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Loch Eye, Easter Ross - a Case Study in Eutrophication

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Summary

Concerns over deteriorating macrophyte communities and the declining trout fishery which prompted this study, parallel trends elsewhere, and eutrophication is identified as the common causal factor. The aims of the work were: to assess the nutrient status of the loch; to provide baseline data for monitoring water quality, wildfowl (especially goose) numbers, and land-use; and to consider, if relevant, methods for ameliorating the effects of nutrient enrichment. It was thus necessary to try to determine the loadings from different sources; in the case of Loch Eye these include agricultural runoff and over-wintering wildfowl in particular. The impact of the eutrophication is determined not only by the concentrations and loads associated with different inputs, but their seasonality and the nature of the supplies as regards the nutrients involved and their ratios.

The influence of increased nutrient loadings on plant-animal interactions is discussed, to emphasise the inexorable links between nutrient supply and the biomass, productivity and species composition of organisms at all trophic levels. The importance of physical factors, particularly the morphometry of the lake basin, and the flushing rate (p), is highlighted as a major determinant of the extent to which different types of plants will capitalise on the enrichment. Indeed, the outcome of the competition between phytoplankton and the communities of attached/rooted plants, determines very much our perception of the effects of lake eutrophication.

The site-specific nature of the various processes linking aquatic organisms to physical and chemical features, indicates the importance of case studies such as the NCC-funded programmes on Loch Eye and other eutrophic sites in the UK. Such works also illustrate the detail necessary to establish an

appropriate, scientifically-based, management strategy for a particular water body. What is more, the investigations feature prominently in global research into the nutrient enrichment problem, because they deal with loadings, and not just in-lake responses to the inputs. The value of the study is enhanced by the shallowness of Loch Eye which is *ca* 1.2 m mean depth, the nature of the eutrophication which appears to be mainly agricultural with little urbanisation, and the international conservation status of the loch - it being a Ramsar Site and a Special Protection Area for Wild Birds.

The present programme focused primarily on a 3-weekly sampling schedule to assess nutrient levels in three main inflows together draining *ca* 88% of the total catchment area of 11.85 km^2. Fieldwork began in September 1986 and finished in June 1988. The nutrients of interest are nitrate-nitrogen (N), various forms of phosphorus (P), and dissolved silica (SiO_2). Loadings of these nutrients were assessed for the period August 1987 to June 1988 and adjusted to give annual figures. Phosphorus budgets were erected, before and after taking account of the possible influences of geese roosting on the loch, and of the release of this nutrient from the sediments. The phytoplankton was investigated as a major indicator of trophic status.

To gain an initial idea of the extent of the impact of the catchment on the loch, basic information on land-use, the catchment-to-loch area ratio, loch volume and rainfall was analysed. As agriculture dominates the catchment, it must be significant in the eutrophication of the loch. However, the drainage area is only *ca* 6 times the area of the loch, although, the loch's shallowness would accentuate any enriching effect. A further qualification is that runoff from the land constitutes a loading controlled entirely by net rainfall but this, in north-east Scotland, is low; records for 1986 to 1988, from two Meteorological Office stations located within a few kilometres of the Eye catchment, suggest a mean annual precipitation of 660 mm per annum, but in this area evapo-transpiration is high at *ca* 455 mm; so, a throughput of water equivalent to approximately only 1.4 loch volumes per annum is predicted. The transport of nutrients is thus unlikely to be outstandingly high, although phytoplankton would have ample time to capitalise on these supplies.

The discharges of the feeder streams and the outflow were monitored to erect a water balance and establish more clearly the flushing rate of the loch. 3-weekly records of stream levels (together with flow measurements made by the Highland River Purification Board) indicate mean discharges of 75 l s^{-1} for the Garrick draining 618 ha, 25 l s^{-1} for the Erracht (350 ha) and 15 l s^{-1} for the Loinnbuie (74 ha). Even these records show the streams to be very 'flashy' with flows varying over two or three orders of magnitude, i.e. from less than 1.0 to 60 l s^{-1} in the Loinnbuie to 150 l s^{-1} in the Erracht and to *ca* 450 l s^{-1} in the Garrick. There were considerable problems with the measurement of low flows. A comparison of the mean discharges based on the 3-weekly sampling schedule, and those derived from more frequent sampling over various periods August 1987 to June 1988, suggested that - assuming the more frequent records are the better description of the outflow regimes - the Loinnbuie discharge was being under-estimated by approximately 8%, the Erracht by 22%, and the Garrick by 28%. By contrast, the outflow exhibited a much smoother hydrograph (illustrating the dampening effect of the loch on short-term fluctuations in flow) and the mean discharge there was estimated to be approximately 124 l s^{-1} from both the 3-weekly records and the measurements taken at more or less daily intervals. The flushing rates derived from the feeder stream discharges are 1.96 loch volumes y^{-1} using the unadjusted, instantaneous flow values, and 2.5 loch volumes y^{-1} assuming they were under-estimating flows as just described; these values are both higher than expected from the earlier considerations, and they exceed the estimates based on rainfall ($p = 1.76$) and the outflow measurements (1.67). As a compromise, a value of 1.7 was used in later considerations about nutrient balances and budgets.

