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Modelling the ecological aspects of bankside reservoirs and implications for management

J.A. Steel[1] & A. Duncan[2]
[1]*5 Fort View, Ardbrack, Kinsale, Co Cork, Ireland*
[2]*Royal Holloway College, University of London, Egham, Surrey TW20 0EX, U.K.*

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Abstract

Bankside storage reservoirs are used as a major water supply resource in the lower Thames Valley, England. They form the link between the River Thames and the water treatment works of the Greater London area. The reservoirs act as both a water reserve in times of low river flows, and a quality 'buffer' between the river and the treatment works. The load on the water treatment works (particulate material, physico-chemical characteristics) primarily reflects the water qualities of the reservoirs. Management of such reservoirs thus seeks to reduce the adverse impacts which would otherwise arise from direct river use, and to ensure as far as possible that the ecological processes within the reservoirs do not introduce new challenges to the water treatment. Reservoir management clearly needs a good understanding of those ecological processes and their interactions, and, hopefully, a means to exploit that understanding in hindcasting to explain past events, in forecasting near- or far-future events, and to help in exploring operational options to ameliorate any foreseeable difficulties. The reservoirs consist of a variety of configurations, physical dimensions and operational circumstances. They have, importantly, basically simple morphologies, known hydraulic regimes and physico-chemical qualities. Nonetheless, they appear to behave essentially as small (1–50 Mm^3), eutrophic lakes; and various aspects of their ecology has been studied for the past 65 years. Their attributes and operational involvement make them ideal candidates for ecological modelling, which has been applied to them in varying extents for the past 30 years. The major conclusion which may be drawn from these studies is that even in such relatively simple water bodies, current (and probably future) models can only encompass their broad ecological characteristics. Detailed operational needs have to be met by a variety of modelling approaches, mainly predicated on the basis of only being able to know a lot about a little or a little about a lot. The operational needs for modelling fall into the following broad types: (a) understanding: why did those events occur, or where is our ignorance greatest? (b) short-term forecasts: how will the current situation develop in the short-term (weeks)? (c) what-if considerations: what would happen if some management facility were employed or used differently? (d) optimisation: what are the optimal volume– quality supply arrangements? (e) long-term prediction: what is the longer-term (years) outlook under foreseeable scenarios? (f) projective evaluation: how would potential, as yet non-existant reservoirs behave under prescribed circumstances? Examples of how these needs have been met are outlined, with examples ranging from simple models of the diatom ecology of the reservoirs to much broader trophic–dynamic descriptions which can allow expression of fish–zooplankton–phytoplankton interactions. This is crucial for present and future management of cyanobacterial phases. It is clear that considerable management insight and control can result from modelling assistance, but only if the appropriate questions are asked. Whilst simple short-term modelling is less demanding, any attempt to model the full complexity of the ecology of even these relatively simple water-bodies is probably doomed to founder on complexity–understanding difficulties, unless these are resolved to much more constrained system aspects. This is particularly so for the qualitative biology. The best that may presently be foreseen is for development of the newer multi-biological type models, with reasonably realistic and dynamic physical and chemical environment sub-models, being able to manifest the general characteristics of the ecosystem in question. Despite such difficulties, new reservoir management insights and approaches will inevitably be founded on critical modelling of those ecosystems.

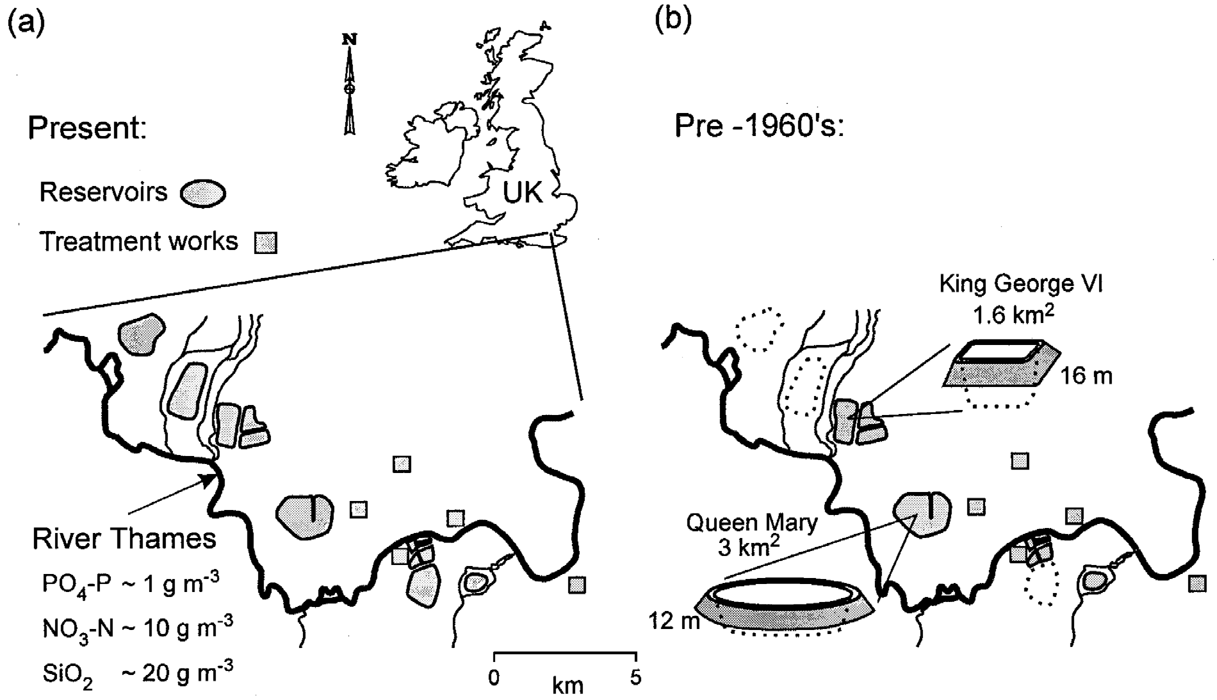

Figure 1. Reservoirs and treatment works in the lower Thames Valley, South-East England. (a) The system at present, and order of magnitude nutrient concentrations in the River Thames. (b) The system in the period before 1960, and the different configurations of Queen Mary and King George VI reservoirs.

Introduction

The reservoirs being considered lie in the south-east of Britain, near London and are distributed along the banks of the River Thames (Figure 1a).

As they are close geographic neighbours, they experience the same climatic conditions. They also all have the same source water, which is the eutrophic River Thames, with nutrient concentrations of around 1 mg PO_4-P l^{-1}, 10 mg NO_3-N l^{-1} and 20 mg SiO_2 l^{-1}. The main reservoirs were built during the past 65 years and, between 1963 and 1974, the total capacity doubled from about 100 to 200 Mm^3. The reservoirs are designed to provide bank-side storage, and they all possess similar features of simple morphologies, steep sides and relatively uniform depths, and are operated with known quantities of throughput water when in supply. The surface areas of the main reservoirs range from 1.5–3.0 km^2; they have volumes of 10–40 Mm^3 and depths of 10–25 m. In general, when in supply, their retention time is within the range 10–100 days.

These reservoirs cannot properly be considered in isolation as single water bodies since they form part of a water supply chain linking a series of different (eco)systems: river–reservoir–treatment works–water supply (Steel, 1972; 1975). It is important to realise that the fundamental ecological processes of these reservoirs are not different from those in lakes. The reservoirs may, of course, differently manifest the effects of those processes because their unique morphological/operational characteristics can influence relative process magnitudes. However, also most importantly, these are bodies of water which are manageable and measurable, which allows considerable opportunity to identify and quantify those fundamental processes.

