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# Which offers more scope to suppress river phytoplankton blooms: reducing nutrient pollution or riparian shading?

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## Abstract

River flow and quality data, including chlorophyll-a as a surrogate for river phytoplankton biomass, were collated for the River Ouse catchment in NE England, which according to established criteria is a largely unpolluted network. Against these data, a daily river quality model (QUESTOR) was setup and successfully tested. Following a review, a river quality classification scheme based on phytoplankton biomass was proposed. Based on climate change predictions the model indicated that a shift from present day oligotrophic/mesotrophic conditions to a mesotrophic/eutrophic system could occur by 2080. Management options were evaluated to mitigate against this predicted decline in quality. Reducing nutrient pollution was found to be less effective at suppressing phytoplankton growth than the less costly option of establishing riparian shading. In the Swale tributary, ongoing efforts to reduce phosphorus loads in sewage treatment works will only reduce peak ($95^{th}$ percentile) phytoplankton by 11%, whereas a reduction of 44% is possible if riparian tree cover is also implemented. Likewise, in the Ure, whilst reducing nitrate loads by curtailing agriculture in the headwaters may bring about a 10% reduction, riparian shading would instead reduce levels by 47%. Such modelling studies are somewhat limited by insufficient field data but offer a potentially very valuable tool to assess the most cost-effective methods of tackling effects of eutrophication.

**Keywords:** river quality model; phytoplankton; nutrients; riparian shading; pollution mitigation; climate change

## 1. Introduction

Implicit in the EU Water Framework Directive (WFD: 2000/60/EC) is the desire for healthy river ecosystems and an avoidance of eutrophication. One of the manifestations of eutrophic conditions is phytoplankton blooms. Dense phytoplankton blooms may cause deterioration in ecosystem services, for example obstructing water abstractions for drinking and industrial purposes (Henderson *et al.*, 2008) impairs the provision of water of a sufficient quality. Phytoplankton also include the toxic cyanobacteria group whose occurrence in UK rivers is extensive (Ferguson, 1997). The specific effects of phytoplankton on dissolved oxygen (DO) (Cox, 2003) can have wider impacts on river ecosystems; fish stocks being endangered by (i) diurnal DO sags associated with night-time respiration and (ii) chronic DO suppression due to increases in river biochemical oxygen demand (BOD) following population crashes. Hilton *et al.* (2006) have described eutrophication as having several stages culminating with the outcompetition of macrophytes, depending on river retention time, by either phytoplankton or periphyton. An underlying, and serious, additional concern is the likely impact of a future changing climate which may favour phytoplankton blooms and reduce DO levels (van Vliet and Zwolsman, 2008; Cox and Whitehead, 2009; Johnson *et al*., 2009; Whitehead *et al*., 2009).

It has been widely assumed, within the scientific community and elsewhere, that eutrophication in surface freshwaters is induced by enrichment of nutrients (nitrogen, phosphorus) which can be controlled by reducing pollution from diffuse (predominantly agricultural) and point (urban wastewater and industrial) sources. Nutrient pollution and eutrophication is currently a key focus for policy decision-making, although such decisions also need to consider other motivations for curtailing nutrient inputs to rivers (e.g. drinking water quality: EC Nitrates Directive). Measures for reducing nutrient pollution and their cost-effectiveness have been extensively explored (e.g. Pretty *et al*., 2003; Cherry *et al*., 2008). Whilst models have successfully described the effect of increases in nutrient levels on specific groups of phytoplankton in lakes (Reynolds *et al*., 2001); for river systems the relationship is less clear. Hydrological response and the complexity of river ecology, coupled with the influence of geology, water temperature and riparian shading implies that significant variation in threshold nutrient levels between different rivers will occur. For example, high flows are likely to suppress levels by flushing phytoplankton downstream into estuaries (Neal *et al*., 2006). Thus in terms of nutrients, the critical threshold concentrations for transition between ecological states cannot be fixed. However, values for phosphorus have been proposed for legislative purposes (UKTAG, 2008), albeit linked to ecological indicators not of total river phytoplankton biomass but rather to the more-specific trophic diatom index (TDI), a measure of benthic algal community composition (Kelly, 1998). Many other biological criteria of river health focus on macro-invertebrates and these, along with TDI, are indices requiring specific taxonomic skills that are not generally available. Furthermore, they cannot be measured frequently for site monitoring, are not useful for understanding short-term changes, and are difficult to relate to specific drivers or pressures.

Concentration of chlorophyll-a pigment is a surrogate for total water column phytoplankton (sestonic algae) biomass; and threshold levels have been proposed for rivers. For example, a robust scheme based on monitoring of an extensive range of temperate North American and European streams was formulated by Dodds *et al.* (1998) with a view to representing quantitatively the filamentous green algal group, excluded from the TDI, which is readily associated with nuisance conditions. In contrast to the taxonomic indices, chlorophyll-a measurements are readily made as part of UK national water quality monitoring schemes and have been related to nutrient concentrations and other environmental variables on a regional basis (Neal *et al*., 2006). Furthermore, including chlorophyll-a in dynamic process-based river hydrochemical models allows phytoplankton growth to be explored mechanistically.

Many river quality models include phytoplankton (e.g. Whitehead *et al*., 1997; Reichert *et al*., 2001; Scharfe *et al*., 2009) and, due largely to a paucity of observations, typically represent them as an assemblage. If supported by monitoring data however, information from controlled experiments (Bowie *et al*., 1985) offers scope to improve model performance by discriminating between the behaviourally well-defined functional groups and their environmental requirements (light, heat, nutrients). Diatoms, dominant in spring, often become growth limited by a third nutrient, silicon, and are succeeded by chlorophyceae (which include the cyanobacteria) during the summer (Garnier *et al.*, 1995). However, in the summer when hotter, sunnier conditions are intuitively most favourable for blooms, sustained low chlorophyll-a concentrations have often been observed (Skidmore *et al.*, 1998; Balbi, 2000) and attributed for example to enhanced grazing by zooplankton (Gosselain *et al.*, 1998). Phytoplankton have fairly well defined optimum light intensities for growth (Bowie *et al.*, 1985), which can be self-regulated to a greater or lesser extent depending on the species through their buoyancy, an essential consideration when modelling lake phytoplankton (Reynolds *et al.*, 2001). Yet, for river models it is debatable how best to embody vertical light attenuation and whether representing stratification at low flows is warranted. In summary, a myriad of environmental, biological and physical factors can potentially enhance or curtail phytoplankton growth.

In using the QUESTOR model (Boorman, 2003a, b), which represents explicitly the flows and chemical inputs from all tributaries and point sources to a network of river channels, the present research seeks to identify to what extent river phytoplankton growth is limited by nutrients. In this way, understanding how riverine nutrient and chlorophyll-a concentrations relate in the context of other environmental variables (flow rate, light/shading, temperature) will be tested. To assist this exercise, in keeping with existing concepts of classification (Dodds *et al.*, 1998; Gordon *et al.*, 2004; Hilton *et al.*, 2006), a scheme using average spring-summer conditions as a means of differentiating four key progressive stages of eutrophication will be applied to enable prediction of trophic state in successive reaches along a river network. Many of the c. 300 rivers used by Dodds *et al.* (1998) to formulate classification were of similar size to that typical of larger UK rivers. This exercise will focus on the Yorkshire Ouse network in north-east England (Figure 1), draining an area of 3530 $km^2$ at the tidal limit 7 km south of York at Naburn Weir (NGR 4593 2446). Particular emphasis will be made on the Swale and Ure tributaries, the main characteristics of which are listed (Table 1).

Upstream of the more urbanised lower reaches (near York and Harrogate) the Ouse catchment is largely rural and includes the Swale, Ure and Nidd tributaries, showing a climatic range, a wide diversity of agricultural and other land-uses, and varied urban coverage. In particular, the Swale sub-catchment has a greater incidence of point sources associated with urban centres. The Ouse has been the subject of large research monitoring programmes (e.g. LOIS: (Leeks and Jarvie, 1998) and considerable modelling effort (e.g. Silgram *et al.*, 2009). Model applications have included studies specifically simulating chemical status for EU legislative purposes (UKTAG, 2008). According to Boorman (2003c), using Environment Agency (EA) data, during 1996-2004, "good" chemical status was achieved throughout the Ouse for DO ($10^{th}$ percentile > 60% saturation), BOD ($90^{th}$ percentile < 5 mg/L) and ammonium ($90^{th}$ percentile < 0.6 mg N/L), although for soluble reactive phosphorus (SRP: mean < 120 μg/L), only parts of the Swale and Ure were compliant. For nitrate, all reaches are classified as "moderately low" or better (mean mg nitrate-N/L < 4.52), which complies with the EC Nitrates Directive legislation. Simulated classifications were largely in accordance with observed data (Boorman, 2003a). Mismatches were only apparent at the extremities of chemical percentile distributions. The model performed well at reproducing flow time series. In terms of biology, current classification based on macro-invertebrates reveals all reaches to be fairly good quality or better, many being in the highest class. Using existing standards, the Yorkshire Ouse is a largely clean river network.

The capability of QUESTOR shown in the Ouse study (Boorman, 2003a) suggests it is suitable for the present study to address the following objectives:

1. Successful simulation of phytoplankton (chlorophyll-a) against measured data using a dynamic river flow and quality model at a daily time-step;
2. Test the hypothesis that residence time is the major control on spatial patterns in peak chlorophyll-a levels;
3. Make indicative predictions of phytoplankton behaviour in response to climate projections as summarised for the Ouse for 2080 (Johnson *et al.*, 2009);
4. Identify mitigation options for different parts of the catchment and predict their effects, in each case carrying out a simple cost-effectiveness analysis.

## 2. Methodology

As the objectives include running scenarios involving perturbation of land-use, a linked modelling approach was applied. The 1997-1999 period was chosen as it had more abundant and spatially widespread chlorophyll-a data than at other times. To represent flows and nitrate concentrations from diffuse sources the NAL-CASCADE model was used (Hutchins *et al.*, 2010) which is sensitive to changes in land management (Hutchins *et al.*, 2009). NAL-CASCADE was linked to QUESTOR, which, using observations from tributaries and effluents (sewage works and industry), represents the diffuse inputs of other substances and point sources. Whilst numerous models exist, covering a range of spatial scales (e.g. Johnes, 1996; Arnold *et al*., 1998; Hutchins *et al*., 2002; Wade *et al*., 2002; Davison *et al*., 2008), changes in phosphorus concentrations from diffuse sources were not to be modelled explicitly. Although it is acknowledged that diffuse phosphorus can be a very important driver of river ecological status in specific cases, the omission is made for various reasons. Recent evidence suggests such diffuse agriculturally-derived components are of less importance than previously thought, accounting for less than 15% of the annual UK riverine P load (Jarvie *et al*., 2006; White and Hammond, 2009). Much of the diffuse phosphorus load is delivered to rivers during high flows in winter when algae are largely absent or being flushed out, and the attribution of cause and effect between land applications and river concentrations is very complex (Neal *et al*., 2010a). Of the mitigation options available, Haygarth *et al.* (2009) show that micro-scale detention of mobilised P (e.g. by buffer zones) is more effective than the almost negligible impact of reducing inputs (e.g. land-use change or reduced fertilisation), which is the primary focus for controlling N.

