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RESEARCH AND APPLICATION TO WATER SUPPLY

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Active reservoir management: a model solution

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Predictions from a functional model of algal growth capacity (maximum achievable algal biomass) and algal species succession for Chew Valley Lake (UK) were compared with observed data for 1989. Initial simulations grossly underestimated algal growth from the late spring onwards. However, inclusion of a pulsed input of phosphorus from the sediment, at a level consistent with field observations, resulted in very good predictions. The model was then applied to a number of hypothetical operational scenarios as examples of the use of such a model as a tool for reservoir management. An example is also given of an extension of the model to predict chemical properties of abstracted and treated water.

Introduction

Several presentations in this conference have outlined the difficulties of using both steady-state and statistically-fitted models when predicting the effects of remedial actions on algal growth. Steady-state procedures, of their very nature, cannot deal with dynamic situations. Statistical models require extensive calibration, and predictions often have to be made for environmental conditions which are often outside the original calibration conditions. In addition, the calibration requirement makes them difficult to transfer to other lakes. To date, no computer programs have been developed which will successfully predict changes in species of algae.

The obvious solution to these limitations is to apply our limnological knowledge to the problem and develop functional models, so reducing the requirement for such rigorous calibration. Reynolds has proposed a model (see later), based on fundamental principles of algal response to environmental events, which has successfully recreated the maximum observed biomass, the timing of events and a fair simulation of the species succession in several lakes. A forerunner of this model was developed jointly with Welsh Water under contract to Messrs. Wallace Evans and Partners, for use in the Cardiff Bay Barrage study. Reynolds (1989a) was able to authenticate this model against a data-set obtained in respect of Slapton Ley, Devon, which like the proposed barrage lake is shallow and extremely discharge sensitive. In this paper we will test a much developed form of this original model against a more complex data-set and, using a simple example, show how it can be applied as an aid in the choice of management strategy for the reduction of problems caused by eutrophication. Some further developments of the model are indicated.

Model description

The mathematical bases of this model have been presented elsewhere (Reynolds 1984, 1989a). A schematic diagram of the operation of the model, in its simplest form, is given in Figure 1. Daily hydraulic flow rates (F_{in}), metabolisable phosphorus concentrations (P_{in}, usually approximated as soluble reactive P) and nitrogen concentrations (N_{in}, usually approximated by the nitrate concentration) are fed into the computer model for each stream, pumped input, sewage discharge or industrial discharge. The model then calculates the phosphorus and

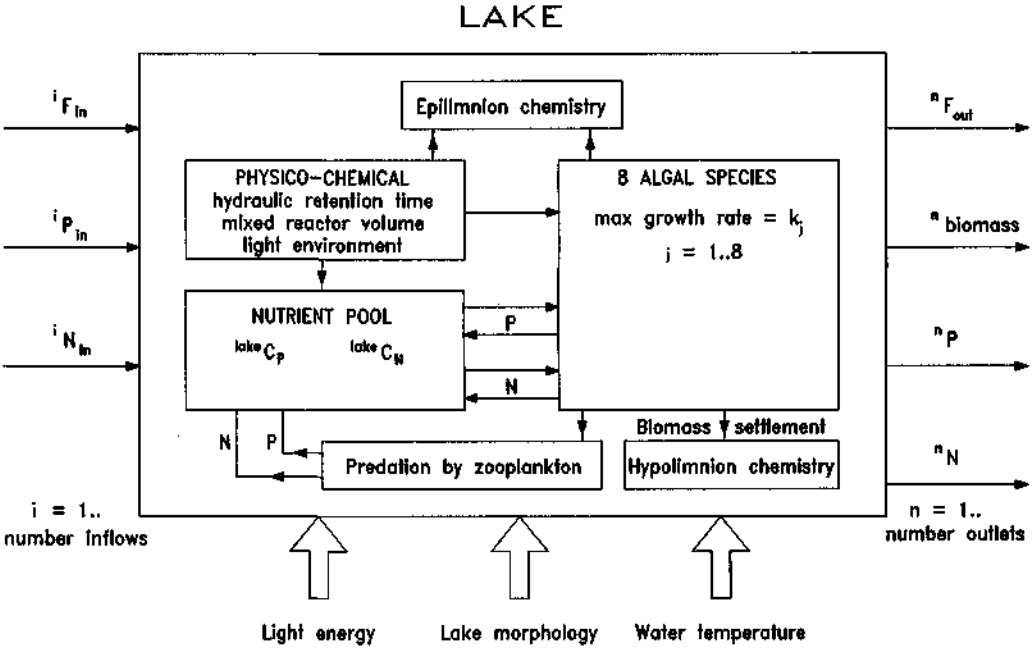


Figure 1. A schematic diagram of the algal growth capacity model. Explanation in the text.

nitrogen concentrations in the lake (C_p and C_n), assuming no utilisation by algae. Using input data identifying the mixed layer volume, depth and mean depth, the light climate in the water column is estimated from the incoming insolation (input). Assuming no limitation by nutrients, inocula of up to eight algal species, characteristic of different types found in the water-body, are then allowed to grow at their own individual maximum rates, adjusted according to the temperature and light conditions in the water. There is linkage between nutrient concentrations and the growth rate only when the availability falls below the growth rate saturating concentration. The model checks to see if there are sufficient nutrients to sustain the calculated amount of growth during the day, assuming that a Redfield ratio of nutrients (Redfield 1934) holds within the cell. Provision is made for more efficient use of nutrients when external concentrations fall. If no limitation is encountered the biomass estimate is retained and the appropriate amount of phosphorus and nitrogen is removed from the nutrient pool. If there is insufficient phosphorus or nitrogen to sustain this amount of growth then the biomass is increased by the maximum amount allowed by the limiting nutrient. The rate-limiting nutrient concentration is reduced to zero and the other nutrient pool is reduced accordingly.