Earlier studies during the International Biological Programme 1966–72 showed that loadings of organic material upon the treatment works was wholly dependent upon the quality of the reservoir water (Steel, 1975; Steel et al., 1972) (Figure 2). In the absence of a reservoir stage in the water chain, the works' loads would be both much greater and more variable. Nevertheless, algal crops could be limited by appropriate reservoir management, despite the presence of concentrations of phosphorus and nitrogen so high in relation to algal growth requirements that early studies showed there was virtually no discernible planktonic nutrient N and P uptake in the reservoirs.

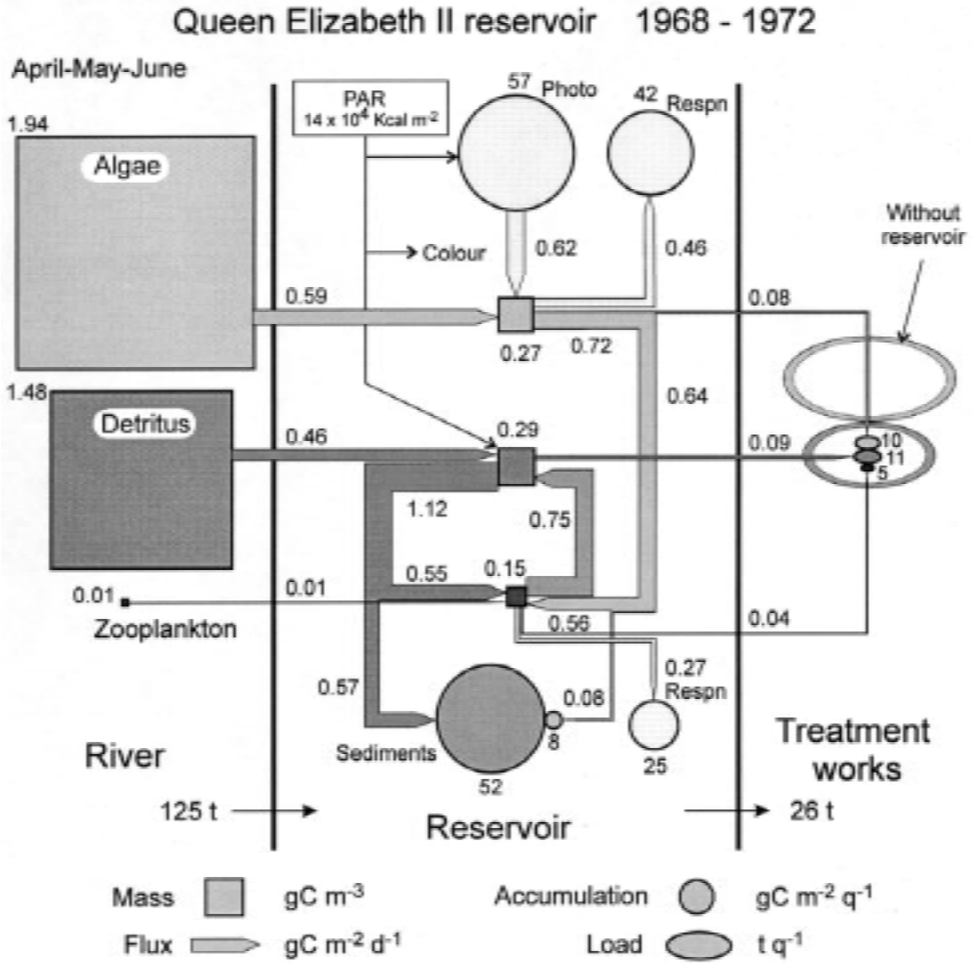

Figure 2. Measured values of various ecosystem quantities, for the productive quarter April–June, 1968–1972.

An ability to control the quantities of plankton plants in the reservoir provides important benefits for treatment technology, investment requirements and ease of operation. The primary overall objective for this research was the absolute need to maintain an adequate supply of potable drinking water. It was clear that both understanding the ecological interactions within the reservoirs and quantifying their magnitudes was, and is, crucial to an informed management. Operational need was therefore the main driving force for the present exercise.

This example of the application of increasing ecological understanding, and its associated modelling approach, is drawn from attempts to answer some major business questions posed during developments that occurred during the past 35 years. In the 1950–60s, consideration of new storage resources (Figure 1b) included questions such as ‘how deep should these reservoirs be?’ In the lower Thames Valley, the main options were to build a basin of large area but shallow depth (as Queen Mary reservoir) or one that was deep and small in area (as King George VI reservoir). For land already owned, the former would be cheaper, but for maximum resource efficiency the latter choice was desirable. In either event there were great reservations about water qualities.

In Queen Mary reservoir, long-term deep deoxygenation was not experienced, despite occasional, transient thermal stratification, but large crops of diatoms and blue–greens were a regular feature. These were treated by either mass dosing with $CuSO_4$(ca 1 mg $l^{-1} \equiv$ c 30 tonnes of $CuSO_4$ in Queen Mary reservoir), a regime of continuous low level $CuSO_4$ dosing (0.1–0.3 mg l^{-1}) or by closure and switch of supply to another reservoir, usually King George VI reservoir. Annual appearance of a stable thermal stratification, with regular de-oxygenation and formation of H_2S was the dominant characteristic of King George VI Reservoir. These were often also accompanied by large algal populations, particularly diatoms, and cyanobacteria and *Ceratium* in the epilimnion under thermally stratified conditions (Figure 3a). This reservoir was mainly managed by enclosure, with relatively occasional use. Despite having multiple, shoreline draw-off facilities, a stable supply of good quality water for abstraction was rarely possible because of internal wave motions associated with the thermal stratification.

In summary, the considerable expenditure required for the extra, deep storage did not ensure a usable supply, mainly because of thermal stratification. Detailed experimental work by White et al. (1955) suggested that thermal stratification would be controlled by internal mixing with submerged water jets. The first of the new reservoirs with such a mixing facility was therefore designed to be fairly deep (17 m), but still similar enough to the current experience, in case mixing was unsuccessful, or introduced unforeseen quality effects. In the event, jet mixing provided an outstanding control of thermal stratification. Figure 3b shows that, in Queen Elizabeth II reservoir, isothermal conditions could be easily maintained throughout the year, with consequent oxygenation of the complete water column and, possibly, the sediment surface.

This raised the next business question: ‘could the remaining new reservoirs be even deeper?’ There was little information in the literature as to what happens to the algae in such deep, mixed basins, with unlimited nutrient supplies. Several important subsidiary questions emerged: would they grow large crops throughout their depths? would there be a changeover to less easily treatable types of algae? Would there occur some alteration in the algal sequencing?

Modelling was seen as a major tool in attempting to provide some answers, at least semi-quantitative, to these questions. As there was a firm conviction that an understanding of the reservoirs’ biological dynamics was sought, an early decision was for an ‘ecologically’ structured model, rather than attempting a multi-parameter regressional type model. As previously indicated, the ecological interactions to be considered were of general applicability and not unique to these reservoirs, although the simple basin morphologies and operational attributes does make them easier to study. This generality also suggested that appropriate lake and marine studies could provide a fruitful beginning.

A simple model and developing ecological understanding

The simple model

Talling’s (1957a; b) algal model was recognised as an entirely appropriate starting point. It considers the population’s photosynthetic carbon gain against its respiratory carbon loss in an homogeneously mixed water column. Initial quantification of the local reservoir values of the variables contained in Talling’s model led to a swift development of a simple, mixed system algal model (Steel, 1972). For diatoms in deep, fully mixed reservoirs, this model predicted that:

(1) algal growths begin later;
(2) their growth rates are constrained;
(3) maximum attainable biomasses could be energy-limited rather than nutrient-limited;
(4) potential maximum biomasses would be significantly reduced.