### *2.1. Simulating water quality*

A land-use dataset (Posen *et al*., submitted), which combines landcover (CEH LCM2000: Fuller *et al.*, 2002) and 2004 Defra Agricultural Census data, was spatially linked to HOST soil class information (Boorman *et al.*, 1995). Values for topsoil nitrate available for leaching (NAL) were calculated at monthly resolution for each combination of land-use and soil (Hutchins *et al.*, 2010). Along with gridded meteorological inputs (daily rainfall and weekly/fortnightly potential evapotranspiration) these NAL values were used in CASCADE, a model of catchment hydrology and diffuse pollution transfer to rivers (Cooper and Naden, 1998). Spatially distributed into hydrological response units (HRUs) (Cooper and Naden, 1998; Hutchins *et al*., 2010) CASCADE represents soils in two layers (topsoil and subsoil), and provides diffuse flows and nitrate concentrations as inputs to the 1-D river quality model, QUESTOR (Eatherall *et al*., 1998; Boorman, 2003a, b). Amongst a range of other determinands, QUESTOR simulates temperature, flow, pH, DO, BOD, nitrate, ammonium, inorganic P, organic P, suspended sediment and chlorophyll-a. In addition to diffuse inputs, QUESTOR considers inputs from point sources, abstractions for water supply, aeration at weirs and in-river biochemical kinetic transformations. A truncated river network was assumed (Figure 1), the upstream ends of the network typically being defined by the location of a monitoring site. Boorman, (2003a) defines equations in QUESTOR for flow and chemical quality.

*2.2. Calculating in-river transformation processes*

Rate coefficients for the in-river transformations (namely nitrification, denitrification, benthic oxygen demand, BOD decay, BOD sedimentation and P mineralisation) were determined by calibration against EA river chemical monitoring data from many sites within the defined network (Figure 1) typically available through fortnightly resolution grab samples. Chlorophyll-a data covering the 1997-1999 period were only available at 13 sites in the upper reaches of the Ouse network (typically with 50-100 observations in total per site). Of these, 5 of the sites are at or above the top of the network and were used to provide the model with input time-series. Data from the other 8 sites aided calibration. Any robust calibration of phytoplankton photosynthetic growth rates is limited due to the lack of downstream sites. However for 1993-1996, LOIS data were available at 4 sites towards the bottom of the network (typically with 150-200 observations per site). These slower flowing lower reaches have some phytoplankton blooms. So, data from 12 sites were used to define network-wide parameter values and assess model performance (Figure 1).

*2.3. Hydrological representation*

CASCADE daily hydrological response was calibrated for the gauging stations at Catterick (27090) and Kilgram Bridge (27034) (Figure 1). These headwater areas, part of the Swale and Ure systems which were chosen for detailed analysis, are also the only suitable gauged sub-catchments that are predominantly rural, permitting calibration of flows from diffuse sources. For the Ouse catchment, the QUESTOR river network consisted of 205 reaches and, as well as the main Ouse channel, represented 8 branches (Figure 1). These consisted of the main tributaries (Swale, Ure and Nidd) and some minor tributaries (Foss, Crimple Beck, Whiske and Cod Beck). The model configuration of QUESTOR as described by Boorman (2003b) was used, as the main objective was to link water flow and quality with phytoplankton response.

The linked model was used throughout the Yorkshire Ouse with the exception of the Swale where, due to the greater relative abundance of urban areas, a slightly different approach was undertaken. Here, to maximise the accuracy of the modelling as a basis for the exploration of proposed measures focusing on cleaning-up of point rather than diffuse sources, observed flow and nitrate data from small tributaries were used where available, instead of applying NAL-CASCADE. Further details are given elsewhere (Ani *et al.*, 2010).

*2.4. Simulating phytoplankton behaviour*

The biological model used in QUESTOR comprises three autotroph types (phytoplankton, benthic algae and macrophytes: each with fixed stoichiometry (chlorophyll-a content c. 1% by mass)) determined by interrelated processes of photosynthesis, respiration, death and predation. All types are active in water column DO and nutrient exchange. Phytoplankton alone contribute to water column chlorophyll-a, and, on death, a BOD and nutrient source. The photosynthetic rate is first order with respect to biomass and is temperature dependent (via the Arrhenius equation: Equation 1). Examples of water temperature data (Figure 2) show annual ranges typical of the case-study area. As Equation 1 shows, maximum photosynthetic rate is limited by a multiplicative formulation of nutrients (f(N): minimum of N and P: a hyperbolic relationship as defined by Michaelis Menten kinetics (Equation 2)) and light (f(L)). For light limitation, (i) attenuation ($\gamma$) with depth is described by the Beer-Lambert law (including effects of suspended sediment (SS) and the phytoplankton (Phy) themselves) (Equation 3), and (ii) photolimitation, with respect to autotroph-specific optimum intensities, is represented by the Steele (1962) formulation (Equation 4). Phytoplankton are assumed to be exposed to depth-averaged light. Limitation factors (where values lie between 0 (full limitation) and 1 (no limitation)) are calculated for every time-step in all reaches. The model is driven by photosynthetically-active radiation determined using weather station data for the Cawood site (NGR 4575 4375) held at the NERC British Atmospheric Data Centre (Figure 2). Water temperature in each river reach is modelled by a weighted mixture of contributions from input sources with additional allowance made for the balance between incoming radiation (using the Cawood data) and outgoing long wave back radiation from the water surface. Modelling macrophyte and benthic algal populations is outside the scope of the present paper which solely considers phytoplankton, treating them as an assemblage rather than discriminating between component groups.

$$k \quad Phy \; k_{ref} \quad {}^{(T \quad T_{ref})} \quad f(N) \quad f(L)$$

**(1)**

where k= photosynthetic rate (/day), Phy (i.e. Chl-a) is a concentration in mg/L, T= temperature (°C), $T_{ref}$= 20°C. f(N) and f(L) hold values between 0 and 1. Other terms defined in Table 3

$$f(N) \quad \min \left[ \frac{N}{N \quad k_N}, \frac{P}{P \quad k_P} \right]$$

**(2)**

where: N= mg N/L (nitrate plus ammonium) and P = mg P/L (inorganic plus organic); other terms see Table 3.

$$_{base} \quad L_{ss} SS \quad L_{Phy} Phy$$

**(3)**

where: SS and Phy (i.e. Chl-a) are concentrations in mg/L; other terms see Table 3

$$f(L) \quad \frac{2.718}{d} \left[ e^{\frac{R_s L_1 L_2}{L_{opt}} e^{\quad d}} \quad e^{\frac{R_s L_1 L_2}{L_{opt}}} \right]$$

**(4)**

where: $R_s$ = radiation at the surface not reflected ($W/m^2$) and d = water column depth (m); other terms see Table 3.

*2.5. Scenario analysis*

The impacts on river phytoplankton concentration of four scenarios were explored:

A. Alter river flow, water temperature and solar radiation in line with UKCIP02 projections to 2080 (Johnson *et al.*, 2009): the "Climate Change Scenario";
B. Reduce SRP concentrations in effluent from sewage treatment works (STWs) by imposing tertiary treatment: the "STW scenario";
C. Abandon all agricultural land to moor/grass/heath: the "Agriculture Scenario";
D. Plant riparian deciduous trees (e.g. alder, poplar, willow): the "Tree Scenario".

All model inputs and parameters not affected by the specification of the scenarios were held the same as for the 1997-1999 model application. Scenarios B, C and D represent an eventual steady state for which attainment, in particular for D, may take a long time.

The "Climate Change Scenario" (A) considers the direct impacts of a changed climate (as opposed to any that may indirectly induce land-use change). Modifications to QUESTOR model inputs were made in accordance with projections for the Ouse (Johnson *et al.*, 2009). These were taken from the UKCIP02 high emission scenario which used the HadCM3 General Circulation Model downscaled in space using the Hadley Centre Regional Climate Model. Seasonal-specific changes relative to 1997-1999 were explored for water temperature and incoming solar radiation. For spring and summer the water temperature increase is projected to be 36% and 25% respectively, whereas for solar radiation respective increases of 9% and 15% are projected. For river flows, a constant reduction of 30% was implemented for all seasons and at all flow levels. This value was in very close accordance with the projections for spring and summer conditions which are of most relevance to phytoplankton growth. Johnson *et al.* (2009) believe that nutrient concentrations will increase, yet this assumption does not account for possible changes in land-use in response to new climate regimes. In this respect there is much uncertainty. In the scenario therefore, present day nutrient concentrations were used.

The "STW Scenario" (B) was implemented in the Swale catchment upstream of Thornton Manor (3/39: Figure 1) where knowledge of nutrient cycling processes and data for model parameterisation is more comprehensive than elsewhere (Bowes and House, 2001) and population density is higher than in neighbouring catchments (e.g. Ure). To simulate implementation of tertiary treatment the concentrations of SRP (represented by inorganic P) and Total Phosphorus (TP: inorganic plus organic fractions) in STW effluents were capped at 1.5 and 2.0 mg P/L respectively (Carey and Migliaccio, 2009), these levels representing a realistic compromise between currently-available technology and avoiding expenses that are disproportionate for small treatment works. The effect of the measure was to cap the P content in over 85% of the total effluent volume discharging into the Swale system. The remaining 15% comes from works that are far from the modelled main river network, and as a result each of these must be represented as a portion of the input from a minor tributary, aggregated with other sources. Representation of such tributaries relies on observations (in the case of phosphorus), so these particular works cannot easily be subjected to hypothetical interventions. In the riverbed, a shift in the equilibrium between water and solid phase following the change in phosphorus loading from

STWs may occur and thereby influence the availability of phosphorus for phytoplankton growth. Data from the Thames in southern England suggest such re-equilibration processes may take many years and in some places sediment dredging may also be necessary (Neal *et al.*, 2010b). Concentrations of other chemical species entering the river from STW effluents remained unaltered.

The "Agriculture Scenario" (C) was implemented in the headwater catchments of the Ure (above 27034: Kilgram Bridge) and Nidd (above 27005: Gouthwaite Reservoir) (Figure 1). The Ure was of particular interest as observed chlorophyll-a levels were the highest in the entire Ouse network in the 1990s (Neal *et al.*, 2006). The change to be represented is a conversion out of production of 200 $km^2$ of arable land and agricultural grassland. For this area, a NAL value for non-agricultural grassland of 5 kg N/ha per year (Silgram *et al.*, 2004) was to be used throughout. A transfer function representing hydrological leaching of solutes (Anthony *et al.*, 1996) was applied to land-uses before and after the change. The relative decrease in nitrate leached quantified the scenario; implemented as a scaling factor applied to the daily CASCADE nitrate concentration outputs from the relevant HRUs, and then fed through into the QUESTOR simulation. Any change in nitrate concentrations in response to land management perturbations may take many years to be seen in soil leachate; taking much longer still to be manifested in rivers where groundwater sources dominate (Hutchins *et al.*, 2010). Against that, any eventual reduction of phosphorus leaching to rivers resulting from the change in management (not included in specification of the scenario) can only be beneficial in reducing phytoplankton. Other diffuse chemical inputs were not altered.