Certain types of algae can be eaten by zooplankton. The model allows the existing population of zooplankton to feed and increase in mass, according to the temperature and the extent to which food concentration meets the minimum survival requirement or saturates its maximum. New terms are calculated each day. As a result of consumption of algae by zooplankton, phosphorus and nitrogen are assumed to be released instantaneously and the appropriate amount is added back into the available nutrient pools. Algal cells, phosphorus and nitrogen in solution are washed out of the system at a rate proportional to the rate of water abstraction and/or compensation flow. The computer program then cycles to calculate values for the next day.

As a result of this approach, in particular the use of a maximum growth rate at a given temperature and light environment, the resulting biomass estimates indicate the maximum level (or capacity) of algal biomass a lake or reservoir can recruit. As has been stated earlier (Reynolds, this volume) this maximum level may not always be observed in reality. Unaccounted natural occurrences, such as infection by chytrids or limitation by an unmodelled nutrient, such as silica, may not allow the populations to achieve their theoretical levels.

Comparison of model simulations with real events

Because the model is based on fundamental ecophysiological processes and the growth rates etc. are fixed, the model needs no calibration to adjust the model parameters. This makes it easy to transport from lake to lake. However, the model can only make reliable predictions in those situations where all the relevant processes are included within the algorithm. The predictive ability of the model is tested by its ability to simulate existing conditions and the resulting responses of populations. In this paper we present real data from Chew Valley Lake and a simulation as an example of the model's capabilities. There are three reasons for this choice.

(a). The first model fit was not completely realistic. As a consequence, the example shows some of the pit-falls of uncontrolled application of the model, and the way in which the model itself can assist the search for improved authenticity in the simulations.

(b). A full chemical data set was available for the modelled year (Hilton *et al.* in prep.).

(c). The model was successfully applied by Bristol Water, as an aid to its decision-making on some proposed operational changes to the lake, operated as a reservoir.

However, it must be said that the examples of possible management options given at the end of this paper are purely exemplary and in no way constitute any part of a commission brief.

Chew Valley Lake

The lake (reservoir) was built in 1953 by the Bristol Waterworks company. It has a surface area of 4.86 km^2 at top water level, a mean depth of 4.27 m and a maximum depth of 11.5 m. Wilson *et al.* (1975a) showed that, under natural conditions, it would stratify intermittently, during appropriate episodes in the summer. At these times, anoxia rapidly develops in the hypolimnion, significantly reducing the quality of abstracted water. As a result of Wilson's work, destratification was introduced to maintain chemical water quality to the treatment works. This equipment is still in continuous operation during the summer. The biology of the lake has been described by Hammerton (1959) and Wilson *et al.* (1975b).

Data on the lake's morphology, volumes of water in the inflowing streams, the volumes abstracted for treatment, the quantity of compensation flow, and nutrient concentrations in the inflows, were supplied to the model for 1989. The model was run with the assumption of a completely mixed lake, producing the predicted algal capacity shown in Figure 2. Comparison with algal biomass measurements (Fig. 2, initial prediction – expressed as chlorophyll-*a*) made in the lake during 1989 showed that the initial spring peak of chl*a* (60 µg l^{-1}) was well reproduced in both timing and maximum concentration. In the year in question the diatoms became limited by a lack of silica (Hilton *et al.* in prep.) and growth by green algae was handicapped by low water temperatures. The real algal biomass did not achieve the predicted maximum capacity. For the rest of the year, the first model run suggested that biomass levels would be low, not exceeding about 15 µg chl*a* l^{-1}. In reality, a significant peak of algae occurred in the autumn.

Although it is quite possible for the model to predict a maximum capacity which is not achieved in reality, the converse, i.e. observed algal levels higher than predicted levels, should