Some of these model effects are illustrated in Figure 4a. Under similar conditions, these effects were largely determined by a combination between mixed depth (Z_{mix}) and light attenuation (ϵ_q), because depth offsets transparency and turbidity offsets depth. In part, what the model predicts is that column maximal algal biomasses (mg chlorophyll-a m^{-2}) would be linearly, inversely related to mixed depth-light attenuation, as plotted in Figure 4b. The field observations in Figure 4b broadly agreed with such a prediction. Occasionally, however, greatly reduced crops were observed, which implied that some other factor or factors needed to be included in the model. Even so, the application of this simple model was already a major step on the path to answering some of the difficult questions previously posed.

Addition of a nutrient term

The most obvious deficiency of the simple model was its supposition of unlimited nutrient availability, although in the nutrient context of these reservoirs, only SiO_2 was remotely likely to cause limitation in diatoms. For completeness, however, nutrient effects were included by simple Michaelis–Menten nutrient characteristics for SiO_2 and phosphorus, with subsequent addition of Droop’s (1965) ‘quota’-type effects (Steel,

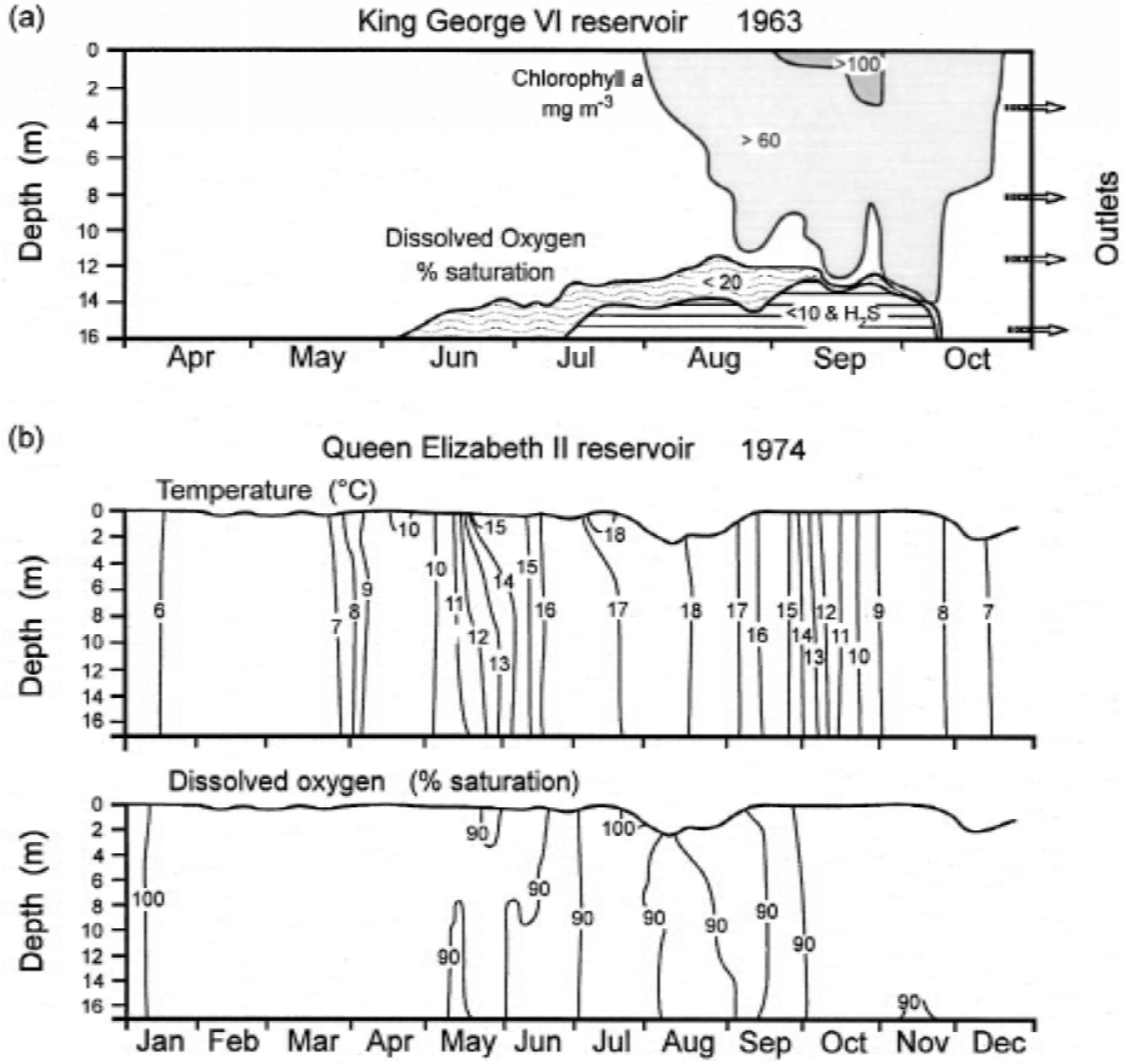

Figure 3. (a) Thermal stratification in King George VI reservoir during the growing season of a representative year, with consequences to algal crops and de-oxygenation. (b) Results of jet mixing in Queen Elizabeth II reservoir in 1974, illustrating the degree of isothermy and oxygenation achieved.

1978). Some indication of the results of this model for typical reservoir waters is given in Figure 5a. It is clear that phosphorus limitation, for example, is extremely unlikely.

In this form, the model indicated that phosphorus-stripping from the existing 1000 mg PO_4-P m^{-3} or so to, perhaps, levels of the order of 10 mg PO_4-P m^{-3} would be necessary to achieve any further, significant reduction in potential algal crops. Even if such a reduction was achievable, the costs involved far out-stripped any potential savings. The business therefore decided not to undertake P-stripping, even of only the reservoir inlet water: another major investment decision. Similarly, SiO_2 was never low enough in the early part of the year to limit diatom crops, and the model indicated that most diatom crops were maximal long before SiO_2 was potentially limiting. Analysis of the experience with $CuSO_4$ treatment showed that it did not reduce either algal growth rates or maximum crops to levels below that achieved by en-