The "Tree Scenario" (D) was implemented in all the tributaries of the Ouse. Whilst some experimental studies, as reviewed by Ghermandi *et al*. (2009), appear to suggest that effects of tree shading on light availability are unlikely to be substantial in rivers as wide as those in the Ouse network, a recent model study (Dewalle, 2008) goes into much greater systematic and quantitative detail, indicating such a view to be unduly pessimistic. Indeed the findings of Ghermandi *et al*. (2009) and others (e.g. Davies- Colley and Quinn, 1998; Boothroyd *et al*., 2004; Caissie *et al*., 2007) are consistent with the Dewalle (2008) model but all these study sites are dissimilar to the Ouse, either in terms of latitude or river width. In the Ouse, the reduction in intensity of solar radiation reaching the water column due to tree shading was assumed to be 39%, as estimated using the method of DeWalle (2008). The method calculates shading as a function of site latitude, river flow direction and the ratio of the height of the riparian vegetation to the width of the riverbank. In this respect a 30m-wide channel with a mature 30m-high mixed canopy of alder, willow and poplar trees was assumed. In the minor tributaries (e.g. Wiske and Cod Beck) the channel is narrower, hence the assumptions allow for an inevitable degree of failure to plant throughout the network length or to always achieve full canopy height. As the river channel flow aspect is predominantly NW to SE, flow direction was considered to be uniformly distributed between the extreme cases of north-south and east-west trending channels. Full reduction in intensity of solar radiation was only implemented between 1$^{st}$ May and 30$^{th}$ November each year when deciduous trees are in leaf. The 39% figure was linearly interpolated to/from 100% transmittance for a 40 day period either side of the duration of full shading. Conceptually, tree planting induces a shading effect and also lowers water temperatures, as accounted for in the model using the dynamic water heading module. Satellite imagery reveals riparian shading to be currently fairly considerable in the catchment. Estimates suggest that this may be at approximately 25% of capacity, but as trees are more abundant in the lower reaches of the Swale and Ure than further upstream the interaction with effects due to residence time is likely to limit the influence of this control on phytoplankton growth. The potential effects of existing shading on chlorophyll-a were estimated in the present study but were not fully accounted for in terms of assessing scenarios.

### *2.6. Analysis of costs*

Costs for removing phosphorus at sewage treatment works in the Swale are based on a method used to support Defra policy on reducing phosphorus inputs to freshwaters (e.g. Defra, 2009). In this method, regressions relate detailed estimates of expenditure for a large sample of UK STWs to the known “people equivalents” served by each works. Calculation of costs for the “Agriculture Scenario” (C) covers the Ure headwaters and assumes that landowners receive compensation equating to the profit achievable from the lost agricultural land.  Profits are assumed to be approximately 25% of the change in farm gross margin (as modelled by Fezzi *et al*. (2010)), the exact fraction depending on farm type (Fezzi *et al.*, 2008). For the “Tree Scenario” (D) costs were estimated for planting at 5m intervals along both banks of all tributaries of the Ouse. The estimates were based on reports provided by Forest Research (Bill Jones, personal communication). It was assumed that labour costs contribute approximately 20% as, due to access constraints, most of the riverbank length would be more appropriately planted by hand than by machine. Other components to the costs were: plants (44%), guards for saplings (32%), infilling subsequent to initial planting (4%).

## 3. Results and discussion

### *3.1. Presentation and interpretation of results*

Time-series model performance was evaluated using the efficiency criterion (E) (Nash and Sutcliffe, 1970). Due largely to the paucity of chlorophyll-a observations in the Ouse network (see Section 2.2), a variety of graphical methods as well as the commonly used time-series plots of observed and simulated values (Figure 3: Swale and Ure) were used. Chlorophyll-a data were pooled together into two groups, one representing 8 upstream sites and the other representing 4 from the lower reaches downstream. Monthly median and 75$^{th}$ percentile (upper quartile) values were calculated for each group, compared with summary statistics from the daily model simulations for the same sites and displayed graphically (Figure 4). Individually for each of the 12 sites, comparison was made of observed and simulated 95$^{th}$ percentile chlorophyll-a values (Figure 5), hereafter referred to as peak levels, which represent elevated concentrations in the spring-summer growing season. For the Swale and Ure tributaries the longitudinal change downstream in modelled median spring/summer concentrations of nutrients and chlorophyll-a (Figure 6) illustrates the build up of flows and development of blooms down the system. Adding the projected plot for chlorophyll-a in 2080 shows a best estimate of how this may change in the future. Finally, time-series plots of model output illustrating the impact of scenarios in the Swale are shown (Figure 7).

### *3.2. In-river transformations and hydrochemical response*

Through the calibration process, QUESTOR modelling gave an indication of in-river transformation rates and their spatial variability in the Ouse (Table 2). Other studies reveal that these rates also vary temporally (Pattinson *et al*., 1998; Ani *et al.*, 2010). The network of site locations in the first column (relating Table 2 to Figure 1) provided the spatial framework for calibration. The calibration process is very necessary as a sensitivity analysis of previous QUESTOR applications in the Ouse network (Deflandre *et al*., 2006) suggested transformations to be at least as important as other elements of model structure revealed to be fundamental, namely a sound description of flow routing constants defining velocity (already optimised: see Boorman(2003c)) and a rigorous characterisation of tributary inputs. As well as being net sources of SRP, most reaches showed net nitrate sources (nitrification predominant) although some reaches in the middle Swale and upper Ure appeared to be net sinks (denitrification dominant). In the lower reaches of all rivers benthic oxygen demand and BOD decay were important. In keeping with previous applications of QUESTOR in the Ouse (Lewis *et al*., 1997; Boorman, 2003c) there was generally good agreement between model and observations for all determinands, correctly classifying chemistry in the majority of cases. Throughout the network, as illustrated for the Swale and Ure (Figure 3a), errors in flow simulation at low flows were mostly overestimates; these impinge on overall chemical model performance (Table 4 and Figure 3b, 3c). Nutrient concentrations may be underestimated for this reason although such errors can also be attributable to omission of a point source and there may be two possible instances of this: in the Nidd and in the Foss.

### *3.3. Biological response*

Optimised values for biological parameters (Table 3) were the same as those used before for the Ouse by Boorman (2003b), with the exception of the maximum phytoplankton growth rate, which was lower previously (3.0). Arriving at a higher value was a compromise, wherein peak levels were overestimated at some sites downstream (in the Swale (3/39) and Ouse (9/19)) whilst isolated moderately high values (approximately 50 μg/L) were underestimated in some of the smaller tributaries (e.g. Crimple Beck (1/8)). The death rate parameter was not altered. Wider impacts of phytoplankton on DO are controlled by whether or not and when the blooms are grazed, die or are flushed out of the freshwater system beyond the tidal limit. Although Whitehead and Hornberger (1984) and Balbi (2000) used phaeopigment data to infer degradation rates, little is known to help constrain uncertainty in death rate. Implications of this are only likely to be substantial in eutrophic rivers.

The expected spring-summer elevation in network-wide monthly values was a lot higher in the pool of downstream sites than in the upstream group (Figure 4). Values of the 75$^{th}$ percentile were closely reproduced by the model for all months in both groups, supporting the hypothesis of residence time being the dominant control on chlorophyll-a concentration. However, there was a systematic overestimation of monthly medians. Though the magnitude of phytoplankton blooms was simulated well, these blooms may be quicker forming and more ephemeral than represented by the model. In the Ouse network, regression analysis suggested that residence time, as represented by catchment area (x), could explain much of the variability in modelled peak chlorophyll-a ($y=0.012x +9.384$, $r^2=0.76$, $n=12$, $p<0.001$). Whilst this supports

Neal *et al.* (2006), much complexity is hidden, and other studies (e.g. Balbi, 2000) identify specific localised factors which may explain scatter in this relationship. At individual sites model performance for peak chlorophyll-a values was reasonable (Figure 5) but mismatches reveal the elusive nature and importance of these local factors. Data from 1993-1996 showed high phytoplankton levels in the Ure, higher than in the Swale and also than in the main Ouse downstream (Figure 5); and management plans covering the river network going back over 15 years (e.g. NRA, 1994; Environment Agency, 2009) have specifically highlighted eutrophication issues in the upper Ure. In contrast, the model simulated similar levels in the Ure and the Swale. Given the apparent sensitivity of phytoplankton growth to shading, the relatively higher concentrations in the Swale of light–attenuating suspended sediments (SS) (Table 1) may offer an explanation. The predominant source of the elevated levels in the Swale has been attributed to bank erosion (Lawler *et al.*, 1999). Furthermore, the middle reaches of the Swale are more densely shaded by trees than those in the Ure. Another plausible explanation is apparent from data on river channel properties (River Habitat Survey: Raven *et al*., 1998) which suggest that overall the Ure may be the widest river in the network, a feature not revealed by the limited data at gauging stations on which model setups to date have been based (Lewis *et al*., 1997). Consequently it is likely that patches of shallow water which heat up markedly during direct sunshine and enhance phytoplankton growth are more prevalent in the Ure than elsewhere. A future research priority should be to pinpoint how surveys of stream channel morphology can best help determine realistic river width and depth parameters for use by 1-D river quality models.

Aside from the effects of dilution due to inputs of low chlorophyll-a waters from short tributaries or STWs, the model simulated a gradual increase in peak levels along the system as residence time increases (Figure 6). This is reflected in the simulated classification of trophic status based on May-September median (Table 5): most of the network was oligotrophic, only changing to mesotrophic in the lower reaches of the Swale, Ure and Nidd and further downstream in the main Ouse. Simulated phytoplankton populations were significant between April and September, and were short-lived, being flushed out of the system by high flow events rather than dying or being grazed. Results from the lower reaches of the network showed that in 1997-1999 there was more than enough N and P for photosynthesis, even in the Ure (Figure 6b). The calculated growth limiting factors did not fall below 0.9 and 0.8 for N and P respectively. The main controls were light (for which the limiting factor frequently fell below 0.6, even in summer) and water temperature. In the headwaters upstream of the dominant STWs there may be slightly more nutrient limitation, as shown for the Ure and Swale (Figure 6, vis-a-vis half saturation constant values in Table 3). Whilst many empirical analyses show strong relationships between TP and phytoplankton both (i) in surveys across many river sites (e.g. Soballe and Kimmel, 1987; van Niewenhuyse and Jones, 1996; Royer *et al*., 2008) and (ii) in long-term high-resolution time-series data at individual sites (e.g. Kinniburgh and Barnett, 2009), this does not demonstrate that phosphorus limitation is critical to phytoplankton abundance. The QUESTOR modelling also revealed a link with TP, but it did not qualify as the major limiting factor.

The magnitude of chlorophyll-a peaks varied considerably from year-to-year. In comparing Figures 2 and 7 the model implied this variation to be strongly related to water temperature (and probably sunshine hours). Levels in 1998 were lower than in the other years modelled. Modelling in the present study did not include silicon limitation, nor did it embody the biological complexity that could result in a mid-summer suppression of phytoplankton growth. Highest values were simulated in July (Figure 4), later than observations suggest (May). However, as data are currently limited, evidence for shortcomings in the model predictive capability due to a lack of discrimination between different phytoplankton groups is not compelling.

