Institute of Freshwater Ecology
Edinburgh Laboratory, Bush Estate, Penicuik
Midlothian, EH26 OQB, Scotland
Telephone 031 445 4343; Fax 031 445 3943

Nutrients, phytoplankton and water clarity in Loch Leven following phosphorus loading reduction

Project Manager: A E Bailey-Watts

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Principal Investigators:

A E Bailey-Watts, BSc, PhD, MIWEM
L May, BSc, PhD
A Kirika

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Contents

Abstract

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1. The fundamental importance of eutrophication in the functioning of aquatic ecosystems is highlighted and a distinction is drawn between the causal factors and processes of nutrient enrichment *per se*, and eutrophication *sensu lato* i.e. the biological manifestations of the enhanced inputs of nutrients - primarily the accumulation of biomass of plants of different types. The results from Loch Leven following phosphorus (P) loading reduction, show that the influence of the weather on changes in physical, chemical and biotic factors, cannot be over-emphasised.

2. The history of Loch Leven studies is reviewed with special reference to long-term shifts in P loading, and the likely importance of these and of changes in the abundance of herbivorous zooplankton, on the population densities and species composition of the phytoplankton. In this connection, factors such as nutrients which determine how many plant cells are produced, are distinguished from other components such as grazers which influence how many cells are observed.

3. The rationale of controlling the supplies of P (as opposed to those of other nutrients) to Loch Leven, and more specifically, the scientific basis for targetting the inputs of industrial P-rich waste, is outlined, mainly with reference to the 1987 report prepared for SDD and others. Thus, the model-based predictions of the effects of the control are summarised in relation to the findings of the work done in 1985, when a total of 20.5 t P (including 8.1 t in runoff, 6.3 t in the industrial effluent and 5.3 t in treated sewage effluent) entered the loch, and mean in-loch concentrations of 62 μg TP l^{-1} and 21 μg chlorophyll *a* l^{-1} were recorded. The analysis suggested that by eradicating the industrial contribution, the annual P and chlorophyll levels would be reduced to *ca* 57 $\mu g\ l^{-1}$ and 17 $\mu g\ l^{-1}$ respectively, assuming a flushing of 1.88 loch volumes y^{-1} (the long-term average); the calculations also predicted that P and algal levels would fall to the lower values within *ca* 0.5 to 1.6 years of the P loading being reduced to the extent envisaged.

The P reduction was amply achieved, such that by late 1989, no effluent was being discharged by the industry previously involved. Thus, during 1990, *ca* 2.0 t P was transported to the loch by the stream formerly receiving the mill waste. This figure is in keeping with conventional agricultural runoff P losses and, in being higher than the figure of 1.7 t measured in 1985, reflects the fact that 1990 was a somewhat wetter year than 1985. To this extent, the reduction in P burden to the loch brought about by diversion of mill effluent, was partly offset by the heavier rainfall in 1990; indeed, as 6.3 t was 'diverted', and an extra 1.6 t in total runoff and direct rain was introduced to the loch, the net reduction is only *ca* 4.6 t ie 22% of the 1985 total of 20.5 t. Nevertheless, a reduction was achieved and in addition to establishing this over the period 1988 to 1989, the purpose of the present study was to assess its early effects (if any) on the P and phytoplankton content of the loch. Continuing 8-daily measurements in 1990 show, however, considerably elevated levels of TP and pigment, ie 80 $\mu g\ l^{-1}$ and 50 $\mu g\ l^{-1}$ respectively. Initially, the findings were considered to be all the more puzzling, bearing in mind that 1990 was a very wet year - a factor likely to suppress algal growth, if not the inputs of P; as a consequence flushing was high, with a value of 2.94 loch volumes, compared to the 1985 figure of 2.57, which, at the time was judged to be fairly exceptional. Much of the report examines the apparent anomaly, and concludes that mean annual figures for e.g. rainfall, flushing and P and phytoplankton concentrations mislead; only when seasonal fluctuations in e.g. precipitation and algal abundance are considered, do the reasons for elevated P and phytoplankton become clearer. The striking contrasts between conditions prevailing in 1985, and those observed in 1990, are described to emphasise the importance of temporal variability.

The main issues are as follows:

- while rainfall (and thus flushing and runoff P loadings) were the higher in 1990, the difference from 1985 is not great. However some 60% of the annual throughput of

water occurred in the first 3-3½ months of 1990, while an additional proportion of only 5-10% was delivered between April and September and the remainder of the year was reasonably dry; contrastingly *ca* 30% of the 1985 rain fell in July and August and the autumn was also wet.

- the coincidence of warm weather and the summer period of low water enewal in 1990, enhanced denitrification which resulted in very low nitrate concentrations, and in turn, led to a rapid and marked release of soluble reactive P from the sediments, to a peak concentration of 78 μg SRP l^{-1} in August; the increase over the very moderate background levels of 6 μg l^{-1} represents an internal loading of *ca* 3.7 t to the water column, an amount equivalent to *ca* 60% of that estimated to have entered the loch from the mill in 1985 and now diverted.

- on the return of somewhat cooler weather and reduced de-nitrification, much of the P released was readsorbed by the sediments, but a proportion was sequestered by phytoplankton which were able to accumulate biomass on account of the relatively low flushing rates prevailing over the remainder of the year.

Even in the absence of a weather regime as unusual as 1990 when conditions appeared to 'conspire' to negate effects of the present phase of P loading reduction, it is likely that more time would be needed to assess these effects. In this connection, we are fortunate to have attracted funding from the Nature Conservancy Council (Scotland), to maintain the limnological surveillance. It is recommended that this be continued until the end of 1993; by then a more informed judgement can be passed on whether further P reduction measures such as tertiary treatment of effluent from the major sewage works, are necessary or appropriate.

1. INTRODUCTION AND BACKGROUND TO THE STUDY

This report discusses the consequences of a recommendation (made in 1987) to stem the input of phosphorus-rich effluent to Loch Leven, in an attempt to reverse eutrophication trends. Changes in the levels of phosphorus (P) in the feeder stream concerned, and in P and phytoplankton in the loch itself, are examined to assess whether the lake has responded to the P-reduction measure. Factors such as weather conditions influencing the response are also considered.