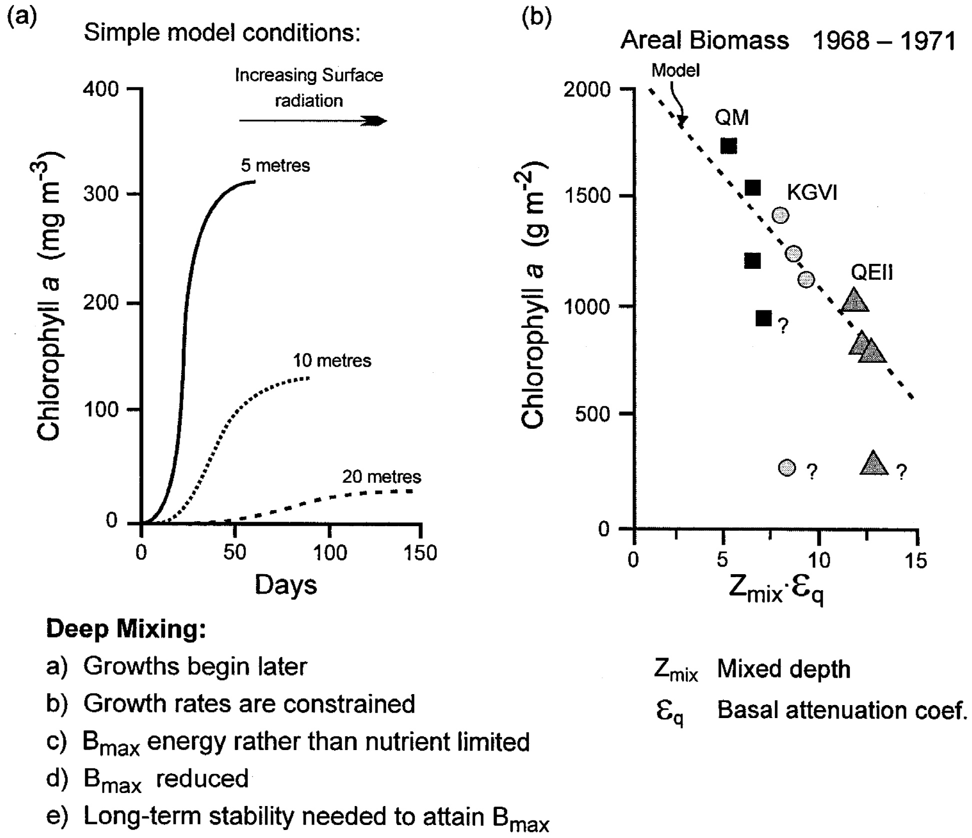

Figure 4. Chlorophyll-*a* concentration outputs from a simple, 'Talling-type' model for typical early season conditions. (a) Algal growth under full-depth mixing conditions in basins of different depth. (b) Inverse linear relationship between maximal algal biomass and the combined mixed depth–light attenuation coefficient, validated by observed data points from three reservoirs. See text for explanation of the exceptional points.

ergy limitation by full-depth mixing. Copper treatment was therefore discontinued as a management tool: yet another major quality decision with significant, beneficial, financial implications. Furthermore, studies were also beginning to show that $CuSO_4$ dosing also had adverse effects on the reservoir cladoceran populations

More detailed model investigations of mixed depth and the water's light attenuation suggested that mixing the potential epilimnetic algal populations through a greater, de-stratified water column offers a way to severely limit the algal crops, and potentially to a far greater extent than due to mere 'dilution' (Figure 5b). So the lesson for the business of water supply is to make reservoirs as deep as it is technically and financially feasible, along with mixing and draw-off facilities at least as efficient as those in the Queen Elizabeth II reservoir.

However, there were still some unresolved questions. Sedimentation studies and modelling suggested that mixing was not reducing sedimentation losses and that SiO_2-stress enhanced sedimentation could potentially remove large diatom crops very rapidly. However, sedimentation was not the reason why, sometimes, virtually no crop appeared, despite SiO_2 being copiously available. As far as could be ascertained, algal parasitism did not appear to be the cause. This raised the question 'could cladoceran grazing be sufficient to suppress the diatom crops; especially if freed from the effects of $CuSO_4$ poisoning?'.

An intermediate model: addition of a grazing equation

On the basis of measurements of zooplankton biomass, feeding and respiratory rates in the reservoirs, particularly in Queen Elizabeth II reservoir (Andrew, 1976; Chalk, 1981; Duncan, 1975), a simple size-

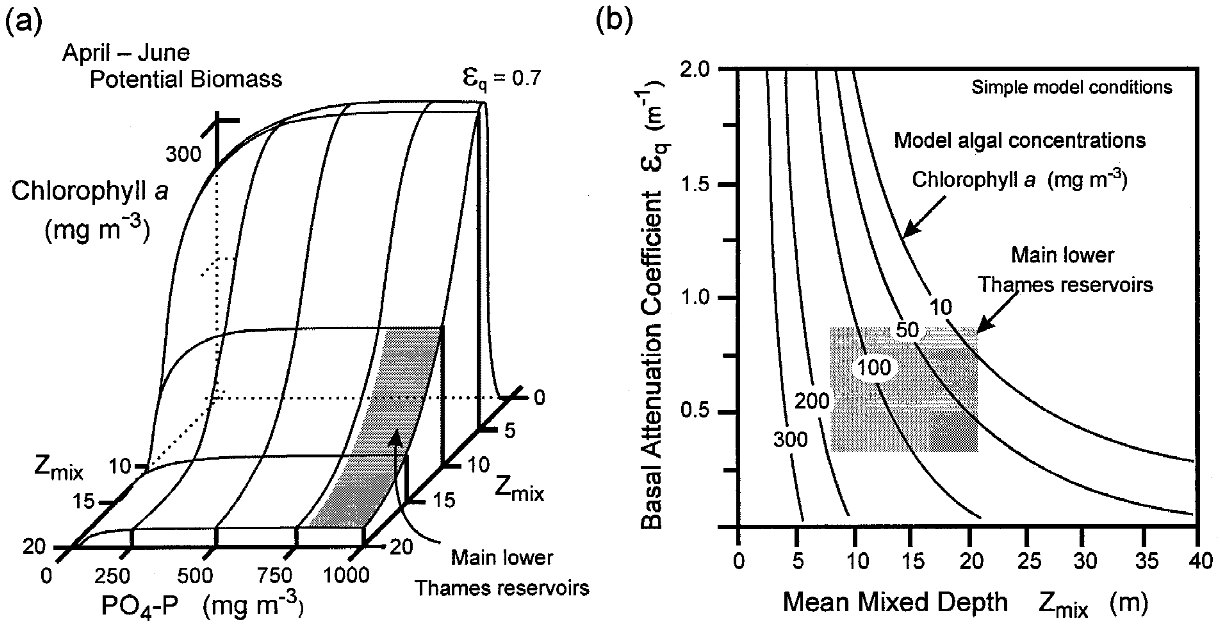

Figure 5. (a) Output from the simple model with a nutrient term for various mixed depths and typical sub-surface light attenuation. Also implied is the degree of P-stripping required to reduce potential algal crops under the conditions of the main lower Thames Valley reservoirs. (b) Output from the simple model of how mixed depth and light attenuation combine to reduce potential epilimnetic algal crops, and the range of conditions in the main lower Thames Valley reservoirs.

related grazing equation was incorporated into the model, based upon a large, 2 mm length daphnid filter feeder, and running simultaneously with the algal growth equation (Steel, 1975). This model, now of intermediate complexity, predicted that the grazeable algae of deep, fully mixed reservoirs would be much more susceptible to a given grazer pressure than if they had been in shallower waters (Figure 6a). This pressure could be continued to quite low algal concentrations due to the daphnids' ability to maintain feeding rates, despite reduction in food concentrations to an incipient limiting level. These grazing effects would be most marked in the deeper mixed reservoirs, where the later and slower algal growths tended to achieve their maximal levels at a time when the daphnid populations could take advantage of them – in late April when the water was warming to 10 °C. If these predicted effects were real, then there should be observable relationships between the algal and zooplankton crops in the deeper, mixed reservoirs.

Figure 6b is a plot of observed algal and zooplankton crops in Queen Elizabeth II reservoir for the period 1968–73, expressed as averages for the quarter April–June which incorporates the population maxima. This plot appears to show grazer-sensitive algal crops, with a changeover from maximal energy-limited algal crops in 1968–69 to markedly reduced crops from 1971 onwards associated with increasing zooplankton biomasses. The 1973 value for zooplankton biomass is the same as that for the 1969 algal crops, which is energetically impossible without some additional zooplankton food source. One source is likely to be the period of intense, small diatom production that occurs in the River Thames during the period April–June. In relatively small surface-area reservoirs, input of riverine water with such algae will add considerably to both the reservoir's productivity and potential daphnid food supply. This is a significant source as the river crop concentration is usually much greater than is possible in the deep, mixed reservoirs – and can be up to an additional 50% in apparent productivity when river crops are large.

