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Ecological principles for management techniques in deeper reservoirs

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Abstract

Eutrophic reservoirs produce many management problems for both water supply and recreational management, which are only expensively solved. A range of eco-technical methods have been used over the past 10 years to control phosphorus and nuisance algal bloom levels. Three case studies are used to show that these methods may be succesful but, if used without careful monoitoring or controls, or if not applied in combination with each other or with other techniques, their practical value and benefit-cost ratio may not be apparent.

Introduction

The need to control excessive growths of phytoplankton algae in reservoirs has been well documented (Lund, 1970 & 1981; Steel, 1972; Youngman, 1975; Collingwood, 1977). Algal interference with water treatment includes taste and odour problems (Hayes & Greene, 1984), reduced filtering capacity and filter runs (Youngman, 1975) (Figure 1), disturbance of flocculation process (Collingwood, 1977; Hayes & Greene, 1984), breakthrough of algal cells into water distribution systems providing a carbon source for animals (Smart, 1989; Smart & Harper, 1999) and presence of cyanobacteria toxins (Zalewski, 1999). These problems all contribute to greater water treatment costs although it is very difficult to estimate what this cost must be for individual water treatment works (Harper, 1992). In lakes and reservoirs with a high amenity or conservation value, algal control is also required to reduce the interference with recreational activities and improve the aesthetic nature of the site. At Rutland Water (U.K.), a very severe bloom of the cyanobacteria, *Microsystis aeruginosa* led to the closure of the whole lake to recreational activities in the late summer of 1989.

Ecological principles for the control of phytoplankton and lake restoration

There are many methods used for controlling phytoplankton growth and eutrophication in reservoirs and deep lakes. Sas (1989), Klapper (1991), Harper (1992) and Cooke et al. (1993) review many of these techniques and report on the degree of success. Three case studies are presented here from different reservoirs in central England, which demonstrate different ecological principles, which can be used for control. The techniques are:

1. Artificial mixing, with the aim not only of preventing stratrification, but also of reducing the underwater light available to algal cells, and subsequent disruption to potential maximum algal growth rates, was carried out at Staunton Harold reservoir in Derbyshire, England.

2. Nutrient reduction by direct ferric sulphate dosing into the reservoir to inactivate the phosphorus in the incoming river water, was carried out between 1990 and 1995 at Rutland Water in Leicestershire, England.

3. Biological control has also been investigated at Covenham reservoir, Lincolnshire, England which is designed on the Thames reservoir design (a simple flow through system consisting of a concrete bowl with an inflow and outflow). Both nutrient reduction by ferric dosing (1990–1993) and artificial mixing have also been carried out at Covenham reservoir during the study period.

Artificial destratification to control phytoplankton in Staunton Harold reservoir

Artificial destratification had been successful for improving water quality by preventing deoxygenation since the early 1950s (Hooper et al., 1953) and many new reservoirs are designed with artificial mixing systems (Steel, 1972; Tolland, 1977). The effects of artificial mixing upon phytoplankton have often been contradictory. The success or failure of artificial mixing in altering phytoplankton biomass and succession would appear to depend upon the timing and intensity of mixing energy which Ridley (1971) noted would be different for each lake. Shapiro (1981) suggested that when increases in phytoplankton occurred after mixing, it was probably due to inadequate circulation with the phytoplankton not being thoroughly mixed throughout the water column, but the nutrient concentrations increased in the euphotic zone. In contrast, when a decrease in phytoplankton biomass was recorded, it was due to complete circulation with the algal cells spending less time in the photic zone and thus, their production was reduced. Low algal biomasses were routinely found in the London reservoirs even though the water in the reservoirs had significant concentrations of soluble reactive phosphorus and nitrate ($>$1mg $l^{-1}$$PO_4$-P and 7 mg $l^{-1}$$NO_3$-N) (Ridley et al., 1966). The phytoplankton biomass were maintained well below the nutrient carrying capacity (20–60 μg l^{-1} chlorophyll a; Duncan, 1990) due to the use of powerful jetted inlet water (Cooley & Harris, 1954; Steel, 1972).

Staunton Harold reservoir forms part of the water supply system for the City of Leicester in the East Midlands. Water was pumped from the River Dove in Derbyshire and used to fill two valley reservoirs – Staunton Harold and Foremark reservoirs. The raw water from the reservoirs was then treated at Melbourne Treatment works before being pumped to Leicester, as part of the potable water supply system (Figure 2). Both reservoirs were considered to be eutrophic and supported summer blooms of cyanobacteria often in excess of 50 μg l^{-1} (Brierley, 1985). The major characteristics of the reservoirs are outlined in Table 1.

Artificial mixing was carried out using a inexpensive and efficient perforated-pipe system (Tolland, 1977; Davis, 1980) in Staunton Harold reservoir. The nearby Foremark reservoir was used as a ‘control’ – the reservoir had been shown to behave similarly with respect to stratification, water chemistry and phytoplankton activity (Severn Trent Water Authority, unpublished). A 40 mm diameter polythene pipe, perforated for the last 150 m, ran out into the deepest part of the reservoir at right angles to the dam. A compressor was connected to the pipe below the dam. The compressed air left the pipe *via* the small holes and rose to the surface as a curtain of bubbles, water being entrained from the lower levels to the surface. Local destratification around the pipe was then followed by mixing of the whole reservoir. The estimated air flow from the compressor was between 27 and 28.8 l s^{-1} during the study period.

Table 1. The physical characteristics of Staunton Harold and Foremark reservoirs

	Staunton Harold	Foremark
Construction completed	1964	1976
Volume (10^6 m^3)	6.36	13.4
Area (hectares)	85	92
Maximum depth (m)	23	32
Mean depth (m)	7.9	14.3
Catchment area	2590	295
Nominal retention time (days) 1981–1983	108	148

The perforated pipe system was able to fully destratify the reservoir in three days during September 1981, and when used continuously prevented Staunton Harold from stratifying in 1982 and 1983. A maximum temperature difference between the surface and bottom of 3.3 °C occurred in Staunton Harold on 7 June 1982 compared to a difference of 13.25 °C in Foremark reservoir (Brierley, 1985). The stability of the water columns, measured as the Brunt-Vasala frequency – N^2 and defined by Phillips (1966) and Boyce (1974) by the equation:

$$N^2 = -\frac{\boldsymbol{g}\mathrm{d}k}{k\mathrm{d}z}\ (\mathrm{sec}^{-2}),$$

where $\boldsymbol{g}$ is the acceleration due to gravity; k is the density of water at 4 °C; dk/dz is the density gradient over a specified depth interval.

High values of N^2 indicate that density gradients are intense and as a result the resistance to mixing is high whilst a low N^2 value indicates weak or low density gradients and that the water column is unstable or well mixed. The N^2 values were calculated from weekly temperature profiles and the changes in N^2 (0–5 m) and (0–15 m) in Staunton Harold and Foremark reservoir are shown in Figure 3.

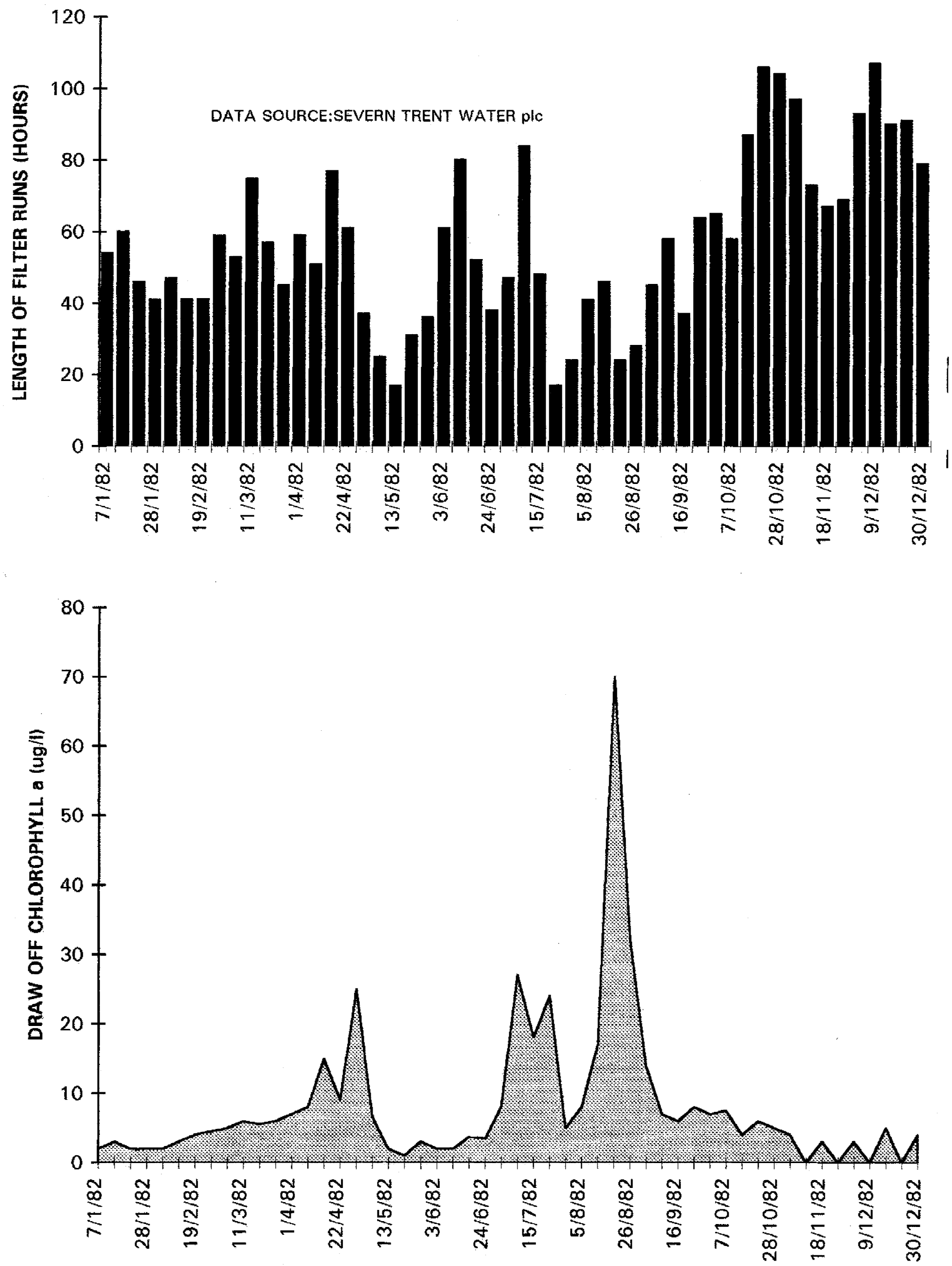

Figure 1. Filter run length at Melbourne Water Treatment works in relation to the development of chlorophyll in Staunton Harold Reservoir.

