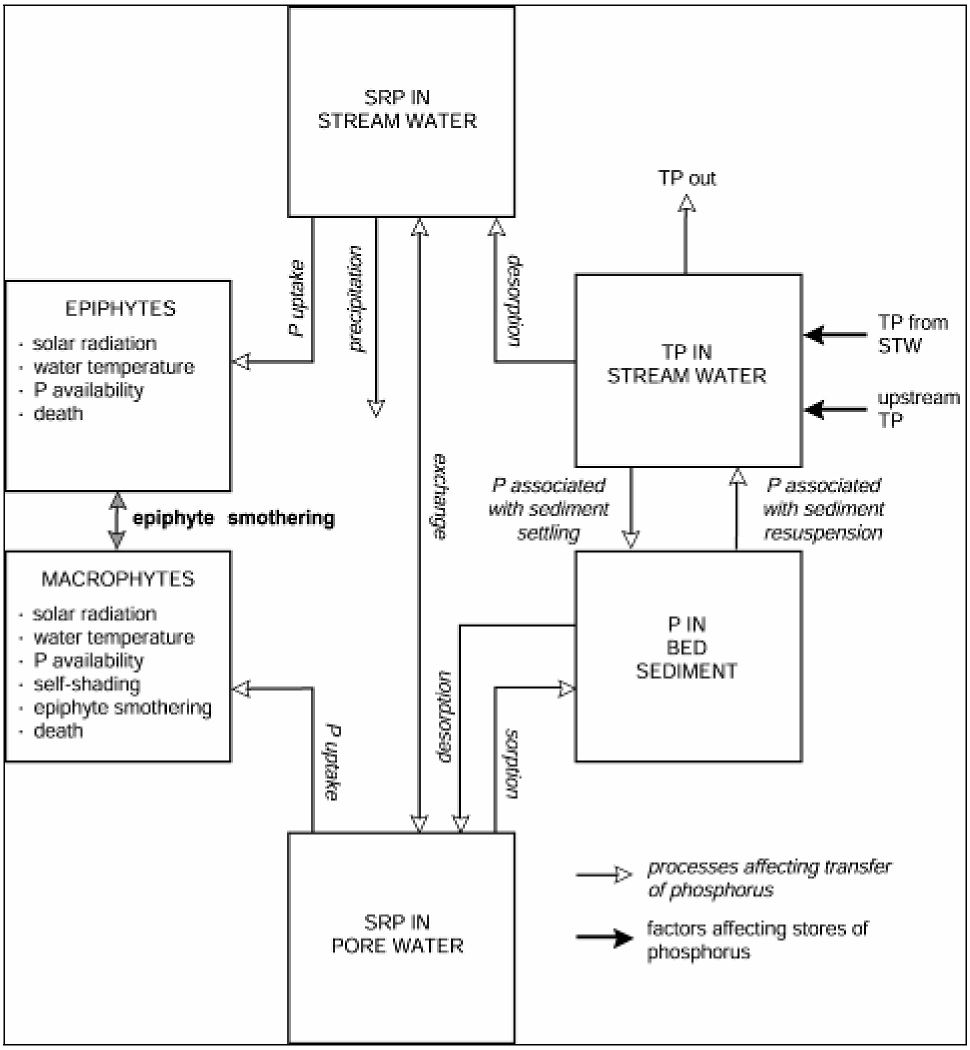

Figure A2.8 The in-stream component of the INCA-P model, showing phosphorus inputs, processes and outputs, together with sediment and ecological interactions.

This simplified representation of the behaviour expected in a real river system was used to simplify the model structure, reduce the time taken for each model run and to minimise the model's data requirements. Given the complex and highly heterogeneous nature of flow pathways, P processes and stores, it is uncertain if building a more realistic representation would improve model performance. Moreover, the hydrological and N process simulations produced by the INCA model, which use the same assumptions and structure, appear adequate (Whitehead *et al.* 1998b, Wade *et al.* 2002a).

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Environment Agency
Rio House
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