Addition of a term for river algal immigration

Incorporation of possible levels of algal immigration into the reservoir during the April–June quarter raised the complexity of the model by another stage. It also introduced a dynamic daphnid-type grazer component which predicts the general form of the empirical algae–zooplankton relation shown in Figure 6b for different possible levels of river phytoplankton

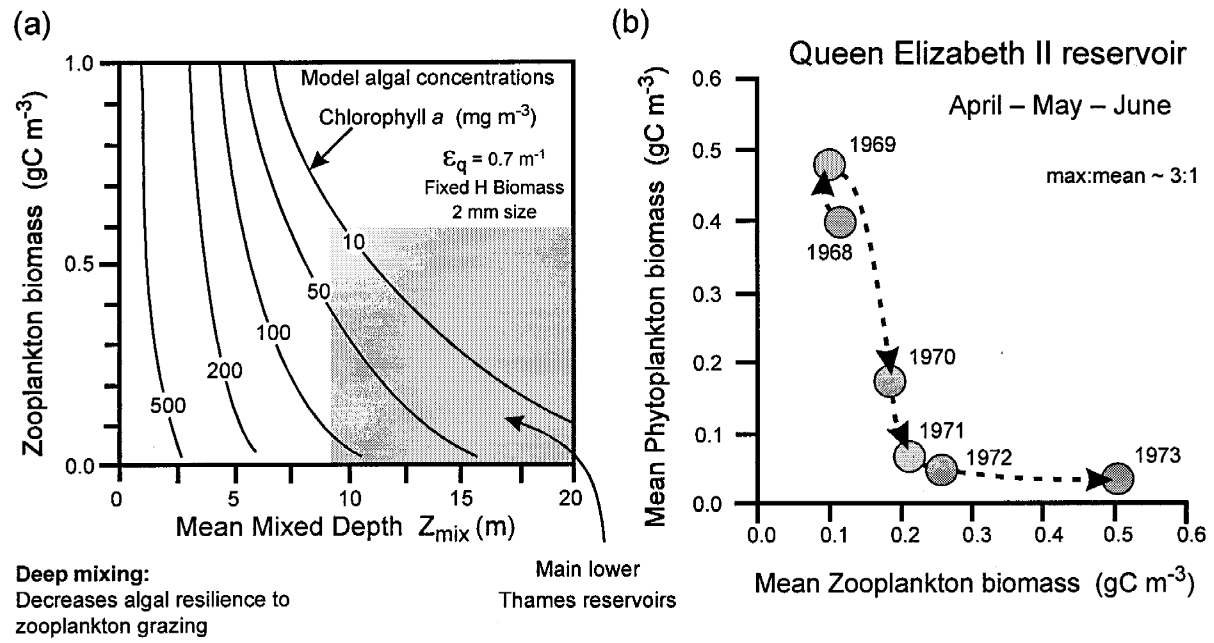

Figure 6. (a) Output from the intermediate model with a simple size-related grazing equation. With deep mixing, algal resilience to zooplankton grazing is reduced. The relevant range of conditions in the lower Thames Valley reservoirs is indicated. (b) The observed relation between phytoplankton and zooplankton biomass means in the jet-mixed Queen Elizabeth II reservoir during April–June 1968–1973.

(Figure 7). Superimposed upon these plots is the observed data set from 1968–1988 which conform more or less to the general prediction. However, careful examination of Figure 7 shows that most data points fall within a characteristic cluster and that the data of 1968, 1969 and 1973 are extraordinary. Large scale, natural fish mortalities occurred in these reservoirs during the later 1960s and it makes it appropriate to ask: ‘were fish–zooplankton interactions yet a further factor in these events?’ To answer this question, and to improve the predictive power of the rough grazer model needed much more information on zooplankton sizes and associated dynamics, coupled with more critical size-related algal modelling.

Detailed observations on the daphnid communities of Queen Elizabeth II reservoir in 1972 showed very high absolute population sizes as biomass, as well as an extraordinary proportion ($>50\%$) of large animals in the population (Duncan, 1975) (Figure 8a). Present-day observations from another deep, mixed reservoir confirmed the same picture of large crops of large-sized daphnids (Santos, 1989) (Figure 8b). This raises a major question: ‘how can these large-bodied grazer populations exist in the reservoir and be supported? To start to address these latter two questions needs considerably more realistic climatic-environmental and biological-ecological detail in the model.

A complex model

At present, the model incorporates these environmental effects by use of sub-system ‘mimics’, together with the facility to impose particular conditions unique to the reservoirs, such as mixing. The advantage of having the sub-systems is that changing a property, such as depth, also causes appropriate, concomitant changes to the thermal and mixing conditions to occur. In its current form, the model outputs can illustrate the general behaviour of various, environmentally adaptive algal groups (Steel, 1995), and the size-related, structured zooplankton populations (Figure 9). It can also be used to explore in more realistic detail many of the ecological interactions previously more simply considered – a thinking tool for the manager or ecologist.

An area of particular concern is: ‘are the zooplankton and/or mixing seriously reducing potential cyanobacterial populations?’. These appear to be much less prolific now, even in those reservoirs which

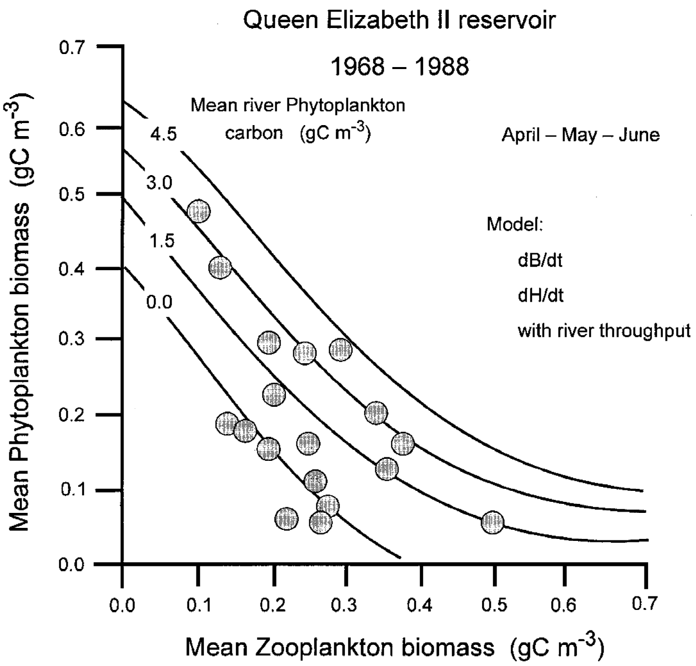

Figure 7. Output from the intermediate model with grazing and an algal immigration term. The observed mean phytoplankton and zooplankton biomasses for the April–June period from 1968–1988 fall between the predicted limits for realistic levels of riverine algal immigration. Apart from three exceptional points (see text), most data points form a cluster with narrow ranges.

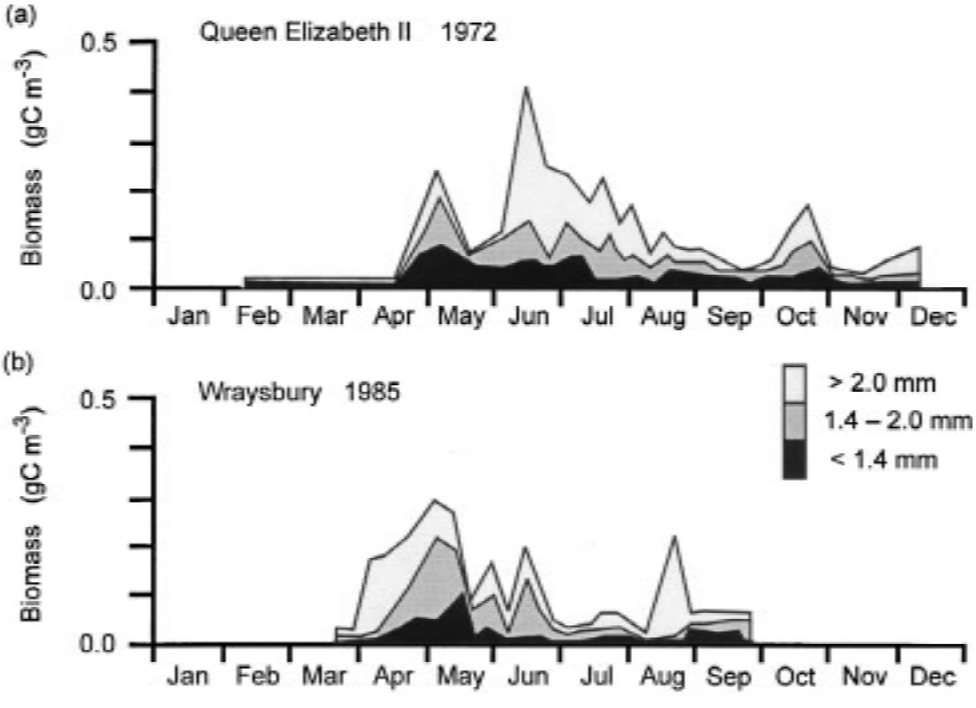

Figure 8. Observed data. The size structure of populations of *Daphnia* species in two jet-mixed reservoirs: (a) Queen Elizabeth II reservoir, 1972; (b) Wraysbury reservoir, 1985.