In 1981, oscillations in both the 0–5 m and 0–15 m N^2 values were similar for both reservoirs. The similarities and low N^2 values were a result of the poor climactic conditions which prevented Staunton Harold from stratifying until late in the season and depressed the metalimnetic layer in Foremark to between 14 and 18 m for most of the summer. The effects of continuous artificial mixing on water column stability in Staunton Harold were very noticeable in 1982 and 1983. N^2 (0–5 m) in Foremark showed large peaks in both years which were related to periods of intense heating of the epilimnion. Generally N^2 (0–5 m) values for Staunton Harold were much lower than those recorded for Foremark. The differences in the

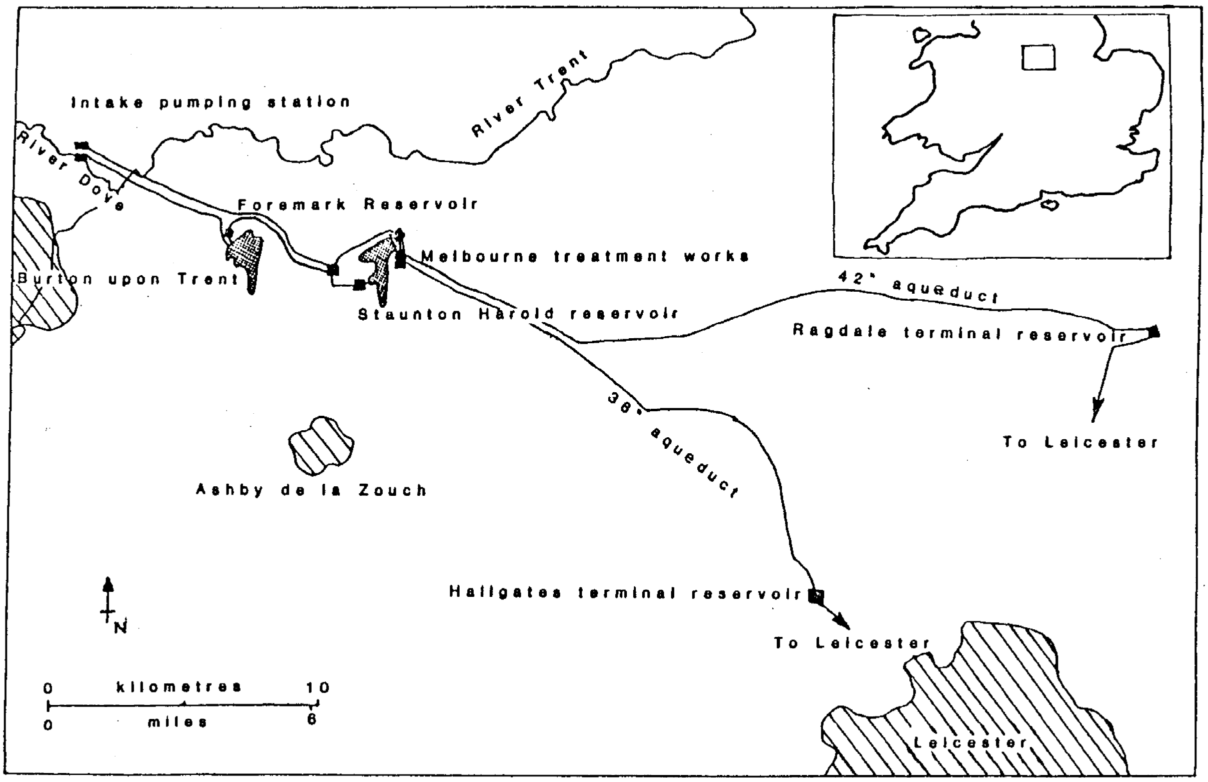

Figure 2. Location of Staunton Harold Reservoir.

water column stability between the reservoirs was further emphasised when using the N^2 (0–15 m) values with Foremark having higher values between May and September in both 1982 and 1983.

The mixing in 1982 and 1983 also prevented any build up of soluble reactive phosphorus and silica in the lower layers, whilst in Foremark typical clinograde distributions of both nutrients occurred for much of the summer period, as the hypolimnion became progressively more anaerobic.

Although the water column remained well mixed throughout most of 1982 and 1983 as a result of the continuous mixing, phytoplankton biomass was observed to stratify (Figure 4 shows chlorophyll *a* isopleths during 1982) when water column stability increased greatly or motile species were present, e.g. *Aphanizomenon flos-aquae* in August 1982. In Foremark, algal stratification occurred in all three summers and the biomass was concentrated in the top 5 m (Figure 5).

Analyses of the individual species' maximal rate of population increases (Table 2) showed that small centric diatoms, *Cryptomonas* spp. and *Rhodomonas* spp. had highest rates during artificially mixed years and lowest rates in stratified years. In contrast, *Aphanizomenon flos-aquae*, *Anabaena* spp and *Microcystis aeruginosa* all had rates of increases which were lowest in artificially mixed years and highest in the stratified years. The seasonal succession in both reservoirs was typical of eutrophic reservoirs with diatoms in spring, followed by colonial chlorococcales in early summer and cyanobacteria dominating in mid to late summer. The phytoplankton assemblages found in the reservoirs, were identical to those found by Reynolds (1984) in eutrophic reservoirs, and the periodicity of these assemblages appeared to be driven by nutrient availability and water column stability. The only major disturbance from the sequence of assemblages as a result of artificial mixing occurred in 1983 when an assemblage not recorded before and consisting of *Chlamydamonas* sp. and a mixture of small chlorophytes. This assemblage became dominant in July and September and it was proposed that it had resulted from the increased nutrient availability in the euphotic zone as an indirect effect of the mixing.