3-weekly sampling at the loch edge, as well as the feeder streams and the outlet, plus 9-weekly measurements in open water in the loch, showed that temperature varied less in the streams than in the loch, i.e. from 1°C to 14°C as against *ca* 0°C to 20°C. Secchi disc readings of 0.8 to 2.5 m, show that much of the bottom of Loch Eye is in the lighted zone for most of the year. Measurements of the attenuation of light in different bands of the spectrum supported this finding; at 1 m depth, the intensity of blue light is commonly reduced to 1% of surface values, but over most of the red and green portions of the spectrum readings of at least 25% of the surface values were usually obtained at this depth. The euphotic depth, which is the level at which an algal cell is likely to be able to balance respiratory losses by photosynthetic gains and so at least sustain growth, varied from theoretical values of 2.0 to 6.5 m. Conductivity of the loch water exhibited little variation with values of 224 to 255 µS cm^{-1} (standardised to 25°C), reflecting proximity to the coast.

The Erracht reflects the predominance of agriculture in its drainage area by being rich in nitrate with 2.5 to 9.5 mg N l^{-1} compared to 0.1 to 6.0 mg l s^{-1} in the Loinnbuie and 0.1 to 2.4 mg l^{-1} in the Garrick. The concentrations are generally highest in winter and lowest in summer. This seasonal pattern is particularly marked in the loch, and the concentrations are similar to those recorded in the Loinnbuie and the Garrick. Minimum levels of *ca* 0.1 mg N l^{-1} in summer are characteristic of many Temperate Zone waters, and illustrate that even in eutrophic situations, nitrogen can be reduced to limiting levels on occasions. It is at this time, that those species of blue-green algae ("cyanobacteria") which can fix atmospheric nitrogen have an advantage over other species.

By contrast to the situation with nitrate, phosphorus concentrations show irregular fluctuations. Moreover, the Erracht is the more dilute in terms of total phosphorus (TP); it rarely exceeds 30 µg P l^{-1} there, whereas many concentrations of 50 µg l^{-1} and more, were recorded in the other streams. Overall, TP in the streams came within the range 10 to 150 µg l^{-1}. Approximately 75% to 85% of the P in these waters is in soluble form. Most of the TP values in the loch came within the range 30 to 40 µg l^{-1}, although a series of maxima of 50 to 60 µg l^{-1} were recorded. There are as many maxima due to increases in the particulate component (PP) as to rises in the soluble fractions (TSP) and, on average, the TP consists of roughly equal amounts of both fractions. There is some evidence of an increase in the concentration of soluble reactive P (SRP - the fraction most immediately available for algal growth), due to sediment release in summer 1987. The concentrations of SRP in the loch are otherwise very moderate - 2 to 10 µg l^{-1} - but the feeder waters also rarely exhibit levels of > 20 $\mu g\ l^{-1}$. Silica concentrations are characteristically less variable than those of N and P, with the range 3 to 10 mg l^{-1} encompassing all the stream values; the mean levels were 4.7 in the Garrick, 6.4 in the Loinnbuie and 7.4 in the Erracht. Another difference between SiO_2 and the other nutrients is in the sharp contrast between stream and loch concentrations; the maximum in the loch was only 2.0 mg l^{-1}. Possible biotic and abiotic causes of the depletion of SiO_2 there (to <0.1 mg l^{-1} on occasions) are discussed.

From the products of flow and nutrient concentration, the estimated total loadings (for the 12-month period August 1987 to July 1988) are 8.6 t nitrate-N, 0.23 t TP (about half of which consists of SRP), and 21 t SiO_2. The catchment N losses are 15 kg ha^{-1} 15 from the Loinnbuie drainage area, 12 kg ha^{-1} from the Erracht and 3 kg ha^{-1} from the Garrick, and overlap considerably with values published for runoff in other well-studied eutrophic catchments. Phosphorus losses (also kg ha^{-1}) are very similar to published values and moreover, virtually the same for all three streams, i.e. 0.18 to 0.22 TP, 0.10 to 0.12 SRP, and all 0.04 PP. The SiO_2 export coefficients vary somewhat more, from 30 kg ha^{-1} *via* the Loinnbuie, 16 from the Erracht and 20 from the Garrick, but the range also overlaps considerably with the (few) values published for areas of generally similar land-use and geology elsewhere. The total burden of phosphorus to the loch is equivalent to a specific areal loading of 0.12 g P m^{-2} of loch surface (1.95 k m^2) - a rate which is less than one-twelfth of that estimated for Loch Leven.