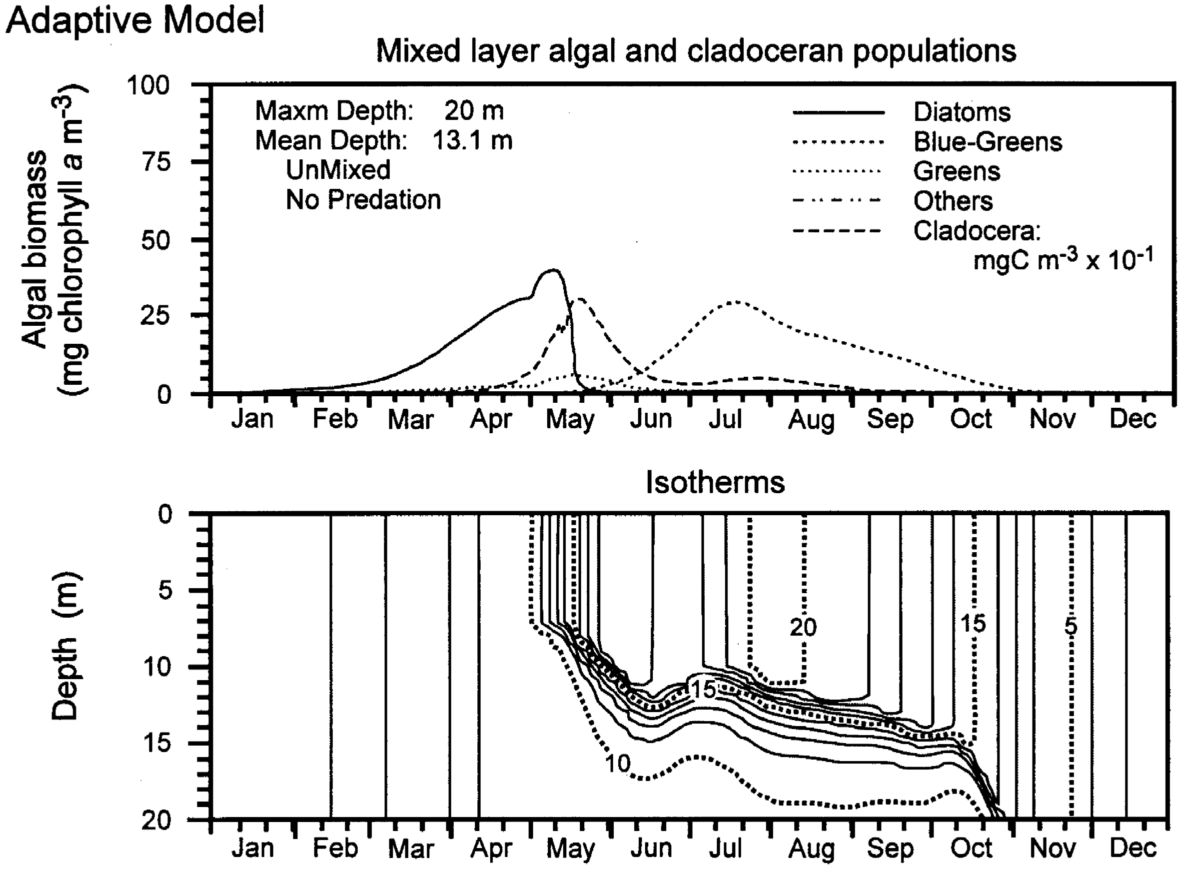

Figure 9. A sample of the output from the complex, environmentally and biologically adaptive model.

were notorious for them 30–40 years ago. What the model suggests in response to this question is that, as long as there is a reasonable, large-sized zooplankton population present, and adequate mixing when the cyanobacterial initiates appear, then sufficient grazing can markedly reduce any later, non-grazeable sized cyanobacteria. However, all this is as yet untested, and will be experimentally difficult directly to support or refute!

Future developments

Fish–zooplankton interaction

Studies during 1993 (Figure 10a) provided a good characterisation of three of the reservoirs' fish populations and associated zooplankton species composition and size structure. (Kubecka & Duncan, 1994; Renton et al., 1995; Seda & Duncan, 1994). The extraordinarily low fish biomass in Wraysbury reservoir is associated with the highest proportion of large-sized daphnids in the zooplankton, in contrast to the much larger fish biomass of Queen Mary reservoir which is associated with a much smaller proportion of large daphnids in the zooplankton, with an intermediate position in Queen Elizabeth II reservoir. There were quality differences also. In Wraysbury, the fish biomass was virtually all perch and ruffe with no cyprinids, whereas in Queen Mary reservoir it was largely bream and roach (cyprinids) plus some perch. The differences in zooplankton size structure applied to species as well as individuals. The large-sized *Daphnia magna* and *D. pulicaria* co-existed all year with the smaller *D. galeata* in Wraysbury reservoir (as in Queen Elizabeth II reservoir, but with different proportions), whereas only *D. galeata* existed in the open water zooplankton of Queen Mary reservoir. It is probable that the total zooplankton biomass data of Figure 10a are not significantly different.