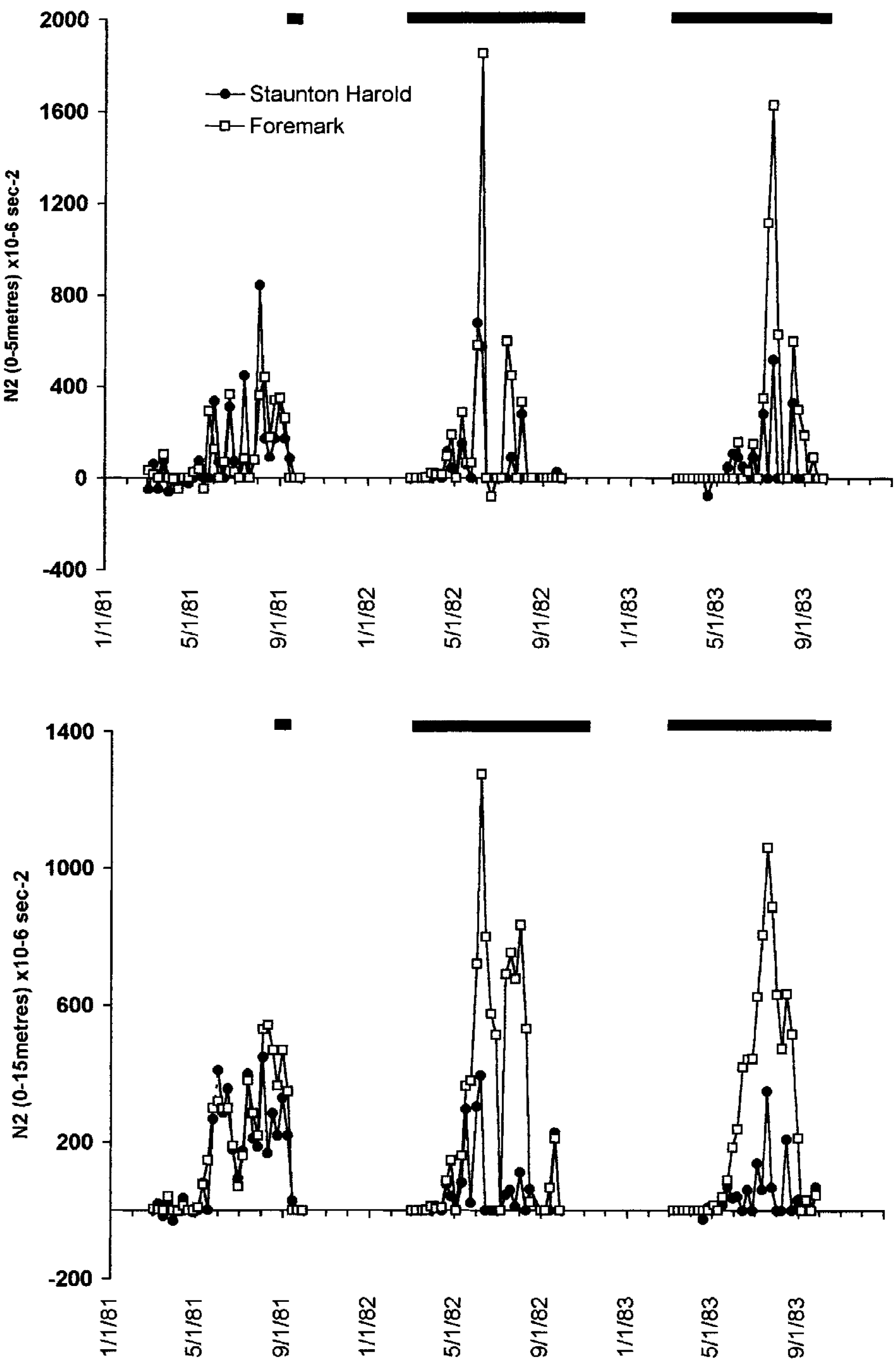

Figure 3. Temporal change in water column stability at 0–5 m and 0–15 m in Staunton Harold Reservoir (closed circles) and Foremark Reservoir (open circles). Solid bars indicate periods of artificial mixing in Staunton Harold.

Mean algal biomass (as chlorophyll *a*) was lower in one year of artificial mixing (1982) and higher in another (1983) when compared to the mean biomass in the naturally mixed year and those in Foremark reservoir which stratified (Table 3), probably due to

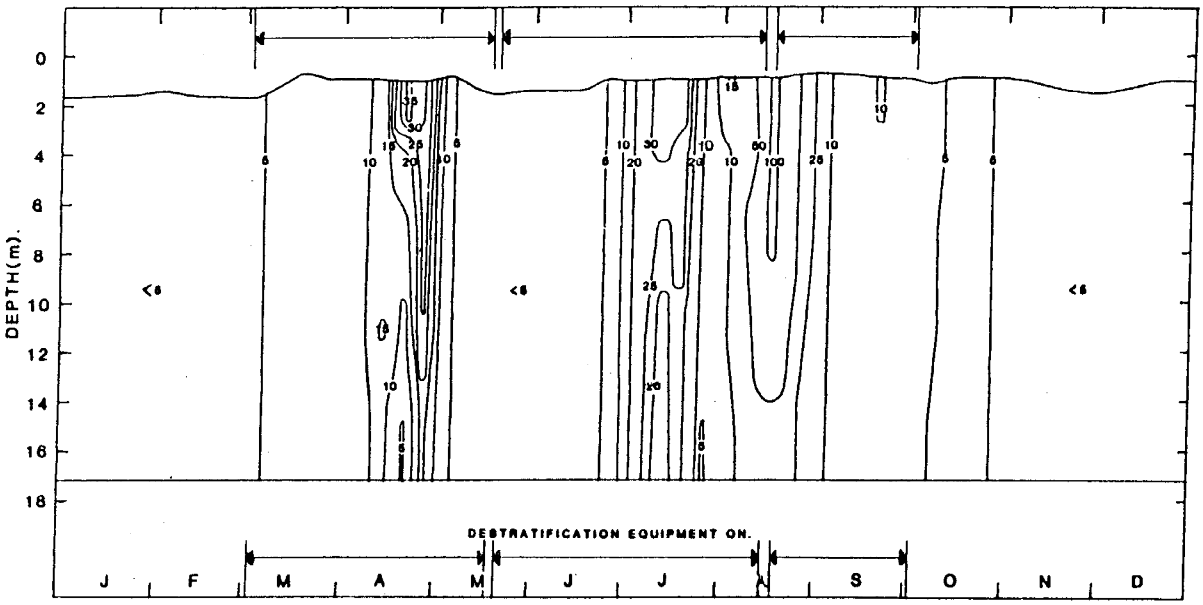

Figure 4. Variations in the vertical distribution of chlorophyll in Staunton Harold. Isopleths in $\mu g\ l^{-1}$.

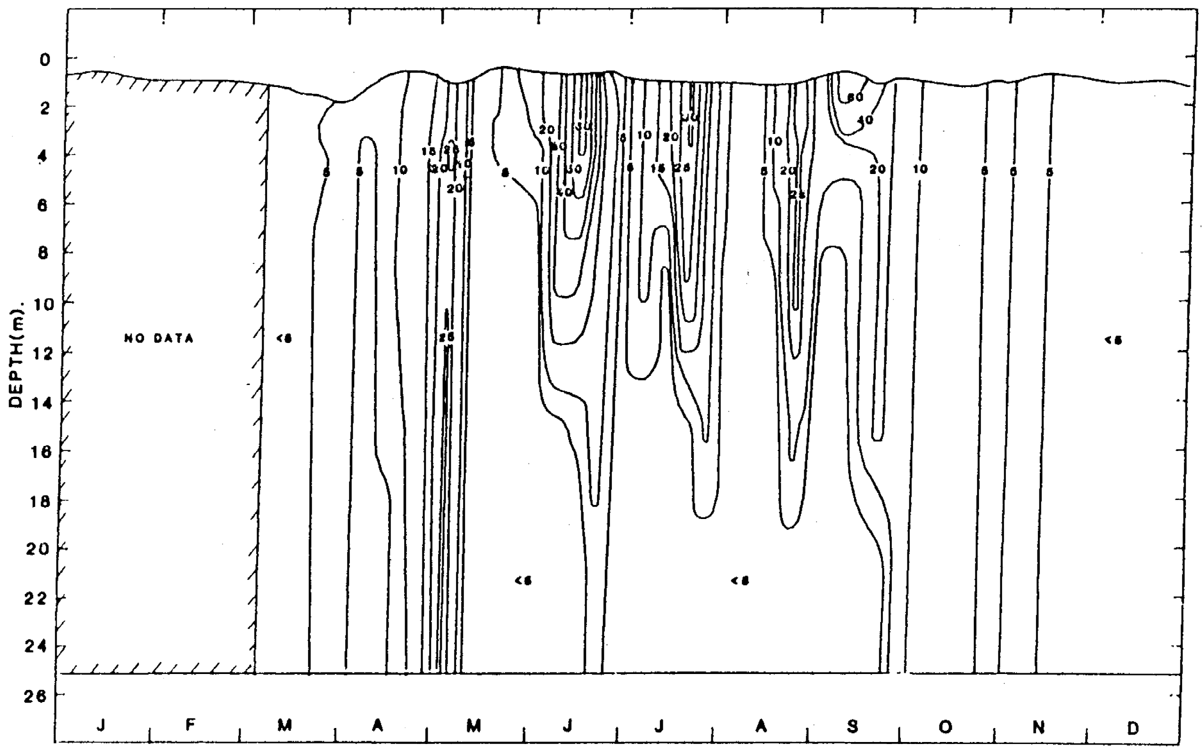

Figure 5. Variations in the vertical distribution of chlorophyll in Foremark. Isopleths in $\mu g\ l^{-1}$.

inadequate mixing and the ability of the phytoplankton to stratify.

The productivity and photo-adaptation of the mixed and stratified phytoplankton populations was

Table 2. The maximum observed summer rates of population increase (kn+ $d^{-1)}$ of selected species of phytoplankton from Staunton Harold and Foremark reservoirs between 1981 and 1983

Reservoir	Artificial mixing		Natural mixing			
	Staunton Harold	Staunton Harold	Staunton Harold	Foremark	Foremark	Foremark
Year	1982	1983	1981	1981	1982	1983
Small centric diatoms	0.38	0.34	0.31	0.3	0.29	0.20
Asterionella	0.27	0.08	-	0.45	0.15	0.15
Cryptomonas	0.35	0.16	0.20	0.06	0.09	0.16
Rhodomonas	0.29	0.42	0.29	0.34	0.28	0.15
Aphanizomenon	0.19	0.13	0.20	0.18	0.31	0.29
Anabaena	0.14	0.18	0.26	0.11	0.23	0.14
Microcystis	0.16	0.19	0.16	0.16	0.17	0.26

Table 3. Annual mean and maximum algal biomass (as chlorophyll *a*) under mixed and stratified conditions in the surface layers of Staunton Harold and Foremark reservoirs

		Chlorophyll *a* (μg l^{-1})	
		Staunton Harold	Foremark
1981	Maximum	114.0*	213.0
	Mean	17.9*	18.4
1982	Maximum	**69.0**	71.0
	Mean	**10.9**	14.7
1983	Maximum	**43.0**	73.0
	Mean	**11.4**	13.7

Figures in bold indicate artificially mixed years, * 1981 was a naturally (wind mixed) year.

measured in both 1982 and 1983 in both reservoirs by light/dark bottles with titration of changes in oxygen concentration. Depth integrated values for photosynthesis (ΣA) which include variation in underwater light with depth, were significantly lower from mixed populations (Table 4). Much of the variance of integral photosynthesis was reflected by the variation in phytoplankton biomass on the different occasions. This was normalised by integrating the depth profiles of photosynthesis per unit biomass to demonstrate that the integral of the specific rate of photosynthesis per unit chlorophyll (ΣP) was significantly lower for mixed populations also (Table 4).