*3.4. Climate change scenario (A)*

In the scenario used here, phytoplankton blooms are liable to increase dramatically in the Ouse network, changing the trophic status considerably (Table 5) despite the generally good chemical status of waters currently seen throughout. Predicted increases of over 150% in peak chlorophyll-a occurred in more than half the reaches. In the Ouse and the lower reaches of its main tributaries these peak values showed a three- to five-fold increase. The change in May-September median levels as flows and water residence time build up along two of the main tributaries (Swale and Ure) are compared with the present day picture (Figure 6). Shift from oligotrophic/mesotrophic status to one dominated by mesotrophic or eutrophic conditions is likely. Re-running the model scenario with present day flows reveals that effects attributable to projections of radiation and water temperature alone represented on average 53% of the future increase in chlorophyll-a on a reach-by-reach basis.

By 2080 the greatly enhanced biomass of phytoplankton and associated nutrient demand could render the levels of nutrient supply (as seen currently) to be of a substantial limiting influence on growth. It appears that under the mesotrophic-oligotrophic conditions of the present day Ouse network, light and heat are the predominant limiting factors, whereas in the projected future state the growth of blooms would, in certain reaches of the Ure, be limited by nutrient supply as well as by light. The effect is only seen in the Ure (Figure 6b), because in the Swale nutrient levels rapidly increase downstream (Figure 6a) due to the more considerable point source contribution. The outcome would result in severe blooms being ephemerally curtailed either by nutrient supply, or, as is the case with the less severe blooms seen currently, by flow events. Therefore it seems that the influence of nutrient limitation would be to prevent blooms from becoming prolonged. However, beneficial effects would be short lived, probably not substantial and have minimal influence on the initial establishment of phytoplankton populations in the spring.

*3.5. Management-related scenarios (B, C and D)*

Carrying out the “STW Scenario” (B), although reducing mean annual SRP by 37% (from 210 to 133 µg/L), will only reduce peak chlorophyll-a by 11% (from 26.1 to 23.1 µg/L) in the Swale lower reaches (3/39) at Thornton Manor (Figure 7). Implementing measures of this type is underway in much of the UK. In the Swale this implementation should make achieving WFD standards for SRP realistic but in itself will probably do little for river ecology. Nevertheless, if TP concentrations from STWs were further reduced to 0.1 mg/L, effectively eliminating point sources, it is interesting to note that though the additional decrease in river SRP at Thornton Manor achieved (to 111 µg/L) would be small, it would be accompanied by a substantial further suppression of phytoplankton (to 20.6 µg chl-a/L, a 21% decrease relative to the baseline). However, if, in addition to cutting TP in STW effluents to 2.0 mg/L, the entire River Swale channel (and its main tributaries) were also to be shaded from incoming solar radiation ("Tree Scenario" (D)), though no further reduction in river SRP would occur, a much larger overall reduction in peak chlorophyll-a at Thornton Manor (44%) would be achievable.

The "Agriculture Scenario" (C) may discernibly reduce peak chlorophyll-a levels, although it is only substantially effective in areas of negligible point source input. A 10% reduction for the Ure at Boroughbridge (4/46) (from 33.3 to 30.1µg/L) is predicted. Although the beneficial effect in the Nidd (2/44) is much less (2%), some reduction was predicted for the main Ouse (2%) despite dilution from other rivers unaffected by the scenario. The greater effectiveness of the measure in the Ure is not surprising given that nutrient levels are currently lower than elsewhere. Alternatively however, implementation of the "Tree Scenario" (D) is far more effective, reducing peak chlorophyll-a by 47% in the Ure (to 17.7 µg/L). A similar level of reduction is seen in the Nidd. The effects of shading on solar radiation reaching the water column in the Swale, Ure and Nidd are less than those projected for the narrower (typically 12m wide) Nete in Belgium (Ghermandi *et al.*, 2009), although the impacts on chlorophyll-a appear to be similar.

If current levels of riparian shading are accounted for in model calculations any reduction in effectiveness of this scenario appears likely to be small (approximately 7%). So, for combating the likely detrimental effects of climate change, it is concluded that riparian shading is the most appropriate of the management strategies considered here. Model estimates suggest shading could prevent 88% of the projected increase in peak chlorophyll-a in the Ure. Shading is also the least costly option (Table 6). Of the £0.27m quoted, approximately £0.15m accounts for planting in the Ure and Swale.

The model study shows that it is important to prioritise mitigation actions specific to phytoplankton in headwater areas where the water travel time in the freshwater environment is longest. In contrast, much attention in the past has focused on the lower parts of catchments near tidal limits which often have intense land-use and most severe pollution. Importantly, phytoplankton levels in the lower reaches of large rivers are heavily dependent and sensitive to management decisions made in upstream headwater areas. Whilst there may be more scope to reduce pollutant leaching and improve river chemistry in the lower parts of catchments, it appears that such effort would have little impact on river phytoplankton.

### *3.6. Summary of modelling outcomes*

- Most reaches in the system are net nutrient sources;
- Simulated chlorophyll-a levels and their increase downstream was broadly in line with the limited observations available;
- Model performance for flow and chemical parameters in the Swale and Ure was acceptable in providing a basis for scenario analysis.
- Under climate change projections, downstream parts of the Ouse network are likely to become eutrophic, roughly half of the increase being attributable to changes in temperature and radiation;
- Reducing the inputs of phosphorus from point sources in the Swale catchment will lower the peak chlorophyll-a levels but probably not by more than 10%;

A similar level of improvement is likely if land in the headwaters of the Ure were to be taken out of agricultural production;
Planting riparian trees is a considerably more promising measure and is more cost-effective than the others tested, reducing peak chlorophyll-a levels by more than 30-40%, and probably one capable of offsetting most of the detrimental impacts of climate change.

**4. Conclusion**

The findings of this study are of UK- and European-wide relevance. In moving towards WFD implementation, UKTAG (2008) state the importance of phytoplankton biomass as an important metric due to its influence on light penetration and oxygen in water bodies. Whilst the predictions should be refined and updated as increasingly-substantiated climate change projections become available, the study highlights the danger of a currently clean river system (in terms of chemical and biological indicators) becoming considerably impacted with phytoplankton blooms by 2080. Greater confidence in our knowledge of the system as represented in models is essential as we face an uncertain climatic future. To this end more comprehensive long-term monitoring will be vital. Continuous monitoring technology for DO (Williams *et al.*, 2000) and chlorophyll-a has become more commonplace recently. Coupled with the possibilities afforded by earth observation to record the development of blooms down a river system (Hunter *et al.*, 2008) this will aid understanding of the wider impacts of phytoplankton, particularly on DO and fish habitats. In this context, a shorter model time-step (e.g. hourly) would be advisable.

The main issues arising from the modelling study are of much wider relevance than just to the Ouse and are as follows:
Whilst between-year variability at any site is likely to be large, residence time is the dominant control on phytoplankton in the Ouse network. However, reconciling model results with observations reveals (i) likelihood of important local controls and (ii) need to consider in more detail the complexity of the biological system;
Phytoplankton growth does not appear to be strongly limited by current levels of nutrient supply;
Substantial nutrient limitation, whereby peak chlorophyll-a is cut by over 20%, is only likely if nutrient loads are cut to levels seen in near-pristine environments where STW and agricultural influence is negligible;
In terms of river quality, management measures required of the scenarios will take many years (decades) to reach full effectiveness;
WFD compliance could be attained more cost-effectively by controlling light conditions through riparian shading rather than curtailing nutrient inputs.
In general, any mitigation strategy related to phytoplankton is more likely to be effective if targeted in headwater areas.

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**References**

Ani EC, Hutchins MG, Kraslawski A, Agachi PS. Development of an advection-dispersion model and its use in identifying variability in nitrogen transformations in large rivers. Rivers, Research and Applications 2010; DOI: 10.1002/rra.1418.

Anthony SG, Quinn P, Lord EI. Catchment scale modelling of nitrate leaching. Aspects of Applied Biology 1996; 46: 23-32.

Arnold JG, Srinivasan, R., Muttiah, RS, Williams, JR. Large area hydrologic modelling and assessment. Part 1: model development. Journal of the American Water Resources Association 1998; 34: 73-89.

Balbi DM. Suspended chlorophyll in the River Nene, a small nutrient-rich river in eastern England: long-term and spatial trends. The Science of the Total Environment 2000; 251-252: 401-421.

Boorman DB. LOIS in-stream water quality modelling. Part 1. Catchments and methods. The Science of the Total Environment 2003a; 314: 379-395.

Boorman DB. Climate, Hydrochemistry and Economics of Surface-water Systems (CHESS): adding a European dimension to the catchment modeling experience developed under LOIS. The Science of the Total Environment 2003b; 314-316: 411-437.

Boorman DB. LOIS in-stream water quality modelling. Part 2. Results and scenarios. The Science of the Total Environment 2003c; 314-316: 397-409.

Boorman DB, Hollis JM, Lilly A. Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom. 126. Report no. 126, Institute of Hydrology, Wallingford, UK, 1995,

Boothroyd IKG, Quinn JM, Langer ERL, Costley KJ, Steward G. Riparian buffers mitigate effects of pine plantation logging on New Zealand streams. 1. Riparian vegetation structure, stream geomorphology and periphyton. Forest Ecology and Management 2004: 194: 199-213.

Bowes MJ, House WA. Phosphorus and dissolved silicon dynamics in the River Swale catchment, UK: a mass balance approach. Hydrological Processes 2001; 15: 261-280.

Bowie GL, Mills WB, Pocella DP, Campbell CL, Pagenkopf JR, Rupp GL, Johnson KM, Chan PWH, Gherini S, Chamberlain CE. Rates, constants and kinetic formulations in surface water quality modelling. US Environment Protection Agency, Georgia USA, 1985,

Caissie D, Satish NG, El-Jabi N. Predicting water temperature using a deterministic model: application on Miramichi River catchments (New Brunswick Canada). Journal of Hydrology 2007; 336: 303-315.