Loch Leven has been the subject of, and formed a main base for scientific investigation into various aspects of pure and applied freshwater ecology over the last 25 years (Royal Society of Edinburgh, 1974; Bailey-Watts, Kirika, May and Jones, 1990). It was chosen as a research site in the late 1960's in recognition of its conservation status; it is a National Nature Reserve, primarily on the basis of its abundant and diverse waterfowl populations. Also, at that time, much of our understanding of the functioning of standing waters rested on studies of deep, stratifying lakes. The balance has been redressed to some extent in the intervening years, with major contributions on the dynamics of plankton populations in shallow systems, and not least, on changes in the species composition and abundance of phytoplankton in relation to nutrient enrichment. By their very nature, shallow waterbodies are likely to respond quicker than their deeper counterparts, to a given burden of extra nutrients. In this regard, eutrophication can be viewed as a most important factor in lake ecology; it provides chemical energy that fuels the production of plant matter which forms the bases of food chains leading to fish and other aquatic vertebrates.

It is important to distinguish between the phenomena and processes controlling nutrient inputs, ie eutrophication *per se*, and the chemical and biological manifestations of accelerated enrichment, ie eutrophication *sensu lato* (Bailey-Watts 1991). Only a very small percentage of the many thousands of publications and reports on 'eutrophication' deal with actual inputs. But the well-documented studies on the Norfolk Broads (Moss, 1983) and Lough Neagh (Gibson, Smith and Stewart, 1988) in particular, feature strongly on the world scene (Sas, 1989). Loch Leven results are now contributing to this (Bailey-Watts and Kirika, 1987) although some major findings still exist only in report form (Bailey-Watts, Sargent, Kirika and Smith, 1987).

Dense algal blooms resulting from enhanced nutrient enrichment, have been recorded at Loch Leven since at least the late 1940's (Rosenberg, 1938). Associated reductions in the species diversity and overall depth distribution of rooted vegetation have also given cause for concern (Jupp and Spence, 1974). Changes in the structure of, and balance between these two plant

communities affects organisms at other trophic levels. The inexorable links between these biological components and the chemical and physical environment are important features of lake systems. They must be borne in mind when assessing the affects of eutrophication, and indeed, when formulating strategies for controlling, reversing or preventing the process (Moss and Leah, 1982; Moss, 1983). This philosophy is reflected in the present report, by considering the influence of e.g. the weather on the dynamics of phosphorus (P) and phytoplankton.

From the considerations in 1985 about the eutrophication models of Dillon and Rigler (1974) it is plain that flushing rate, which is controlled by rainfall, has a large influence on P loadings and the response of the receiving waterbody to these burdens. While that publication indicated that this has been realised for nearly 2 decades, it is only the data analyses carried out by Bailey-Watts, Kirika, May and Jones (1990) as an extension to the 1985 Leven study, that emphasise the much broader influence of P, and of associated weather factors on lake dynamics. For this reason, the present report pays more attention than hitherto on weather conditions such as air temperatures.

There was considerable concern over the eutrophication of Loch Leven in the late 1960s and early 1970s. Chlorophyll concentrations commonly exceeded 250 $\mu g\ l^{-1}$ (Bailey-Watts, 1974, 1978) equivalent to the theoretical maximum of *ca* 400 mg m^{-2} of the euphotic zone (Bindloss, 1974, 1976). As a result and on the basis of a crude estimate of the P loading (Holden and Caines, 1974), discharges of P-rich industrial effluent were decreased. While a reduction in overall phytoplankton abundance was observed, it is difficult to attribute this wholly to assumed reduction in P inputs; the abundance of herbivorous zooplankton increased dramatically in 1971 (Bailey-Watts, 1978). In any event, algal biomass increased again in the early 1980's, and classic bloom-forming, blue-green algae (cyanobacteria), became more prominent once more. This led to a desk study which indicated that P burdens to the loch had also increased (Bailey-Watts, 1983), and that this was primarily due to the industrial contribution; the mill effluent involved was very rich in total soluble P (TSP) which is readily hydrolysed to soluble reactive P (SRP) - the fraction most immediately available to phytoplankton.

Field observations and laboratory experiments suggested that the availability of P now determines the biomass of phytoplankton, for most of the time in Loch Leven (Bailey-Watts, 1988, 1990). During the earlier years of the study, P supplies tended to exceed the requirements of even the densest crops, and light, through the phenomenon of 'self-shading' (Talling 1960) was the ultimate limiting factor (Bindloss, 1976). Also, but usually in summer

only, shortage of nitrogen rather than phosphate is important. Logically therefore attempts to suppress algal growth - or more correctly, prevent such marked accumulations of biomass in the first place - needed to concentrate on P.

Before any decisions could be made on what P reduction options were available and were the most appropriate, a full re-assessment of the loading, and an investigation of the contributory sources was necessary. As a result, the Department of Agriculture and Fisheries for Scotland, the Natural Conservancy Council, the Scottish Development Department, and Tayside Regional Council funded a two-year programme (1985-86). This assessed the loadings attributable to the industrial source, 4 sewage treatment works (STWs), agricultural runoff from the catchment of 145 km^2, over-wintering geese, and rain falling on the loch surface (13.3 km^2). Twenty-seven sampling points were visited at 8-daily intervals. This was backed up by Forth River Purification Board records of stream flows, and Meteorological Office rainfall data. The programme estimated that approximately 20 t P entered Loch Leven in 1985. The majority (8.1 t) of the total load entered in runoff; a further 6.3 t emanated from the industrial source, while 5.3 t originated at STWs. The contributions from direct rainfall and from geese were minimal by comparison.

A broad strategy for P reduction proposed that the main point-sources be targeted - first, at least. Runoff is likely to be an important factor in the eutrophication of Loch Leven. However, only in years as extraordinarily wet as the records showed 1985 to be (but see below), would such high loads from agricultural land be expected. Loadings of industrial and sewage P also vary from year to year - due to shifts in manufacturing schedules and sizes of sewer-served populations - but the variability is likely to be less than the 3- to 4-fold differences characteristic of annual, rain-dependent runoff. In any event, too, agricultural loads would be more difficult to control because of their diffuse nature; the other sources tend to enter the system very largely in well-defined pipes. This does not mean, however, that no thought should be given to controlling diffuse sources of nutrients. Firstly, eutrophic sites exist where agricultural runoff appears to be by far the major source of nutrients (Bailey-Watts and Kirika, 1991), and secondly, many lakes have been observed not to recover as rapidly as originally predicted despite eradication or diversion of point-sources (see Section 5). This is commonly because nutrients can be recycled from material that has accumulated in the bottom sediments over the decades of enrichment. Then, consideration has to be given to stemming 'internal' sediment release and/or 'external' runoff: in large lakes, the latter is likely to be the more feasible option.