Productivity experiments showed that phytoplankton cells from different depths in the stratified populations appeared to be light/shade adapted, whilst those from mixed populations were tolerant to a wide range of irradiances (Brierley, 1985). It would appear that this tolerance was a result of a selection of species which have either evolved to favour turbulent conditions, or to control carbon assimilation through excretion of photosynthates and photorespiration, or regulation of their own light climate by vertical movements. These populations also had lower assimilation rates compared to the stratified populations.

Changes in phytoplankton periodicity as a result of mixing have been widely described, although changes in productivity have been rarely investigated. Haynes (1971) found a large decrease in *Aphanizomenon flos-aquae* during mixing was due to the redistribution of filaments throughout the water column, rather than a decline in the population. Reynolds et al. (1983, 1984) imposed an artificially imposed cycle of mixing and restratification on the phytoplankton in large limnetic enclosures (Lund tubes) in Blelham Tarn and that the phytoplankton responded in a predictable way with species of winter/spring phytoplankton (eg. diatoms) were selected for and became dominant during mixing episodes, whilst summer assemblages were favoured by and increased during restratification periods. The growth and dominance of slow growing, late summer specie such as *Microcystis aeruginosa* and *Ceratium hirudinella* were delayed. Reynolds (1980) also predicted that the average biomass would be maintained

Table 4. The mean photosynthetic characteristics (+/- s.e.) of mixed and stratified populations of phytoplankton in Staunton Harold and Foremark reservoirs

Photosynthetic characteristic	Mixed populations	Stratified populations
Integral photosynthesis, $\Sigma A(mgO_2\ m^{-2}\ h^{-1})$	296.5 (± 48.8)	596.2 (± 72.4)
Integral of specific photosynthesis, $\Sigma P\ (mgO_2\ mg\ chl^{-1}\ m^{-2}\ h^{-1})$	17.52 (± 2.09)	26.68 (± 3.16)

below the nutrient carrying capacity if mixing was carried out. Ryback (1985) had also found that species more typical of winter/spring assemblages were favoured by mixing.

Steel (1972) concluded from work on the Thames reservoirs that the phytoplankton biomass was lower than expected due to the reduced underwater light that the phytoplankton received as a result of the powerful vertical mixing. The reduced gross photosynthetic rates and maintained respiration rates then led to a lowering of net photosynthesis (Talling, 1957, 1965, 1971) and thus phytoplankton biomass. These changes in rate processes clearly occurred in a full-scale mixing experiment.

Artifical mixing is widespread in lowland reservoirs, but is frequently used in a continuous mode – to maintain well-oxygenated, unstratified water. Its use could, however, not give unimportant control over algal succession and productivity in those reservoirs where water managers have the vision to integrate it with their ecological monitoring results.

Phosphorus inactivation by ferric sulphate dosing and nutrient reduction by changes in the pumping regimes at Rutland Water, 1990–95

The excessive blooms and scums of *Microcystis aeruginosa* at Rutland Water in 1989, with deaths of sheep and dogs (NRA 1990) and the subsequent closure of the reservoir to leisure activities, led to both national and international media attention. In June 1990, Anglian Water Services plc started to directly dose Rutland Water with ferric sulphate. This followed an experiment at Foxcote reservoir (see Daldorph, 1999) where large stands of macrophytes were recorded after ferric dosing successfully reduced the phytoplankton biomass (Young et al., 1991). Initially, dosing at Rutland Water was seen as a short term measure to reduce

Table 5. The main physical characteristics of Rutland Water

Construction completed	1975
Volume ($m^3 \times 10^6$)	136.9
Area (hectares)	1255
Maximum depth (m)	34
Mean depth (m)	10.9
Nominal retention time (days) – 1993	730
Natural catchment area (hectares)	7400

the soluble reactive phosphorus and potentially lower the peak and mean algal biomasses.

Rutland Water was the largest man-made, potable, pump-storage reservoir in Europe when constructed in 1976 (Harper, 1982) (and is still the largest in the U.K.). Water is pumped from the eutrophic rivers, Welland and Nene, into the reservoir and enters the reservoir in the south arm via submerged jets. These rivers usually provide between 60 and 90% of the total water input. The main physical characteristics of the site are shown in Table 5 and the location of the reservoir in relation to the catchments is shown in Figure 6.

Water from the reservoir is treated at Wing water treatment works (or it may in emergencies be directly used for river regulation). The water is then put into the water supply system together with two other reservoirs, Grafham and Pitsford, which between them supply up to 1.5 million customers with up to 545 l $\times\ 10^6$ day^{-1} potable water in Northampton, Peterborough, Milton Keynes, Bedford, Kettering and Corby (AWS, unpublished). The reservoir has two multiple depth draw-off towers and the main body is mixed during the summer months (usually May–October) to prevent stratification using Helixor air guns (Harper, 1982).

Recreation and conservation interests were included in the development of the site from early on in the planning and building of the reservoir (Knights,

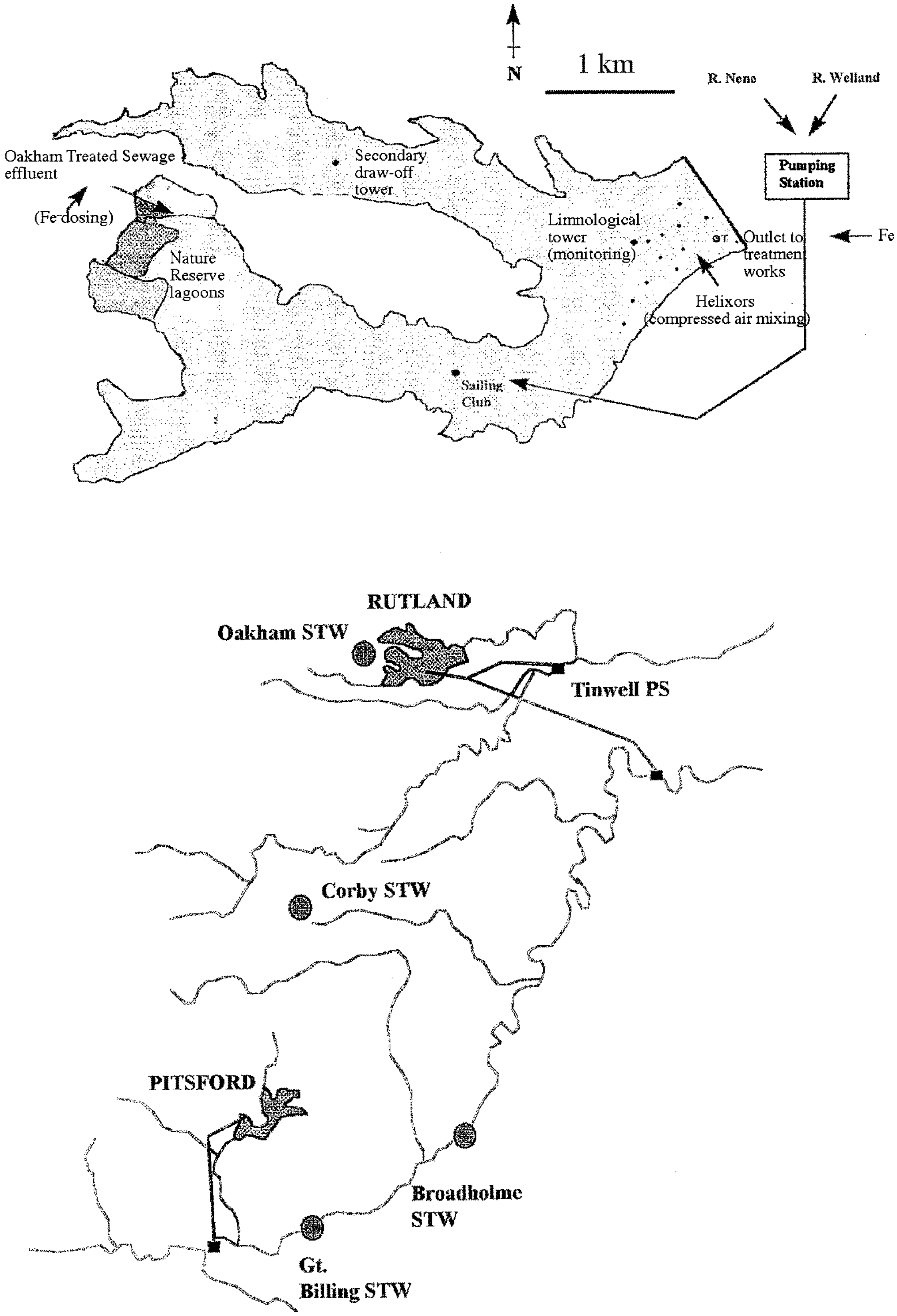

Figure 6. Rutland Water and its catchment area, showing major sewage works location (STW).

