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LOCH LEVEN: PAST AND CURRENT WATER QUALITY AND OPTIONS FOR CHANGE

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Summary

1. Only very broad patterns are evident in the timing and densities of algal population maxima in the eutrophic Loch Leven; seasonal and inter-annual variation are hallmarks because the broad, shallow waterbody is very sensitive to the weather and this is highly variable.

2. Rainfall and wind patterns are especially important, as they control, respectively, the flushing rate and mixing patterns, which both have a considerable bearing on phytoplankton composition and production, and shifts in other aspects of water quality such as nitrate depletion and the release of phosphate from the sediments.

3. Even major weather events and their effects are impossible to predict to any meaningful extent, and although the long-term database allows their chances of occurrence to be estimated, these assume future climate regimes will be the same as those of the past.

4. A good example identified by the long-term research, concerns the correspondence between long, dry (low-flushing) and calm summers, and significant releases of soluble reactive P from the sediments, the accumulation of this nutrient in the water column, and the predominance of large, potentially toxic, blooms of cyanobacteria (blue-green algae).

5. Such developments have characterised a number of recent years, including 1992, and have more or less completely masked water quality improvements that were expected, after reducing the external loading of the major growth-limiting nutrient, phosphorus (P), by *ca.* 6.5t y^{-1} in the late 1980s; this reduction is equivalent to 30% of a total external loading measured in the wet year of 1985.

6. The perceived severity of an algal growth is largely a function of (i) the spatial distribution, rather than the overall lake-wide biomass *per se*, and (ii) how many people are present to witness the blooms.

7. There is good evidence to suggest that in years with wet summers, the sediment-derived, internal loading of P, is negligible, and that annual mean, in-lake P and phytoplankton levels relate to the external P inputs, according to classic eutrophication models.

8. If the average P concentration of the effluents from the Milnathort STW and the combined Kinross North and South STW were reduced to 2 mg l^{-1} (from figures of *ca* 10 mg l^{-1}), annual mean concentrations in the loch, of approximately 20μg P l^{-1} (at a flushing rate of *ca.* 1.0 loch volumes per year) to 35μg P l^{-1} (*ca.* 2.5 loch volumes per year), and *ca.* 13μg chlorophyll l^{-1} (more or less irrespective of the annual mean flushing), are predicted. These predictions rest on the assumption that the changes in **concentration** equate to an 80% reduction in the **loadings** of P from these works as measured in 1985-86. The predictions also assume that internal loading (sediment release) of soluble reactive P (SRP, which is the form of the nutrient most immediately available to algae) is negligible, although this plainly has not been the case in approximately 50% of the years for which data are available. Even then, however, the relative importance of this SRP *per se* (as opposed to the associated low-flushing, warm, relatively calm conditions), on the success of bloom-forming cyanobacteria is not known.

9. Following such a P reduction programme, the diffuse, runoff-derived loading to the loch, approximating to 3.5t in a low-flushing year, and *ca.* 9t in a wet year, would always outweigh the remaining total input of *ca.* 1.5t y^{-1} from the 3 STW in the Leven catchment.

10. Even setting aside the exacerbating effects of long, dry summers on the trophic status of the loch itself, the total external P loading even in a dry year would equate to some 0.35g P m^{-2} y^{-1} which is approximately 3 times the figure considered by OECD to be 'critical', i.e. likely to result in appreciable algal crops; over the long-term, however, the populations can be expected to be less dense than those experienced at present.

11. The degree to which bio-available P is released from the sediments, will continue to be controlled in large part by the weather, but there are reasons to be optimistic about lowering the algal crops, and eventually reducing the pool of phosphate in the upper layers of the deposits, by cutting back the external P load. Lower crops will result in improved water clarity and this could enhance the growth of rooted vegetation and increased competition with the phytoplankton for nutrient and light resources.

12. A wide variety of other lake restoration and algal control techniques are also reviewed with special reference to their applicability to the Leven situation, and in the light of a recently agreed plan to introduce Rainbow Trout (*Oncorhynchus mykiss*) to the loch, some speculations on the impacts of this fish on algal quality are presented.

Contents

1. INTRODUCTION

The aim of this report is to develop and assess options for improving nutrient and phytoplankton quality of Loch Leven water. Even after reducing the total annual external loading of phosphorus (P) to this loch by *ca.* 6.3t P y^{-1}, dense blooms of cyanobacteria (blue-green algae) have been manifest. In part, this can be attributed to the fact that although the reduction is equivalent to some 30% of a total figure of 21t as measured in a reasonably wet year (1985 - Bailey-Watts, Sargent, Kirika and Smith 1987), the loading is still *ca.* 1.1g P m^{-2} y^{-1}; this exceeds the 'critical' value as defined by OECD (1982) i.e. one likely to result in dense algal blooms, by approximately 8 times. Also, while the cutback is reported to have been achieved by the end of 1987 (Mr B D'Arcy, personal communication), the authors' laboratory recorded high concentrations of P occasionally in 1988 and 1989 (Bailey-Watts, May and Kirika 1991), and the effluent pipe was not dismantled until the end of September 1989. Above all, however - as in many previous years - the success of blue-green algae has undoubtedly been exacerbated by the long, dry (low-flushing) periods.

This report attempts to explain the complex ecological issues that have been considered in developing a control strategy and, moreover, in adjudging the various consequences (likely 'success') of any measures proposed. The main option considered here concerns the reduction of the P content of the two major sewage treatment works (STW) presently discharging P-rich effluent to the loch i.e. Kinross and Milnathort. The general cycles of algal populations are described, with special reference to those that discolour the water ('blooms'), and current conditions with respect to nutrient and chlorophyll levels, species appearances and water clarity, are described in the context of records extending back to the late 1960s. The report emphasises throughout, the erratic behaviour of this loch. This is manifested in the dynamics of many physical, chemical and biological features. While these are known to relate to the highly vagarious nature of the weather, they render the prediction of many events very difficult, even where sufficient data exist to facilitate estimates of the chances of an event taking place.

2. THE CYCLE OF ALGAL BLOOMS IN LOCH LEVEN

A major feature is the highly variable phytoplankton, with different sequences observed over long-term, inter-annual, seasonal and short-term scales. Overall abundance (expressed as chlorophyll *a* concentration in **Figure 1**, which uses an 18-year run of data) shows no consistent trend, nor do changes in the population density of individual species (**Figure 2** gives examples for the period 1968 to 1976).

However, with 'blooms' i.e. populations visibly discolouring the water, some patterns emerge, although again, the actual densities of the organisms and the timing of their appearance also vary. The most consistent feature concerns dense crops of unicellular centric ('pill-box') diatoms coming to peak numbers usually in late-winter or early spring (**Figure 3**). Even these comprise assemblages of different species from year to year (Bailey-Watts, 1988a,b) but the genus *Stephanodiscus* is usually a major component. Sparser, more mixed, crops of algae follow the collapse of these populations, but by various growth strategies, certain species may build up biomass to form blooms between say, June and October. Of these, the 'summer' growths of blue-green algae are the most noticeable, because many of the species are of millimetre dimensions and thus discernible with the naked eye; in addition, they often exercise an advantage over other algae, in being able to move up through a water column to form the characteristic surface aggregations ('flowering'). Indeed, the larger the species, the faster they can rise to the surface (Stokes Law, see e.g. Reynolds 1984).

The severity of the blooms is largely a function of our perception of them, i.e. how many people are around at the right (or wrong!) time. The spring diatom maxima often produce more chlorophyll and biomass than the summer cyanobacteria, but the diatoms attract much less attention, because (i) they remain distributed throughout the water column, imparting a general cloudiness, rather than forming a surface cover, (ii) less people wish to paddle in the loch, or visit the castle in March than in the summer months, and (iii) the diatoms do not persist in large numbers for more than a few weeks, whereas the cyanobacteria, albeit sometimes involving a succession of species, can persist for months.

3. THE POSITION NOW COMPARED TO EARLIER YEARS

Inter-annual differences in the timing and extent of blooms at Leven cannot be over-stated, and it is important in the context of comparing 'now' with 'then'. Such a comparison would imply a consistent (long-term) change or trend from the early years of study to recent times - and this is not the case with Loch Leven. However, from records extending back more than 25 years, the following two features have been selected to illustrate some major shifts. One of these relates to the period 1967 to 1971, which differs from all subsequent years. The other concerns 1985 as an example of a 'good' quality, and fairly recent year, prior to the latest (1987) cut-back of P-rich mill effluent, compared with a number of more recent years - which have been very 'bad' but similar to other runs of years from e.g. the mid- to late 1970s.

3.1 Algal blooms pre-1971 ('no' *Daphnia*) compared with later years

The differences in general algal character between these two periods, corresponds to a shift in the abundance of the water-flea, *Daphnia* (here, *D. hyalina* var *lacustris* Sars). This change is superimposed on the more 'natural' year-to-year differences in species and biomass. From May 1967 to approximately half-way through 1971) algal biomass was extremely high (**Figure 1**). The annual mean chlorophyll *a* concentrations (calculated from weekly sampling) for the last 4 of these years, ranged from $65 \mu g\ l^{-1}$ to $95 \mu g\ l^{-1}$ (Bailey-Watts 1978). By chance, the value for 1972 was *ca.* $85 \mu g\ l^{-1}$, but with a very different seasonal distribution of algae, and the densest population of *Anabaena* for many years (**Figures 1** and **2**). This contrasts with the highest values recorded since 1972 - *ca.* $50 \mu g\ l^{-1}$, in 1990 and 1992. Examples of much lower levels recorded during this period, are those of 32, 41 and 36 $\mu g\ l^{-1}$ in the 3 years 1977 to 1979 (Bailey-Watts 1982), while similar concentrations of 21, 33 and $42 \mu g\ l^{-1}$ were recorded in 1985, 1988 and 1989 respectively (Bailey-Watts, May and Kirika 1991).

Up to 1971, the algae were able to capitalise, as always, on the ideal growth conditions afforded by this shallow, nutrient-rich, and not too-rapidly flushed, loch. They were thus able to produce a lot of cells. These populations were also able to build up considerable biomass virtually unchecked (i.e. with few losses), because *Daphnia*, which is a major grazer of phytoplankton, and an animal normally found in the vast majority of the World's freshwater lakes, was to all intents and purposes, absent (Bailey-Watts 1973; Johnson and Walker 1974). The reason for this is not known, although poisoning by dieldrin insecticide has been considered. The species was recorded in Loch Leven at the end of the 19th century (Scott 1899), and it was also present in 1954, for example (Johnson and Walker 1974). Indeed, there is evidence (Bindloss 1976) that algal crops during these early years increased in biomass until they shaded themselves to the extent that their photosynthetic efficiency was impaired; the maximum population densities were thus light-limited rather than nutrient-controlled. By contrast, in subsequent years, starting with an immense explosion of the *Daphnia* population in June 1971 (Bailey-Watts 1973; Bailey-Watts, Kirika, May and Jones 1990), the loch shifted to the state with which we are now more familiar i.e. with algal crops that are dense, but rarely as dense as those recorded in 1968-1971. It is also not clear why this animal reappeared; the hatching of resting eggs (ephippia) known to have been present in the sediments is one possibility, while an innoculum of the species from other waterbodies is another.

This very significant change in algal abundance was not the only impact of the rise in *Daphnia.* **Figure 4** (from Bailey-Watts, Kirika, May and Jones 1990), shows how the sizes of the dominant phytoplankton organisms during the 'pre-*Daphnia*' years, contrast with those of 1972 to 1976: relatively small species (including blue-green algae) predominated in the summers of 1968, 1969 and 1970, while the year's relatively large forms became a feature of subsequent summers. This size shift is attributed to the feeding activity of *Daphnia.* The feeding mechanism is very complex (Fryer 1991), but essentially, the animal appears to select the smaller elements of the plankton - leaving larger species such as the notorious, bloom-forming cyanobacteria, to proliferate. The crustacean attained a peak density of *ca.* 70 individuals l^{-1} at the end of June 1992 - when these large algae were abundant.