At Loch Leven it was considered likely from the outset, that the mill effluent would be a

prime target for control - as it comprised a single outfall. We have also now established that its contribution exceeds that of the 2 main STWs combined. However, thought had to be given to the likely affects of the whole range of feasible action, and the extent to, and time over which the loch would respond. To assess the likely effects and responses, the results of the loading study were examined to establish whether the estimates of loadings, the amount of P passing down the outflow, and aspects of the water balance related to each other according to 2 predictive models. A major finding was that the Leven data fitted these equations very closely. The first model examined is that of Dillon and Rigler (1974) which predicts the annual mean, in-lake total P concentration ($[TP]_l$) from:

$$[TP]_1 = \frac{L(1-R)}{p.z}$$

where L is the areal loading (mg P m^{-2}), R is the P retention coefficient ie the difference between the amount of P entering the lake and that leaving it, expressed as a fraction of the load, p is the flushing rate (loch volumes y^{-1}), and z is the mean depth (m). The 1985 estimates are: 1540 for L, 0.60 for R, 2.57 for p and 3.9 for z. These predict a mean in-lake concentration of 61.5 mg m^{-3}, compared to the observed value of 62.7 mg m^{-3}. The other equation referred to, is the empirical model of Kirchner and Dillon (1975) which aims to predict R from water balance data alone. It uses a factor, q_s - the 'areal' water loading in metres - which is the total volume of water entering the loch (in one year), divided by the surface area of the lake; the volume coming into Loch Leven in 1985 was 134.7 x 10^6 m^3 ie 2.57 times the loch volume of 52.4 x 10^6 m^3), and the area is 13.34 x 10^6 m^2. q_s is thus 10.11 m. R is related to q_s according to

$$R = 0.426e^{(-0.271q_s)} + 0.547e^{(-0.00949q_s)}$$

Where q_s is 10.11, R is 0.58, ie very close to the value of 0.60 measured.

As the data fit so closely to the models, the Dillon and Rigler equation was used again, to predict $[TP]_l$ values commensurate with the loadings resulting from a range of possible P reduction strategies. Information on variation in $[TP]_l$ due to inter-annual fluctuation in p was also taken into account - this being facilitated by the some 150 years' records of the discharge of water from Loch Leven (Ledger and Sargent, 1984). This part of the predictive exercise assumed, however, that the frequency distribution of future annual flushing rates, is the same as that exhibited for past values. Using the data for 1936-1985, an essentially normal distribution was found with a mean rate of 1.89 loch volumes y^{-1}, and 95% limits of 1.25 and 2.45 loch volumes y^{-1}. The range of possible P reduction situations considered in the earlier

report (Bailey-Watts, Sargent, Kirika and Smith, 1987), included (i) removal of 80% of the P from the industrial effluent, and (ii) 80% from the waste waters issuing from the 2 major STWs discharging to the Leven system. These more-or-less correspond to the 2 phases of the P-reduction strategy eventually formulated on the basis of this work. The main difference is that effectively all, not just 80% of the P in mill effluent has been removed. If the industrial waste water alone was targeted, annual mean concentrations of TP in the loch would be expected to fall to between 55 (corresponding to the 95% *p* value) and 61 (5%) $\mu g\ l^{-1}$, and to 57 $\mu g\ l^{-1}$ if the long-term mean flushing rate of 1.88 loch volumes y^{-1} held. Equivalent values, assuming the sewage effluents from the two Kinross STWs alone were treated, are 62, 72 and 65.

In the event, the programme formulated, recommended that attention be given to the industrial effluent first and, depending on its effects, consider the STWs second. The execution and monitoring of the effects of controlling the industrial effluent, are the subjects of the present report.

Another important consideration is the likely time over which it would be appropriate to monitor the situation, after P reduction, in order to assess its effects. The OECD (1982) model was used to do this, as it predicts the 95% response time, ie the time by which P levels in a waterbody are likely to fall to within 5% of the concentrations predicted for the new equilibria corresponding to the particular reduction strategy adopted. The model predicts that the 95% response time will approximate to $3/p$; so, at the long-term average p value of 1.88 loch volumes per year, the period is ie 1.6 y. The same model suggests a correction be made for the P retention coefficient R; then, the 95% response time is $(1-R).3/p$. Taking the 1985 R value of 0.6, the predicted response time was 0.64 y. However, as suggested by Marsden (1989) referring to the work of Sonzogni, Uttormark and Lee (1976), the use of p to predict the response time is strictly only appropriate where conservative substances are involved - but P behaves as a non-conservative element. Then, the rate of change is more accurately estimated by reference to the P residence time; this, assuming steady state conditions, is calculated by dividing the mean annual content of a lake (ie. mean concentration x lake volume) by the annual loading. Taking the 1985 mean P concentration of 63 $\mu g\ l^{-1}$, the loch volume as $52.4 \times 10^6\ m^3$, and the loading as 20.5 t, the P residence time is 0.15 y. As the 95% response time is equivalent to 3 P residence times, this estimate of the period during which 95% of the expected change is likely to take place, is 0.48 y., i.e. less than one-third of the value predicted on the basis of water residence time.

2. INVESTIGATIVE METHODS

All sampling and laboratory analytical methods relating to stream P and algae, and to rainfall and water balances, are as described in the earlier report on Loch Leven P loadings (Bailey-Watts, Sargent, Kirika and Smith, 1987). However, because the only new information required on loadings concerned the industrial input on the South Queich, the number of sites sampled was much smaller than that assessed in 1985. Indeed this was the only inflow sampled and a maximum of only 3 sites were visted: above, at and below the point of entry of the effluent; since late 1989, when the pipe ceased to flow, only the points above and below this were sampled. Where loadings of P from the other sources - ie STWs, agricultural land (runoff), direct rain and wildfowl - were needed, these were calculated on the basis of the 1985 data corrected where appropriate for any differences in, for example, rainfall (which should affect runoff), and the sizes of sewer-served populations (affecting inputs from the STWs). As in 1985, the loch was sampled at one point near its outflow in the south-eastern corner, and at another site off the public pier on the north shore of the west bay. As in 1985 too, an 8-day schedule was operated.

Water clarity and sediments - aspects which were not covered in 1985 - were investigated during the present study. To maintain a check on clarity in open water, Secchi disc transparencies were occasionally measured. Surveys of the P content of the sediments were undertaken, in order to assess the P release potential of these deposits. Cores were collected with a Jenkin Surface Mud Sampler. Subsamples were taken of water immediately above the mud and of interstitial water and the deposits themselves at 1-cm intervals down to *ca* 10 cm into the cores. They were analysed for a range of dissolved and particulate fractions of P. As the samples were not kept under inert, or nitrogen gas during handling, values obtained for interstitial phosphate ions are likely to under-estimate the actual levels.