Carey RO, Migliaccio KW. Contribution of wasterwater treatment plant effluents to nutrient dynamics in aquatic systems: a review. Environmental Management 2009; 44: 205-217.
Cherry KA, Shepherd M, Withers PJA, Mooney SJ. Assessing the effectiveness of actions to mitigate nutrient loss from agriculture: A review of methods. The Science of the Total Environment 2008; 406: 1-23.
Cooper DM, Naden PS. Approaches to delivery modelling in LOIS. The Science of the Total Environment 1998; 210/211: 483-498.
Cox BA. A review of dissolved oxygen modelling techniques for lowland rivers. The Science of the Total Environment 2003; 314-316: 303-334.
Cox BA, Whitehead PG. Impacts of climate change scenarios on dissolved oxygen in the River Thames, UK. Hydrology Research 2009; 40: 138-152.
Davies-Colley RJ, Quinn JM. Stream lighting in five regions of North Island New Zealand: control by channel size and riparian vegetation. New Zealand Journal of Marine and Freshwater Research 1998; 32: 591-605.
Davison P, Withers PJA, Lord EI, Betson MJ, Stromqvist J. PSYCHIC - a process based model of phosphorus and sediment mobilisation and delivery within agricultural catchment. Part 1: Model description and parameterisation. Journal of Hydrology 2008; 350: 290-302.
Deflandre A, Williams RJ, Elorza, FJ, Mira J, Boorman, DB. Analysis of the QUESTOR water quality model using a Fourier amplitude sensitivity test (FAST) for two UK rivers. The Science of the Total Environment 2006; 360: 290-304.
Defra, 2009. Consultation on Update to the 2005 Detergents Regulations: Changes to derogation procedures and the implementation of a ban on inorganic phosphates in domestic laundry cleaning products. Department for Environment, Food and Rural Affairs, London. October 2009. Accessed at: http://www.defra.gov.uk/corporate/consult/environment.htm (22nd March 2010)
DeWalle DR. Guidelines for riparian vegetative shade restoration based upon a theoretical shaded-stream model. Journal of the American Water Resources Association 2008; 44: 1373-1387.
Dodds WK, Jones JR, Welch EB. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen and phosphorus. Water Research 1998; 32: 1455-1462.
Eatherall A, Boorman DB, Williams RJ, Kowe R. Modelling in-stream water quality in LOIS. The Science of the Total Environment 1998; 210/211: 499-517.
Ferguson AJD. The role of modelling in the control of toxic blue-green algae. Hydrobiologia 1997; 349: 1-4.
Environment Agency. River Basin Management Plan, Humber River Basin District, 2009.
Fezzi C, Rigby D, Bateman IJ, Hadley D, Posen P. Estimating the range of economic impacts on farms of nutrient leaching reduction policies. Agricultural Economics 2008; 39: 197-205.
Fezzi C, Hutchins MG, Rigby D, Bateman IJ, Posen P, Hadley D. Integrated assessment of Water Framework Directive nitrate reduction impacts. Agricultural Economics 2010; 41: 123-134.
Fuller RM, Smith GM, Sanderson JM, Hill RA, Thompson AG. The UK Land Cover Map 2000: Construction of a parcel-based vector map from satellite images. The Cartographic Journal 2002; 39: 15-25.

Garnier J, Billen G, Coste M. Seasonal Succession of Diatoms and Chlorophyceae in the Drainage Network of the Seine River - Observations and Modeling. Limnology and Oceanography 1995; 40: 750-765.
Ghermandi A, Vandenberghe V, Benedetti L, Bauwens W, Vanrolleghem PA. Model-based assessment of shading effect by riparian vegetation on river water quality. Ecological Engineering 2009; 35: 92-104.
Gordon N, Finlayson B, McMahon T, Gippel C. Stream Hydrology: an Introduction for Ecologists. Wiley and sons, 2004.
Gosselain V, Viroux L, Descy JP. Can a community of small-bodied grazers control phytoplankton in rivers? Freshwater Biology 1998; 39: 9-24.
Haygarth PM, ApSimon H, Betson M, Harris D, Hodgkinson R, Withers PJA. Mitigating Diffuse Phosphorus Transfer from Agriculture According to Cost and Efficiency. Journal of Environmental Quality 2009; 38: 2012-2022.
Henderson R, Chips M, Cornwell N, Hitchins P, Holden B, Hurley S, Parsons SA, Wetherill A, Jefferson B. Experiences of algae in UK waters: a treatment perspective. Water and Environment Journal 2008; 22: 184-192.
Hilton J, O'Hare M, Bowes MJ, Jones JI. How green is my river? A new paradigm of eutrophication in rivers. The Science of the Total Environment 2006; 365: 66-83.
Hunter PD, Tyler AN, Willby NJ, Gilvear DJ. The spatial dynamics of vertical migration by Microcystis aeruginosa in a eutrophic shallow lake: A case study using high spatial resolution time-series airborne remote sensing. Limnology and Oceanography 2008; 53: 2391-2406.
Hutchins MG, Anthony SG, Hodgkinson RA, Withers PJA. Modularised process-based modelling of phosphorus loss at farm and catchment scale. Hydrology and Earth Systems Science 2002; 6: 1017-1030.
Hutchins MG, Deflandre-Vlandas A, Posen P, Davies H. How do river nitrate concentrations respond to changes in land-use? A modelling case study of headwaters in the River Derwent catchment, North Yorkshire UK. Environmental Modelling and Assessment 2010; 15: 93-109. DOI: 10.1007/s10666-009-9218-2
Hutchins MG, Fezzi C, Bateman IJ, Posen PE, Deflandre-Vlandas A. Cost-effective mitigation of diffuse pollution: setting criteria for river basin management at multiple locations. Environmental Management 2009; 44: 256-267.
Jarvie HP, Neal C, Withers PJA. Sewage-effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus? The Science of the Total Environment 2006; 360: 246-253.
Johnes PJ. Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. Journal of Hydrology 1996; 183: 323-349.
Johnson AC, Acreman MC, Dunbar MJ, Feist SW, Giacomello AM, Gozlan RE, Hinsley SA, Ibbotson AT, Jarvie HP, Jones JI, Longshaw M, Maberly SC, Marsh TJ, Neal C, Newman JR, Nunn MA, Pickup RW, Reynard NS, Sullivan CA, Sumpter JP, Williams RJ. The British river of the future: How climate change and human activity might affect two contrasting river ecosystems in England. The Science of the Total Environment 2009; 407: 4787-4798.
Kelly MG. Use of the trophic diatom index to monitor eutrophication in rivers. Water Research 1998; 32: 236-242.

Kinniburgh JH, Barnett M. Orthophosphate concentrations in the River Thames: reductions in the past decade. Water and Environment Journal 2009; online 3. Mar 2009:

Lawler DM, Grove JR, Couperthwaite JS, Leeks GJL. Downstream change in river bank erosion rates in the Swale-Ouse system, northern England. Hydrological Processes 1999; 13: 977-992.

Leeks GJL, Jarvie HP. Introduction to the Land-Ocean Interaction Study (LOIS); rationale and international context. The Science of the Total Environment 1998; 210/211: 5-20.

Lewis DR, Williams RJ, Whitehead PG. Quality simulation along rivers (QUASAR): an application to the Yorkshire Ouse. The Science of the Total Environment 1997; 194/195: 399-418.

Nash J, Sutcliffe J. River flow forecasting through conceptual models: part 1 - a discussion of principles. Journal of Hydrology 1970; 10: 282-290.

Neal C, Robson AJ. A summary of river water quality data collected within the Land-Ocean Interaction Study: core data for eastern UK rivers draining to the North Sea. The Science of the Total Environment 2000; 251/252: 585-665.

Neal C, Hilton J, Wade AJ, Neal M, Wickham H. Chlorophyll-a in the rivers of eastern England. The Science of the Total Environment 2006; 365: 84-104.

Neal C, Jarvie HP, Withers PJA, Whitton BA, Neal M. The strategic significance of wastewater sources to pollutant phosphorus levels in English rivers and to environmental management for rural, agricultural and urban catchments. The Science of the Total Environment 2010a; 408: 1485-1500.

Neal C, Jarvie HP, Williams R, Love A, Neal M, Wickham H, Harman S, Armstrong L. Declines in phosphorus concentration in the upper River Thames (UK): links to sewage effluent cleanup and extended end-member mixing analysis. The Science of the Total Environment 2010b; 408: 1315-1330.

NRA. Rivers Swale, Ure and Ouse catchment management plan consultation report. National Rivers Authority, Northumbria and Yorkshire Region, 1994.

Pattinson SN, Garcia-Ruiz R, Whitton, BA. Spatial and seasonal variation in denitrification in the Swale-Ouse system, a river continuum. The Science of the Total Environment 1998; 210/211: 289-305.

Posen PE, Hutchins MG, Lovett AA, Davies HN. Identifying the catchment size at which robust estimations of agricultural land use can be made and implications for diffuse pollution modelling. Applied Geography, submitted.

Pretty JN, Mason CF, Nedwell DB, Hine RE, Leaf S, Dils R. Environmental costs of eutrophication in England and Wales. Environmental Science and Technology 2003; 37: 201-208.

Raven PJ, Holmes NTH, Dawson FH, Everard M. Quality assessment using River Habitat Survey data. Aquatic Conservation: Marine and Freshwater Ecosystems 1998; 8: 477-499.

Reichert P, Borchardt D, Henze M, Rauch W, Shanahan P, Somlyody L, Vantolleghem P. River Water Quality Model no. 1 (RWQM1): II. Biochemical process equations. Water Science and Technology 2001; 43: 11-30.

Reynolds CS, Irish AE, Elliott JA. The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). Ecological Modelling 2001; 140: 271-291.

Royer TV, David MB, Gentry LE, Mitchell CA, Starks KM, Heatherly T, While MR. Assessment of chlorophyll-a as a criterion for establishing nutrient standards

in the streams and rivers of Illinois. Journal of Environmental Quality 2008; 37: 437-447.
Scharfe M, Callies U, Blocker G, Petersen W, Schroeder F. A simple Lagrangian model to simulate temporal variability of algae in the Elbe River. Ecological Modelling 2009; 220: 2173-2186.
Silgram M, Kirkham F, Mole A, Hutchins MG. Nitrogen losses from non-agricultural land in England and Wales: a review. Final Report for Defra 2004. ADAS Wolverhampton.
Silgram M, Schoumans OF, Walvoort DJJ, Anthony SG, Groenendijk P, Stromqvist J, Bouraoui F, Arheimer B, Kapetanaki M, Lo Porto A, Martensson K. Subannual models for catchment management: evaluating model performance on three European catchments. Journal of Environmental Monitoring 2009; 11: 526-539.
Skidmore RE, Maberly SC, Whitton BA. Patterns of spatial and temporal variation in phytoplankton chlorophyll a in the River Trent and its tributaries. The Science of the Total Environment 1998; 210: 357-365.
Soballe DM, Kimmel BL. A large-scale comparison of factors influencing phytoplankton abundance in rivers, lakes and impoundments. Ecology 1987; 68: 1943-1954.
Steele JH. Environmental control of photosynthesis in the sea. Limnology and Oceanography 1962; 7: 137-149.
UKTAG. UK Environmental Standards and Conditions (Phase 1). UK Technical Advisory Group of the Water Framework Directive, Final report April 2008, 2008.
van Niewenhuyse E, Jones JR. Phosphorus-chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fish and Aquatic Sciences 1996; 53: 99-105.
van Vliet MTH, Zwolsman JJG. Impact of summer droughts on the water quality of the Meuse river. Journal of Hydrology 2008; 353: 1-17.
Wade AJ, Whitehead PG, Butterfield D. The Integrated Catchments model of Phosphorus dynamics (INCA-P), a new approach for multiple source assessment in heterogeneous river systems: model structure and equations. Hydrology and Earth System Sciences 2002; 6: 583-606.
White PJ, Hammond JP. The source of phosphorus in waters of Great Britain. Journal of Environmental Quality 2009; 38: 13-26.
Whitehead PG, Hornberger GM. Modelling algal behaviour in the River Thames. Water Research 1984; 18: 945-953.
Whitehead PG, Williams RJ, Lewis DR. Quality simulation along river systems (QUASAR): model theory and development. The Science of the Total Environment 1997; 194/195: 447-456.
Whitehead PG, Wilby RL, Battarbee RW, Kernan M, Wade AJ. A review of the potential impacts of climate change on surface water quality. Hydrological Sciences Journal 2009; 54: 101-123.
Williams RJ, White C, Harrow ML, C. N. Temporal and small-scale spatial variations of dissolved oxygen in the Rivers Thames, Pang and Kennet, UK. The Science of the Total Environment 2000; 251-252: 497-510.