If these observations, and associated experiments, had not been made, the phytoplankton changes would most probably have been attributed entirely to an earlier reduction in the external phosphorus loading (also a cutback of mill P). These remarks are not meant to suggest that this phase of P control contributed nothing to the general lowering of algal biomass over the 1970s (see e.g. Bailey-Watts, Kirika, May and Jones 1990), but it did not start until 1972.

3.2 Algal blooms in the late 1980s and early 1990s compared to 1985

A comparison of the algae found in a number of years since 1985, with those recorded in 1985, also highlights the hallmark of inter-annual variability of the Leven system. The phytoplankton was much denser in 1992 than in 1985 (**Figure 5**). However, elevated summer crops dominated by bloom-forming cyanobacteria as indicated for 1992, are not new for this loch (Bailey-Watts 1978, 1982 and 1986; Bailey-Watts, Kirika, May and Jones 1990; Bailey-Watts, May and Kirika 1991), although it is possible that this particular *Anabaena-Microcystis* phase lasted longer than many others with these species dominating.

As noted above, the mean annual chlorophyll concentration for 1992 could be much the same as that for 1990 i.e. $50\mu g\ l^{-1}$ (Bailey-Watts, May and Kirika 1991), and thus as high as any value since 1972, although then, some $85\mu g\ l^{-1}$ was measured. The major concern in 1992, however, is not with the overall pigment levels, but with the species of algae present, and their spatial distribution patterns on rather few days. *Anabaena* increased from approximately 3000 small tangles ml^{-1} in early May to bloom proportions in June and July. The pair of high pigment values shown for mid-June 1992 in **Figure 5** are derived from scums at the loch edge, and not the loch as a whole. The highest lake-wide concentrations were equivalent to *ca.* 95 μg chlorophyll l^{-1} and recorded from the 10-site and 3-site sampling programmes carried out on 16 and 24 July respectively. On the first of these, the population was very patchy - in keeping with the prevailing calm conditions. A much more uniformly distributed crop was found on the 24th which was a very windy day. The *Anabaena* population then decreased, to be fairly swiftly succeeded by *Microcystis*. The numbers of this second cyanobacterium of major importance this year, peaked in August. It too, exhibited patchy distributions alternating with more regular dispersion patterns, in line with changing weather conditions. Generally, however, the chlorophyll levels remained above $60\mu g\ l^{-1}$. The chlorophyll maximum of *ca.* 100 $\mu g\ l^{-1}$ in September 1992 (**Figure 5**) consisted primarily of centric diatoms. This population soon collapsed, and since that time, the concentrations have remained between $5\mu g\ l^{-1}$ and $25\mu g\ l^{-1}$, while total P levels have fluctuated around double this value.

4. FACTORS CONTROLLING GROWTH

It is important to distinguish between the factors controlling growth *per se* i.e. the rate of increase of an algal population, and those controlling the biomass achieved. Commonly, it is the amounts of material present at any one time which determine the extent of the problem. [In this connection, it should be pointed out that elevated concentrations of nutrients *per se* are usually of little concern]. Especially when dealing with populations of buoyant cyanobacteria, care also needs to be taken over identifying real growth, as opposed to a re-distribution of the biomass.

The observed sequences of algae and shifts in their abundance, are the outcome of the relative 'abilities' of each species to cope or contend with, or capitalise on, the ever-changing suite of environmental conditions facing them. Many factors influence the algae, but the current focus at Leven is on light and nutrient availability as the main controls of growth potential, and on flushing rate and processes such as sinking of material onto the sediments, and grazing by zooplankton, which determine the biomass achieved.

The importance of each factor varies through the year. Indeed, in spite all nutrient requirements being met, and a relative lack of (active) grazers, early in the year, phytoplankton crops are rarely dense because days are short; light energy is then at a premium. Contrastingly, with lengthening days, the spring diatom maximum is controlled by nutrient availability; P, or sometimes SiO_2, is depleted, while N in the form of nitrate, usually remains at high, non-limiting levels (Bailey-Watts 1988b). As already described, these diatoms can build up to prodigious numbers, yet, being heavy, silica-containing cells, they suffer continual losses through sinking; if this were not the case, we would see many more of them (Bailey-Watts 1990b).

Later in the year, but always depending on flushing conditions which determine the time available for algae to capitalise on light and nutrients, other species start to predominate. If it is a wet and windy summer/autumn, forms capable of fairly rapid growth or requiring turbulent mixing to maintain themselves in the column, appear. A host of such types has been recorded e.g. *Dictyosphaerium* in 1969 and *Steiniella* in 1970 (Bailey-Watts 1974), *Asterionella* and centric diatoms in 1985 (Bailey-Watts, Smith and Kirika 1989).

By contrast, in warm, dry (low-flushing) summers, a sequence of blue-green algae is more probable. Firstly, when it is warm, much of the bottom of Loch Leven i.e. the surface sediments even to depths of 5m, reach the same temperature as the overlying air mass. This leads to accelerated bacterial activity, and in particular the microbial process of de-nitrification (Johnston, Holding and McCluskie 1974), which results eventually (by mid-June, if not earlier) in the virtual complete disappearance of nitrate from the water (Bailey-Watts, Kirika, May and Jones 1990). Our recent data on nitrate need to be checked because of an analytical problem, but first impressions are that the concentrations fell below 0.1mg N l^{-1} by mid-June, and to less than a few microgrammes per litre in August. During the summer of 1985, the concentrations remained above 0.5mg l^{-1}. Our data and those of the FRPB also suggest that the winter concentrations of nitrate were considerably lower in 1992.

Nitrate depletion to e.g. <100, possibly 50 μg N l^{-1}, may not be evident for long, but it seems to have two major consequences. Firstly, it limits the growth of many algae excepting some of the blue-green species capable of fixing the gaseous form of N dissolved in the water. Secondly, because the nitrate molecule contains oxygen, its disappearance reduces the oxygen tension at the sediment-water interface, which could also lead to the release of bio-available SRP from the deposits. This is what has happened in a number of years in the past (Bailey-Watts, 1985, 1986; Bailey-Watts, Kirika, May and Jones 1990), and more recently (Bailey-Watts, May and Kirika 1991, 1992). Indeed, P maxima of *ca* 60 μg l^{-1} in 1989, and *ca* 80 μg l^{-1} in 1990 are major features of **Figure 6** and are attributed to release from the sediments, since inputs from outside the loch would have to been so massive to effect the rapid increases in P concentration observed. 1991 also saw values of *ca* 60 μg l^{-1}, while figures of 100 to 150 μg l^{-1} were recorded in 1992. It is possible that the actual maxima (which would require daily sampling to detect) may have been higher than these recorded values. Winter SRP levels in these last few years are generally low, but further analysis of the data is necessary to establish whether this reflects reduced P loadings, or greater utilisation of the nutrient by algae.

It is also thought that in some years - including 1992 - the high pH induced by the photosynthetic activities of the algae themselves, effects the mobilisation of P from the sediments. *Microcystis* in particular, can fix carbon efficiently at the sort of pH values that we have recorded this year i.e. >8.5 units.

It is usually accepted that the SRP released from the sediments fuels further algal growth - the nutrient having been reduced to just a few microgrammes per litre by late May in 1992, for example. The question still remains, however, as to whether sediment P release, or the warm, and calm, low flushing conditions associated with this process, is the more influential in promoting the cyanobacterial blooms. SRP levels over the loch as a whole, decreased by *ca.* $50\mu g\ l^{-1}$ over 10 to 15 days following the time of the recorded peak level of this nutrient in June 1992. Meanwhile, the concentrations of algal pigment increased by a similar amount - and it is quite common for algal cells to contain similar amounts of chlorophyll and P, although many algae are able to sequester P in excess of immediate needs and redistribute it at successive cell divisions. Little is yet known, however, about the P status of the 1992 blue-green algae, and research in these areas continues, as does work attempting to establish what levels of nitrate, water temperature and flushing rate need to be attained, and for how long, to effect the P releases. It appears that once cooler conditions return and higher oxygen levels are established, a good proportion of the SRP returns to the sediments by re-adsorption onto iron compounds with which much of the phosphate in the Leven deposits is associated (Bailey-Watts, May and Kirika 1991). The peak SRP level in early June 1992 may have been checked by a spell of cool weather during which the water temperature fell from 17 to *ca.* 14°C. The loch returned to the higher temperature within another week, but thereafter, it gradually decreased to 12°C by September, and to 5°C during October.

In contrast to events that appear to have more or less immediate effects on the algae, the influence of some environmental changes may not be evident until many months later. In this connection, the triggers of the *initial* increases in the population density of organisms that we tend not to detect until they have attained quite considerable numbers, is a major, neglected area of phytoplankton ecology. However, long series of observations on the Leven phytoplankton, suggest that within the Cyanobacteriales, for example, there are some species that can build up biomass very rapidly, and others which take longer to attain bloom proportions. *Oscillatorias* e.g. *O. redekei*, *O. agardhii* and *O. rubescens* appear to flourish best in summers preceded by a dry (i.e. low-flushing) winter; a good example is 1976. Contrastingly, *Microcystis* and *Anabaena* - both of which have been prominent this year - can capitalise very quickly on prevailing conditions, by being able to start a population from resting cells in the sediments i.e. they do not require a large over-wintering innoculum. The rapidity with which *Microcystis* 'replaced' *Anabaena* at the end of July 1992, is a good example of this. In common with all algae, however, even these opportunists require at least some time to grow, and the period over which they can do this is determined by the flushing regime. The two genera differ in other respects. *Anabaena* can fix atmospheric nitrogen, while *Microcystis* apparently cannot (but see Paerl, in Sandgren 1988). *Microcystis*, which replaced *Anabaena* very quickly in late July 1992, can probably tolerate lower light availability (Reynolds 1984).

Bailey-Watts, Kirika, May and Jones (1990) attribute the variability in the phytoplankton of Leven to shifts in physical conditions (e.g. flushing rate, water temperature and mixing), wrought through a very erratic and variable weather regime. For example, surface blooms will only form as long as calm weather prevails, and the duration of a bloom depends ultimately on the weather. Indeed, it is highly probable that had not the breeze come from the east on the day of the SNH 'launch' (13 June 1992) at Loch Leven, the very heightened perception of the bloom problem would not have come about. The importance of the weather in dominating a number of key water quality features, is further highlighted in the following section which develops a eutrophication control strategy. Here, the main problem is the unpredictable nature of important releases of phosphate from the sediments. Furthermore, there is evidence to suggest that the weather in 1985 - the year in which the P loading was assessed and on which much of the thinking on the restoration of this loch is based - was unusual; as will be explained, however, this has turned out to be very fortunate.

5. THE CONTROL OF PHYTOPLANKTON ABUNDANCE AT LOCH LEVEN BY REDUCTION OF P INPUTS FROM SEWAGE TREATMENT WORKS

The main reason for having to re-consider the possibility of improving the chemical, and especially the biological, quality of Loch Leven water, is that the weather has masked the improvement that it was reasonable to expect (from the 1985 study by Bailey-Watts, Sargent, Kirika and Smith 1987) following a significant reduction in the external P loading to the loch by the end of 1987. The total external loading in 1992 is likely to be *ca* 14t (Bailey-Watts, May and Kirika 1991), whereas that measured in 1985 was 21t. Yet, 1992 has seen the acute and classic manifestations of eutrophication i.e. releases of soluble reactive P (SRP) from the deposits, and dense cyanobacterial blooms, while 1985 passed without these features. As shown above, neither of these classes of problems is new to this loch, however.