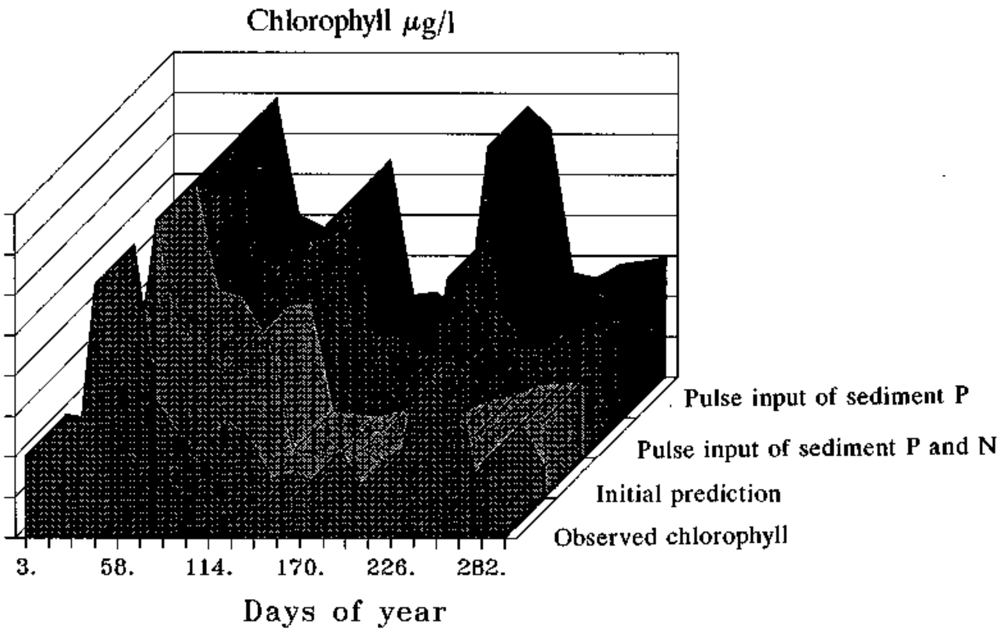


Figure 2. A comparison of observed chlorophyll-*a* concentrations (μg l⁻¹) with algal capacities predicted with different amounts of sedimentary nutrient input to Chew Valley Lake.

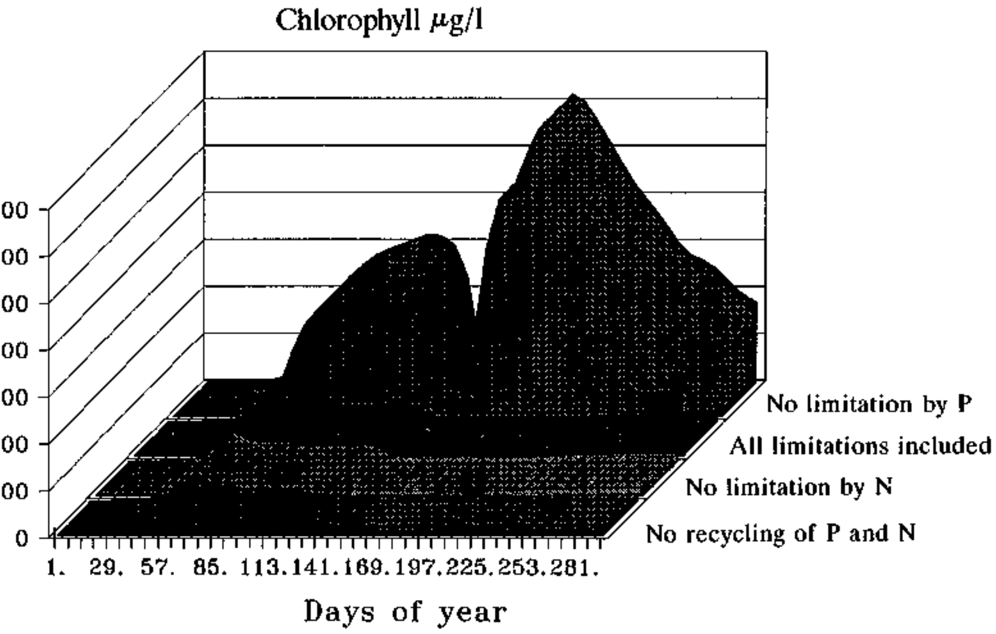


Figure 3. A comparison of predicted algal capacities (chlorophyll-*a*, μg l⁻¹) in Chew Valley Lake after removal of selected limitations.

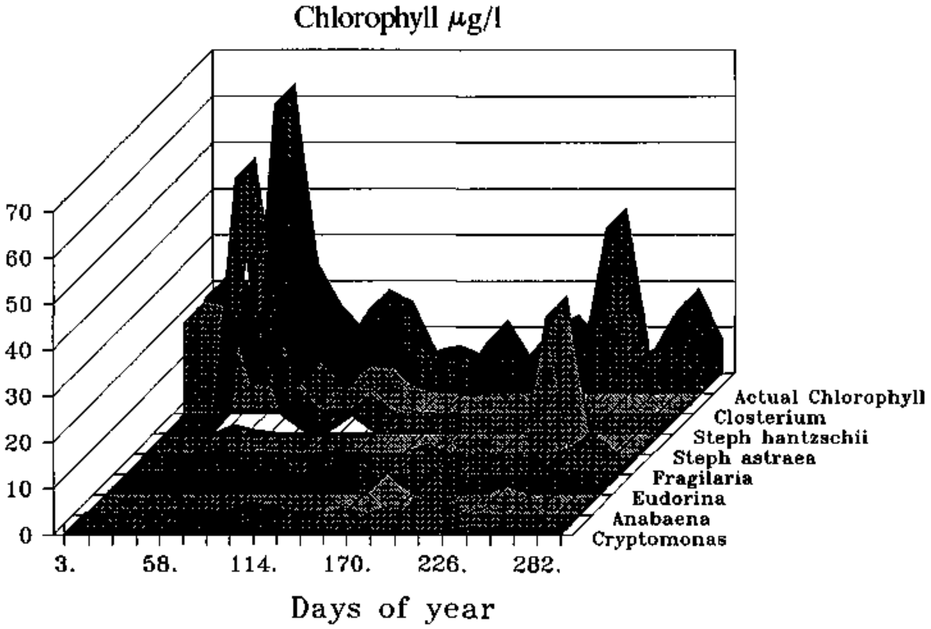
not be possible if the model is viable. In order to find possible causes of the model's failure to predict accurately, the limitations on algal growth imposed by each of the three constraining variables, (1) phosphorus concentrations, (2) nitrogen concentrations and (3) zooplankton nutrient recycling, were removed one at a time. The resulting predicted levels are shown in Figure 3. Removal of either the capacity limitation by nitrogen or the lack of nutrient recycling by zooplankton had only a small effect on standing crops, in both cases reducing biomass predictions, compared to the case of full limitation. However, removal of capacity limitation by phosphorus allowed the predicted capacity to rise to over 600 μg l^{-1} of chl*a*, with a significant peak in algal concentration at the end of summer. Hence, a possible cause of the poor predictive ability at the end of the summer in the original simulation could have been an unrecognised source of phosphorus.