Meteorological Office rainfall records have been used to estimate flushing rate and runoff loadings assuming that the linear relationship found between runoff discharge and P loading in 1985, still holds. Unfortunately, Forth River Purification Board data on outflow discharge are not available at the time of writing; that alternative means of calculating the flushing rate is thus not possible. Another calculation, based on South Queich flow information is possible, however; Smith (1974) found that water runoff from this sub-catchment represents *ca* 25% of the total streamflow into Loch Leven. For reasons already outlined, Meteorological Office data on air temperature have also been examined.

3. RESULTS

3.1 Phosphorus reduction

3.1.1 *Decreases in the South Queich*

The volume and phosphorus content of the mill effluent were reduced following discussions, started in 1987, with the factory Director. The change was gradual at first, but in late 1989 the process producing the P-rich effluent was discontinued. Since then, nutrient-laden sludge has been tankered away from the Leven catchment. As a consequence, P levels in the lower South Queich have decreased such that they are now virtually identical to those measured at a site some 100 m upstream of the original point of outfall. **Figure 1** illustrates the results for total soluble P - this being the major component of the total P (TP) content of the effluent. Of the gaps in the record shown, those representing actual missing values are for February on, in 1986, and the whole of 1987 (when no work was done on the loch), and for parts of December 1988, January 1989 and 1990, and of July 1990. The other gaps relate to dates on which the concentrations exceeded, by a considerable margin, the upper limit of the range of values plotted, i.e. 120 $\mu g\ l^{-1}$; an axis accommodating these would result in none of the patterns illustrated being discernable. The maximum in 1985 was 10 $mg\ l^{-1}$, while another two peaks came between 1 and 2 $mg\ l^{-1}$, and the remainder were between 0.2 and 0.4 $mg\ l^{-1}$. Currently therefore, P levels right down to the mouth of this stream are characteristic of an agricultural catchment. **Table 1** summarises the year-to-year shifts in the concentrations of TP, TSP (ie TP minus the particulate fraction - PP), and SRP.

Table 1. Concentrations of total phosphorus (TP), the total soluble fraction (TSP) and the soluble reactive form (SRP) in the South Queich above and below the point of entry of mill effluent, and of the effluent itself - 1985 figures are compared to those for 1988, 1989 and 1990.

Sampling Site Phosphorus fraction		**stream above mill** ($\mu g\ P\ l^{-1}$)			**the effluent** ($mg\ P\ l^{-1}$)			**stream below mill** ($\mu g\ P\ l^{-1}$)		
		TSP	**TSP**	**SRP**	**TP**	**TSP**	**SRP**	**TP**	**TSP**	**SRP**
Year: **1985** (n=47)	**min**	**8**	**1**	**1**	**0.04**	**0.03**	**0.02**	26	18	19
	mean	31	15	11	14.6	9.8	0.9	538	475	55
	max	136	50	40	>100	89.2	6.4	10267	9837	782
1988 (n=38)	**min**	10	6	3	0.3	0.08	0.02	15	7	2
	mean	**39**	**21**	**14**	**0.9**	**0.7**	**0.2**	**57**	**35**	**17**
	max	376	112	75	8.5	7.7	0.6	408	35	73
1989 (n=44)	**min**	11	6	2	0.1	0.04	0.02	12	7	3
	mean	**30**	**18**	**13**	**0.6**	**0.4**	**0.2**	**208**	**22**	**14**
	max	185	92	69	1.1	1.0	0.6	7220	99	74
					(n = 20)					
1990 (n=46)	**min**	11	5	1	-	-	-	11	5	1
	mean	**36**	**20**	**13**	-	-	-	**40**	**20**	**12**
	max	183	86	55	-	-	-	204	88	59

Figure 2 compares the changes in water discharge for 1985, 1989 and 1990. The values correspond to the days of chemical sampling, and each is a mean figure derived from 48 ½-hourly points on the stream level recording chart. The total range of values is similar in each of the years with recorded minima varying from 89 to 194 l s^{-1} and maxima from 4.2 to 4.6 $m^3 s^{-1}$. The annual means differ considerably, however, largely due to the contrast between the very dry 1989 - 734 l s^{-1} - and the very wet 1990 - 1023 l s^{-1}. Indeed, the 1990 figure is greater than that recorded in 1985 (913 l s^{-1}) which at the time was considered to be especially wet. Comments by Bailey-Watts, Sargent, Kirika and Smith (1987), on the low chance of occurrence (< 5%) of a higher flushing rate than that calculated for 1985 reflect this. As established in the case of the 1985 data, however, the present results make sense when considered alongside information on rainfall and the size of the stream catchments. Firstly, the contrasting patterns of rainfall in 1985 and 1990 each mirror those of flow in the South Queich (**Figure 3**). Also, in 1990, the estimated total, evaporation-corrected volume of water runningoff the whole Leven catchment - which is 145 km^2 equivalent to 4.3 times that of the South Queich drainage area - is 133.2 x 10^6 m^3, which is 3.8 times the 34.7 x 10^6 m^3 derived from the mean of all the continuousdischarge values recorded on the South Queich (1099 l s^{-1}). The mean discharge of this feeder stream was 1.27 times that obtained for the North Queich. The ratio of their drainage areas is 1.29:1 in favour of the South Queich. What Figures 2 and 3 illustrate so well is the sharp contrast in the seasonality of water inputs - even between the two wet years; flow values over much of the first quarter differ by almost an order-of-magnitude - with 1990 being the wetter - while differences of approximately 10-fold are observed also for most of the third quarter of the year - but this time 1990 is by far the drier. Such contrasts are extremely important when it comes to discussing responses to P reduction, and interpreting the observed changes in nutrientsand phytoplankton in the loch (see Section 4).

The product of the TP concentrations (C_i) in the South Queich and the mean instantaneousflow for each of the chemical sampling days (Q_i) in 1990, are compared to the loading data for 1985 and 1989 in **Figure 4**. There are two important points to stress. The first is that the amount of P transported by this stream in 1990 is considerably less than that carried to the loch in 1985. Indeed, the mean of the 44 products of C_i and Q_i is equivalent to 2004 kg y^{-1}, which is 20% larger than the 1669 kg measured in the somewhat drier 1985, and attributable to the portion of the inflow upstream of where the industrial effluent entered. The second consequence of the cutback concerns loading seasonality. In 1985, the industrial, rainfall-independent influx of P dominated the loading picture, and the patterns of loading differed from those of flow. In contrast, now that runoff-related P loadings dominate, their changes tend to parallel the shifts in flow more closely.