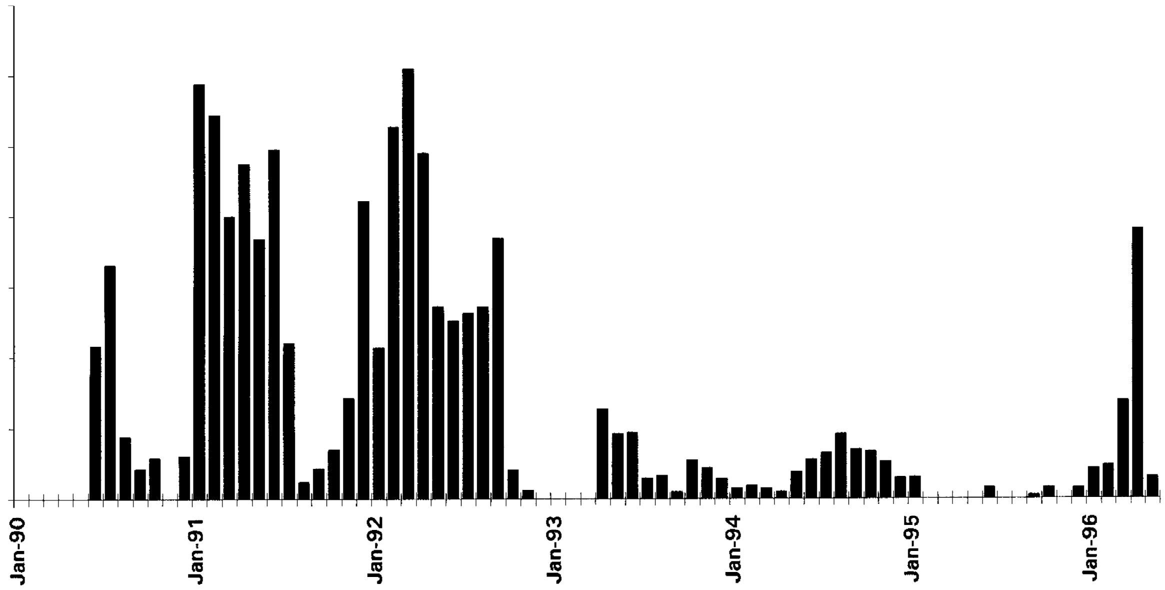

Figure 7. Quantity of ferric sulphate dosed into Rutland Water.

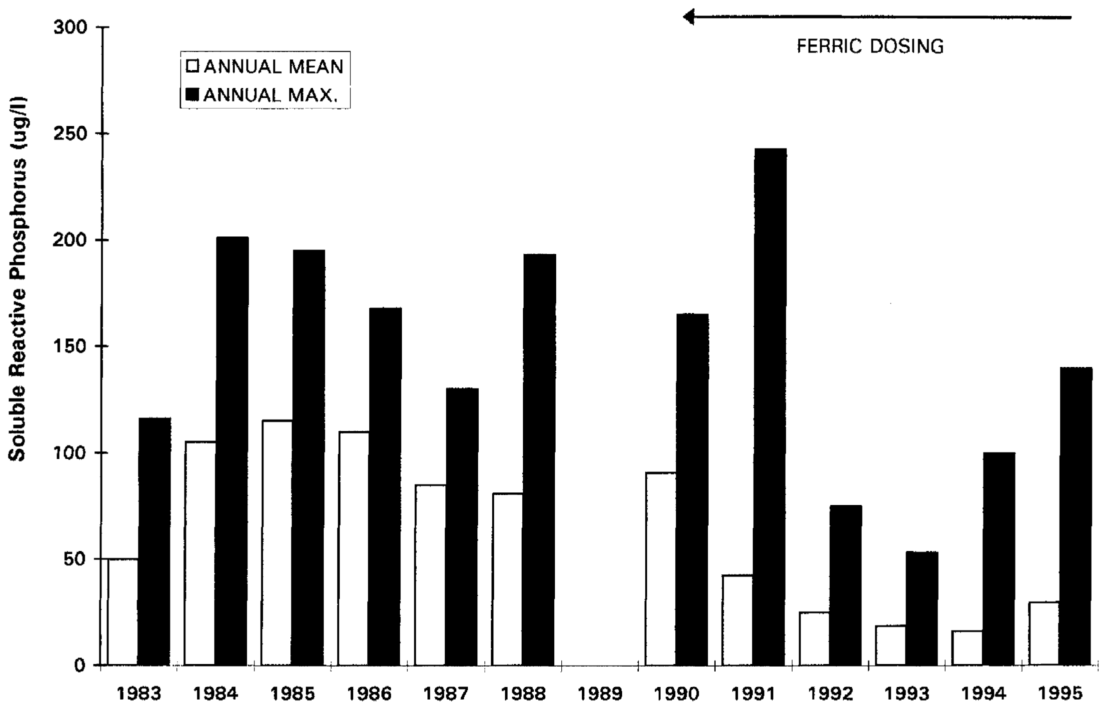

Figure 8. Annual mean and maximum soluble phosphorus in Rutland Water, 1983–85.

1982), with walking, sailing, surfboarding, cycling, picnic areas and conservation centres being some of the attractions. The site was developed as a 'put and take' trout fishery with an international reputation. The site is of national and international importance for resident or visiting birds, and has been designated a Site of Special Scientific Interest, a RAMSAR site and a Special Protection Area under the EU Birds Directive.

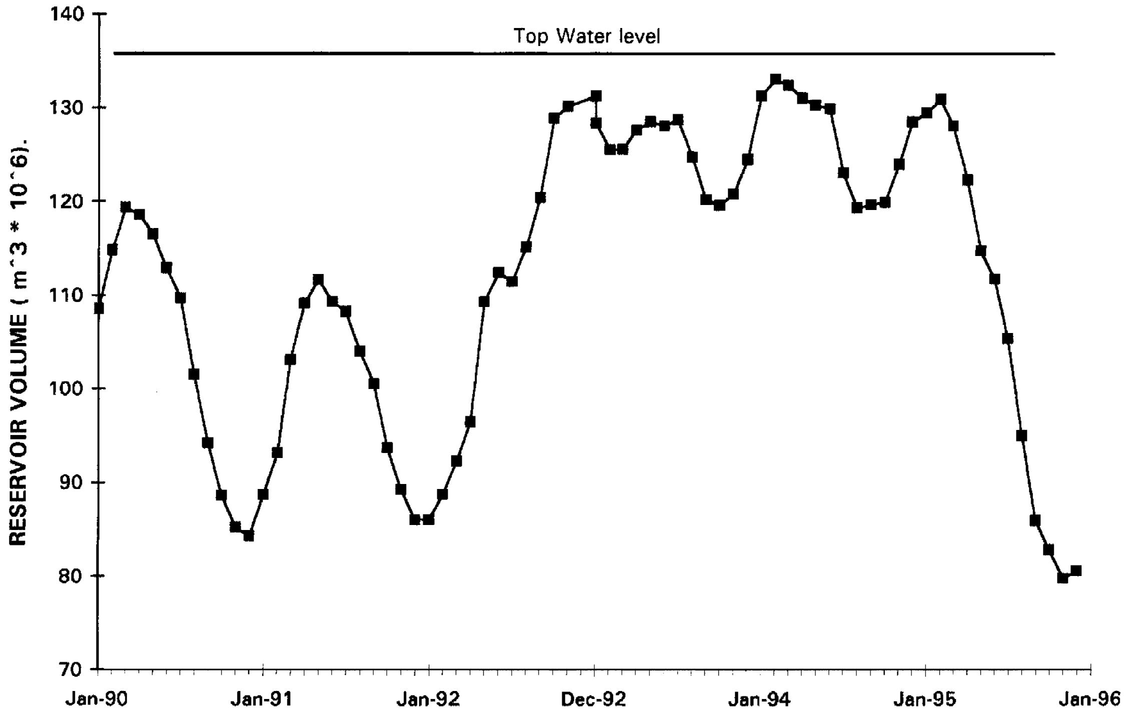

Figure 9. Seasonal changes in reservoir volume, Rutland Water, 1990–95.

Detailed limnological research and monitoring of the reservoir has been carried out since the reservoir started filling in 1975 (Harper & Bullock, 1982). It is eutrophic, with phytoplankton periodicity typical of eutrophic lakes (Ferguson & Harper, 1982). Phytoplankton biomass was initially maintained well below the nutrient carrying capacity, probably as a consequence of inadvertent discontinuous mixing (see above). The continuous operation of the 'Helixors' may be successful in reducing algal biomass as stable conditions persist during anticyclonic weather and full mixing would occur during windy weather. Thus a pattern of intermittent mixing with alternate mixed and stratified water columns results, as predicted by Reynolds et al. (1984).

The Environment Agency and its predecessor, the National Rivers Authority (NRA), have monitored weekly changes in the physical, chemical and biological (phytoplankton and zooplankton) nature at up to five sites at the reservoir since June 1990. Benthic and littoral invertebrates have been monitored, on a less frequent basis, since September 1990. Full results of the invertebrate monitoring will be reported elsewhere. The sites monitored weekly and the main features of the reservoir are shown in Figure 6 .

Ferric sulphate dosing of the reservoir started in June 1990 and continued until 1996. The timing and magnitude of ferric dosed are shown in Figure 7. The annual mean concentration of soluble reactive phosphorus at the Limnological Tower declined from 1990 to 1994 (Figure 8) but increased slightly in 1995. The annual maxima appeared to decrease from 1991 to 1993 but increased in 1994 and 1995. The lower mean and maximum concentrations of soluble reactive phosphorus cannot be attributed solely to the ferric dosing as there were major changes in two other aspects of reservoir management over this period. The dosing period coincided with two major drought periods (1990–1992 and 1995–1997) which led to major changes in the input pumping regimes (because of the minimum flows required in the rivers Welland and Nene). The changes in the reservoir volumes during this period are shown in Figure 9. Also the Helixor air guns used for artificially mixing the reservoir, were upgraded in 1992 (Daldorph, pers. comm.).