The release of P from sediments ('internal loading') may be viewed as alleviating P limitation in an otherwise essentially P-limited system. As outlined above, particular algae may occasionally be limited by shortages of e.g. nitrate, silica or light, but overall, P is the most important factor, and there are three classes of eveidence supporting this view. Firstly, dense algal populations often collapse when, or soon after, the levels of this nutrient are very low (e.g. Bailey-Watts 1988b). Secondly, factorial nutrient enrichment experiments with natural plankton, indicate seasonal patterns of P shortage in line with the observed changes in phytoplankton and nutrients in the loch (see e.g. Bailey-Watts, 1990). Thirdly, while loadings estimated for 1969 to 1972 by Holden and Caines (1974), and for 1973 to 1978 (unpublished data) are not based on such an extensive and intensive sampling programme as that operated by Bailey-Watts, Sargent, Kirika and Smith (1987), the values are generally mirrored by the in-lake phytoplankton densities measured as chlorophyll *a* (**Figure 7**, from Bailey-Watts 1985). However, the relationship is not as simple as it may first appear, and inter-annual differences in weather (flushing rate in particular) are likely to contribute to this. As examples, annual mean concentrations of plant pigment in 1974 and 1975 are very similar, while the loading in the former is approximately double that of the latter. Considerably denser phytoplankton crops were recorded in 1973 than 1974, in spite of comparable loadings. The point plotted for 1976 is also misleading, in that the (very hot) summer was dominated by the cyanobacterium *Oscillatoria rubescens* which is characteristically very low in chlorophyll per unit cell volume (Bailey-Watts 1978, but probably not *O. agardhii* as reported there); there was thus considerably more biomass than suggested in **Figure 7**. It is interesting to consider the data for 1985 and 1990 in the context of **Figure 7**: y_{1985} 21, x_{1985} 1.54 (Bailey-Watts, Sargent, Kirika and Smith 1987) and y_{1990} 50, x_{1990} 1.00 (Bailey-Watts, May and Kirika 1991).

P limitation is thus an important part of the thinking about the restoration of Loch Leven. It is also relatively easy to reduce the *external* loadings of P to this lake, even though measures for stemming the *internal* loads are far less easily contemplated. [Nevertheless, many other eutrophication control options - including sediment treatment/removal - have been considered (**Appendix I**)].

In years such as 1985, when the release of SRP from the sediments is not important in the total P budget, Loch Leven behaves in a manner predicted by a number of classic eutrophication models (Vollenweider 1968, 1975; Dillon and Rigler 1974; OECD 1982). Consider the fit of the Leven data to three of these models:

i) the Dillon and Rigler (1974) formulation relating the P loading (**L**, in mg P m^{-2} y^{-1}) to the in-lake P level, actually at 'spring overturn', but taken here as the annual mean value, ($[\mathbf{TP}]_l$, in μg l^{-1}), the P retention coefficient ($\mathbf{R_p}$ - the fraction of P entering the system, that is retained), the mean depth of the loch (**z**, in metres), and the flushing rate (**p**, in loch volumes y^{-1}):

$$[\mathbf{TP}]_l = \frac{\mathbf{L\,(1 - R_p)}}{\mathbf{z \cdot p}} \qquad (1)$$

Our data for 1985 based on a very thorough coverage of the catchment and 8-daily sampling of some 25 stream sites (Bailey-Watts, Sargent, Kirika and Smith, 1987) are as follows: **L**, 1540 mg m^{-2} y^{-1}; $\mathbf{R_p}$, 0.60; **z**, 3.9m; **p** is 2.57 if the inflow volume is used, or 2.50 if the outflow volume is used; the 2 values predicted for $[\mathbf{TP}]_l$ are 61.5 μg l^{-1} and 63.3 μg l^{-1}. The measured value was 62.7 μg l^{-1}. When $\mathbf{R_p}$ was predicted according to the equation of Kirchner and Dillon (1975), a value of 0.58 was obtained - which is very close to the measured figure of 0.60.

ii) the OECD (1982) equation based on all of the OECD data, which included Loch Leven:

$$\mathbf{[TP]_l} = 1.55\ [\mathbf{[TP]_i})/(1 + \sqrt{\mathbf{T}_{(w)}})]^{0.82} \tag{2}$$

where:

$\mathbf{[TP]_l}$ is as in equation [1], $\mathbf{[TP]_i}$ is the mean influent concentration of total P - 153μg l^{-1}, and $\mathbf{T}_{(w)}$ is the water residence time in years - 0.39 (i.e. the reciprocal of **p**); then, the predicted $\mathbf{[TP]_l}$ is 64.4 μg l^{-1} which is also very close to the measured value of 62.7 μg l^{-1}.

iii) the OECD (1982) model based on the Shallow Lakes and Reservoirs Report, which also included information from Loch Leven:

$$[\mathbf{chl}\ \boldsymbol{a}] = 0.43\ [\mathbf{[TP]_i}/(1 + \sqrt{\mathbf{T}_{(w)}})]^{0.88} \tag{3}$$

where:

[chl *a*] is the annual mean chlorophyll *a* concentration, while the other parameters are as defined in (2); the advantage of considering pigment levels is that these focus on the organisms which, rather than P itself, cause the problems associated with eutrophication. By inserting the values for 1985 already quoted for $\mathbf{[TP]_i}$ and $\mathbf{T_w}$, this equation predicts a value of 23.5 μg chl *a* l^{-1}, while 21.02 μg l^{-1} was measured. As it happens, this equation predicts higher pigment values per unit of P load, than other OECD models - although only by *ca* 10%. It may thus be especially relevant to the situation at Loch Leven, because this lake appears to be very efficient at converting its P loading to phytoplankton biomass.

It was not until after the completion of the 1987 report, that this author appreciated just how valuable 1985 had been for carrying out the loading study; the fit to these models is due to the very fact that the weather in 1985 was not conducive to significant releases of SRP from the sediments (**Figure 6**). The 1987 report pointed out, however, that (in 1985):

'there is no evidence yet, to rule out the possibility that considerable amounts of P were released from the sediments; if correspondingly high fluxes of P back into the deposits occurred, no net gain of P by the water column would be detected. Long-term observations on physical and chemical factors at Loch Leven, however, suggest that release of the nutrient would not be important in 1985. Firstly, for rapid releases, high temperatures and low nitrate concentrations and calm conditions must prevail (Bailey-Watts 1986; Bailey-Watts, Wise and Kirika 1987). In 1985, the maximum temperature was only 15.9 ^{0}C and on this and only one other occasion - in July - were temperatures of >15.0^{0}C recorded. Secondly, in some hot summers, massive and rapid increases in SRP in the loch are recorded, but the nutrient subsequently decreases, mainly due to re-adsorption by the sediments. For example, no remarkably dense populations of phytoplankton were observed in 1980, following an internal loading of *ca.* 2.4 tonnes SRP within a two-week period in August.'

The options considered below, for lowering P and algal levels, focus on the possibility of reducing the external P loading by cutting back the total P content of the effluents from the following sewage treatments works (STW), to 2mg l^{-1} or 1mg l^{-1} (from levels of around 10 mg l^{-1}):

(a) the combined Kinross North and Kinross South
(b) Milnathort
(c) both (a) and (b).

The model-predicted consequences assuming no internal P loading, are examined first. Separate attention is paid to the likely effects of internal loading and whether, because of these, it is futile to contemplate the restoration of this problem loch. Thought is also given to whether it would take e.g. 3 years or 30 years to achieve the predicted improvements in algal levels, species, blooms and water clarity.

As, in 1985, the effluent from the Milnathort works, for example, commonly contained 10 mg total P l^{-1}, the following calculations assume that the reductions in **concentration** to 2mg P l^{-1} or 1mg P l^{-1} are equivalent to lowering the P **loading** by 80% and 90% respectively. Should this not prove to be the case - in the event of more reliable data being obtained, or measurements being made - the values discussed below, can easily be re-calculated. The chances are, however, that effluent quality is likely to be better, rather than worse, than that indicated above. Then, the percentage improvements presently assumed are likely to be somewhat optimistic.

5.1 Assuming no internal recycling of P

Figures 8a to **8d**, taken from Bailey-Watts, Sargent, Kirika and Smith (1987), show how the *annual*, external loadings of total P and SRP, and the *annual* mean levels of total P and chlorophyll *a*, are likely to vary according to a range of P reduction situations, and each, over the spectrum of flushing values measured for the 49 years up to 1985. While the 1985 value for **p**, was 2.54 loch volumes y^{-1} and 4th in the array of 49 figures, and thus had a chance of <5% of being exceeded in the next 49 years, the values for 1988 and 1990 did exceed this - not least, due to very wet first quarters.

The slopes of the lines in **Figures 8a** and **8b** are based on the assumption - drawn from 1985 data (Figure 29 of Bailey-Watts, Sargent, Kirika and Smith 1987) - that annual inputs of runoff-derived P are linearly related to runoff i.e. flushing. If the lines in these Figures were extended to the **y** axis where **x** (the **p** value) is zero, the point of intercept on the **y** axis gives the total loadings derived from point sources plus over-wintering wildfowl - i.e. inputs reaching the loch in negligible amounts of water, and independent of runoff regimes.

New loading predictions rest on two assumptions. The first, is that 100% of the P from industry (the woollen mill) has been eliminated (this being a larger reduction than those of 50% and 80% thought possible in 1987, and as described by lines C and D in **Figures 8a-d**. The other assumption is that the combined Kinross Nth and Sth STW is now in operation but, due to extra housing, the *loading* is likely to have changed little since 1985 when two separate works existed. The 1992 situation for TP is thus, as indicated by the upper line in **Figure 9a**. Under this loading regime, the predicted, annual mean TP concentrations in the loch itself are still high (**Figure 9b**), that is, even without additional inputs from the sediments. At *ca.* 53μg l^{-1}, for a low flushing year (**p**=1.0), they are not markedly lower than that predicted (and measured) prior to the elimination of the mill P, for the high flushing year of 1985 (**p**=2.5). Similar comments apply to the predicted chlorophyll *a* levels (**Figure 9c**).

The three P reduction possibilities under consideration lead to the following conclusions:

(i) if 80% of the P considered (from measurements made in 1985-1986) to emanate from the combined Kinross STW, was eliminated, a reduction of *ca* 2.77t would result; this is indicated by the third line from the top in **Figure 9a**. Note that following this control strategy, the predicted annual mean levels of P and chlorophyll deviate little from 40μg l^{-1} and 15μg l^{-1} respectively.

(ii) by eliminating 80% of the P derived from the Milnathort STW, instead of Kinross, the total external P loading would be reduced by *ca.* 1.4t (2nd line from the top in **Figure 9a**). This would result in annual mean P and algal levels of *ca.* 46 μg l^{-1} and 18μg l^{-1} respectively.

(iii) if both of these works were targeted in this way, the situations described by the bottom lines in **Figures 9a to 9c** are predicted; these correspond to loadings of between 5.2t P y^{-1}, and annual mean levels of 20 to 35μg P l^{-1} and *ca.* 13μg chlorophyll *a* l^{-1}.

As increasing amounts of P derived from point-sources are eliminated, plainly the ratio of point-source to diffuse-source inputs in the total P loading is reduced. This means that the ratio of P supplied independently of rainfall (runoff) to that supplied as a function of rainfall, also decreases. The changing form of the lines relating predicted P and chlorophyll concentrations in the loch, to flushing rate, reflect this (**Figures 8c, 8d, 9b and 9c**). Thus, where point-sources dominate (upper portions of the graphs for the 'pre-mill' cutback situation in particular), the nutrient and pigment levels decrease with increased flushing, largely due to the diluting effect of the incoming water. At the same time, chlorophyll concentrations decrease, primarily because at high flushing, the algal populations have less chance to accumulate biomass (regardless of nutrient conditions), before being washed out of the system. Where diffuse sources outweigh point-sources (lower portions of the graphs) the annual mean levels of P and chlorophyll are not so sensitive to **p** (although if there were no point-sources and no flushing, nothing would enter the system other than dry-deposited P from the atmosphere and wildfowl); the models reflect this, in the increase (though very gradual for the control situation being considered now) in P and chlorophyll, with increasing **p**.