## Tables

| | Swale: at site 3/39 (see Figure 1) | Ure: at site 4/46 (see Figure 1) |
|---|---|---|
| [1]Dominant substrate | Cobbles, sand bank (gravel further upstream) | Gravel/pebble (cobble further upstream) |
| [1]Macrophyte presence | Some submerged/emergent | Largely absent |
| [2] Mean velocity at (i) mean flow (ii) low (Q90) flow (m/s) | (i) 0.56 (ii) 0.22 | (i) 0.59 (ii) 0.20 |
| [3]Bank width (m) | 15 | 35 |
| [4]Alkalinity (meq/L) | 2.97 | 2.50 |
| [4]Suspended sediment (mg/L) | 26.6 | 13.9 |

Sources: 1) Raven *et al*. (1998), Pattinson *et al*. (1998). 2) Velocity calculations were made along river stretches starting upstream at sites 27090 (Swale) and 27034 (Ure) respectively (see Figure 1). 3) Raven *et al*. (1998). 4) Neal and Robson (2000)

Table 1: Characteristics of the Swale and Ure river stretches.

| Reach | Nitrification | Denitrification | Benthic Oxygen demand | BOD decay | BOD sedimentation | P mineralisation |
|---|---|---|---|---|---|---|
| 3/10 | 2 | 0 | 0 | 0 | 0 | 0 |
| 3/17 | 4 | 0 | 0.5 | 0 | 0 | 1 |
| 3/32 | 2 | 1.25 | 1.25 | 0 | 0 | 0 |
| 3/39* | 5 | 0 | 2.5 | 0 | 0 | 12 |
| 6/7 | 12 | 0 | 4 | 2.5 | 0 | 7.5 |
| 7/2 | 0 | 0.75 | 0 | 0 | 0 | 0 |
| 7/3 | 6 | 0 | 0 | 0 | 1.5 | 8 |
| 7/4 | 7 | 0 | 0 | 0 | 2 | 8 |
| 4/8 | 0 | 0 | 0 | 0 | 0 | 0 |
| 4/12 | 4 | 8 | 0 | 0 | 0 | 0 |
| 4/19 | 10 | 0 | 0 | 0 | 0 | 0 |
| 4/31 | 15 | 0 | 0 | 1 | 0 | 0 |
| 4/38 | 15 | 0 | 0 | 1 | 0 | 1 |
| 4/46 | 15 | 0 | 0.5 | 0 | 0 | 0 |
| 2/5 | 10 | 0 | 0.25 | 6 | 0 | 0 |
| 2/8 | 15 | 0 | 3.5 | 0 | 0 | 0 |
| 2/9 | 7.5 | 0 | 2.5 | 0 | 0 | 0 |
| 2/10 | 7.5 | 0 | 3.5 | 0 | 0 | 4 |
| 2/12 | 0 | 0 | 0 | 5 | 0 | 6 |
| 2/21 | 8 | 0 | 1 | 0 | 0 | 8 |
| 2/25 | 8 | 0 | 1 | 1 | 0 | 6 |
| 2/27 | 5 | 0 | 0 | 1 | 0 | 2 |
| 2/31 | 5 | 0 | 0 | 1 | 1 | 2.5 |
| 2/44 | 7.5 | 0 | 1 | 1.5 | 0 | 6 |
| 1/10 | 6 | 0 | 2 | 1.5 | 0 | 10 |
| 8/2 | 0 | 0 | 3 | 0 | 0 | 4 |
| 8/3 | 0 | 0 | 0 | 0 | 0 | 4 |
| 8/10 | 0 | 0 | 4 | 0.75 | 0 | 4 |
| 9/4 | 2 | 0 | 1 | 0 | 0 | 0 |
| 9/19 | 15 | 0 | 1.5 | 0.75 | 0 | 6 |

*deamination at 3 day$^{-1}$ (ammonium source and organic N sink). Absent elsewhere.

Table 2: Water quality transformation rate coefficients for river stretches (day$^{-1}$). Stretches comprise one or more reaches. The denoted reach (see Figure 1) is the downstream end of the stretch.

| Parameter | Value | Units/comments |
|---|---|---|
| Maximum phytoplankton growth rate ($k_{ref}$) | 4 | $day^{-1}$ |
| Maximum phytoplankton respiration rate | 0.005 | $day^{-1}$ (as fraction o f $k_{ref}$) |
| Maximum phytoplankton death rate | 0.005 | $day^{-1}$ (as fraction o f $k_{ref}$) |
| Half-saturation constant for N in phytoplankton ($k_N$) | 0.1 | mg N/L |
| Half-saturation constant for P in phytoplankton ($k_P$) | 0.01 | mg P/L |
| Arrhenius factor for temperature dependencies ($\theta$) | 1.08 | |
| Stoichiometric ratios in phytoplankton biomass | 1:50:10:1 | Chl-a:C:N:P |
| Optimum light intensity for phytoplankton ($L_{opt}$) | 60 | W $m^{-2}$ |
| Light extinction coefficient with depth in clean water ($\gamma_{base}$) | 0.01 | $m^{-1}$ |
| Light attenuation with depth due to suspended sediment ($L_{SS}$) | 0.01 | $m^{-1}$ $mg^{-1}$ L |
| Light attenuation with depth due to phytoplankton ($L_{Phy}$) | 10 | $m^{-1}$ $mg^{-1}$ L |
| Fraction of incoming radiation that is visible light ($L_1$) | 0.5 | |
| Fraction of visible light useful for photosynthesis ($L_2$) | 0.5 | |
| Fraction of light reaching water surface not reflected | 0.6 | |

Table 3: Values of global biologically-related parameters (source: Bowie *et al*., 1985)

| | observed | simulated | Time-series model performance (E value) |
|---|---|---|---|
| Swale: 90th percentile mg BOD/L | 2.7 | 2.4 | BOD: -0.242 |
| Swale: 10th percentile mg DO/L | 8.5 | 8.9 | DO: 0.474 |
| Swale: median mg $NO_3$-N/L | 4.4 | 3.8 | $NO_3$-N: 0.087 |
| Swale: 90th percentile mg $NH_4$-N/L | 0.22 | 0.12 | $NH_4$-N: 0.066 |
| Swale: median mg SRP/L | 0.21 | 0.13 | SRP: -0.700 |
| Ure: 90th percentile mg BOD/L | 2.3 | 1.8 | BOD: 0.249 |
| Ure: 10th percentile mg DO/L | 8.4 | 10.0 | DO: 0.355 |
| Ure: median mg $NO_3$-N/L | 2.6 | 1.8 | $NO_3$-N: -0.927 |
| Ure: 90th percentile mg $NH_4$-N/L | 0.09 | 0.10 | $NH_4$-N: -4.483 |
| Ure: median mg SRP/L | 0.06 | 0.07 | SRP: 0.056 |

Table 4: Water quality model performance at Swale (3/39) and Ure (4/46) sites

| | **oligotrophic** | **mesotrophic**; c.f. meso-eutrophic (Hilton *et al*, 2006) | **eutrophic** | **hyper-eutrophic** |
|---|---|---|---|---|
| May-Sept median chl-a (after Dodds *et al*. 1998) | <10 | 10-30 | 30-80 | >80 |
| Median annual chl-a (after Gordon *et al*., 2004) | <4 | 4-10 | 10-25 | >25 |
| Number of reaches in Ouse network: current baseline | 154 | 51 | 0 | 0 |
| Number of reaches in Ouse network: 2080 projection | 96 | 74 | 35 | 0 |

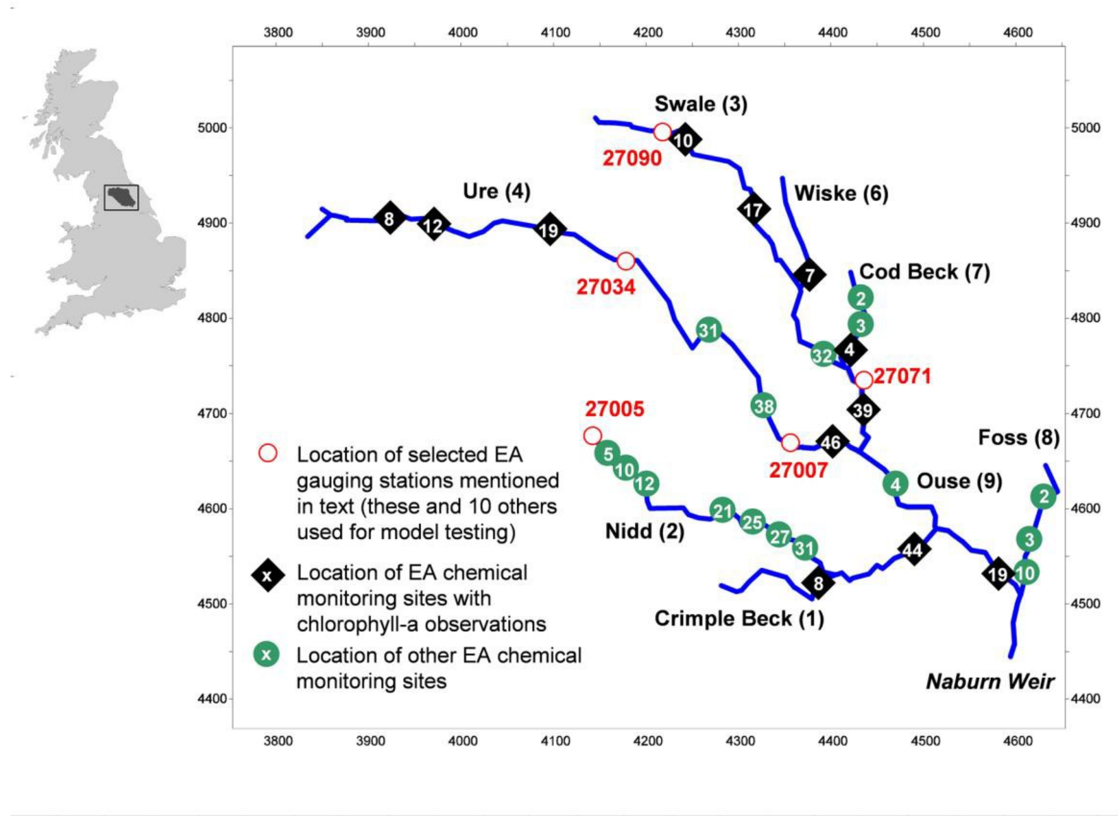
Table 5: Classification scheme for trophic status based on chlorophyll-a concentration (μg/L), and use of the system based on May-Sept median in the Ouse network modelling.

| Scenario | Initial cost (£m) | Rwming cost (£m/yr) | Total cost over a 20 year period (£m) |
|---|---|---|---|
| B. "STW scenario" | 14.5 | 0.68 | 28.1 |
| C. "Agriculture scenario" | | 1.0 | 20.0 |
| D. "Tree scenario" | 0.27 | | 0.27 |

Table 6: Economic costs of the management-related scenarios in the Yorkshire Ouse