There was, thus, a decrease in the quantity of river water pumped into the reservoir (Figure 10a). The phosphorus load entering the reservoir from these main river sources also declined during this period (Figure 10b).

The response of phytoplankton to the decreases in phosphorus as a result of the reduced phosphorus, inputs from the main rivers, and ferric dosing are illustrated by the annual mean and maximum chlorophyll concentrations at the Limnological Tower in Figure 11. The annual mean concentration did not exceed 20 μg l^{-1} from reservoir filling in 1975 to 1995. Since 1990 there were three years when the annual mean was at the lowest recorded levels, this was no doubt a response to the lower soluble reactive phosphorus concentrations. In 1993 a peak of over 120 μg l^{-1} chlorophyll was recorded and in 1994 the highest chlorophyll level measured during routine monitoring was found at buoy S12 and exceeded 220 μg l^{-1}. These high peaks are thought to have been in response to the increased loads of nutrients as the reservoir was being refilled during 1993 after the drought (see Figures 9 and 10) together with an unknown quantity of phosphorus released from the reflooded littoral area. There had been a drawdown greater than 30% in January 1992 which created >370 ha of exposed sediments. Large parts of this exposed area were colonised by terrestrial plants and so when the reservoir refilled there was the potential for release of phosphorus from both plants and sediments. The release of nutrients from exposed and shallow littoral areas is one area of study which needs to be carried out.

Anglian Water Services plc also instigated a longer term remedial measure in 1993. Experimental trials on phosphorus stripping at major sewage works in the River Nene catchment, started prior to the legal requirement to carry out phosphorus stripping under the EC Urban Waste Water Treatment Directive. These trials have been monitored by The Environment Agency (Rose & Balbi, 1997).

In response to the problems encountered at Rutland Water and the complex issues involved in the management of such a site, English Nature, Anglian Water Services and the Environment Agency set up a tripartite group in 1995 which had the task of developing an 'Action Plan' for the cost-effective, sustainable management of the reservoir. To date, the group has developed the framework in which the roles and responsibilities of the three organisations were laid out and both legal and other targets have been proposed. Steps still to be carried out include the determination, appraisal, assessment of costs and benefits and of the impacts of control measures. Then recommendations for the preferred options along with monitoring requirements will then be put out for consultation. The Action Plan is due for completion in late 1998.

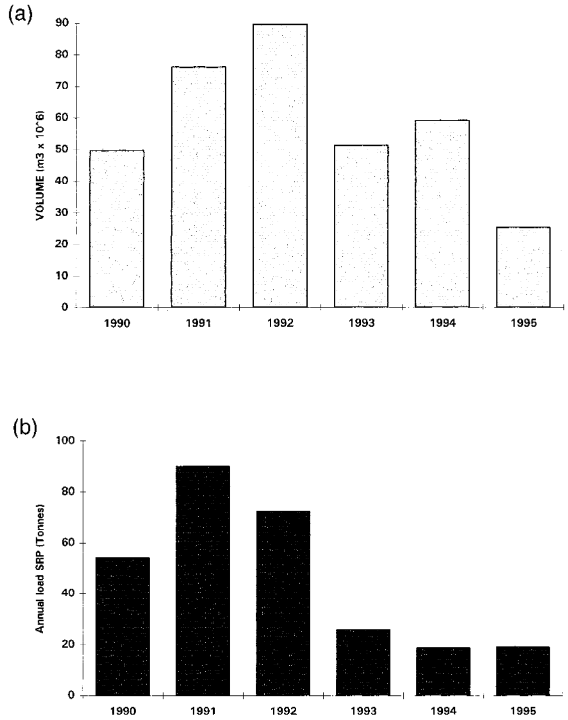

Figure 10. Annual volumes of water (10a) and load of soluble reactive phosphorus (10b) pumped into Rutland Water, 1990–95.

Biological control, direct ferric dosing, reduction of phosphorus loads and artificial mixing at Covenham reservoir in Lincolnshire

The design of Covenham reservoir is based upon that of the London reservoirs – a steep-sided concrete bowl with a jetted inlet. The main physical features are shown in Table 6 and the location in Figure 12. Water is pumped from the Louth Canal into the reservoir *via* the inlet jets, and then is abstracted to the water treatment works which are adjacent to the reservoir for treatment and distribution into the supply network. Direct ferric sulphate dosing was carried out at the reservoir between 1990 and 1993, and dosing of the

Table 6. The main physical characteristics of Covenham reservoir

Construction completed	1970
Volume (m^3 $\times$ 10^6)	11.37
Area (hectares)	80
Maximum depth (m)	15.2
Mean depth (m)	14
Nominal retention time (days)	224

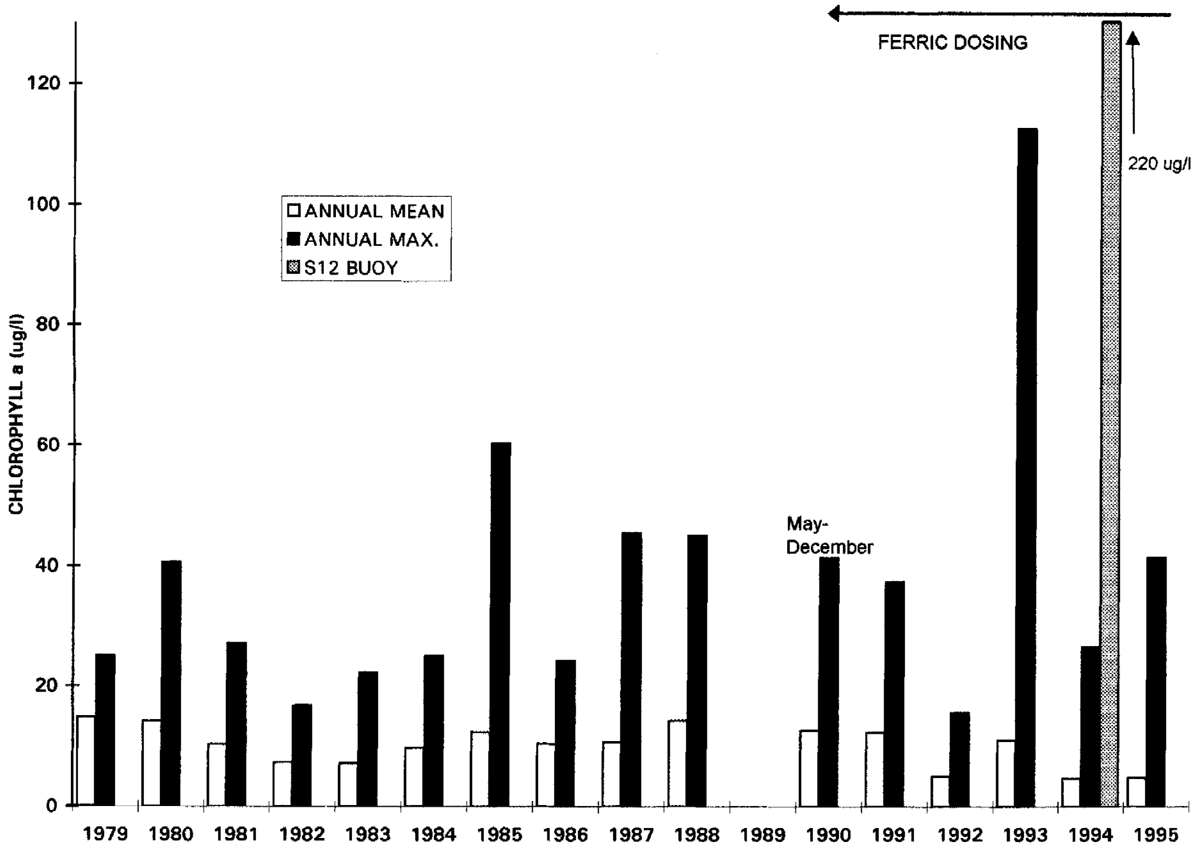

Figure 11. Annual mean and maximum chlorophyll, Rutland Water, 1979–95.

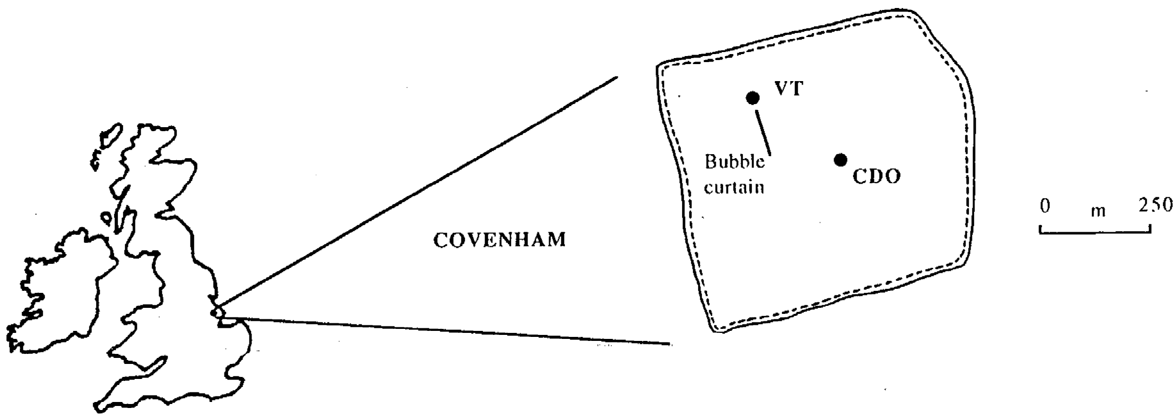

Figure 12. Location and shape of Covenham Reservoir.

effluent from Louth sewage treatment works, which is upstream of the reservoir intake, was undertaken from 1990 onwards. A perforated pipe system was installed in 1990 to aid mixing.