On the basis of this analysis, Loch Leven would be expected to function as indicated by the lower lines on the graphs, even if the P coming out of just the Kinross works was reduced by *ca* 80%. However, attention should be given to cutting back the contribution from the Milnathort STW as well, because we can never be as precise as we would like, in predicting events - even on the basis of the extensive data at our disposal here. Also, if STW P can be virtually eliminated, then it should be eliminated even if the models suggest that this would constitute an 'overkill' (but see below) - because we will still have to contend with the internal loading, the severity, duration and timing of which are impossible to predict with any certainty, as these features are driven by weather events.

If both works were attended to, the external P load would be reduced by *ca.* 4.2 tonnes per year. More importantly, approximately only 1.45t would be entering from the main, SRP-rich, fairly constant, point sources (STW). The major external source of P would then be the land, the runoff from which is somewhat less concentrated in immediately available P, and by its very nature, accompanied by diluting (and generally flushing) water. However, in varying with rainfall, this land-derived source could reach 9t in a high flushing year, although less than 4t in a dry year (see **Figure 9a**).

These models suggest that, following 80% reduction of the effluent P levels in both works, in-loch P and phytoplankton levels would both decrease by *ca* one-third, i.e. from the lowest current levels, to between approximately 25μg P l^{-1} and 35μg P l^{-1}, and 13μg chlorophyll l^{-1}. The algal level is equivalent to less than two-thirds of that measured in 1985, and although that was a remarkably 'good' year, it should be borne in mind that not all the other years have been as bad as the last 4 or 5.

On the basis of other models developed by Vollenweider (OECD 1982), the loch would be expected to reach its new equilibrium with respect to these P and algal levels, within 3 years, and certainly 30 years, but - as amply demonstrated by our data for 1987 on (Bailey-Watts, May and Kirika 1991) - the actual rate of improvement will rest on not just the annual flushing regimes, but the seasonal distribution of rainfall and runoff too.

5.2 Assuming internal recycling of P (of the magnitude experienced in e.g. 1990 and 1992).

In spite of considerable attention paid to the factors controlling, and the mechanisms of, sediment phosphate release (Marsden 1989), the relative importance of low oxygen tension at the sediment surface, or algae-induced high pH, in Leven is not well established. However, as already indicated, the (weather) conditions that appear to initiate the releases are known. So too, are the loadings that this recycling generates. For example, data for 1990 (included in **Figure 6**, taken from Bailey-Watts, May and Kirika 1991), shows a peak SRP concentration of 78μg l^{-1}; this level has been exceeded to any significant extent in only two previous summers i.e. July 1973 with 96μg l^{-1}, and June 1976 with 97μg l^{-1}, although values exceeding 100μg SRP l^{-1} have been recorded this year.

Of the 78μg l^{-1} measured in 1990, some 72μg l^{-1} is considered to be due to release of phosphate from the sediments, while the remaining small proportion (6μg l^{-1}) constitutes the background level which characterised the loch previous to the release event. The release would equate to a flux of *ca.* 3.7t P, if the values from the two monitoring sites (Sluices and Kirkgate Pier) can be taken as representative of the loch as a whole. By similar inference, the input due to the release in 1992 amounts to some 5.2t. Chlorophyll *a* levels exceeded 50μg l^{-1} over much of the last half of both 1990 and 1992. They also exceeded 75μg l^{-1} on 14 sampling occasions in 1990, and rather less often in 1992 over the lake as a whole - contributing to the two annual mean concentrations of *ca.* 50μg l^{-1} quoted above, following values of just 21, 33 and 42μg l^{-1} in 1985, 1988 and 1989 respectively.

6. CONCLUDING THOUGHTS ON SEASONAL ASPECTS AND THE P REDUCTION OPTIONS FOR IMPROVING THE CHEMICAL AND BIOLOGICAL QUALITY OF LOCH LEVEN WATER

Without reliable forecasts of rainfall distribution and wind events, it is impossible to predict how the concentrations of P and algae, and the types of algae ('succession') will vary over a particular year. Already, however, some very low concentrations of phytoplankton, and correspondingly high readings for water clarity, have been recorded, i.e. outwith periods of low flushing - even in years such as 1992 which have been otherwise characterised by dense algal blooms. In August 1992, Secchi disc values ranged from <0.5m to 1.5m, mainly as a result of the patchy distributions of the dominant blue-green algae. **Figure 10a** uses data for 1980 to 1983, and **Figure 10b** does the same for 1992, to illustrate the relationship between water clarity and chlorophyll.

Except under calm conditions when a population is concentrated at the loch surface, the water can be quite clear even - indeed, especially - when large blue-green algae dominate the total biomass. Even sparser crops of smaller algae, can reduce the clarity of the water to a greater extent. Hence, the low Secchi disc readings (*ca.* 0.5m) associated with the dense crops of small unicellular diatoms in spring, and some of the populations of picoplanktonic cyanobacteria (Bailey-Watts and Komarek 1991) in 1968 and 1969, for example. Differences in algal quality (size, colour etc.) may explain the wide range of Secchi readings over a relatively narrow band of chlorophyll values around $20\mu g\ l^{-1}$ (**Figures 10a** and **10b**). Only by reducing algal levels beyond the point of inflection of the lines in these Figures, can marked improvements in water clarity (and, thus, the performance of rooted vegetation which could then compete more effectively with the phytoplankton for light and nutrient resources) be reasonably expected.

Even setting aside the effects of additional P from the sediments, the annual loadings corresponding to the situation where Kinross and Milnathort STW are treated to the extent of 80% P removal described above, would range from *ca.* 0.39 to $0.87g\ m^{-2}$ of loch surface - the variation being very largely due to long-term differences in annual flushing rate. If an additional 10% of the P content of these effluents was removed, the specific areal loadings would range from *ca.*

0.36 to 0.83 g m^{-2} y^{-1} - i.e. some 10% lower than the previous figure which relates to a low-flushing year (e.g. **p**=1), but <5% less than the upper limit of the range, which corresponds to a high-flushing situation (e.g. **p**=2.5). Similarly, if the external supplies of P from all the STW in the Loch Leven catchment were eliminated, the specific areal loads would range from approximately 0.29 g m^{-2} (**p**=1.0) to 0.65 g m^{-2} (**p**=2.5).

Even the lowest of all of these figures (0.29 g m^{-2}) exceeds by far, the value of 0.13 g m^{-2} defined by Vollenweider (1968, 1975) as 'critical' i.e. likely to result in dense algal crops, for lakes with a mean depth of 5m or less, and Loch Leven, with a mean depth of only 3.9m, is especially efficient at converting its nutrient income to plant matter.

On the basis of this analysis, Loch Leven would certainly continue to exhibit eutrophic features, following the execution of the P control programmes envisaged. As a result, especially if the problems exacerbated by internal loadings of P are taken into account, there may be a tendency to cancel any further targeting of the STW. However, there are a number of other considerations, and these are based on the assumption that conditions would at least improve if these controls are set.

If the spring algal growths were stemmed by P loading reduction, water clarity in early spring should improve. Current understanding indicates that the production of macrophytic vegetation (albeit possibly with associated epiphytic algae) would increase. In reducing the production of (heavy) diatoms in particular, the loading of mineralisable organic matter onto the sediments would decrease. In the long-term, this would be expected to stem the potential of summer releases of P from the deposits. The records show that the extent of recycling in any one summer, is primarily weather-driven, and not a direct consequence of the level of diatom production in the previous spring. Conceptually too, one would expect that de-nitrifying bacterial activity (which appears to enhance P re-mobilisation), would decrease. However, it is likely that the amount of carbon already in the sediments could fuel this process for many years to come - but, again weather events will determine when, and to what degree it occurs.

As with virtually all aspects of the chemical and biological dynamics of this loch, ideas on the chances of occurrence of a particular event can be gained, but it is not within our power to accurately predict whether a particular year will see that development. The accrued knowledge on how this loch functions is not inconsiderable, and this institute is well on the way to developing dynamic sub-models of a number of the components. But in the absence of reliable long-term weather forecasts, I cannot yet envisage a model that could predict, for example, whether 1993 will see an SRP release of the magnitude of that recorded this year, or in which month the peak will occur. We do, however, have an idea of the chances of such an event (see below), although even this rests on the assumption that future weather patterns will resemble those of the recorded past - and this may not be the case. Similarly, if the current spell (late 1992) of wet weather equates to rapid flushing of the loch, blue-green algae of the type seen this year, rather than those of the *Oscillatoria-Planktothrix* group, are more likely to become prominent next summer - **if** it is dry.

There are major problems with the internal loadings - particularly those of the magnitude suggested for 1990 and 1992 - because they, in combination with the conditions promoting the releases, are so conducive to enhanced algal growth and biomass accumulation. Firstly, they constitute an addition of P to the water column without any extra water to either dilute the nutrient, or increase the throughput, and the flushing from the loch, of the algae and the P itself. Secondly, the form of P released (SRP) is the most biologically-available. Even sewage effluent is not as pure in this form of P. Nevertheless, out of the 22 years for which reliable data exist since 1968, 11 exhibited P releases resulting in summer concentrations of >50 μg SRP l^{-1}. There is also a suggestion that these releases, accompanied by dense algal blooms, and thus the associated spells of warm, dry weather, occur in runs of 3 or 4 years, rather than at random.

In spite of all the wealth of good-quality, published and on-going research on the assessment and management of eutrophic waters, little is known about whether the classic summer blooms are due primarily to the elevated levels of SRP resulting from sediment fluxes, or to the associated low flushing and other physical conditions. Both contribute, but if flushing rates could be increased at the critical times, planktonic plants at least, would not have time to capitalise on any nutrient resources, and increase so markedly in biomass. However, the chances of engineering what would need to be a seasonally flexible scheme of transferring massive volumes of water, are very small.

The possibility of reducing nutrient inputs to this particularly special loch even further, by means of buffer strips, wet meadows etc., may have to be considered. **Appendix I** refers to these strategies among a number of others, including restorative techniques based on bio-manipulation. In large part, this class of water quality control is considered inappropriate for Loch Leven - not on grounds of cost, but in terms of the risk involved. In some ways, however the (planned) introductions of more fish to Loch Leven (see below) constitutes bio-manipulation. Elsewhere in Europe, including Britain e.g the Norfolk Broadland, experiments have been carried out by removing trout from lakes, with the view to elevating the biomass of organisms such as *Daphnia*, and so increasing the grazing pressure on the phytoplankton. We know rather little about the relationship between the trout populations and *Daphnia* in Loch Leven, but - as outlined in Section 3.1 - there is considerable evidence on the likely impact of the invertebrate on the nature of the algal assemblage.

There are serious financial implications of doing nothing about the fish at Leven. Equally, however, there are unknown risks attached to the introduction of more fish, and as in the case of the rainbow trout exercise planned for Loch Leven, a new genus and species (*Oncorhynchus mykiss* (Walbaum). The concern stems primarily from the fact that there are no data on the current numbers, age structure, food preferences and feeding rates, of the indigenous *Salmo trutta* population. The degree to which the numbers etc., of the introduced stock, compare with the resident population are thus unknown. A situation can be envisaged, where it will be very difficult to explain future shifts in the composition and abundance of the zooplankton and phytoplankton. A reduction in the P loading should lead to 'bottom-up' impacts such as a general lowering of phytoplankton biomass and possibly a change in species composition due to decrease in the P to N loading ratio. Meanwhile, however, the introduction of a new fish species has the potential to effect 'top-down' changes, especially if it increases the grazing pressure on the daphnids. One outcome of this could be an increase in algal biomass, albeit with a shift towards dominance by small algal species, and away from the large bloom-forming types. This would be in keeping with the sequences recorded in the late 1960s, but hopefully, with not anything like the overall biomass levels seen in those years.

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Appendix I.