3.1.2 *Effects on the total input of phosphorus*

As found with the 1985 data on each of the major inflows, in 1990 the annual mean discharge of the South Queich, based on the mean flows on the days of chemical sampling, is within 5% of that based on the flow records for all 365 days. Since P loadings attributable to runoff depend on the flow regime, the estimate discussed in Section 3.1.1 is likely to be reasonably accurate. For present purposes therefore, it is sufficient to accept that P entering *via* runoff to Loch Leven in 1990 is *ca* 20% greater than the total of 8.1 t measured in 1985 (see Section 1) i.e. 9.7 t. Assuming that inputs in direct rainfall in 1990 exceed those found in 1985 (0.42 t) by a similar percentage, but those from STWs (5.3 t) and *via* geese ($\leq$ 0.37 t) have remained largely unchanged, the total burden in 1990 is likely to have been 15.9 t. This represents an overall reduction of 4.6 t from the 1985 total of 20.5 t. In other words, while the 6.3 t formerly entering in mill effluent has been eliminated as intended, the reduction has been partly offset by enhanced runoff. On the basis of the situation in 1990, therefore, the reduction in P burden represents only 22% of the 1985 value.

3.2 **Phosphorus fluctuations in the loch**

While, due to extraordinarily high rain-runoff in 1990, the total loading of P was not as low as reasonably predicted, it did represent an improvement on 1985 figures. However, the levels of TP and PP in the loch in 1989 and 1990 were considerably higher than in 1985 (**Figure 5, Table 2**). Contrastingly, the mean annual SRP concentrations were only marginally higher than those measured in 1985. Indeed, excluding the marked peaks which were observed during August in each of the last 2 years (**Figure 6**), and which were undoubtedly due to release of phosphate ions from the sediments (see also Section 3.3) mean SRP levels would have been nearer 5 $\mu g\ l^{-1}$.

These pulses represent 'unused' P, and the decreases from the maxima are probably due as much to re-adsorption of the ions onto the sediment, as to algal growth. Nevertheless high levels of PP were maintained over much of the last 4 months of both years; these may have been initiated by the elevated levels of bio-available SRP in August.

3.3 **In-stream and loch phosphorus levels compared: retention of P by the loch and release of the nutrient from the sediments**

The ratio of the concentrations of P in the water entering the loch to those in the loch itself indicates (i) the degree to which the nutrient-supply is utilised within the loch, (ii) the net

amount retained (by incorporation into the sediments, for example) and (iii) the importance of the release of P from the sediments.

As described in Section 1 the situation in 1985 suggested that the amount of P leaving the loch *via the outflow* was equivalent to 42% of the external loading. Thus, the measured P retention coefficient, R, was 0.58, ie virtually the same as the value of 0.60 predicted by the equation of Kirchner and Dillon (1975) from hydraulic considerations alone; this is due not only to the high quality of the hydrological and chemical data collected, but also because conditions in 1990 were not conducive to massive releases of P from the sediments - which would result in a lower retention coefficient. Since the annual input of water in 1990 was *ca* 12% higher than that recorded in 1985, the areal water loading is 11.32 m y^{-1} *cf* 10.11 m y^{-1} in 1985. The value for R for 1990, predicted by the equation of Kirchner and Dillon (see p. 4) is thus slightly lower than 0.60 ie 0.54. Although 1990 data on the discharge of water at the outflow

Table 2. Concentrations of phosphorus in Loch Leven, 1985 and 1988 to 1990. L is the site near the outflow at the eastern end of the loch, and PP is located at the northern end of the west bay. TP is total phosphorus, and TSP and SRP are the total soluble and soluble reactive fractions, respectively.

Year	Site		TP	TSP	SRP
1985 (n = 47)	L	min mean max	29 60 134	8 20 45	1 9 38
1988 (n = 29)	L	min mean max	15 61 114	11 21 37	1 10 25
1989 (n = 44)	L	min mean max	33 84 278	9 25 78	1 12 56
1989 (n = 39)	PP	min mean max	40 86 210	10 29 87	1 15 65
1990 (n = 46)	L	min mean max	27 81 132	9 24 96	1 11 79
1990 (n = 46	PP	min mean max	27 77 165	9 27 135	1 13 118

of Loch Leven are not yet available, it is reasonable to assume that the volume leaving the loch over the year equals the estimated input. *R* can then be estimated from:

$$[TP]_{in}/[TP]_{outflow}$$

where $[TP]_{in}$ is the flow-weighted mean concentration of P entering the loch, and $[TP]_{out}$ isthe equivalent value for the outflowing water. By definition, $[TP]_{out}$ is the same as the annual mean TP level observed in the loch itself. As with the determination above of q_s and predicted *R* for 1990, the determination of *R* from the observed P concentrations in 1990 is best illustrated by comparison with the calculations using 1985 data. Bailey-Watts, Sargent, Kirika and Smith (1987) found that $[TP]_{in}$ was 2.4 times the mean, in-loch concentration of TP (and, thus $[TP]_{out}$) ie 151 $\mu g\ l^{-1}$ *cf* 63 $\mu g\ l^{-1}$. The 151 $\mu g\ l^{-1}$ is derived from the total loading of 20.5 t TP divided by the total volume of water entering the loch - 135 x 10^6 m^3. As the concentrations incorporated here are both 'flow-weighted' their ratio (2.4:1) is also the ratio of the amount of P entering the loch to that leaving it. *R* is then (2.4-1)/2.4 ie 0.58 as discussed above. $[TP]_{in}$ for 1990 is the new loading of 15.9 t TP divided by a total water volume of *ca* 12% greater than the 135 x 10^6 m^3 measured in 1985, ie 151.2 x 10^6 m^3; the new $[TP]_{in}$ is thus 105 $\mu g\ l^{-1}$. $[TP]_{out}$ is 81 $\mu g\ l^{-1}$ (see Table 2). The ratio of flow-corrected concentrations is thus 105:81 or 1.3:1. From these figures *R* is (1.3-1)/1.3 = 0.23 which is - in sharp contrast to the situation found in 1985 - quite different from the value predicted from the hydrological balance (*ca* 0.54). Since the observed value is considerably smaller than that predicted, the loch would appear to have exported much more P than expected.