Trial simulations (not shown) using a range of constant input rates of phosphorus throughout the summer period suggested that an amount equivalent to 3μg P l^{-1} in the whole lake, added each day, would be sufficient to give the observed chlorophyll levels in the autumn. Comparison of these phosphorus loads with measured sources showed that underestimation of the input load from streams was unlikely and that roosting birds were present in insufficient numbers to generate the required influx of phosphorus (Hilton, unpublished). By deduction, release from the sediments was implicated. The simulations of biomass, assuming an extra, constant summer input of phosphorus, also suggested that the missing input probably occurred in a pulse or series of pulses, rather than at a constant rate.

A study of the phosphorus concentrations measured in the lake water (Hilton *et al.* in prep.) showed that in May there was a rapid increase of about 200 μg l^{-1} in both soluble reactive phosphorus and total phosphorus in the water, which was maintained until the autumn. By an iterative process it was found that if 50% of this released phosphorus was made available to the algae for growth, then the late summer peak in algal biomass was reproduced by the model at observed levels (Fig. 2, sediment-P pulsed input; any reference to this sediment input in the rest of this paper assumes that only 50% is utilised). The major increase in phosphorus in the lake water occurred during a storm, following a hot, calm period of several weeks duration and was probably due to release from the sediments.

Having established that the model can simulate the correct quantities of total algal biomass, when given phosphorus inputs which satisfy the observations, we may consider the observed and predicted distribution of species. The changes observed in the dominant algal species in the lake are shown in Figure 4. Notable concentrations of the green alga, *Closterium*, were observed in early January. The spring peak was dominated by a small diatom, *Stephanodiscus hantzschii*, with a small contribution from large diatoms, including *Stephanodiscus astraea*, and *Closterium*. Mid-summer algal populations, although much lower than the spring peak, consisted of cyanobacteria and cryptomonads, with a significant contribution from nitrogen-fixing species in mid to late summer (day 170). The late summer population was dominated by *Stephanodiscus astraea*, with a small contribution from *Fragilaria* and the nitrogen-fixing cyanobacterium, *Anabaena*.

Figure 4 also shows species composition predicted by the model, including a pulsed input of sediment phosphorus. The large *Closterium* population observed at the start of the year is not simulated, as it results from the cumulative conditions of the previous year and input data were only available for 1989. The model predicts a spring peak of diatoms, both large and small species, with a small input contribution from large green algae such as *Closterium*. As stated earlier, the early summer populations of diatoms were, in reality, limited by silica, so other species became dominant, but the present model does not allow for silica limitation. The mid to late summer growth of nitrogen-fixing cyanobacteria is well represented by *Anabaena*.



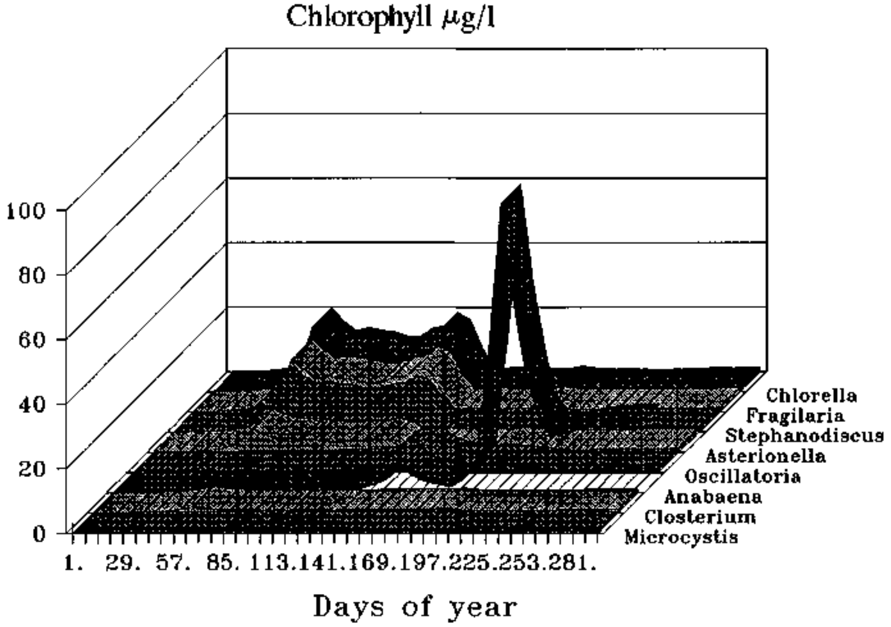


Figure 4. Above: observed composition of algal species in Chew Valley Lake in 1989. Below: predicted composition of algal species from the model, assuming an input of phosphorus from the sediments. Algal biomass is expressed as chlorophyll-*a* ($\mu g\ l^{-1}$)