THE STEMMING OF EUTROPHICATION TRENDS AT LOCH LEVEN

A very wide variety of strategies exist for preventing eutrophication or dealing with its impacts, and the following list is taken from a document prepared by the present author, for the Loch Leven Area Management Advisory Group (LLAMAG), to aid its decisions on possible options for stemming both the chemical and the biological aspects of eutrophication at Loch Leven. The options are grouped on the basis of a number of the limnological issues about the structure and functioning of lake systems considered in this paper. While all of the options are possible, the list indicates those which this author would give priority consideration (***) even if ultimately they are viewed as inappropriate for reasons of cost or effort, and those that involve an unacceptable level of risk (**R**), because, on the basis of present knowledge, they have not proved to be reasonably and consistently, successful elsewhere; in this connection, the individual nature of the way in which a lake 'behaves' is an important consideration. Options not classified as either priority or risky, should still be (and have been) discussed.

A. Options aimed at reducing levels, loadings or availability of nutrients which fuel the growth of planktonic algae

- stemming external supplies

(i) complete diversion (remember effects on water discharges and flushing where large volumes are involved) ***
(ii) phosphorus stripping of e.g. STW effluent ***
(iii) nitrate removal
(iv) 'buffer' strips e.g. reed beds ***
(v) ion exchange beds/columns on inflows
(vi) de-ionising
(vii) dilution with relatively nutrient-poor water ***
(viii) increased flushing (links with vii) ***
(ix) 'bio-discs or -beds
(x) chemical complexing/adsorption with e.g. ferric salts, alum **R**
(xi) remove plant material e.g. overgrowths of aquatic weeds, swathes of filamentous algae and thick blooms ***

- stemming internal supplies e.g. release of phosphorus from the sediments

(i) as (x) above, but including biogenic calcite **R**
(ii) sediment removal **R**
(iii) maintenance of high oxygen levels in the surface sediments
(iv) reduce chironomid densities
(v) maintain higher nitrate levels throughout the summer **R**

B. Options aimed at reducing light availability and thus photosynthetic potential

(i) add 'dark' substances or solutions - such as humic compounds found in peat-stained waters **R**
(ii) cover (partially) with e.g. black table tennis balls **R**
(iii) maintain full circulation of the water over the whole depth of the loch
(iv) deepen the loch (considerably) **R**

C. Options aimed at reducing the time available for phytoplankton growth and biomass accumulation (which could, however, enhance the success of rooted vegetation and attached algae)

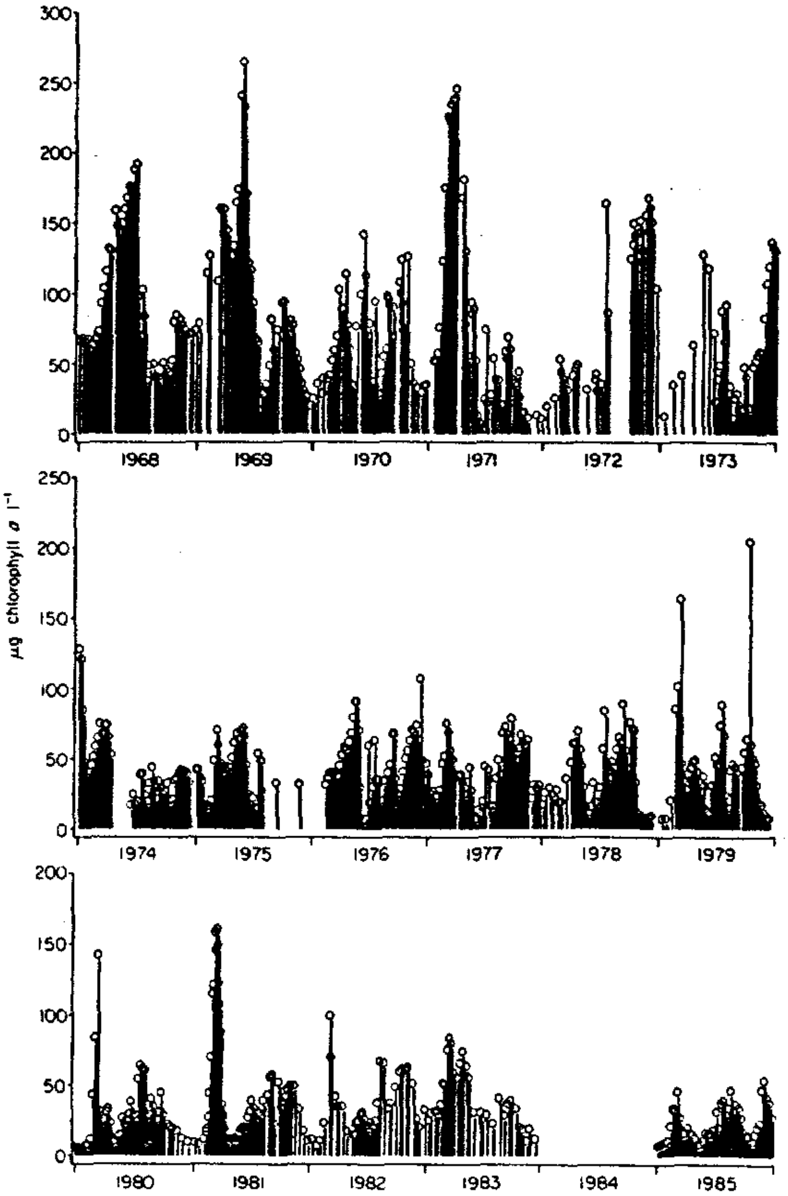
(i) increased flushing - at least on a seasonal basis ***

D. Options aimed at removing/eradicating an existing 'bloom'

(i) increased flushing ***

FIGURES

Figure 1. Fluctuations in total phytoplankton biomass, expressed as chlorophyll *a* concentration, at Loch Leven 1968 to 1985.



μg chlorophyll a l⁻¹
300
250
200
150
100
50
0
1968
1969
1970
1971
1972
1973
250
200
150
100
50
0
1974
1975
1976
1977
1978
1979
200
150
100
50
0
1980
1981
1982
1983
1984
1985

Figure 2. Population densities (plotted on logarithmic scales) of selected algae in the plankton of Loch Leven 1968 to 1976. Ranges in the y axes vary with species.

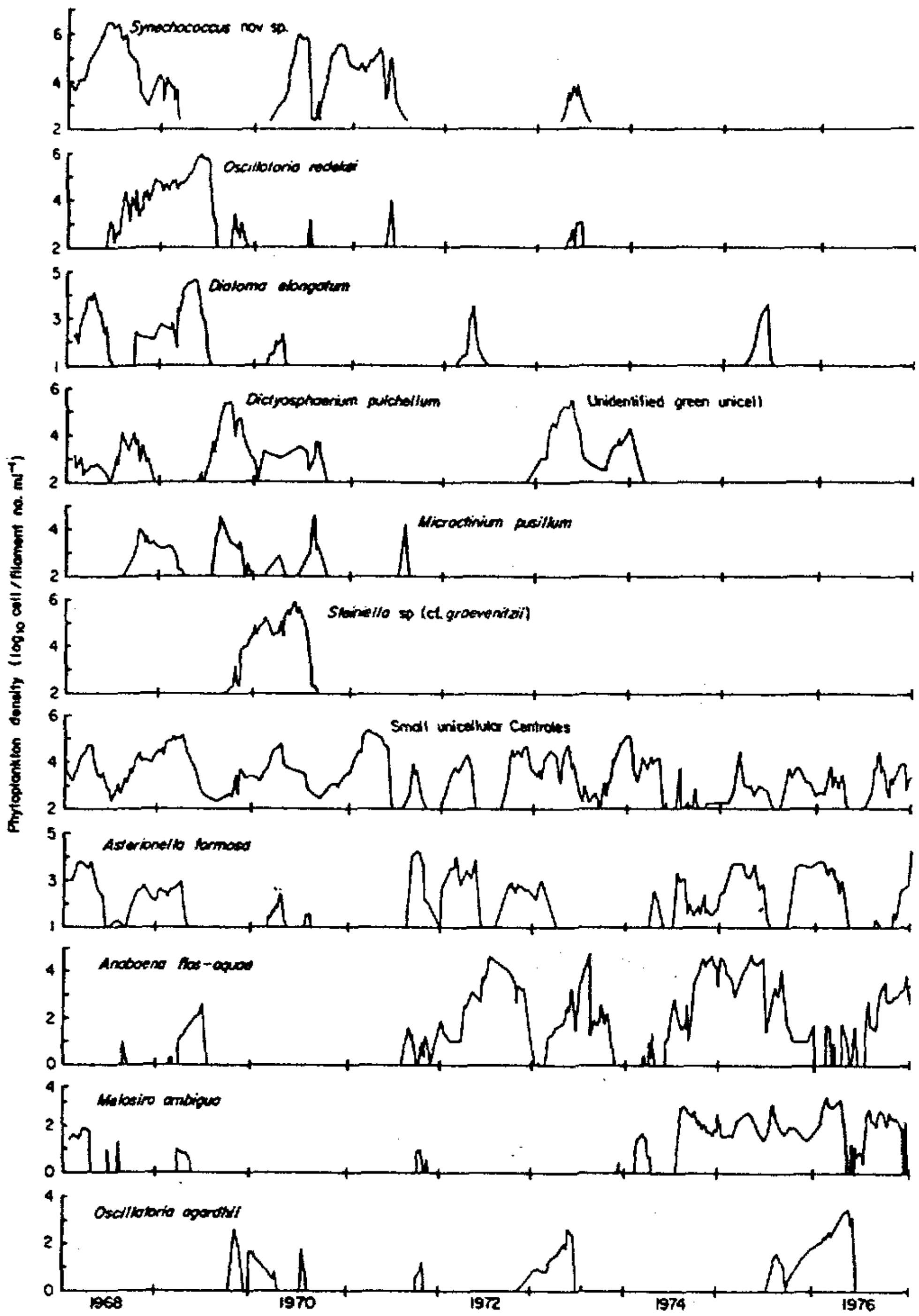
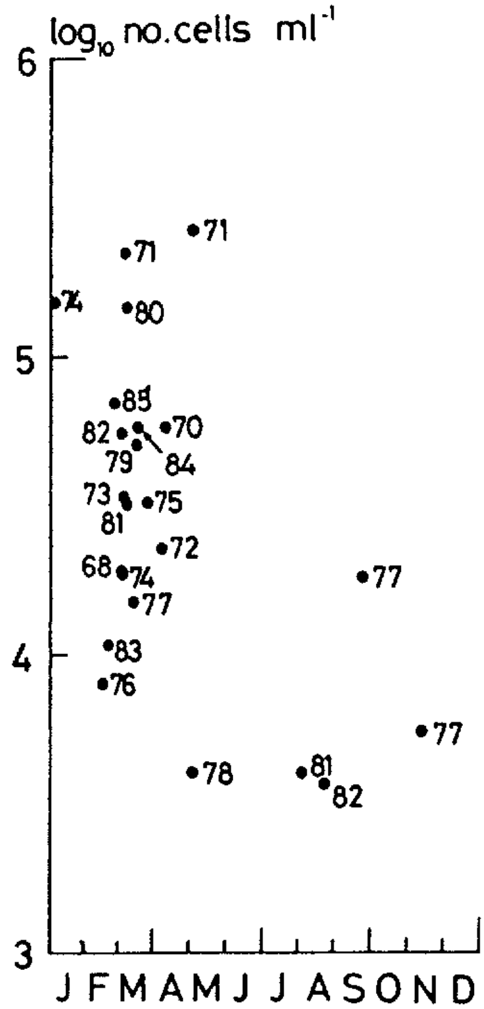

Synechococcus nov sp.
Oscillatoria redekei
Diatoma elongatum
Dictyosphaerium pulchellum
Unidentified green unicell
Microactinium pusillum
Steinella sp (cf. graevenitzii)
Small unicellular Centrales
Asterionella formosa
Anabaena flos-aquae
Melosira ambigua
Oscillatoria agardhii
Phytoplankton density (log10 cell / filament no. ml-1)
1968
1970
1972
1974
1976

Figure 3. (Upper graph). Unicellular centric diatom maxima (vertical axis - logarithmic scale) and their time of occurrence (horizontal axis) in Loch Leven 1968 to 1985.