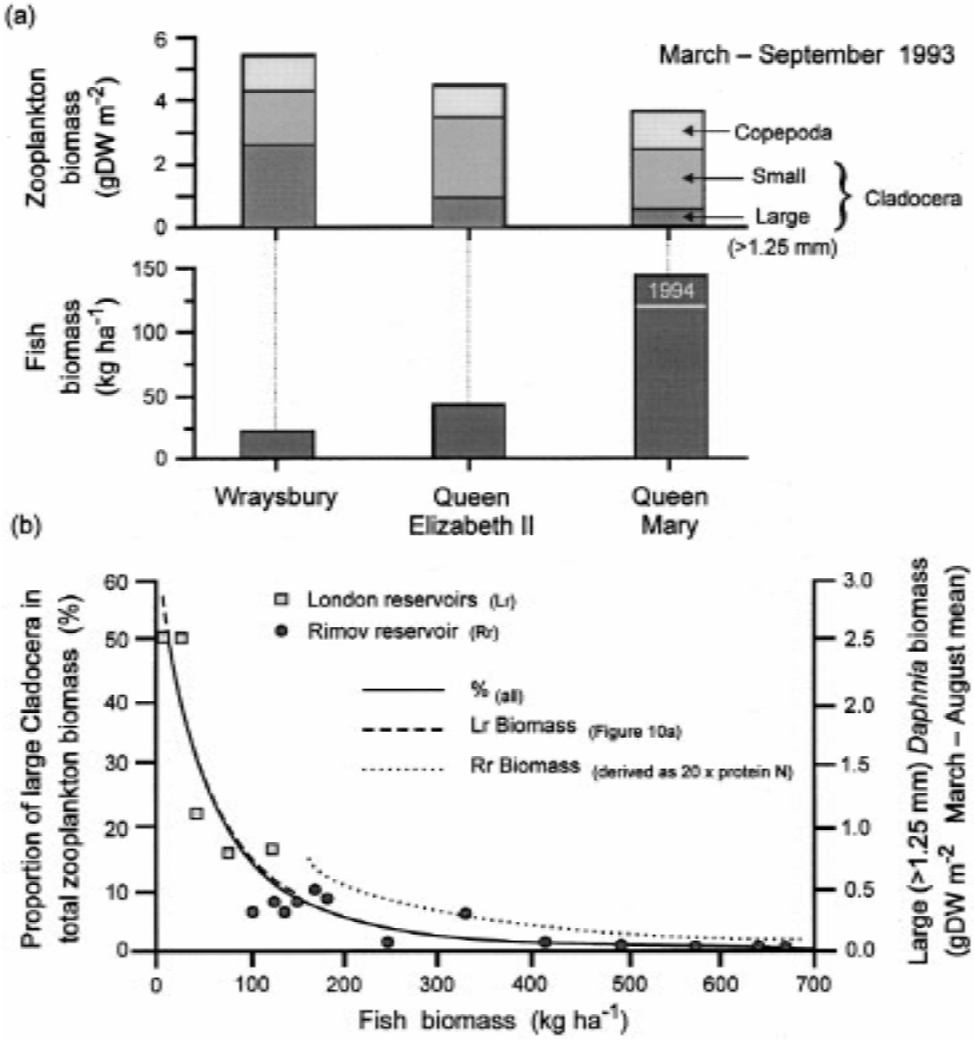

Figure 10. Observed data on the fish-zooplankton interaction. (a) The fish and size-structured zooplankton biomasses determined simultaneously in three well-mixed reservoirs during 1993. (b) The observed relation between the size-structured zooplankton biomass and fish biomass in three lower Thames Valley reservoirs and the Rimov reservoir, Czech Republic.

The lower fish biomasses of the present reservoirs fit well with a similar set of data available for the Czech Rimov reservoir (Kubecka, 1989; Seda et al., 1989) (Figure 10b) in which the fish biomass was deliberately reduced over a period of 14 years from its highest levels (650 kg ha^{-1}) down to 100–50 kg ha^{-1}, below which it proved impossible to go. Figure 10b also contains a rough estimate of large cladoceran biomass derived from nitrogen measurements given in Kubecka and Duncan (1994). The combined data suggest that there might be a crucial fish biomass of the order of 100 kg ha^{-1} above which the large cladocera are under such severe predation pressure they can no longer constitute a substantial proportion of the zooplankton biomass. A secondary consequence seems to be a significant diminution of large ($>$1.25mm) cladoceran biomass. It is interesting to note in these reservoirs of similar trophy, the comparable form and disposition of the proportion and large cladoceran biomasses, which is suggestive of relative constancy (of the order of 5 g dw m^{-2}) in the total zooplankton biomass, despite the large range of fish biomasses involved.

A combination of these measurements with simultaneous measures of algal crops in the three reservoirs and the graphical model in Seda & Duncan (1994) al-

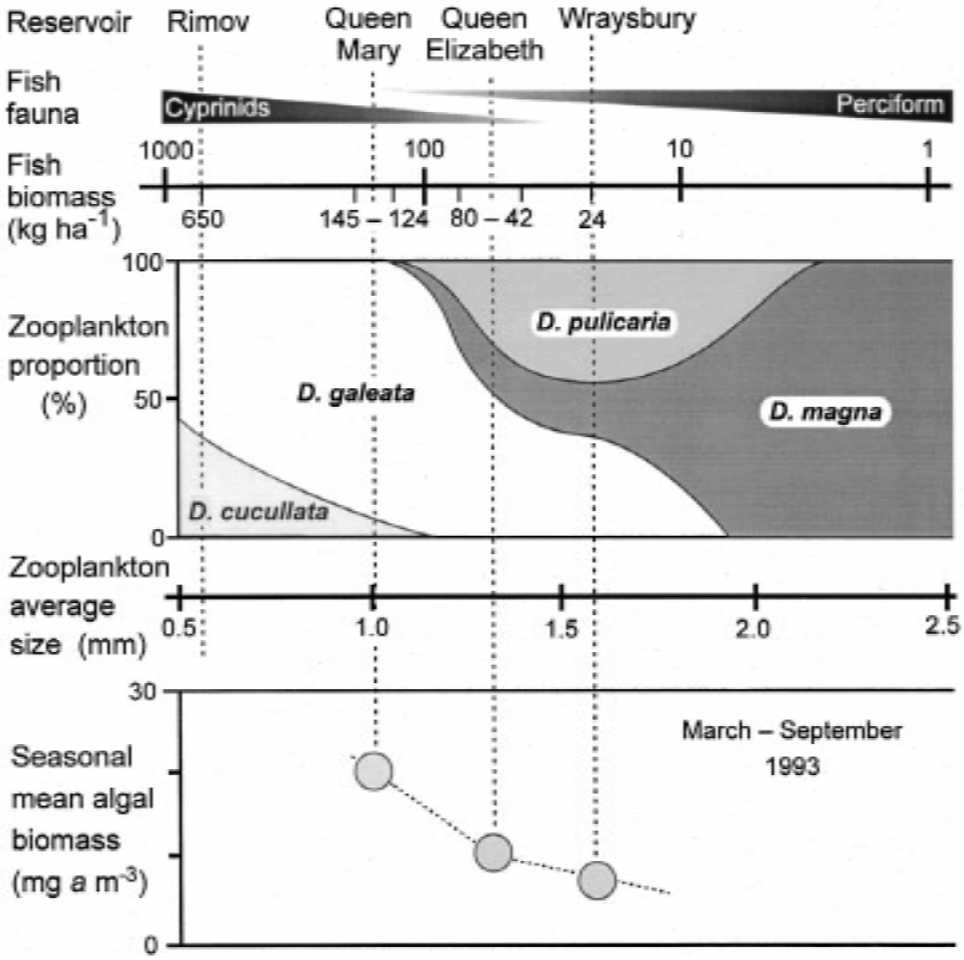

Figure 11. Quantification of top-down effects in the trophic cascade hypothesis in three deep, well-mixed reservoirs. Data from the Czech Rimov reservoir is included.

lows a first quantification of the top-down effects of the trophic cascade hypothesis (Carpenter & Kitchell, 1993) in these deep, well-mixed basins of similar trophic status with natural fish populations, and is shown in Figure 11. The seasonal maximum:mean algal crops are of the order of 3:1. Importantly, the daphnid speciation implies within it a particular adult size, characteristic of the species. The average size covers all individuals, not just adults.