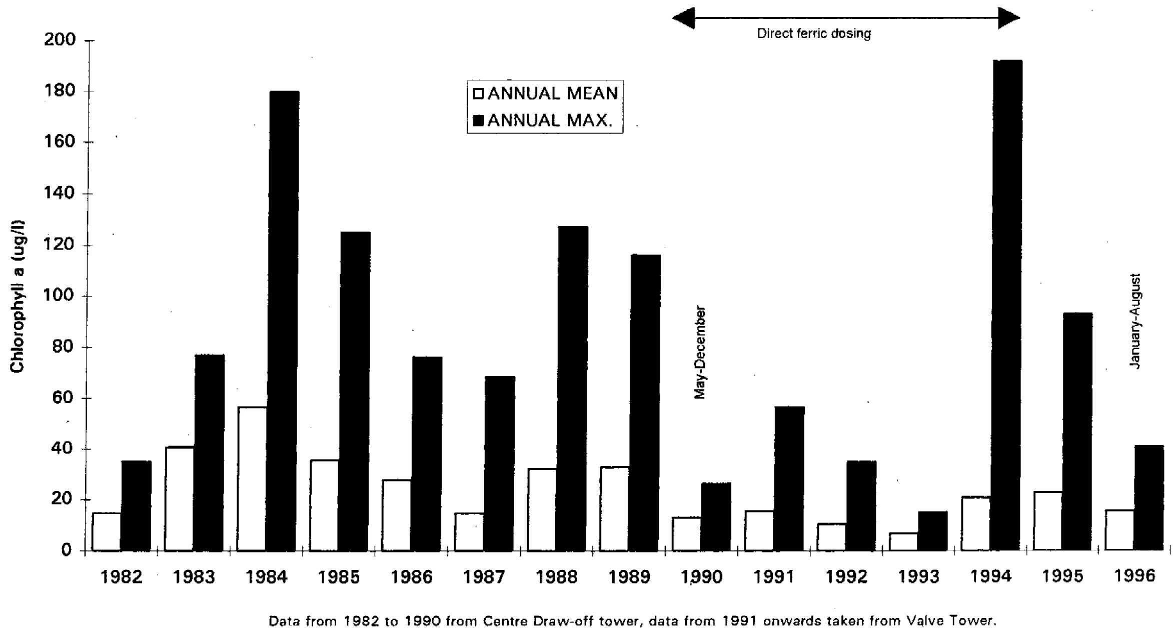

Figure 13. Annual mean and maximum chlorophyll, Covenham Reservoir, 1982–96.

Large phytoplankton biomasses and cyanobacterial blooms and scums have occurred at Covenham reservoir (Anglian Water, unpublished). Figure 13 shows the annual mean and maximum chlorophyll levels. Unlike the London reservoirs (Duncan, 1990), the design alone does not result in low algal biomasses and high water transparencies.

The Environment Agency has been monitoring the limnology of Covenham reservoir since 1990, when Anglian Water started direct ferric dosing into the reservoir. Dosing stopped in October 1994. Artificial mixing using a perforated pipe is used in the summer months to prevent thermal stratification. In 1995, a fish survey to estimate the biomass and community composition was commissioned by Anglain Water and the Environment Agency.

The combination of management control options, which have been undertaken between 1990 and 1995, have led to difficulties in clearly analysing the data to establish which techniques are most successful in reducing phytoplankton. Soluble reactive phosphorus concentrations, expressed as annual mean and maxima, showed a reduction after direct and indirect ferric dosing had been implemented (Figure 14). Both total and soluble reactive phosphorus decreased within one year from the start of dosing and soluble reactive phosphorus remained low (Figure 15). During this period, the phytoplankton did not show such any major reductions in biomass (Figure 13) but there were changes in the successional pattern (Figure 16). The cyanobacterium *Aphanizomenon flos-aquae* was recorded in December 1990 and again in December 1991, together with *Pediastrum*, *Coelastrum*, *Oocystis* and *Anabaena.* In 1992, an early bloom of *Aphanizomenon* and *Anabaena* occurred in May, and the late summer biomass peak was dominated by *Cosmarium* and *Ceratium,* rather than the more typical cyanobacterial genera. *Aphanizomenon* was also present with small centric diatoms in March 1993 and March 1994. A large bloom of *Aphanizomenon* occurred in July 1994. The phytoplankton biomass increased slightly but the succession did not return to the pre-dosing pattern once dosing had ceased.

Covenham reservoir, like the London reservoirs, supports three coexisting species of *Daphnia.* The largest of these is *Daphnia magna*, with *D. pulicaria* intermediate and *D. galeata* the smallest. The changes in the dominance of these three species is shown in Figure 17. *D. magna* dominated the cladoceran zooplankton in May and June, 1991, whilst in 1992 *D. galeata* was dominant from May through to October. In 1993, no single species was clearly dominant throughout the summer. *D. pulicaria* peaked in February and early April and remained the dominant species

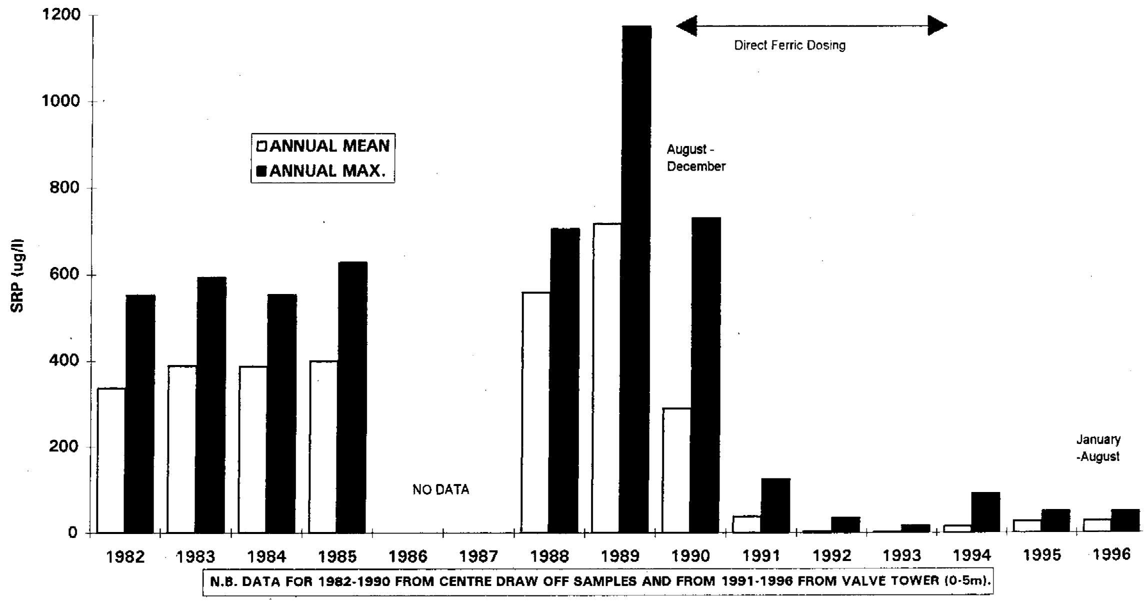

Figure 14. Annual mean and maximum soluble reactive phosphorus, Covenham Reservoir, 1982–96.

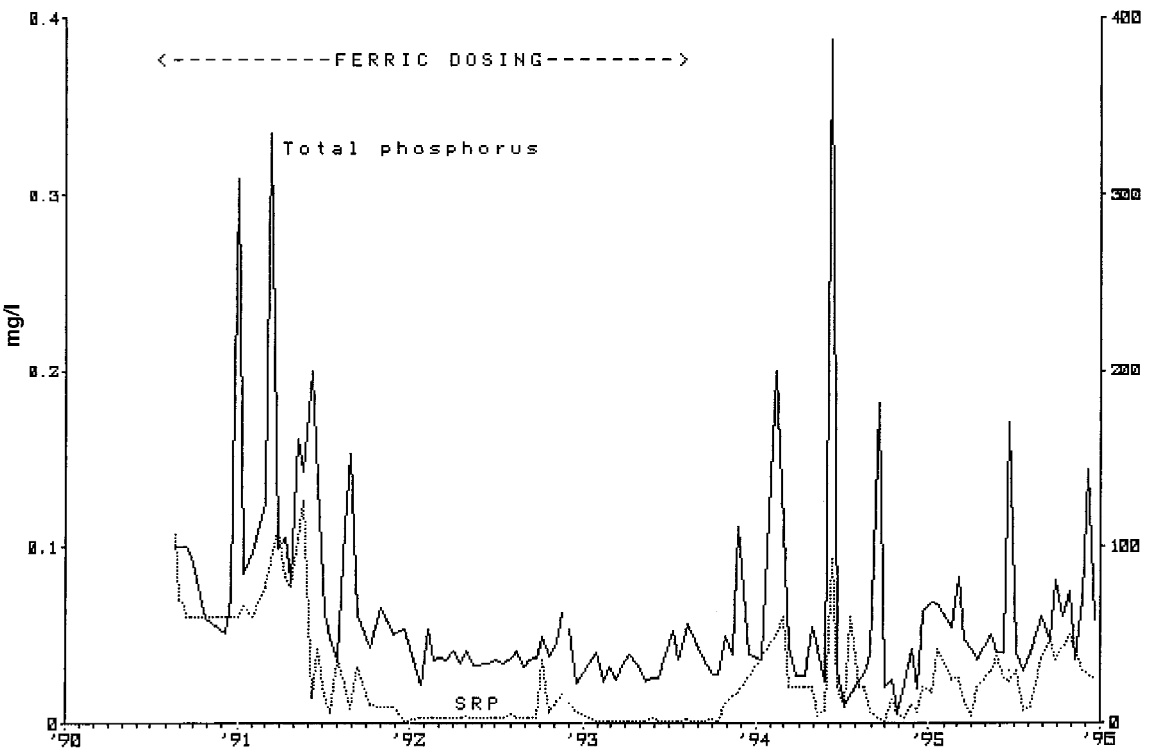

Figure 15. Phosphorus concentrations in relation to ferric sulphate dosing, Covenham Reservoir, 1990–95.