Figure 4. (Lower graph). Mean volume of algae dominating the phytoplankton of Loch Leven 1968 to 1976, to illustrate the contrast in the seasonality of algal size, especially between the years prior to 1971 and those following it.


log10 no.cells ml-1
6
5
4
3
71
71
74
80
85
82
70
79
84
73
75
81
72
68
74
77
77
83
76
77
78
81
82
J F M A M J J A S O N D

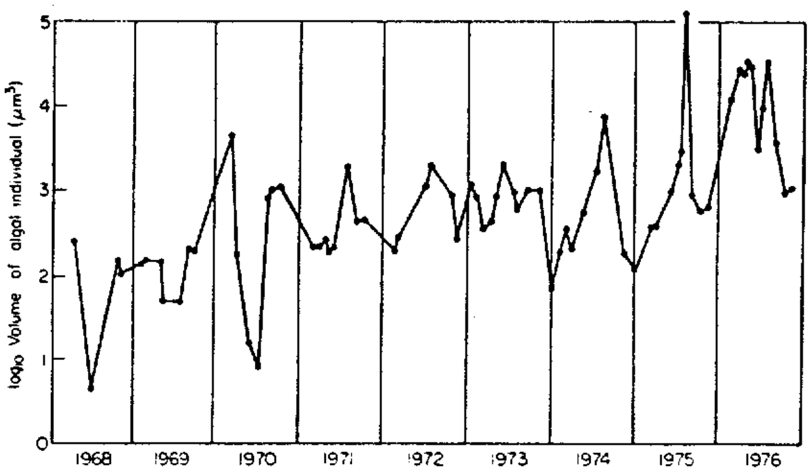
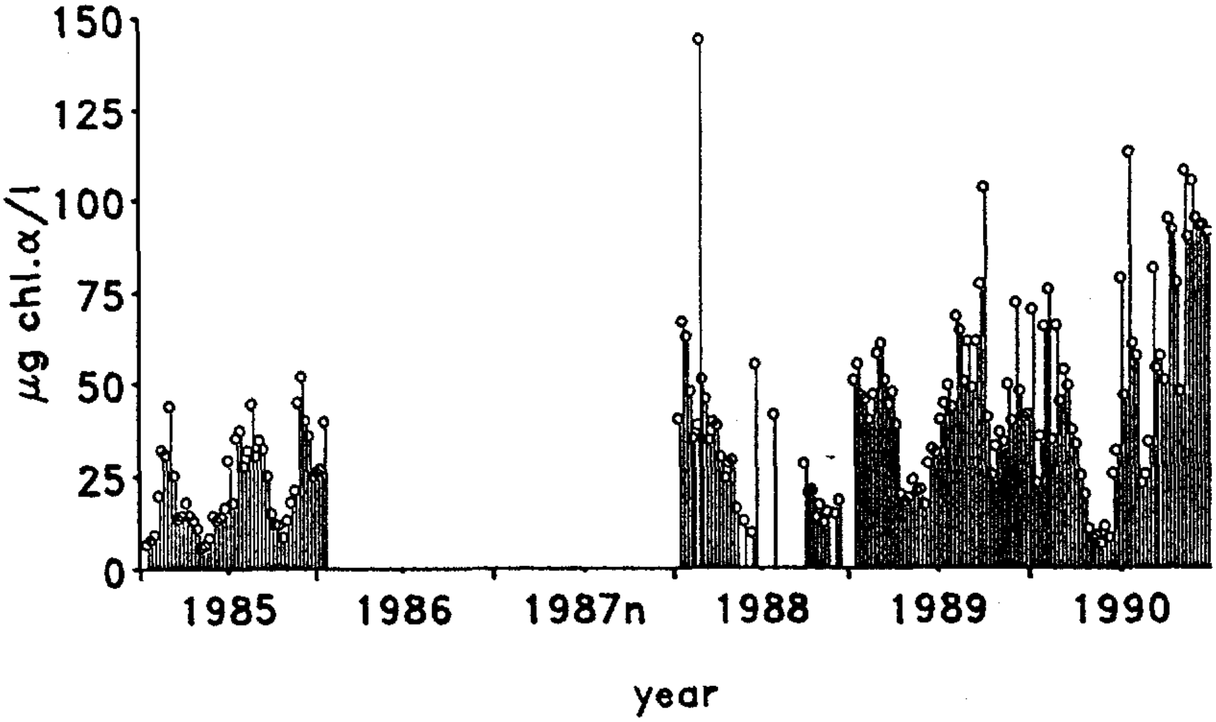

log10 Volume of algal individual (μm3)
0
1
2
3
4
5
1968
1969
1970
1971
1972
1973
1974
1975
1976

Figure 5. As Figure 1 for 1985 to 1990 (upper graph - 'n' denotes no data for 1987), and 1992 (lower graph).

Chlorophyll α nr. outflow of L. Leven
1985 to 1990 – mill effluent ceased late 1989



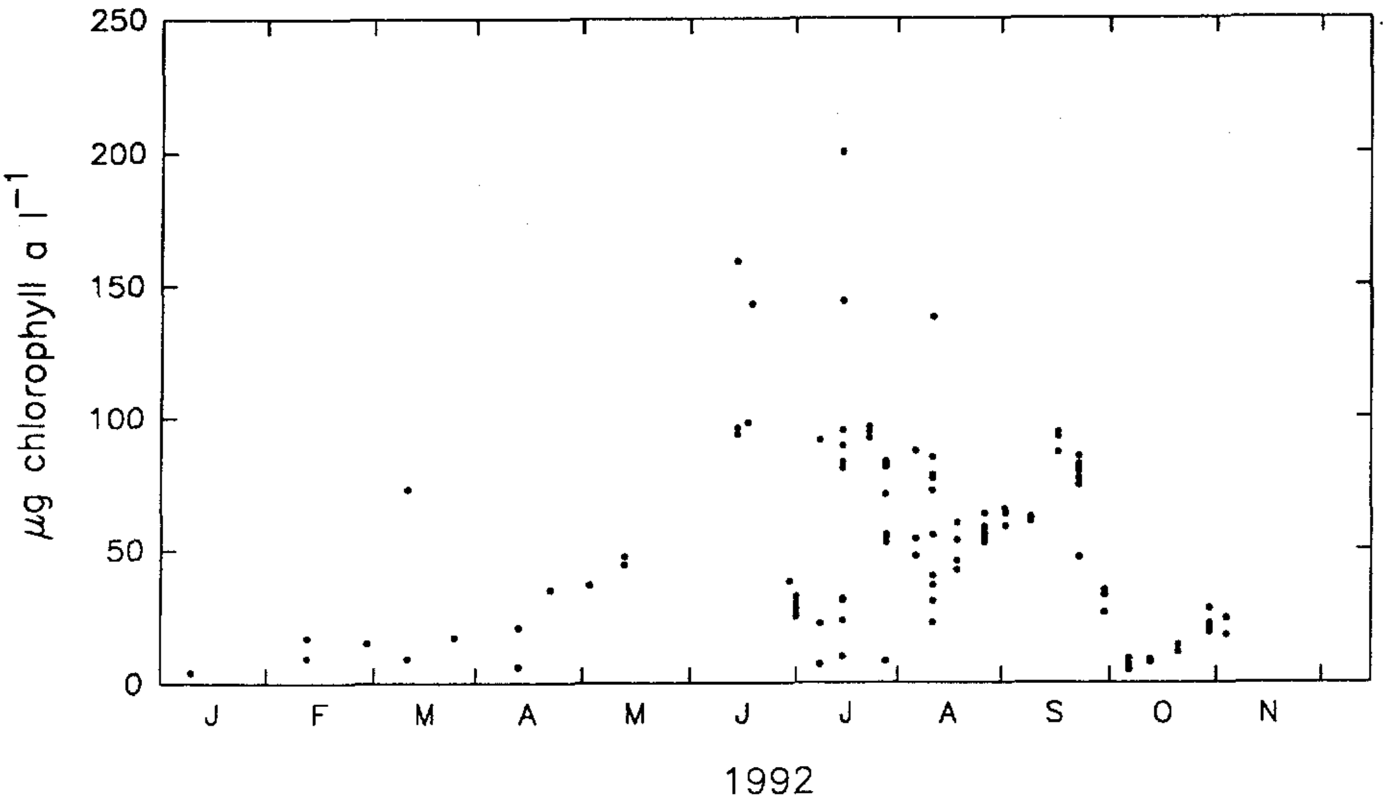
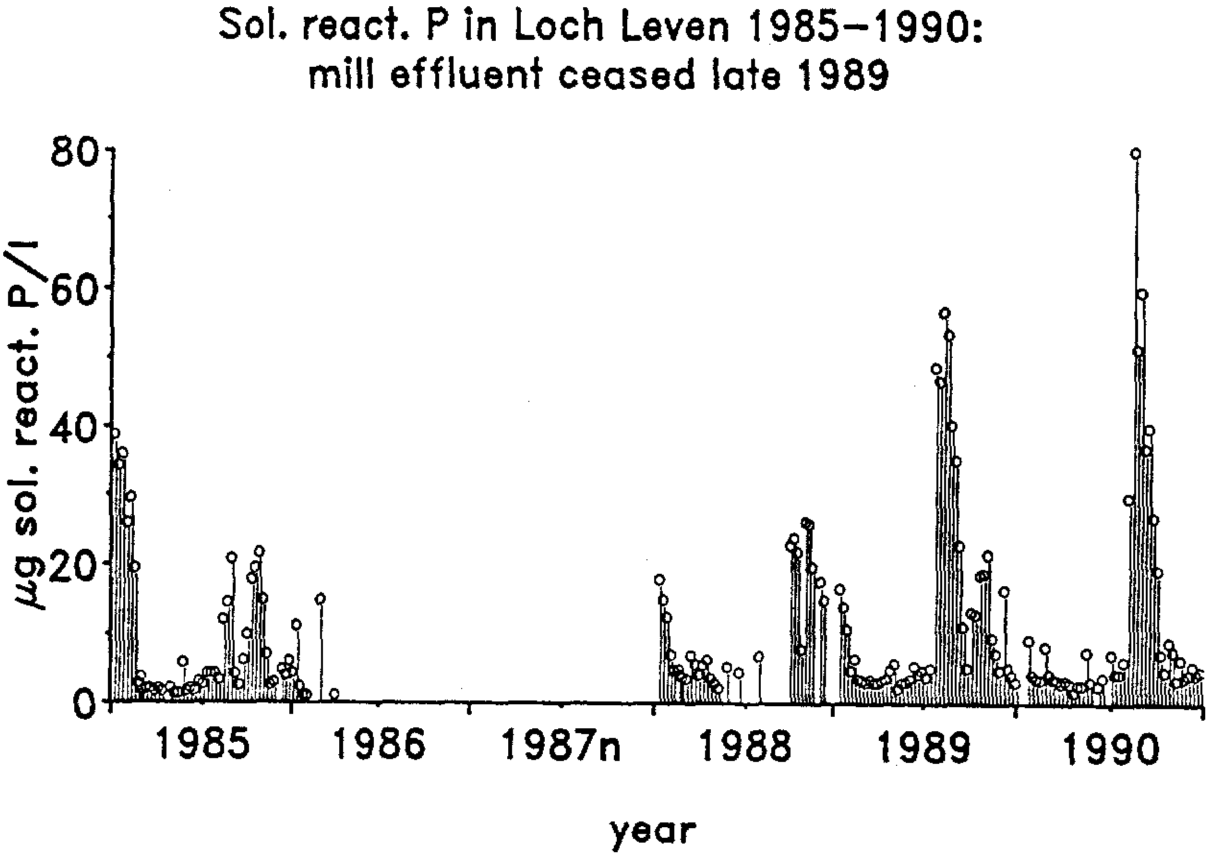


Figure 6. Fluctuations in the levels of soluble reactive phosphorus in Loch Leven; note the massive concentrations due to sediment release in the summers of 1989, 1990 (upper graph) and 1992 (lower graph).