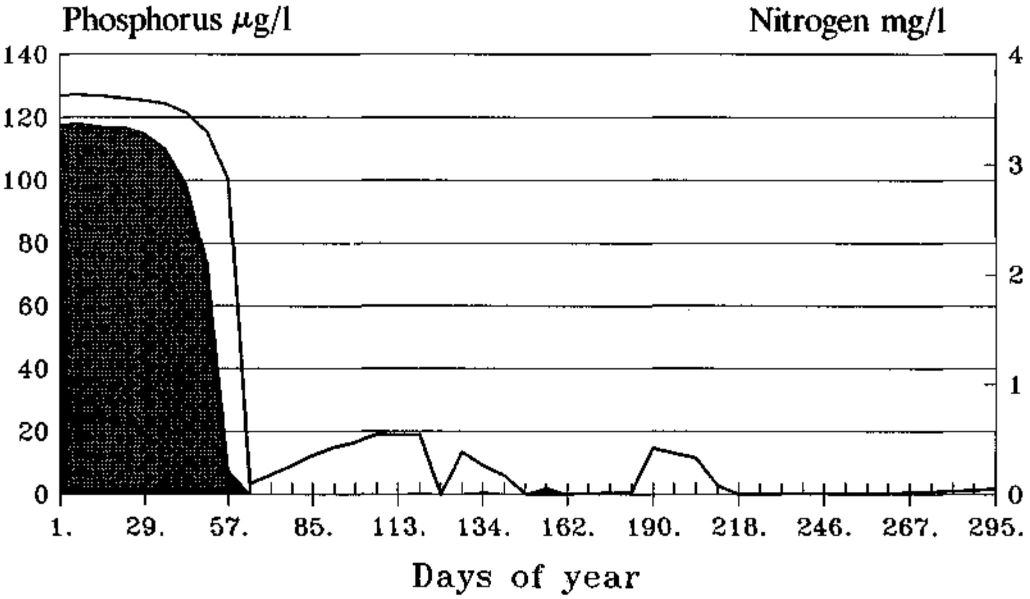


Figure 5. Dissolved nitrate-nitrogen (mg l^{-1}, —) and reactive phosphorus concentrations (µg l^{-1}, shaded areas) predicted by the model, assuming a sedimentary phosphorus input to Chew Valley Lake.

However, the late summer peak is completely dominated by nitrogen-fixing cynaobacteria, rather than the observed diatoms

The appearance of nitrogen-fixing cyanobacteria is symptomatic of a lack of nitrogen in the system. This is borne out in Figure 5 which shows that, under the model conditions, nitrogen levels would have been very low during the periods around day 168 and from day 218 onwards. Hence, although nitrogen was not limiting the total algal biomass, more nitrogen (in addition to the quantities brought in by the inflows and included in the model) must have been introduced into the real system. Otherwise, dominance by the nitrogen-fixers would have occurred in reality as well.

A rapid reduction in nitrate-nitrogen concentrations was observed in the water column. It coincided with the large increase of phosphorus in the water column (around day 155). So the nitrate chemistry was predicted correctly, resulting in the prediction of the dominance of nitrogen-fixing blue-green algae around this period. However, in mid June (approximately day 168), about six weeks after the large release of phosphorus, ammonia levels in the water column were seen to rise rapidly from about 100 µg l^{-1} to about 700 µg l^{-1}. These increased levels were observed for the next six weeks. Again a search of the external inputs could locate no reason for the increase. Other data (Hilton *et al.* in prep.) were consistent with this event also being a release from the sediments.

When an amount of nitrogen equivalent to this release was introduced into the model, the predictions of biomass capacity remained essentially unaltered (some reduction was observed due to the less efficient use of phosphorus at higher nitrogen concentrations) but the species dominance in the late summer changed from nitrogen-fixing cyanobacteria to a mixture of almost all the species introduced in the theoretical inoculum (Fig. 6). Both *Anabaena* and *Fragilaria* were seen in significant quantities in the autumn. Since growth rates of *Stephanodiscus astraea* were not included in the model its dominance could not be predicted directly. However, all the algae which show significant growth in the model are efficient in