An important question, relevant to the business, now is: ‘are these ecological conditions stable without further intervention’? The Wraysbury fish appear unlikely to develop a significant cyprinid fauna because the reservoir does not have suitable spawning sites (Duncan & Kubecka, 1995; Kubecka & Duncan, 1994). The same limitation applies to Queen Elizabeth II reservoir with its similar concrete margins, but spawning sites are presently enhanced by the net-sides of empty fish cages. In Queen Mary reservoir, however, bream and roach are already breeding and growing, so this reservoir may well be able to support a much larger cyprinid population. Earlier observations (1975) show that the zooplankton of Queen Mary reservoir containing a 40:60 proportion of *Daphnia cucullata* (an even smaller daphnid species) to *Daphnia galeata* (from Figure 11) was indicative of perhaps 500 kg ha^{-1} fish biomass and was associated with a seasonal mean chlorophyll-*a* of 40 mg chlorophyll-*a* m^{-3} (equivalent to 135 mg chlorophyll-*a* m^{-3} maximum). Allied to a deepening of Queen Mary from 12 to 15 m, this suggests potential future seasonal algal crops of 30-35 mg chlorophyll-*a* m^{-3} (100–120 mg chlorophyll-*a* m^{-3} max.). As these are likely to be cyanobacteria populations, meas-

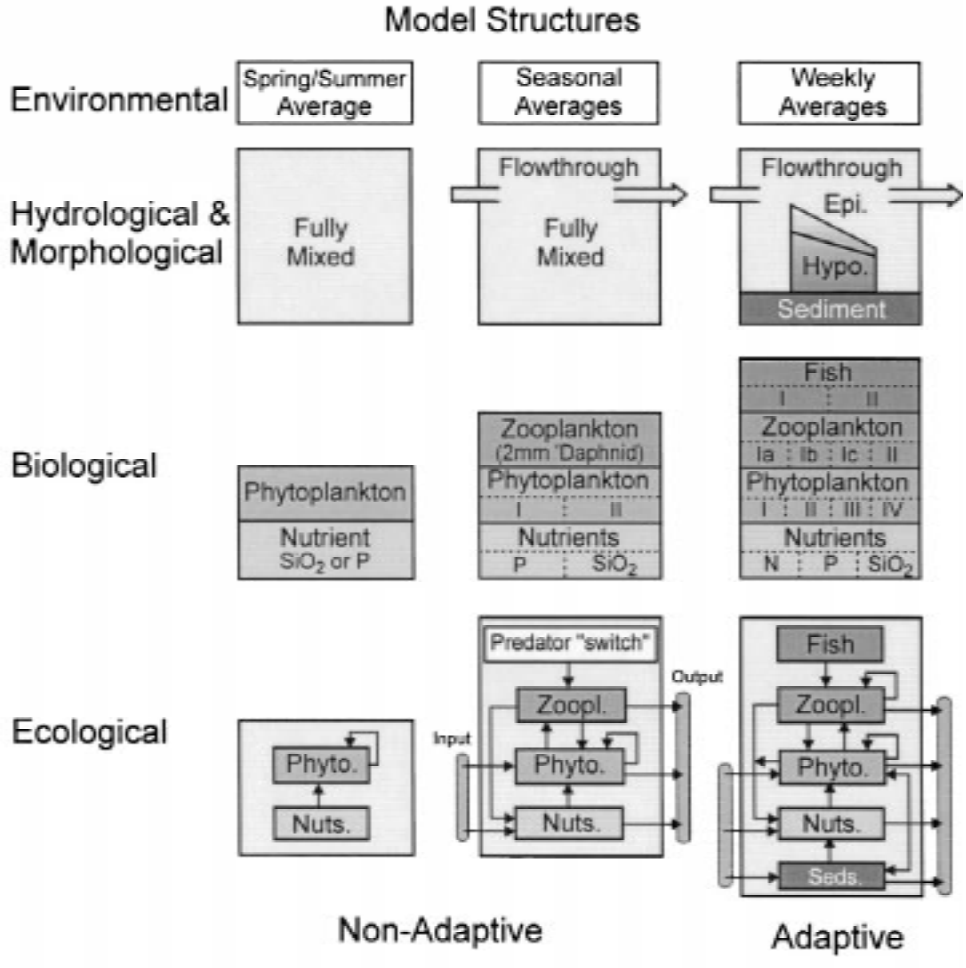

Figure 12. Diagram illustrating the historical evolution of the simple algal model and its environment towards greater complexity as ecological understanding developed in parallel.

ures would also be needed to prevent troublesome downwind aggregations being drawn into the reservoir outlets.

The empirical relationship between the seasonal mean proportion of large cladocerans in the total zooplankton (or biomass of *Daphnia* larger than 1.25 mm) and fish biomass illustrated in Figure 10b provides a start to the possibility of modelling a fish predation term. For Queen Mary reservoir, whose fish biomass in 1994 was near to the threshold value of 150 kg ha^{-1} and whose water quality was worsening, the management implications are reduction of the cyprinid fish by biomanipulative measures (Renton et al., 1995). The present development of the complex model is incorporating not only some of the direct, quantitative information of Figure 11, but also as much of the qualitative effects as current information allows. Again, at this level of development, gaining information of the necessary type and quality becomes increasingly difficult.

Conclusion

The developments of the reservoir models have reflected the management questions which have arisen throughout the past 30 years or so, the measurements which it has been possible to make and the growing understanding of the ecosystems in question. Figure 12 gives a diagrammatic summary of the model structure and nature, the main constituents and some of their interactions, over that time. Even the earliest, most simple models gave answers of considerable worth to the water supply business, and showed where ecological ignorance was particularly acute. The models currently being developed are, unavoidably, substantially more complex than their predecessors. Whilst the models may be of significant help in further understanding and, perhaps, quantification of these ecosystems, their increasing complexity can cause considerable difficulty to their full interpretation and resulting credibility. Experience has shown that there is no one, 'best' model, and the reservoir management has simply adopted the least complex model which will serve the immediate needs.

The development of the ecological knowledge of the lower Thames reservoirs and its application has been an iterative process. Figure 13 illustrates some of the complementary aspects of modelling reservoir quality dynamics and the growth in ecological knowledge. In this particular instance, the initial impetus came from management questions. In the beginning, ignorance about the problems addressed in those questions was great, but relatively straight forward experimentation and measurement rapidly gave considerable insight into the ecological factors involved. This led to simple modelling which nonetheless could greatly enhance the understanding of the system by providing quantified, testable prediction about the system. That enhanced understanding could then be translated into an answer to the original question posed.

Inevitably, once the simple questions have been addressed, the more complex issues become even more pressing, and the main cycle is traversed again. Figure 13 tries also to illustrate some of the limitations involved. For example, more complex business questions may be considered as the problem of ignorance is reduced ('what is it, in ecosystem terms, that is implied by this particular question': 'in the present instance, what do we need to know to answer the question about whether the current reservoir algal conditions are long-term stable'?). However, such ignorance will never be entirely eradicated. Also, as the more complex questions are addressed, there can be exponential increase in the difficulty of the associated, necessary measurement and modelling. This increased complexity does not however necessarily translate into similarly enhanced understanding of the system in question. There is probably a limit to the system insight that can be attained. Overly complex, or ill-conceived models may even so cloud the whole issue that credibility in the capacity of the models to provide insights into the system is diminished – to the detriment of the overall enterprise!

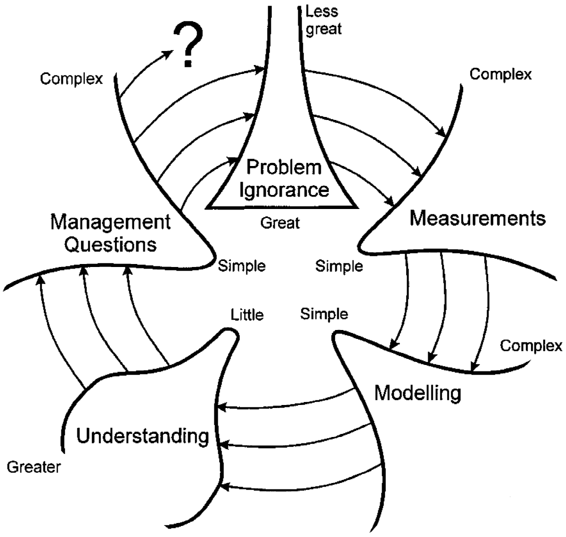

Figure 13. Diagrammatic representation of the main inter-related elements and pathways involved in the development of a predictive model, ecological understanding and management application. Degrees of difficulty and magnitudes are also indicated.

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