Assuming that the true retention coefficient is 0.54, the discrepancy is equivalent to 31% ([0.54-0.23] x 100) of the external loading, ie 5.3 t P. The following calculations show that this could be largely accounted for by the sharp increase in SRP observed in August, and attributed to release of phosphate from the sediments. Firstly, instantaneous admixture of 5.3 t P into Loch Leven (volume 52.4 x 10^6 m^3) would raise the average loch-wide SRP concentration by 101 $\mu g\ l^{-1}$ - assuming no concurrent sequestration by eg algae. The peak measured in 1990 is 78 $\mu g\ l^{-1}$, some 72 $\mu g\ l^{-1}$ of which is likely to constitute the actual increase - there being a background level of *ca* 6 $\mu g\ l^{-1}$ present before the concentrations changed. Already therefore, > 70% of the likely maximum extra influx of P can be accounted for by the changes taking place during August. Secondly, assuming that (i) sediment release is the causal factor, and (ii) fluxes from the deposits were limited to the 16-day period over which net increases in SRP were recorded, the calculated release rates come well within the ranges of values published for other eutrophic waterbodies. An increase of 72 $\mu g\ l^{-1}$ in the water column of Loch Leven (3.9 m average depth) represents an average daily release of 4.5 $\mu g\ l^{-1}$ or 17.6 mg m^{-2} over the whole sediment surface. The potential for the sediments to release

such amounts of P has been realised from observations on mud cores kept at different temperatures in the laboratory. As an example, the rate of release at 25°C, calculated from the results shown in **Figure** 7 is 25 mg P m^{-2} d^{-1}. Certainly there is a large reservoir of P in the sediments, even though P represents only 0.1 - 0.2% of the sediment on a dry-weight basis (**Figure 8**). This is common for eutrophic muds, however, (see e.g. survey by Bostrom, Jansson and Forsberg, 1982) and in Loch Leven approximately one-half of this consists of the iron-bound, soluble P (**Figure 9**) which is likely to be rapidly released under suitable physical and chemical conditions. Indeed, while concentrations of e.g. 200 µg TSP l^{-1} in the sediment interstitial water are normally found (**Figure 10**) it can be shown that the desorption of P from the particulate matter, and its upward flux into the overlying water is the important factor - and not the standing stock of P measured in the interstitial water at any instant.

During the few weeks following the recorded maximum, SRP levels some 40 µg l^{-1} above the background values were maintained. It is unlikely that no P was released over that time - simply that fluxes from the sediment did not exceed the rates of P due to re-adsorption onto the sediments, and uptake by phytoplankton.

3.4 Phytoplankton and water clarity

The phytoplankton has not responded as predicted. Indeed, the 1990 concentrations are generally considerably higher than the values obtained in 1985, even though the flushing regimes of the two years were somewhat similar (**Figure 11**). This is perhaps not surprising, however, considering the changes already reported for P, and that the performance of aquatic plants constitute a useful indicator of shifts in nutrient supply.

The mean chlorophyll concentration measured ([chl]) - 21.1 µg l^{-1} - is close to the value predicted from $[TP_{in}]$ according to the OECD (1982) model which incorporates a correction for the water residence time (T_w in years - the reciprocal of p) as follows:

$$[\mathrm{chl}] = 0.43 \left[[\mathrm{TP}]_{in}/(1 + \sqrt{T_w}) \right]^{0.88}$$

Thus, substituting 151 µg l^{-1} for $[TP]_{in}$ and 0.39 for T_w, [chl] becomes 23.5 µg l^{-1}. Interpolation of the 1990 figures of 105 µg TP l^{-1} and 0.34 for T_w, predicts 17.2 µg l^{-1} for [chl] - a figure reflecting both the lower loading and the slightly higher flushing in 1990 compared with 1985. However, the observed concentration was 51 µg l^{-1} (**Table 3**).

Table 3. Chlorophyll values in Loch Leven, 1985 and 1988 to 1990. L is the site near the outflow at the eastern end of the loch, and PP is located at the northern edge of the west bay.

Year	Site	Chlorophyll concentration ($\mu g\ l^{-1}$) mean	minimum	maximum
1985	L	21.1	4.0	49.9
1988	L	33.3	7.9	142
1989	L	42.4	15.4	101
	PP	40.3	15.3	145
1990	L	51.4	4.9	111
	PP	48.0	7.9	131

The summary values in the Table show that the pigment levels taken at one end of the loch are similar to those taken at the other. Nevertheless, there are occasions when readings at one end are quite different from those at the other (**Figure 12**). The maximum value recorded at site L in 1990 - 111 $\mu g\ l^{-1}$ - corresponds to a value of only 30 $\mu g\ l^{-1}$ at site P, and a day when fierce winds blowing to the eastern end of the loch were stirring up shallow sediments and chlorococcalean green algae such as *Scenedesmus* species in the process. Other instances of marked differences between the two sites were occasioned by wind-blown aggregations of buoyant cyanobacteria. Back calculations using the OECD equation suggests that for this average level of chlorophyll to be achieved under the hydrological regime of 1990, a $[TP]_{in}$ figure of 350 $\mu g\ l^{-1}$ is necessary. This is equivalent to a load of > 50 t P, which is more than double the external loading plus what is considered to arise from the sediments. It would thus appear that the relationships postulated by the model do not hold where the internal loading is significant. The information reviewed by Marsden (1989) supports this view.

In spite of the generally high pigment levels prevailing over the last 2 to 3 years, Secchi disc readings rarely dipped much below 1 m, and in 1990 for example, a number of values >3 m were recorded (**Figure 13**). The main period of clear water in May 1990 corresponds to the trough in chlorophyll concentration shown in Figure 11.

4. DISCUSSION WITH SPECIAL REFERENCE TO THE TIMING OF FACTORS INDUCING SEDIMENT PHOSPHORUS RELEASE

4.1 General considerations

There is plainly an anomaly regarding our understanding of the way Loch Leven functions, and of how if responds to eutrophication trends. On the basis of the 1985 data - which fitted very closely to models relating the P loading to the mean annual in-lake concentration - it was predicted that (i) P levels would fall in response to the reduction in loading of P from the mill, (ii) that 95% of the expected decrease would occur within 0.48 and 1.60 years, depending on the method of calculation. In the event, P concentrations and chlorophyll levels actually rose. Moreover they did so in a year of high flushing - which would be expected to ameliorate the algal manifestations of enrichment. The estimated *p* for 1990 is 2.7 loch volumes, which is a higher value than any recorded over the period 1936 to 1985 (Bailey-Watts, Sargent, Kirika and Smith, 1987). It is also considerably higher than the mean value of 1.88 for that period - the figure on which the 95% response times were based. Why then, have P and chlorophyll concentrations not decreased, or even remained at pre-reduction levels? Assuming (i) that the loading has actually decreased (ie in spite of the diversion of industrial P being partly offset by increased runoff P), and (ii) that sufficient time has elapsed since the reduction, for lower concentrations to be expected.