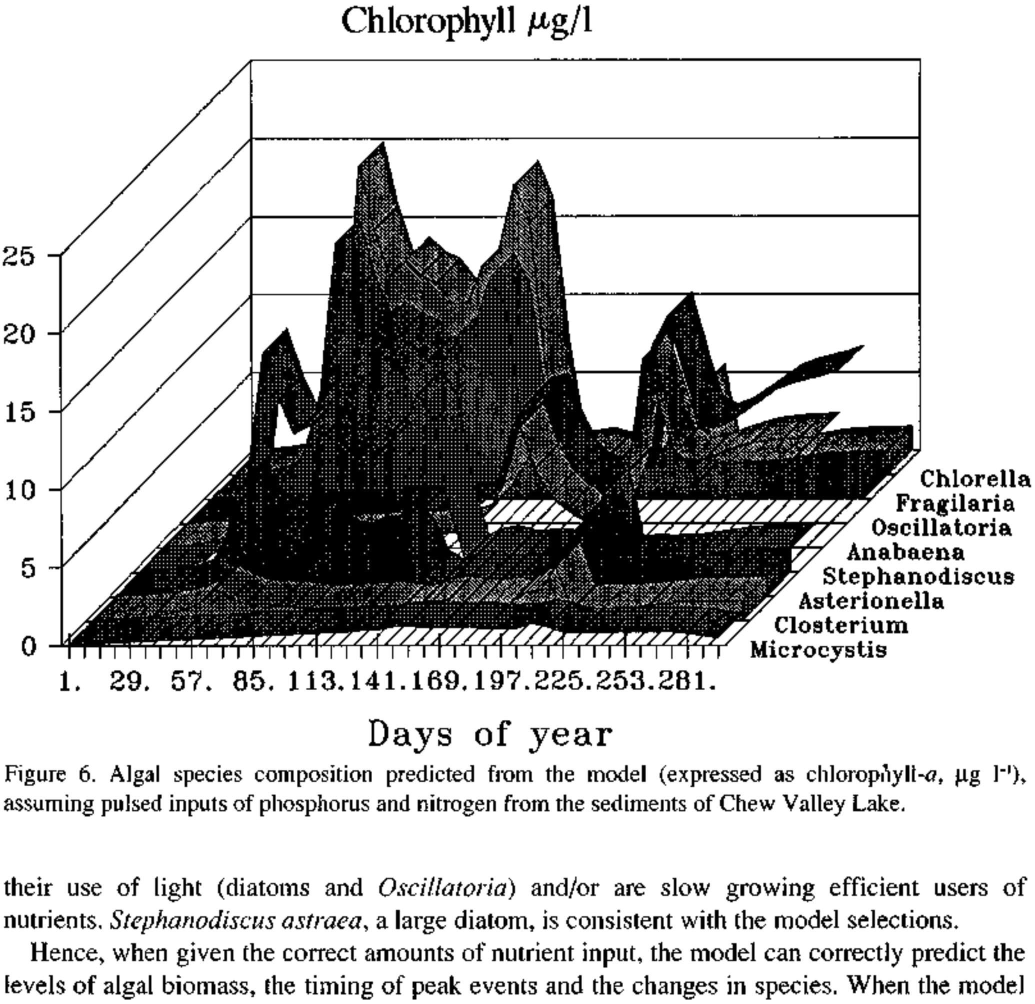


Figure 6. Algal species composition predicted from the model (expressed as chlorophyll-*a*, µg l^{-1}), assuming pulsed inputs of phosphorus and nitrogen from the sediments of Chew Valley Lake.

their use of light (diatoms and *Oscillatoria*) and/or are slow growing efficient users of nutrients. *Stephanodiscus astraea*, a large diatom, is consistent with the model selections.

Hence, when given the correct amounts of nutrient input, the model can correctly predict the levels of algal biomass, the timing of peak events and the changes in species. When the model is used in conjunction with a good chemical and algal data-set it can direct the user to a greater understanding of the lake system, which in itself is an aid to better decision making. Since it would be unrealistic to include growth rates for all algal species in the model, the results will never predict the actual species dominance in all cases. Experience is required to interpret the results in terms of the conditions (low light, low nutrient supply, etc.) which are selecting the dominant species. However a major limitation to the use of the model, without direct comparison with actual data, is our ability to predict phosphorus and nitrogen release from the sediments.

Management strategies

Having checked both the completeness of the input data for the model and the similarity of simulations to real life, the model can then be used in a predictive mode. Suppose that the peak levels of algae in the lake were considered to be unacceptably high, possibly as the result of significant down time of the treatment plant due to filter clogging. Would phosphorus removal from the inflows be a viable policy ? There are seven inflows to Chew Valley Lake, of which the River Chew is by far the largest, constituting at least 90% of the total incoming volume of water. The obvious policy would be to treat water in the River Chew only. If the concentration of phosphorus in the River Chew is reduced to zero in the model, the output shown in Figure 7