Sol. react. P in Loch Leven 1985-1990:
mill effluent ceased late 1989
µg sol. react. P/l
0
20
40
60
80
1985
1986
1987n
1988
1989
1990
year

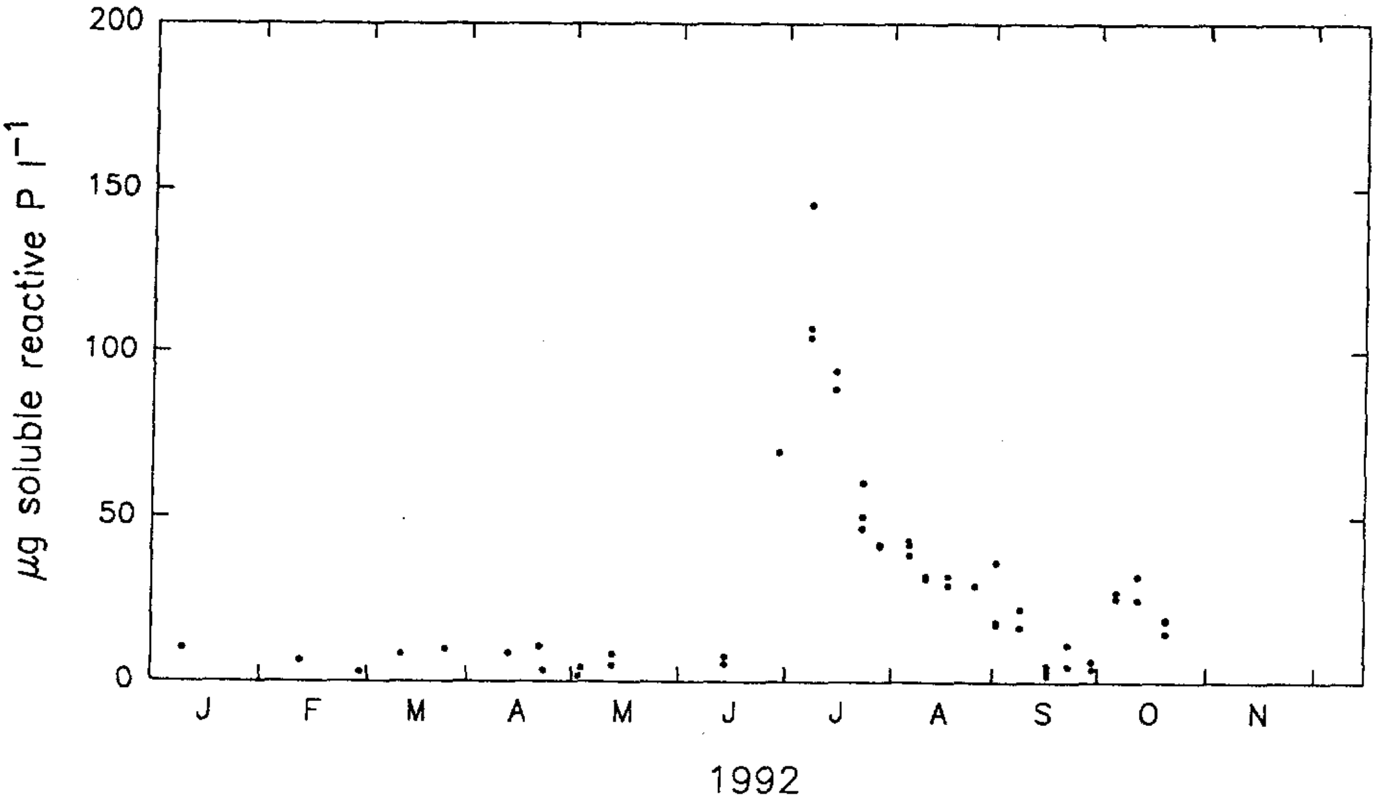
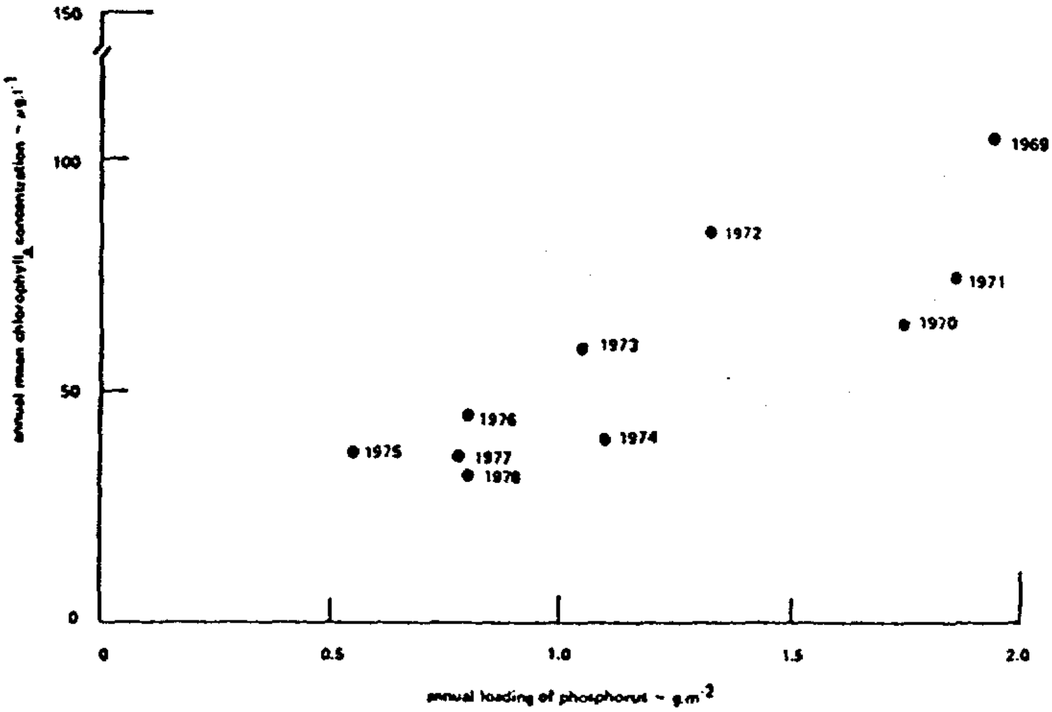

µg soluble reactive P l⁻¹
0
50
100
150
200
J
F
M
A
M
J
J
A
S
O
N
1992

Figure 7. The relationship between the annual external phosphorus loadings to, and the annual mean chlorophyll concentrations in, Loch Leven 1969 to 1978 (see text).



150
100
50
0
0
0.5
1.0
1.5
2.0
annual mean chlorophyll a concentration ~ µg.l⁻¹
annual loading of phosphorus ~ g.m⁻²
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978

Figure 8. (a, top left). Variation in the loading of total phosphorus to Loch Leven assuming a linear relationship between runoff P inputs and flushing rate, over the long-term (1936-1985) spectrum of flushing rates, and for a number of phosphorus reduction situations: A, assuming no change from the situation as measured in 1985; B, removal of 80% of the loading derived from the Kinross North and South sewage treatment works (STW); C, removal of 50% of the loading contributed by the woollen mill; D, removal of 80% of the loading derived from the woollen mill; E, B + C; F, B + D; and G, assuming eradication of all mill and STW P; the chances of occurrence of a flushing rate equal to, or less than that shown on the x axis, are also indicated.

(b, top right). As (a) for soluble reactive P.

(c, bottom left). As (a) for the annual mean in-lake P concentration as predicted by the two models referred to in the text.

(d, bottom right). As (c) for the annual mean in-lake chlorophyll *a* concentration used as an index of total phytoplankton biomass, and for the OECD model referred to in the text.

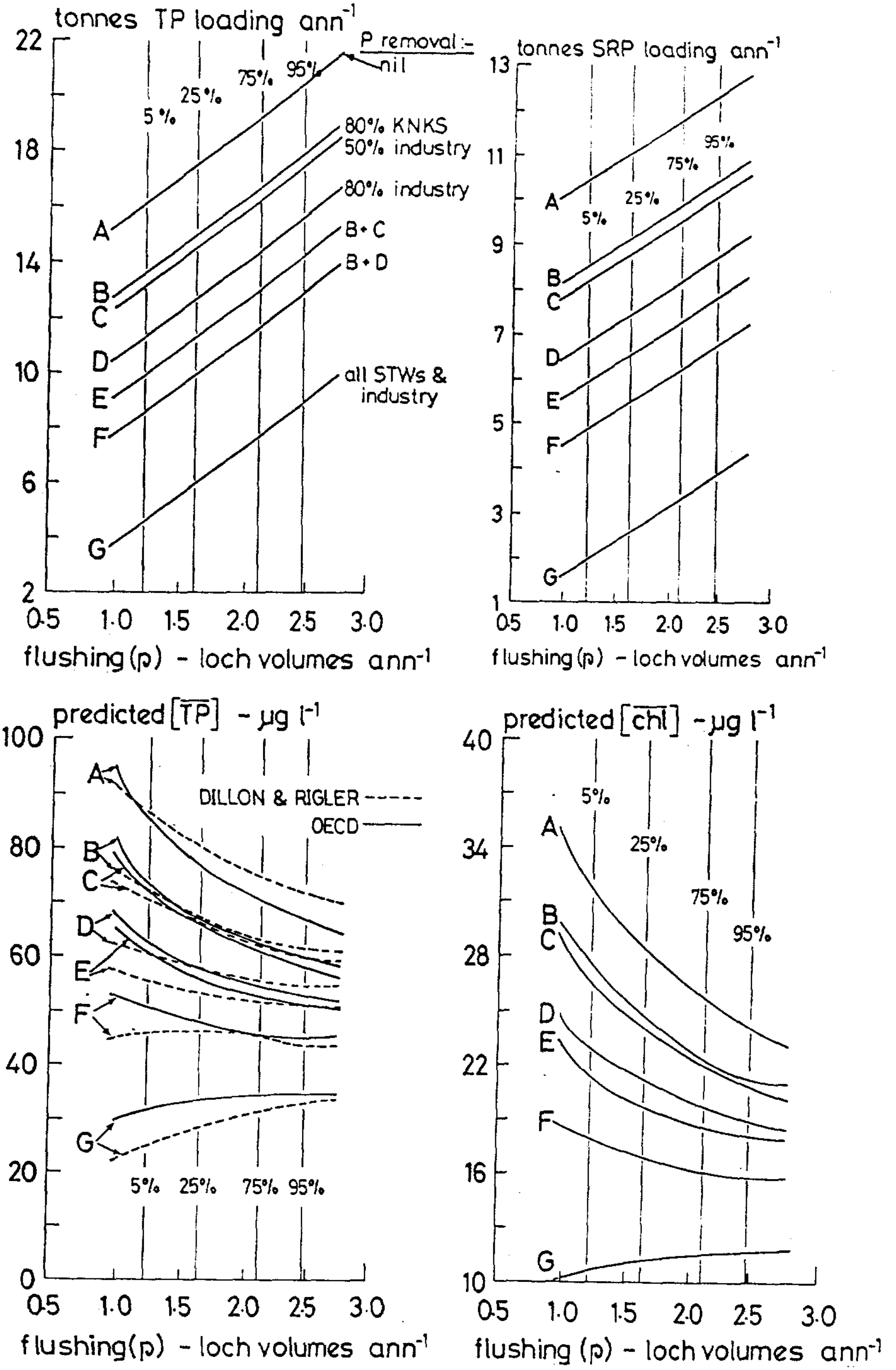
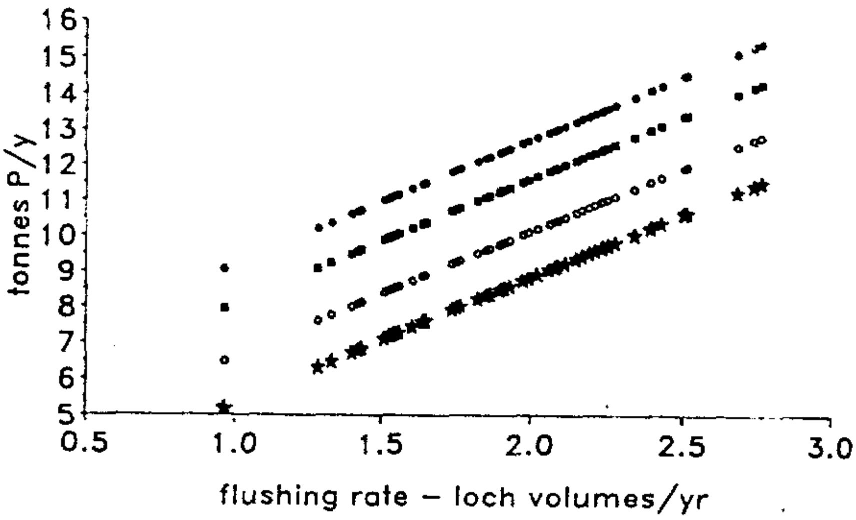
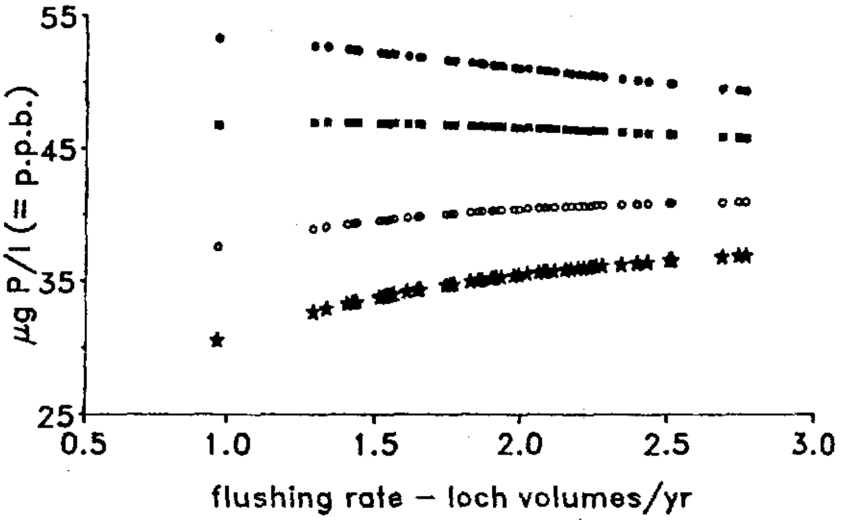