The following section considers other factors that are likely to have caused the apparently anomalous response. In large part, this concerns, or stems from the release of P from the sediment. As will be seen, seasonality in the controlling factors is as significant as annual mean figures.

4.2 Factors influencing sediment release

Previous studies, including those on Loch Leven (Bailey-Watts, Kirika, May and Jones, 1990) and Coldingham Loch, Berwickshire (Bailey-Watts, Wise and Kirika, 1987) and the experiments on sediment cores referred to in Section 3.3 (see also Marsden 1989) show that associated with, and apparently enhancing sediment P release, are warm weather, low flushing rates and low nitrate levels. The periods of interest regarding the current state of Loch Leven in this connection, are late July-early August of both 1989 and 1990 (see Figure 6).

The water was very warm at these times with temperatures of *ca* 17°C (**Figure 14**). These actually correspond to periods of cooling from temperatures of *ca* 20°C in mid-July of both

years, but the water was considerably warmer than during the same time in 1985. Meanwhile, although actual data on flushing rate are not available, information on flows in South Queich, (see Figure 2) and on rainfall (Figure 3) suggest that very little water was runninginto, or passing through the loch during late July-early August in either 1989 or 1990; what is more, very little new water had been introduced in the few weeks prior to these periods - hence the very moderate rise in cumulative discharge in the South Queich (**Figure 15**). The full array of results on nitrate levels in the loch are not yet available, but the samples analysed so far include late July 1990, and at this time the concentrations were near the minimum value of *ca* 0.05 mg N l^{-1} detectable by the method used (**Figure 16**).

4.3 Summary of events leading to high algal crops

Weather conditions during 1990 exhibited striking seasonal patterns that resulted in a high annual mean level of phytoplankton in this first full year following virtual eradication of P inputs from the woollen mill. The extraordinarily high yearly average flushing rate value (*ca* 2.9 loch volumes) was due primarily to a very wet first third of the year. Indeed, some 60% of this high annual throughput of water was delivered during those 4 months. By contrast, the following 5 months saw an input equivalent to only 10% of the eventual annual total, and in the remaining quarter the throughput was moderate - approximately 10% per month. Comparatively dense phytoplankton crops prevailed over much of the second half of the year. These were initiated by the conditions occurring during a critical period in July. The water was warm, and the throughput was minimal; this led to a drawdown of nitrate, and subsequent release of phosphate ions from the sediments. At first, a number of green algae, e.g. *Scenedesmus* spp, commonly associated with shallow systems with nutrient-rich deposits, developed. Later, with nitrate levels being resumed to *ca* 1 mg N l^{-1} but under a flushing regime permitting algae to accumulate biomass before being washed out of the system, a complex sequence, including the following algae was observed: diatoms e.g. *Asterionella formosa* Hass., and a range of *Stephanodiscus* and *Cyclotella*, and blue-green algae, e.g. *Aphanothece clathrata* W. *et* G.S. West, *Microcystis aeruginosa* Kütz, *emend*, and *Gomphosphaeria lacustris* Chod.

5. THE FUTURE SURVEILLANCE OF LOCH LEVEN, AND THOUGHTS ON EUTROPHICATION CONTROL

It is essential that the aspects discussed so far, on the quality of water in Loch Leven be monitored over a time-scale appropriate for establishing the effects of the latest phase of P reduction. Many systems which have been subjected to restorative measures, have taken longer than predicted to achieve, e.g. 95% responses in P levels (Welch, Spyridakis, Shuster and Horner, 1986; Malueg, Brice, Schults and Larsen, 1973; Ahlgren, I. 1978; Cullen and Forsberg, 1988); also in the case of Loch Leven, the upper limit of the estimated 95% response time (*ca* 1.6 y) has not yet elapsed since the envisaged P reduction was achieved. It should be borne in mind too, that the restoration phase under consideration here represents a reduction in P loading of only *ca* one-third at most of the total burden of 21 t measured in 1985. This ignores the actual increase (over 1985 levels) of the runoff component, and the possibility of increases in housing and/or wildfowl aggregations, that would offset the decrease, and thereby lead to a reduced/delayed response: see also Cullen and Forsberg, (1988), regarding experiences in the restoration of Swedish lakes. Indeed, while Richards (1985) has calculated that a 30% shift is sufficient to statistically demonstrate a reduced non-pointsource loading to Lake Erie, the most successful eutrophication control exercises have involved reductions of at least 60% e.g. Waupaca Lakes, Wisconsin (Garrison and Knauer, 1983), and even 90% reduction, e.g. Kootenay Lake, B.C. Canada (Parker, 1976), Schlachtensee, Berlin (Chorus, Ewald and Klein, 1988) and Lake Malaren, Sweden (Willen, 1987). It is necessary therefore to 'wait and see'. The same philosophy is expressed by Utkilen (1989) in relation to programmes attempting to reverse eutrophication trends in the large, deep L. Mjosa, Norway. Following the arguments presented by Cullen and Forsberg (1988) for Swedish lakes, there is no doubt that a reduction in P loading to the currently, essentially P-limited Loch Leven will lead to a reduction in phytoplankton levels (e.g. chlorophyll concentrations).

In all these considerations, and as already demonstrated by the latest information on Loch Leven, a number of factors interacting in a complex manner, may delay the response. In large part, these factors relate either to the weather, or to the reservoir of nutrients that have accumulated during the decades of enrichment - with the weather controlling the rates and extent of many processes, including nutrient fluxes. The individual behaviour of lake systems further complicates matters, and inter-lake differences in the dynamics following nutrient reduction constitute a special research challenge (eg Smith and Shapiro, 1981; Ryding, 1981). Nevertheless, Loch Leven is known to respond very rapidly to changing weather conditions (Bailey-Watts, Kirika, May and Jones, 1990) and rainfall data shown in Figure 3, reflect the vagaries of the Scottish climate.