Ratios of loadings to flushing-corrected standing stocks of the nutrients in the loch (using a flushing rate of 1.7 y^{-1}), indicate that SiO_2 is altered to the greatest extent - ratio 20.4:1 - followed by SRP with a value of 8.5:1, and N with 2.5:1. The ratio for PP is approximately 1:1 (ie. the mean stream concentration of 13 $\mu g\ l^{-1}$ is very similar to the average level of 14 $\mu g\ l^{-1}$ recorded in the loch); this reflects the net result of losses of PP in stream-borne detritus to the loch sediments, and gains due to incorporation of P into phytoplankton.

A first attempt at a P budget assumed that the external loading consisted of no more than the stream-borne inputs, plus the contribution in rain falling on the loch surface. A comparison of this loading (227 kg TP y^{-1}) with the amount exported *via* the outflow (215 kg y^{-1}) suggests a net retention of only 5%. However, eutrophication models, and findings from studies on other lochs, indicate that retentions of *ca* 80% are more likely. The discrepancy between the observed and predicted figures amount to a shortfall of many hundreds of kilogrammes of P.

Of the possible sources other than the feeder streams, rain falling directly on the loch surface is estimated to supply *ca* 26 kg. There appear to be no seepages that might have been previously overlooked. It is possible that recycling of P *via* release of SRP from the sediments is important, and the P and SiO_2 status of the deposits is considered in a later section of the report; however, an examination of the water column data, leads to the conclusion that anaerobic release contributed very little, i.e. 25 kg during summer 1987. Nothing is known about aerobic release of material when the sediment surface is disturbed by wind-induced water mixing. There was, however, a possibility of assessing the likely impact of over-wintering geese which roost on the loch and, following work on the Loch of Strathbeg and Loch Leven, these birds were identified as an important focus at the inception of the project. Calculations using published rates of excretion by geese, indicate that more than 300 kg P could be introduced annually by the Loch Eye populations. This is equivalent to *ca* one-and-a-half times the runoff loading, and goes a considerable way to accounting for the discrepancy in the earlier budget.

The potential impact of the sediments on loch water P and SiO_2 levels is discussed. Although concentrations in the interstitial waters reach 100 μg SRP l^{-1} and 10 mg SiO_2 l^{-1}, instantaneous releases of the pools of these nutrients in the uppermost centimetre of sediments would have a negligible effect. By contrast, while P constitutes only 0.1 to 0.2% of sediment dry weight, fluxes of P *via* desorption from particles, could be very important. Dissolution of diatom remains could also affect the overlying concentrations of SiO_2 to a considerable degree.

Phytoplankton abundance is very moderate in comparison to the levels found in many other eutrophic sites. The average chlorophyll concentration over the whole 2½-year period was 17 μg l^{-1} (*cf* 13 μg l^{-1} for the 12-month period covered by the nutrient loading determinations). Fluctuations in abundance were very erratic, in keeping with the highly changeable physical and chemical environment. Population densities were commonly between 10^4 and 10^5 individuals ml^{-1}; the high numbers are not inconsistent with the relatively low pigment concentrations, as many of the species are small and/or low in chlorophyll content, e.g. chrysoflagellates and the cyanobacteria *Oscillatoria* (= *Planktothrix*), and *Aphanothece*. The mean individual algal size is < 10 μm on many occasions, but the value increases in summer, probably as a result of herbivorous zooplankton cropping the smaller elements. More than 200 species of algae have been recorded from the plankton, although many of these are apparently also associated with the bottom sediments. Arrays of 50 randomly-chosen individuals, often contained > 20 species. In addition, the species composition changes rapidly and some of these changes are very marked. For example, of the 24 species recorded in the sampling array in July 1987, only 17 were also recorded in September; by November, only 18 of the species recorded in July and/or September were detected.

A concluding discussion of the ecology of the loch in relation to its nutrient status, suggests that while the algal plankton appears not to be extremely P-limited, competition for nutrients from the other plant communities must be considerable. Yet, while the burden of *ca* 0.5 t TP is only one-fortieth of the loading to Loch Leven, the flushing-corrected volume of Loch Eye is one twenty-fifth of that of Loch Leven.