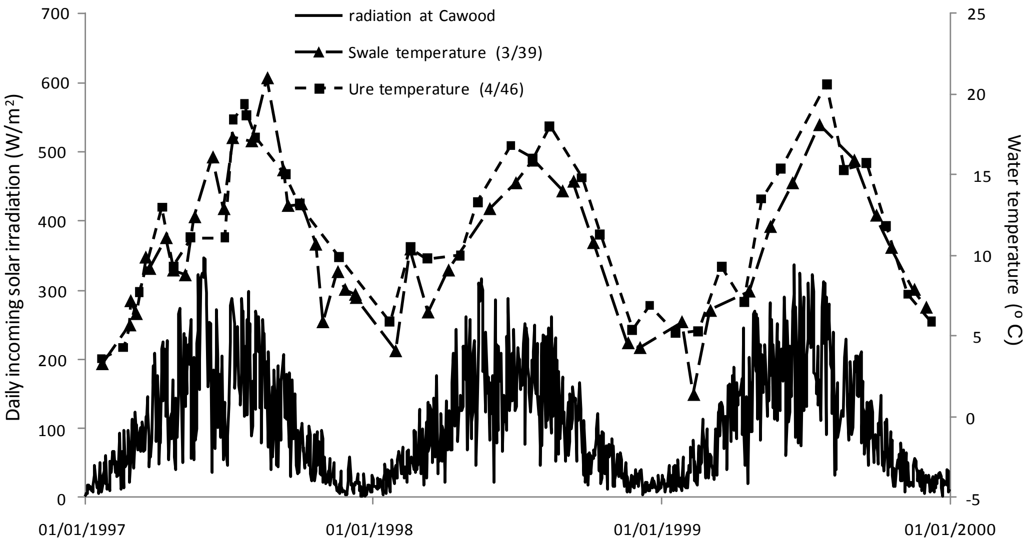
**Figure captions**

Figure 1: Map showing the Ouse river network, the location of gauging stations and water quality monitoring sites. As an example of notation the most downstream monitoring site on the Ure is 4/46.
Figure 2: Examples of time-series data used as model input (radiation and water temperature)
Figure 3: Time series of observed and simulated data (a) flow in the Swale (27071) and Ure (27007) (b) nutrients, DO and BOD in the Swale (at 3/39) (c) nutrients, DO and BOD in the Ure (at 4/46)
Figure 4: Comparison of observed and simulated monthly (a) median and (b) 75$^{th}$ percentile level chlorophyll-a concentrations. Upstream values represent pooling of data from sites 1/8, 3/10, 3/17, 4/8, 4/12, 4/19, 6/7 and 7/4; downstream values represent pooling of data from sites 2/44, 3/39, 4/46 and 9/19.
Figure 5: Observed and simulated chlorophyll-a 95$^{th}$ percentile values for each site (IDs relate to those in Figure 2)
Figure 6: Simulated 1997-1999 downstream changes in nutrient and chlorophyll-a concentration (together with the 2080 projected chlorophyll-a concentrations under the “climate change scenario” (A)) along (a) Swale and (b) Ure. Possible areas where nutrient limitation may occur are indicated by shaded ellipses.
Figure 7: Baseline and scenario (B (“STW scenario”) and D (“tree scenario”)) time-series model response of SRP and chlorophyll-a in the Swale at Thornton Manor (site 3/39).



Swale (3)
27090
Ure (4)
Wiske (6)
27034
Cod Beck (7)
27071
27005
27007
Foss (8)
Ouse (9)
Nidd (2)
Crimple Beck (1)
Naburn Weir
Location of selected EA gauging stations mentioned in text (these and 10 others used for model testing)
Location of EA chemical monitoring sites with chlorophyll-a observations
Location of other EA chemical monitoring sites


Figure 1



700
600
500
400
300
200
100
0
Daily incoming solar irradiation (W/m²)
25
20
15
10
5
0
-5
Water temperature (° C)
radiation at Cawood
Swale temperature (3/39)
Ure temperature (4/46)
01/01/1997
01/01/1998
01/01/1999
01/01/2000


Figure 2

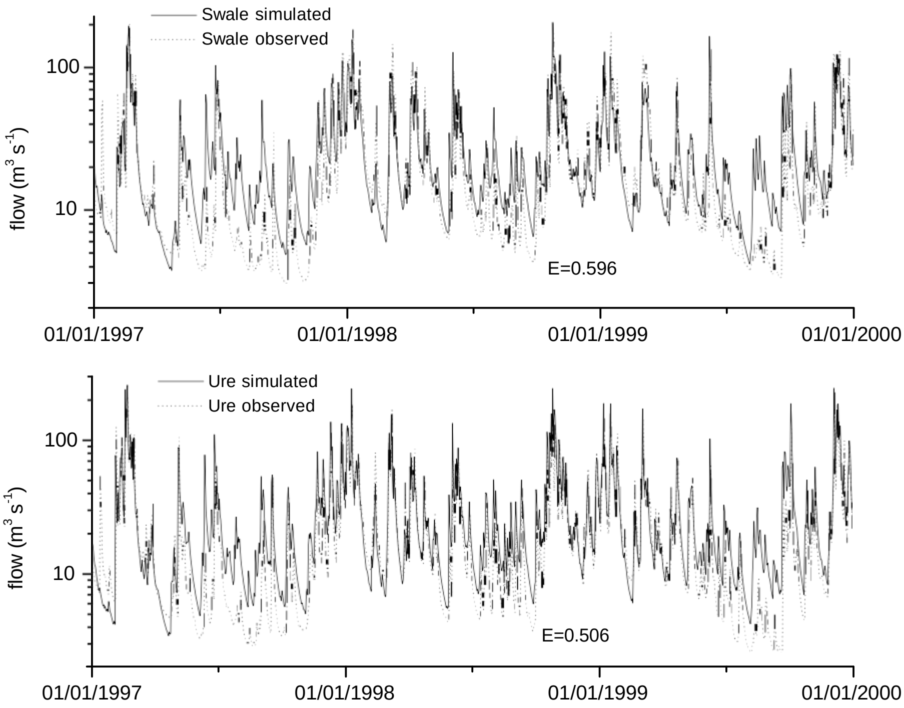

Swale simulated
Swale observed
100
10
flow (m$^3$ s$^{-1}$)
E=0.596
01/01/1997
01/01/1998
01/01/1999
01/01/2000
Ure simulated
Ure observed
100
10
flow (m$^3$ s$^{-1}$)
E=0.506
01/01/1997
01/01/1998
01/01/1999
01/01/2000

Figure 3a

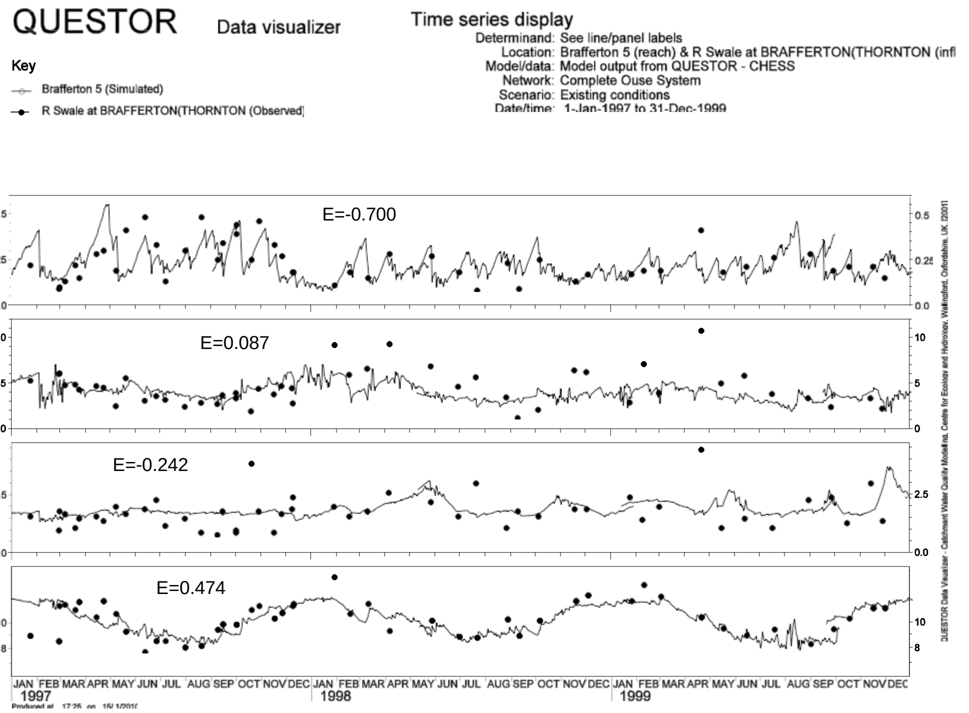
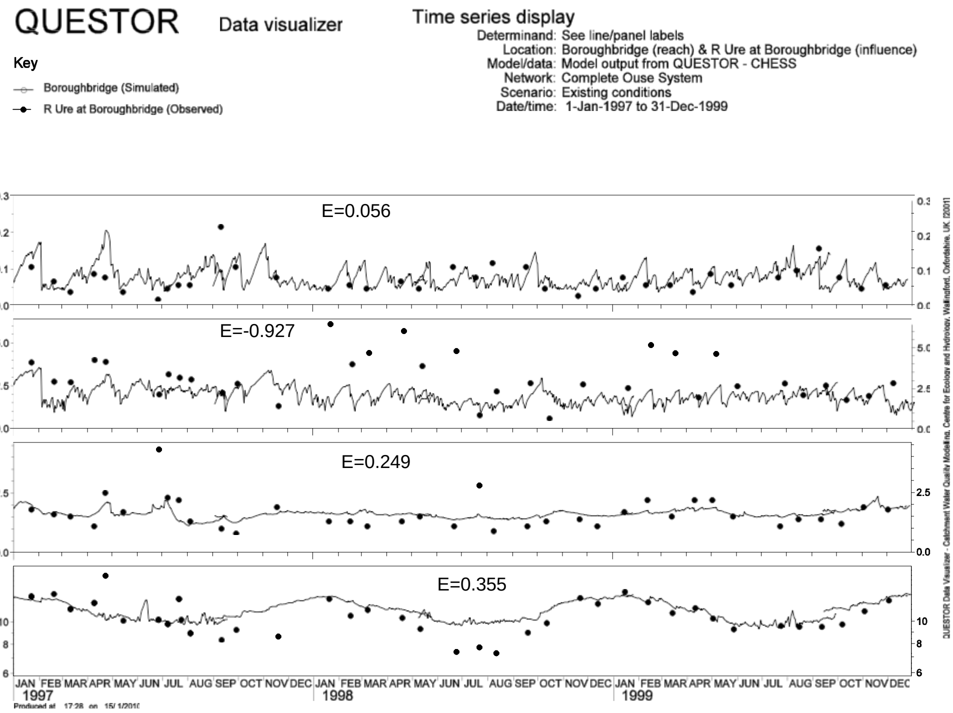

QUESTOR
Data visualizer
Time series display
Key
Brafferton 5 (Simulated)
R Swale at BRAFFERTON(THORNTON (Observed)
Determinand: See line/panel labels
Location: Brafferton 5 (reach) & R Swale at BRAFFERTON(THORNTON (infl
Model/data: Model output from QUESTOR - CHESS
Network: Complete Ouse System
Scenario: Existing conditions
E=-0.700
E=0.087
E=-0.242
E=0.474
BOD (mg l⁻¹)
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
1997
1998
1999