The degree of nutrient(especially P) release from the bottom deposits, is a major consideration in adjudging the efficacy of P reduction programmes; see e.g. Marsden (1989) for a general review, Phillips and Jackson (1990) for the Norfolk Broads experience, Edmondson and Lehman (1981) for Lake Washington, (and Chapra and Reckhow, 1983, for a criticism of the Lake Washington analyses), Welch, Spyridakis, Shuster and Horner (1986) for Lake Sammamish, Malueg, Brice, Schultz and Larsen (1973) for Lake Shagawa, and Burgi, Ambuhl, Bührer and Szabo (1988) for Swiss lakes. It is plain from the data presented in Section 3, that P release is also an important aspect of the present work at Loch Leven. However, the recent performance of planktonic animals has not been followed, even though it is known that daphniids in particular, are likely to affect phytoplankton crops and in this way influence the interpretation of the effectiveness of nutrient control on levels of chlorophyll, for example (Bailey-Watts, 1978, 1982). Unfortunately, programmes of lake surveillance - regardless of whether they are concerned primarily with eutrophication control - rarely include the zooplankton. There are notable exceptions, however, and the observations on zooplankton made in relation to P reduction studies, on e.g. the Norfolk Broads (Moss and Leah 1982, and Schlachtensee (Chorus, Ewald and Klein, 1988) highlight the homeostasis and resilience of aquatic systems - properties that can also explain the delayed responses to even quite marked perturbations, such as major reductions in external nutrient supplies (see also Lerman and Hull, 1987). **Appendix I** summarises the new surveillance programme on Loch Leven - this commencing in early 1991 and including a zooplankton sampling schedule.

Plainly, it remains to be seen whether further measures to reduce the external inputs of P to the loch are warranted. However, a year of relatively high flushing and a seasonality of water renewal like 1985 is necessary, for establishing more definijtely the effects of the removal of mill effluent. Ironically, spells of calm, warm weather inducing releases of P from the sediments, alternating rapidly with periods of very high flushing are needed - if a net export of P from the loch is to be achieved in the relatively short-term. Indeed, seasonal flushing (as used as in the restoration of Sebasticook Lake, Maine, USA (Rock, Courtemanch and Hannula (1984), is perhaps one of the only feasible alternatives to further reduction of the external burden of P, to reverse eutrophication trends at Loch Leven; even at Sebasticook, however, the management of the hydraulic regime was needed, despite previous removal of P from sewage works effluent (and of N from woollen mill waste). The option of further control of point-sources of P to Loch Leven should thus be retained. The findings of the new surveillance programme which it is hoped can be maintained until 1993, will determine whether such an option should be exercised. Control of point-sources is currently almost universally considered to be the most feasible and effective means of stemming P inputs (Forsberg, 1987; Ahlgren, G., 1978; Gachter, 1987). However, good progress is being made

in the treatment of large, nutrient-laden inflows by filtration following precipitation of P with eg ferric chloride; the main experience comes from work on Loosdrecht Lakes, Netherlands (van Liere, Parma, Mur, Leentvaar and Engelen, 1984), the Wahnbach Reservoir, Germany (Clasen and Bernhardt 1987) and Schlachtensee, Germany (Chorus and Wesseler, 1988). Another approach to reversing eutrophication trends by controlling nutrient inputs (as opposed to simply ameliorating the effects of the continued enrichment - see below), concerns the banning of P-rich detergents; this was included in the efforts to restore Onondaga Lake, New York State, USA (Effler, Field, Meyer and Sze, 1981) and was found to be an 'economically favourable' option for the Laurentian Great Lake Erie (Hartig, Trautrim, Dolan and Rathke, 1990), and it was used for Ontario into which Lake Erie flows (Stevens and Neilson, 1987).

By contrast, the following methods which may cure the problems of eutrophication but not stem the root cause of enrichment, are for one reason or another probably not feasible or appropriate for restoring Loch Leven:

- sediment dredging, which is appropriate for small, and usually very shallow (*ca* 0.5 m) systems e.g. Louisiana lakes studied by Knaus and Malone (1984), and the Norfolk Broads (Moss 1983).

- mixing (artificial aeration) primarily to suppress cyano-bacterial blooms, as in e.g. Waupaca Lakes, Wisconsin, USA, (Garrison and Knauer 1983) and the 8 ha Coldingham Loch, Berwickshire, Scotland (Bailey-Watts, Wise and Kirika, 1987, and lakes cited by them).

- addition of nitrate (eg calcium nitrate tetra-hydrate) using the Ripl (1978) method, also to small systems, e.g. White Loch, N.I. (Foy, 1986), Long Lake, Minnesota (Willenbring, Miller and Weidenbacher, 1984).

In conclusion, it should be borne in mind that the restoration of any lake subjected to many years of nutrient enrichment, needs time to succeed. In addition, it must be remembered that, while meteorological and anthropogenic factors 'conspired' in 1990 to temper the improvement expected for Loch Leven, (i) it is unlikely that the predicted reduction in P concentration would have occurred so soon after diversion of mill effluent, and (ii) even the 1990 P and chlorophyll levels are considerably lower than those recorded when regular studies commenced here in 1968 (Bailey-Watts, 1978, 1982; Bailey-Watts, Kirika, May and Jones, 1990). The loch is also considerably more dilute (with respect to nutrients and algal plankton) than e.g. many of the Norfolk Broad systems and the Loosdrecht Lakes which exhibit annual P loadings and mean, in-lake P and chlorophyll levels, 2-4 times those found in Loch Leven.

As such, Loch Leven no longer occupies top position in Scotland as far as phytoplankton biomass is concerned. However, as elsewhere, if conditions conducive to vigorous growth and/or accumulation of algal biomass persist in one year, for the very short time necessary for one cell division more than usual, algal problems are likely to occur (Bailey-Watts, 1991); hence the high profile attained by the cyanobacteria throughout UK and much of Europe during the last 2 years which were characterised by hot summers (see eg NRA 1990). Nevertheless, the more or less continual persistence of 'background' populations of small blue-green algae, and a diverse mixture of green forms, reflects the continuing eutrophic nature of this loch. Also, nitrate levels here continue to increase with winter maxima of > 3.5 mg N l^{-1} in recent years (Figure 16) contrasting with values of < 1.0 mg N l^{-1} in the late 1960s. Any reduction in P burden is thus likely to result in elevated NO_3.N to SRP ratios. The effects of this change on algal species composition is likely to be complex, but a suppression of atmospheric N-fixing cyanobacteria in favour of other blue-green algae, as well as green algae and diatoms, can be expected. Other changes associated with nutrient diversion (and not always involving just P as here, however) eg from dominance by *Oscillatoria agardhii* to crops of e.g. *Gomphosphaeria*, and *Microcystis* have been reported (Ahlgren, G., 1978; Dokulil, 1987). Even in these cases the additional influence of inter-annual variation in weather conditions cannot be ignored.

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