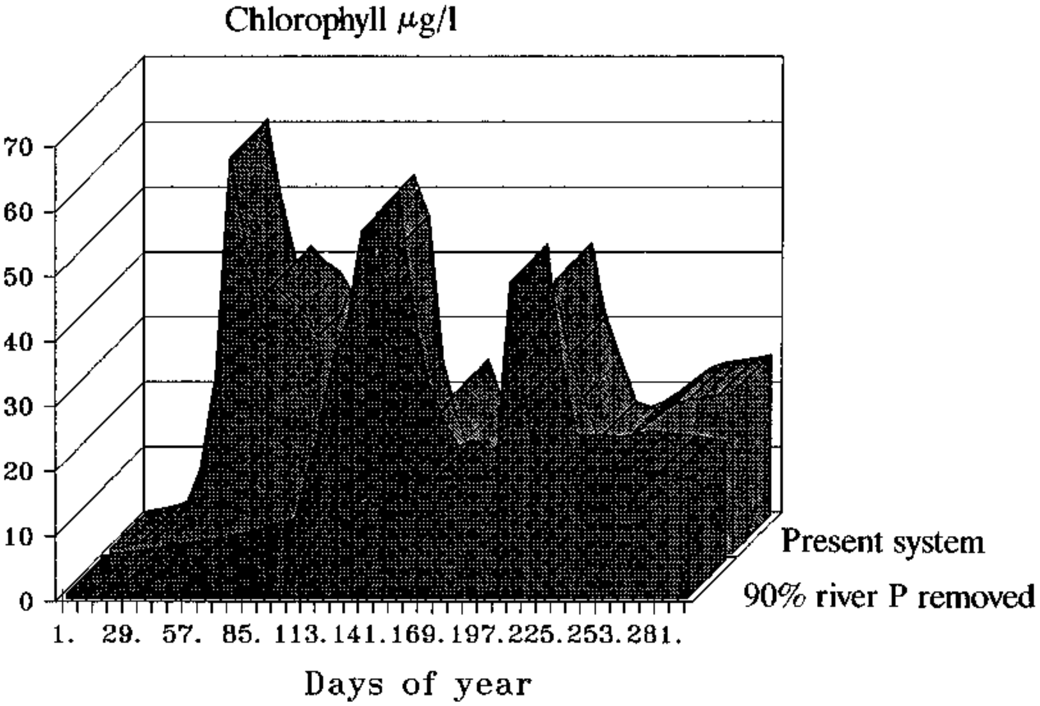


Figure 7. A comparison of predicted algal capacities (chlorophyll-*a*, μg l^{-1}) in Chew Valley Lake with phosphorus input as at present and after its removal from the major inflow.

is obtained. As can be seen, the early spring peak of algal biomass is almost completely removed. Since this is one of the times of peak algal production it would have a beneficial effect. However, removal of phosphorus from the major inflow would give no significant reduction in the algal biomass from late spring onwards, so that no benefit would accrue during this period. The sediment releases, resulting from many years of phosphorus inputs, dominate the system at this time and, as a result, reductions in observed biomass levels during the late spring and summer would take many years to achieve without a programme to remove or stabilise sedimentary phosphorus. They would still be expected to improve after an unspecified number of years, as the internal load reduces, through gradual equilibrium and eventual burial of the phosphorus-rich sediment. Hence, if the treatment problems consistently occurred only in the spring then the phosphorus-removal option may be appropriate (the actual decision will also rely on relative costs). If the problem is typical of the late spring to summer period then alternative or joint removal of phosphorus and sediment-treatment strategies would be required.

As a further example it would be interesting to see if the destratification, which is used to improve hypolimnetic water quality, has any beneficial effect on algal biomass levels. Before destratification was introduced a thermocline often existed at about 7-m depth. If we introduce a permanent thermocline at this depth into the model then the biomass predicted in Figure 8 would probably result. It is not significantly different from the destratified situation. The species composition also shows only minor changes from the destratified situation (Fig. 9). We must conclude that the only benefit (although sufficient in itself) of destratification in this reservoir is the removal of anaerobic conditions in the bottom waters, improving the quality of water abstracted for treatment.

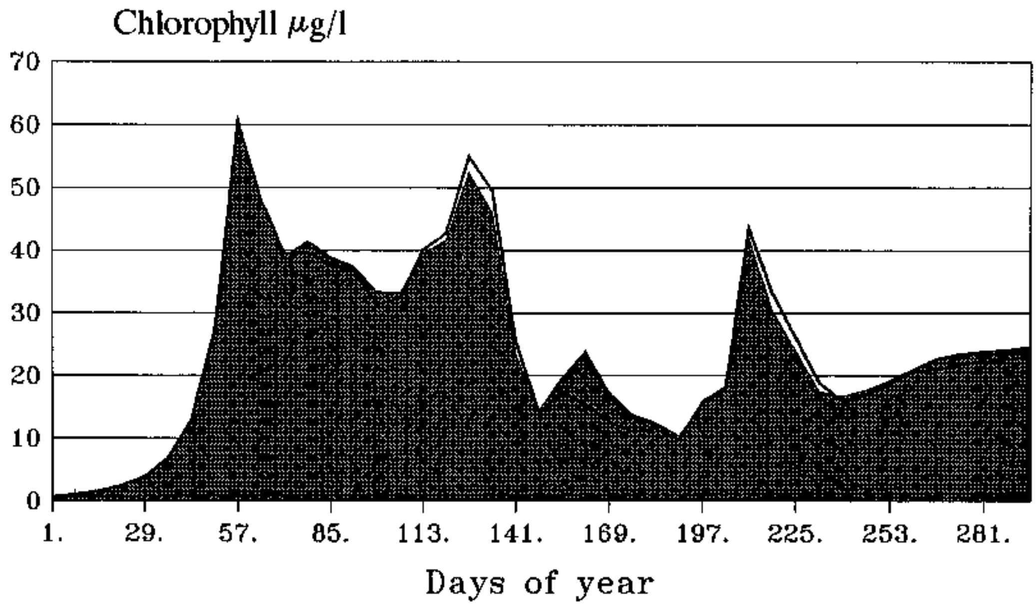


Figure 8. A comparison of (theoretical) predicted algal capacities (chlorophyll-*a*, µg l⁻¹) in Chew Valley Lake with input as at present but assuming either complete mixing (no thermocline; shaded area) or a continuous summer stratification at a depth of 7 m (—).