throughout most of the summer. *D. magna* reappeared during August along with *D. galeata*. The difference seen in the zooplankton communities during these years would tend to indicate top-down control by fish

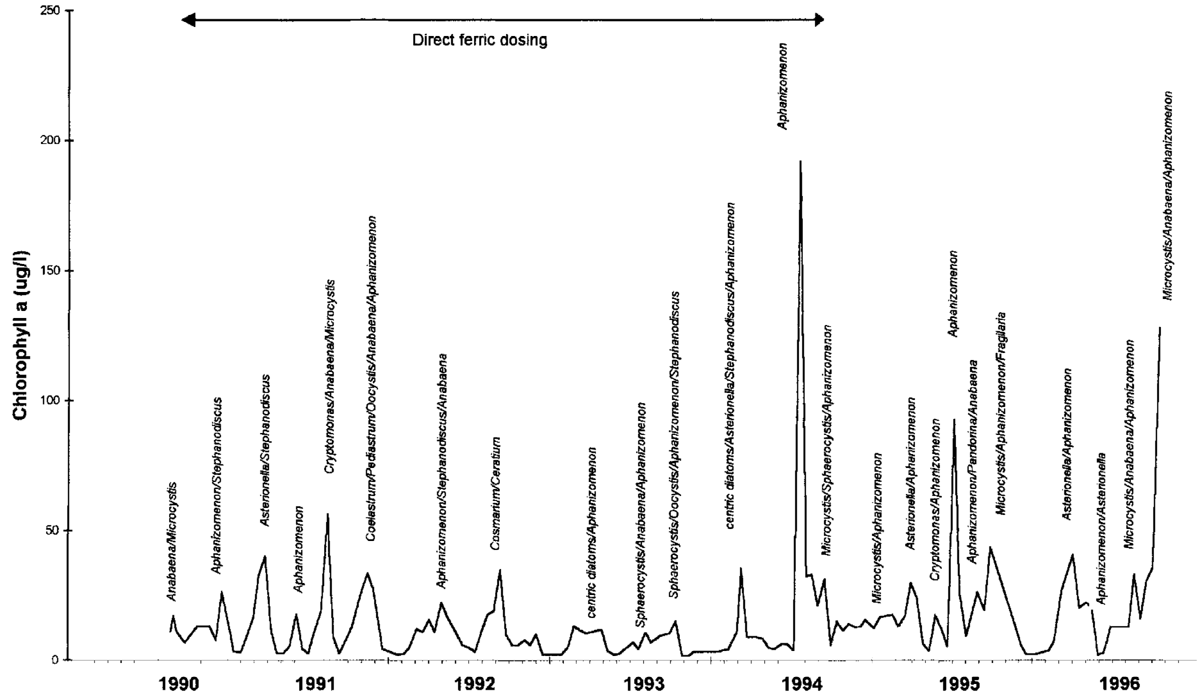

Figure 16. Temporal chlorophyll fluctuations at Covenham reservoir showing the dominant taxa, 1990–96.

predation, possibly with higher recruitment in 1992 of planktivorous fish and fry. The presence of large bodied *Daphnia*, as recorded from Covenham reservoir, would also indicate low fish predation pressure upon the zooplankton. The fish survey, carried out in 1995, showed an extremely low fish biomass of only 4.9 kg ha^{-1}. This is lower than any other biomass recorded for this type of reservoir (Duncan, 1995). The low biomass of fish present, again similar to the London reservoirs, is due to the lack of spawning sites (Duncan, 1990). However, it is unclear as to the reason for the changes in the dominance of the *Daphnia* during this period and it is not certain whether it is a result of top-down control, bottom-up control by mixing and reductions in phosphorus due to ferric dosing, or to a combination of the three.

It is possible that this reservoir is ecologically unstable, with decreasing phosphorus concentrations leading to changes in the phytoplankton which, together with the low fish grazing pressure has prevented a stable annual cycle of zooplankton appearing. Analysis of the zooplankton, phytoplankton and chemical/physical limnology led Harper et al. (1997) in a review of the feasibility of biomanipulation in reservoirs and deeper lakes, to conclude that Covenham should be further studied to ascertain the reason for annual fluctuations, and then whether any biomanipulation is neeeded. In contrast, the natural valley shape of Rutland Water means that cyprinid reproduction will be successful and that any biomanipulation will require tight control of spawning success. Since this cannot be carried out by water level management, due to its pumped storage nature and manipulation of the total fish stocks is impossible because of the economically and recreationally important trout fishery, the use and removal of artificial spawning sites has been recommended on a large-scale trial basis.

Conclusions

These three examples show that, although a variety of methods exist for assisting the management of eutrophic lowland reservoirs, the principal factor of importance is its original design and the second its hydrological regime (cf. Steel & Duncan; Zalewski, this volume). The methods available for reducing algal crops in valley reservoirs with minimal control over inflow and outflows are limited and will only succeeed if their purpose and effects are understood and their

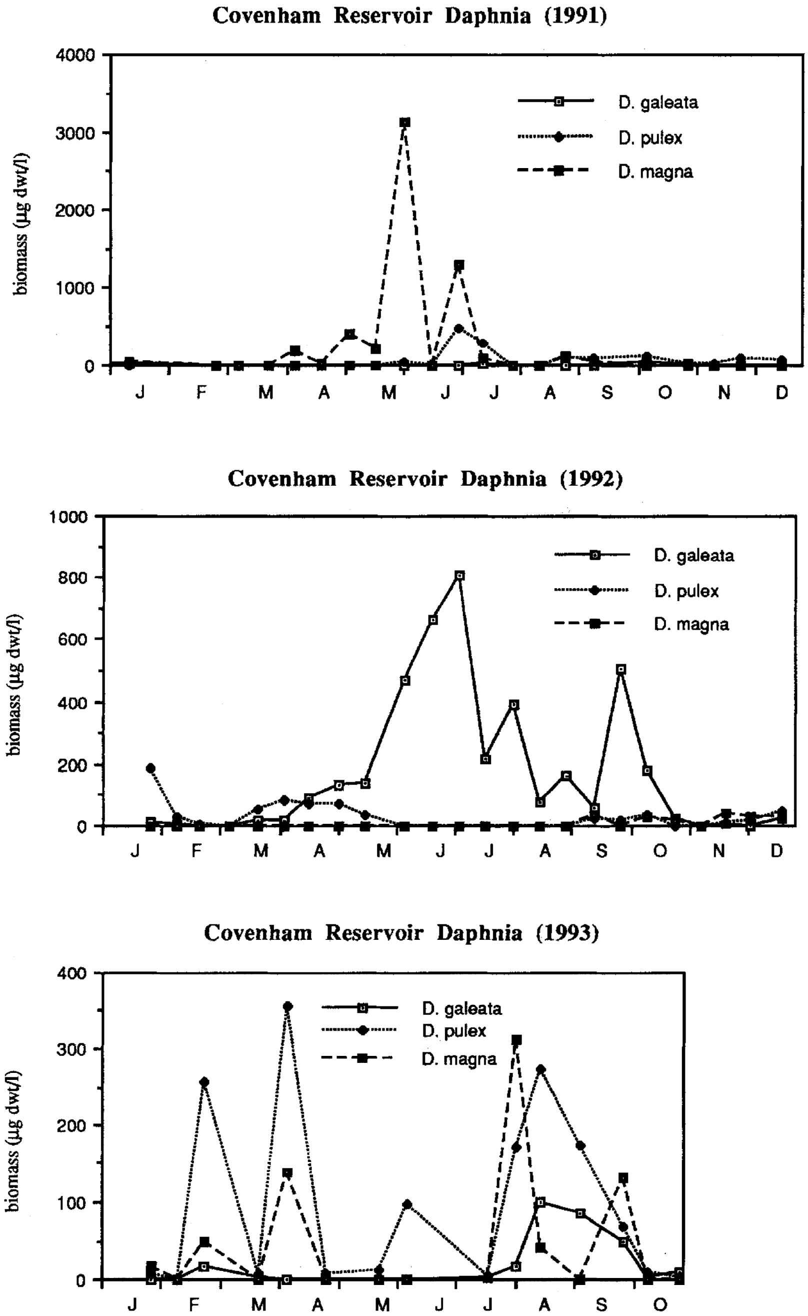

Figure 17. Development of *Daphnia* species in Covenham reservoir, 1991–93.

ecological targets clear. This is a task for the 'sustainable management' to which the Envionment Agency in England and Wales is now committed.

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