tonnes TP loading ann⁻¹
P removal :-
nil
5%
25%
75%
95%
80% KNKS
50% industry
80% industry
B + C
B + D
all STWs & industry
A
B
C
D
E
F
G
22
18
14
10
6
2
0.5
1.0
1.5
2.0
2.5
3.0
flushing (p) - loch volumes ann⁻¹
tonnes SRP loading ann⁻¹
13
11
9
7
5
3
1
flushing (p) - loch volumes ann⁻¹
predicted [TP] - μg l⁻¹
DILLON & RIGLER -----
OECD ——
100
80
60
40
20
0
flushing (p) - loch volumes ann⁻¹
predicted [chl] - μg l⁻¹
40
34
28
22
16
10
flushing (p) - loch volumes ann⁻¹

Figure 9. As Figure 8, for predicted total P loadings (upper graph), annual mean, in-lake total P concentrations (middle graph), and annual mean, in-lake chlorophyll levels (lower graph), assuming complete eradication of mill P, and the following P reduction situations: no removal of STW P, 80% reduction of the P from the combined Kinross North and South STW, 80% reduction of the P from the Milnathort STW, and 80% reduction of the P from both these works.

Predicted TP loadings: pre-reduction of STW P (•)
& after 80% reduction at Kinross (°), Milnathort (▪) or both (★).



Predicted TP concentrations: pre-reduction of STW P (•)
& after 80% reduction at Kinross (°), Milnathort (▪) or both (★).



Predicted pigment levels: pre-reduction of STW P (•)
& after 80% reduction at Kinross (°), Milnathort (▪) or both (★).

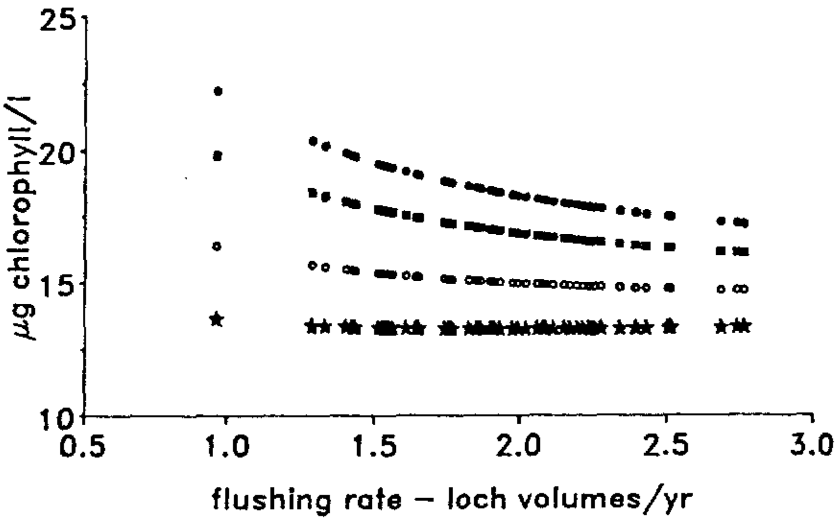
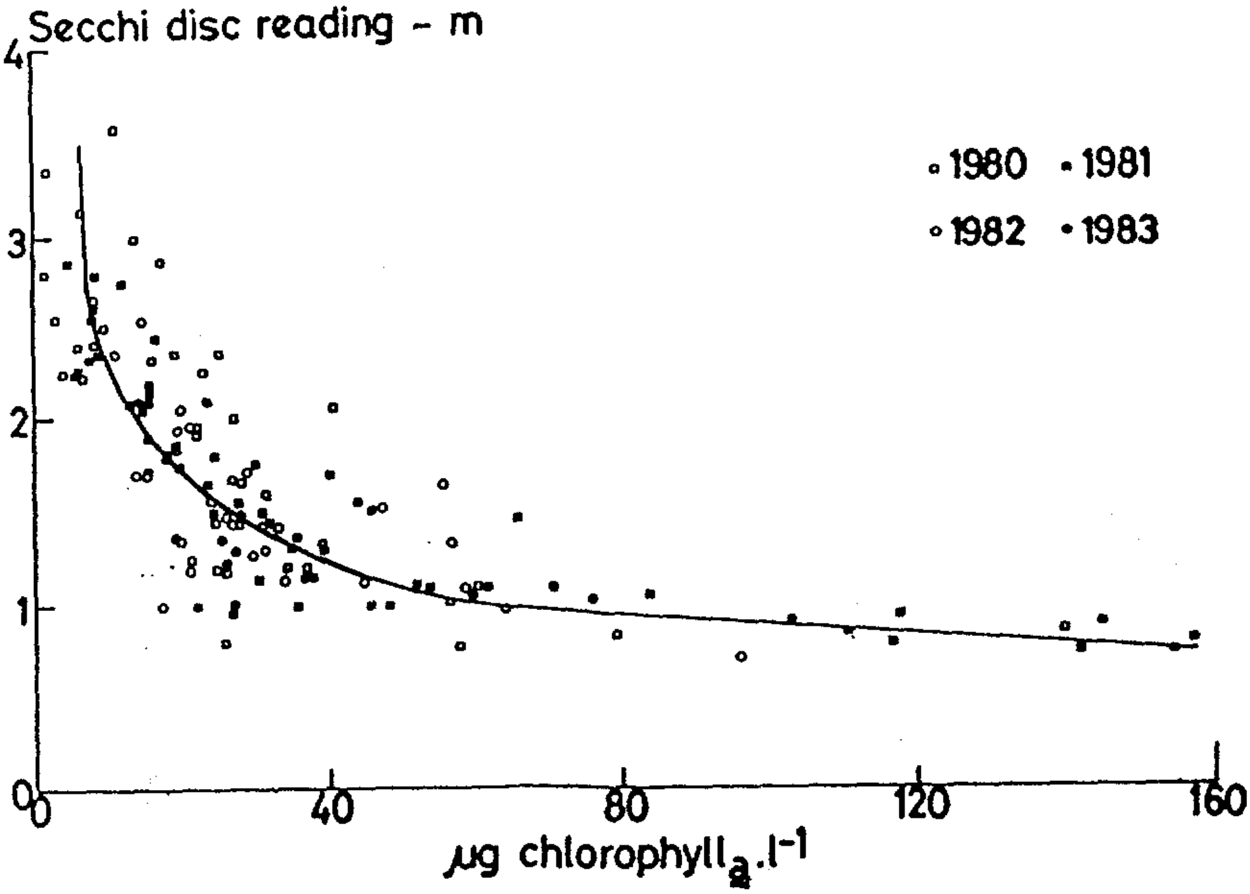
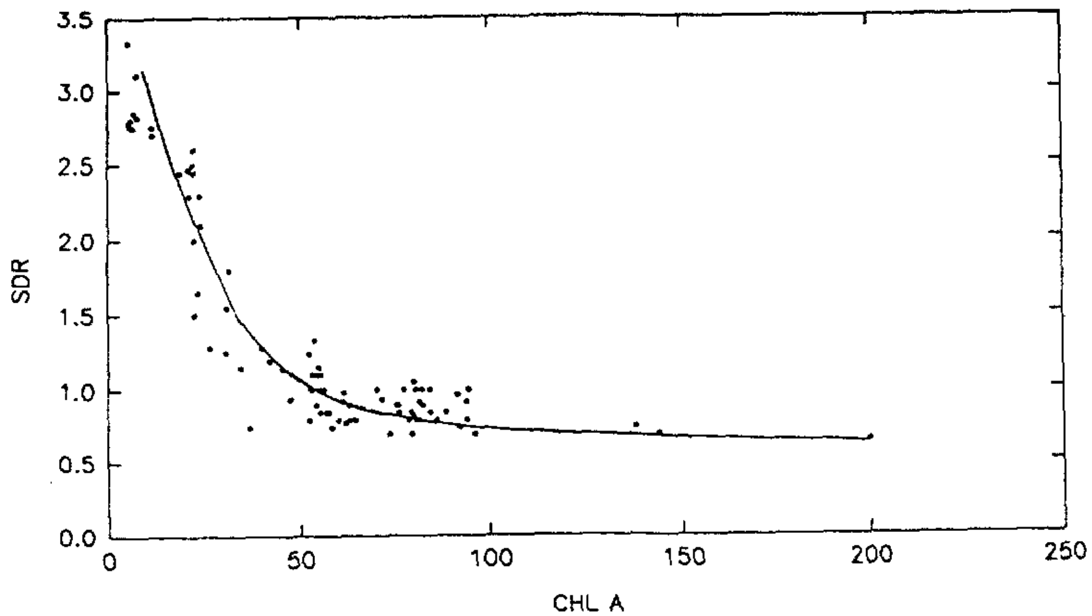


Figure 10. The relationship between water clarity measured as Secchi disc transparency, and phytoplankton abundance measured as chlorophyll *a* concentration: upper graph, data for 1980 to 1983, lower graph, data for all sampling stations for the period July to December 1992.


Secchi disc reading - m
4
3
2
1
0
0
40
80
120
160
µg chlorophyll a .l-1
1980
1981
1982
1983


3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SDR
0
50
100
150
200
250
CHL A