Figure 3b



QUESTOR
Data visualizer
Time series display
Determinand: See line/panel labels
Location: Boroughbridge (reach) & R Ure at Boroughbridge (influence)
Model/data: Model output from QUESTOR - CHESS
Network: Complete Ouse System
Scenario: Existing conditions
Date/time: 1-Jan-1997 to 31-Dec-1999
Key
Boroughbridge (Simulated)
R Ure at Boroughbridge (Observed)
E=0.056
E=-0.927
E=0.249
E=0.355
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
1997
1998
1999


Figure 3c

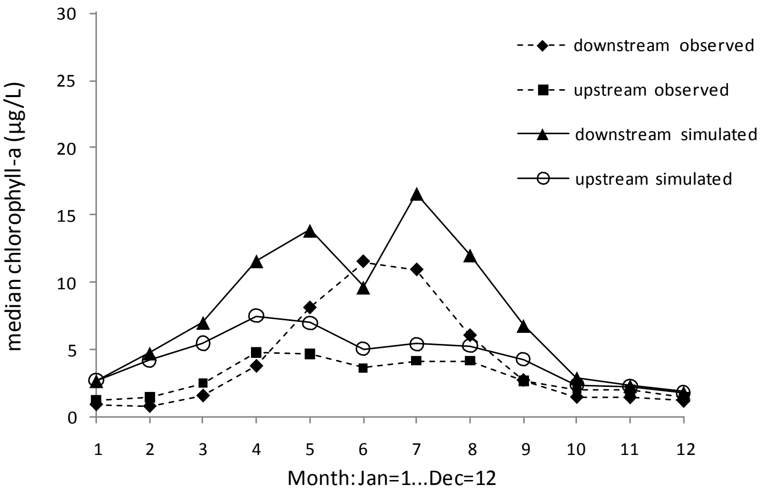

30
25
20
15
10
5
0
median chlorophyll-a (µg/L)
downstream observed
upstream observed
downstream simulated
upstream simulated
1
2
3
4
5
6
7
8
9
10
11
12
Month: Jan=1...Dec=12

Figure 4a

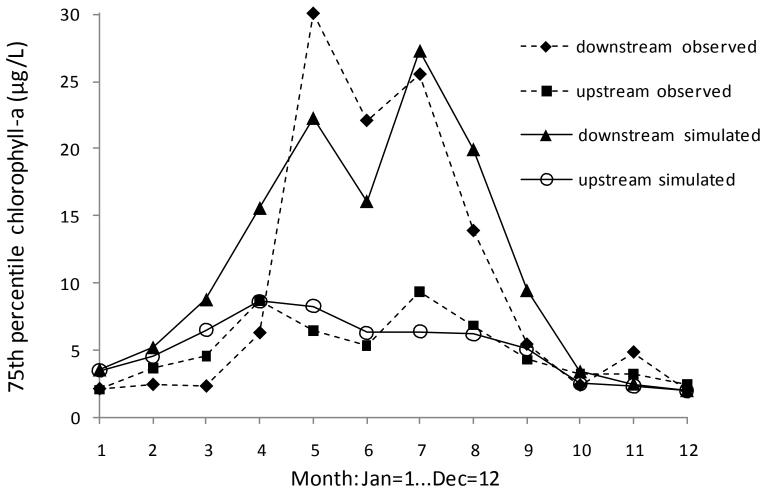

75th percentile chlorophyll-a (µg/L)
30
25
20
15
10
5
0
1
2
3
4
5
6
7
8
9
10
11
12
Month: Jan=1...Dec=12
downstream observed
upstream observed
downstream simulated
upstream simulated

Figure 4b

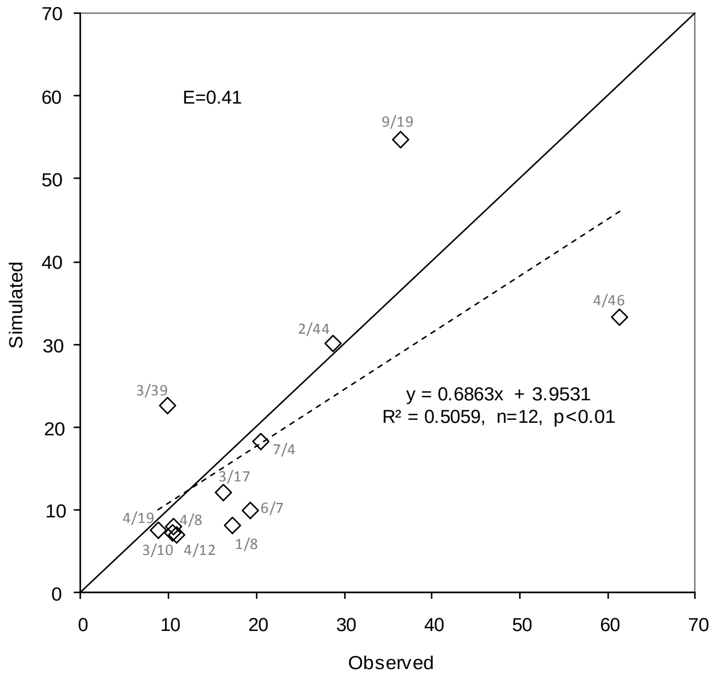

70
60
50
40
30
20
10
0
E=0.41
9/19
4/46
2/44
3/39
y = 0.6863x + 3.9531
R² = 0.5059, n=12, p<0.01
7/4
3/17
6/7
4/19
4/8
3/10
4/12
1/8
Simulated
0
10
20
30
40
50
60
70
Observed

Figure 5

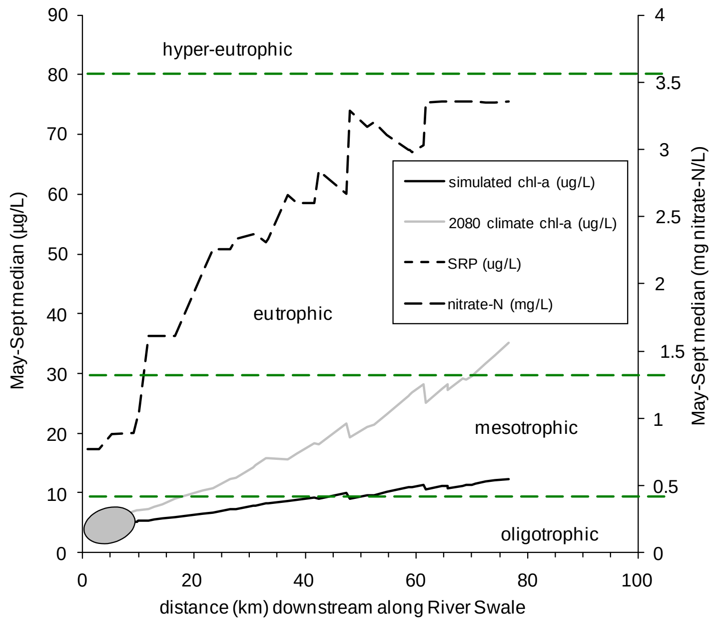
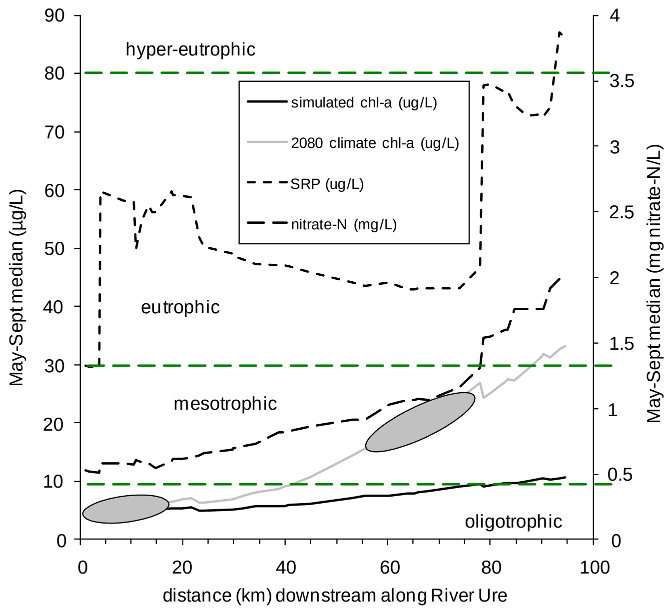

hyper-eutrophic
eutrophic
mesotrophic
oligotrophic
simulated chl-a (ug/L)
2080 climate chl-a (ug/L)
SRP (ug/L)
nitrate-N (mg/L)
May-Sept median (µg/L)
May-Sept median (mg nitrate-N/L)
distance (km) downstream along River Swale
0
10
20
30
40
50
60
70
80
90
0.5
1
1.5
2
2.5
3
3.5
4
100

Figure 6a



hyper-eutrophic
eutrophic
mesotrophic
oligotrophic
simulated chl-a (ug/L)
2080 climate chl-a (ug/L)
SRP (ug/L)
nitrate-N (mg/L)
May-Sept median (µg/L)
May-Sept median (mg nitrate-N/L)
distance (km) downstream along River Ure


Figure 6b

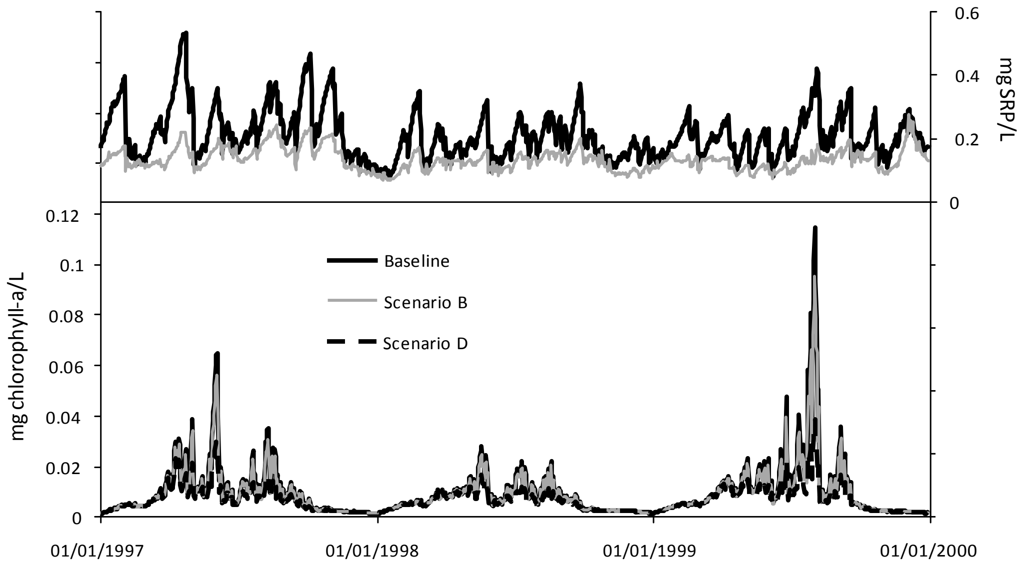

0.6
0.4
0.2
0
mg SRP/L
0.12
0.1
0.08
0.06
0.04
0.02
0
mg chlorophyll-a/L
Baseline
Scenario B
Scenario D
01/01/1997
01/01/1998
01/01/1999
01/01/2000

Figure 7