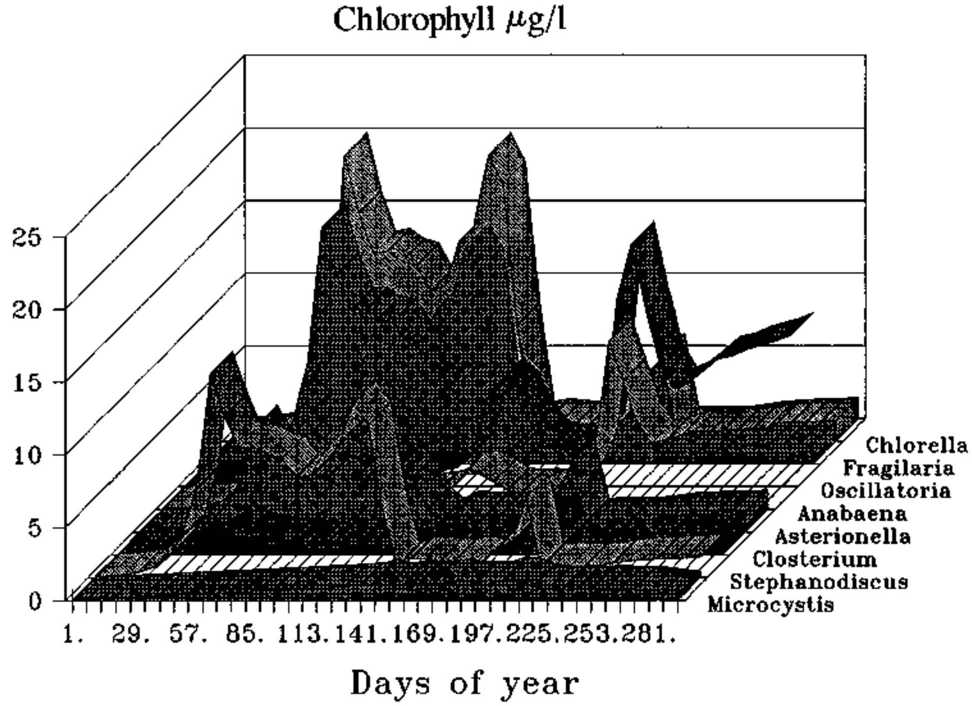


Figure 9. Species composition (chlorophyll-*a*, µg l⁻¹) predicted from the model assuming a pulsed sedimentary input of phosphorus and nitrogen into Chew Valley Lake under conditions of continuous summer stratification.

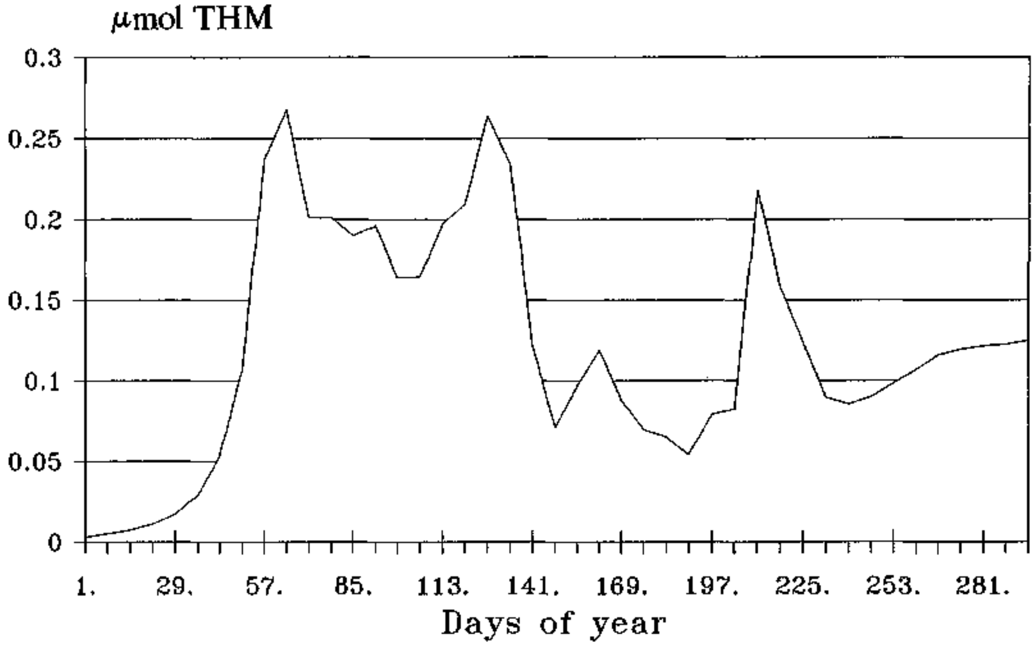


Figure 10. Predicted maximum concentrations of tri halo methanes (μmol l^{-1}) in Chew Valley Lake after treatment, assuming that chlorine is allowed to come into contact with algal cells.

Further developments

Many chemical processes in lakes and reservoirs are either driven by, or strongly influenced by, the algal carbon produced by photosynthesis. If the simulations from our algal growth model are used as a data input to a chemical model it is possible to make reliable predictions of many chemical processes. For example, if algal cells are exposed to chlorine then tri halo methanes (THM) are formed. By using Hoehn *et al.*'s (1980) relationship between algal biomass and THM production it is possible to predict the worst case level of THM production after treatment for the present reservoir (Fig. 10). Using more rigorous chemical relationships (Stumm & Morgan 1981) we have accurately predicted epilimnetic pH levels and the dynamics of hypolimnetic water chemistry changes (Hilton *et al.* in prep.). There is even a good possibility that sediment phosphorus influx could be predicted by combining the hypolimnetic model and the epilimnetic pH model. Similarly reliable prediction of nitrate removal by reservoirs is also within grasp.

Conclusions

The application of a realistic, functionally-based model of algal growth can benefit many areas of limnological research and application. Choices between different management strategies can be made on a more rational basis and, by identifying those areas where the model is still limited, research topics for the future can be identified.

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