A Vollenweider and OECD model predicting the annual chlorophyll of a lake from the flow-weighted TP concentration in the supplies, is used to explore whether the Loch Eye chlorophyll results make sense in relation to the P budget. If the loading of 227 kg y^{-1} is used, a chlorophyll value of 9.2 μg l^{-1} is predicted; where the loading of 553 kg (i.e. from runoff and geese) is used, a figure of 20.4 μg chlorophyll l^{-1} results. This also suggests that substantially more P is reaching the loch than is entering in the streams. But perhaps the excess is nearer the middle of the range 227 to 553 than actually 553 as indicated above. However, two factors are important in interpreting these findings and model-based predictions. One concerns the phytoplankton, which because it is a minor component of the plant biomass competing for phosphorus in Loch Eye, would probably not be as abundant as the models suggest. An alternative possibility is that only part of the goose-derived P is readily available to phytoplankton. Then, in effect, the 553 kg TP and the chlorophyll concentration of 20.4 $\mu g\ l^{-1}$ derived from this loading, could both be considered as overestimates. The fate of nutrients from geese is unknown. However, the droppings could contribute to the particulate P measured in the loch, and at the outflow. They certainly make up part of the sediment. Indeed, it is possible that they influence macrophyte performance, and this is why one of the areas recommended for further attention involves goose exclosure experiments.

Phytoplankton composition reflects the environment in a number of ways. As an example, *Aphanothece* which is prominent here, is also common in the Norfolk Broads where its success is associated with periods of low flushing; also, the cells are embedded in a gelatinous matrix which render the species less susceptible than many algae to losses by grazing and sinking.

If it is reasonable to assume that geese are a major factor in the ecology of Loch Eye, a major source of its eutrophication is very interesting; inputs from birds constitute a diffuse source of nutrients but they resemble point-sources in being largely flushing-independent.

Changes in the population density and roosting schedules of geese, are two of the items recommended for future work to improve existing aspects of the research and management of Loch Eye. Other areas warranting further attention are the water balance and the bio-availability and growth-promoting potential of the sediments. In addition, means should be sought for maintaining observations on macrophyte performance, monitoring the activities of the Loch Eye Angling Association, and obtaining a detailed description of land-use in the catchment. New studies are also recommended, and in the main, these are of the direct, practical management or management research-oriented types; they include biological removal of nutrients from inflows, and exclosure experiments to assess the impact of goose droppings on sediment chemistry. In this respect, Loch Eye offers a very exciting opportunity for conservation-directed scientific research on an appropriate field scale. Production studies on organisms at all trophic levels would be of great scientific value, but first attention should be paid to the algal populations associated with surfaces of different types, i.e. epiphytes on higher plants, epipsammic forms on sand, and epipelic species on muds, are all liable to be extremely important in the bio-energetics and production ecology of this loch.

INTRODUCTION

1.1 Background to the study and its aims

The Loch Eye study was initiated by concerns expressed over reductions in the species diversity and general cover of macrophyte communities in the loch (Charter 1988). There is also concern over reduced angling returns, with fishing having been a major recreational amenity and revenue-earner in earlier years (Loch Eye Angling Improvement Association - Mr A Prickett, personal communication). The trout population appears to have decreased over the last few decades in spite of various stocking programmes. Generally similar changes in aquatic plant and fish populations are reported for other waters world-wide, with the developments at the Norfolk Broads, England (Moss 1979, Moss and Leah, 1980) and Loch Leven, Kinross, Scotland (Bailey-Watts and Maitland 1984) being amongst the best-documented for the UK. Accelerated nutrient enrichment (eutrophication) appears to be a feature of all these waters.

Following the above considerations, the Nature Conservancy Council commissioned a study to identify the causes of deterioration in the 'condition' of the loch and a programme of work was set up by the Freshwater Ecology Group of the Institute of Terrestrial Ecology (now the Edinburgh Laboratory of the Institute of Freshwater Ecology). Field sampling concentrated on catchment characteristics, physical features and chemical - especially nutrient - aspects of the loch water, the inflows and outflow, and the abundance and species composition of the phytoplankton. Water and plankton were collected at approximately 3-week intervals from September 1986 to June 1988 and the P content of open water sediment was analysed. These investigations aimed to:-

- assess the nutrient status of the loch;
- provide base-line data and a set of methods for monitoring water quality, bird numbers and land use in the catchment;
- consider, if relevant, means of ameliorating the nutrient enrichment of the loch.

Before embarking on the results of this work, some general concepts regarding eutrophication and the links between macrophytes, algae and fish are discussed. There is already an enormous literature on eutrophication, so questions about the importance of research on these issues and on the need for a specific programme on Loch Eye are also addressed.

1.2 General concepts: eutrophication and aquatic plant-animal interactions

Much of the focus on freshwater conservation concerns macrophyte communities. This is not surprising, considering the aesthetic appeal of stands of emergents such as *Phragmites* and *Polygonum*, and of submerged forms such as *Callitriche, Lobelia* and *Chara*. In addition, these communities provide living quarters, breeding areas and feeding grounds for many fish, birds and mammals, and in this connection the microflora and invertebrate fauna associated with surfaces of rooted plants are extremely important.

Macrophytes are thus of major significance in their influence on the structure and functioning of aquatic ecosystems at all trophic levels. Yet, they continue to be threatened. The pressures which alter the abundance and species diversity of the plants can be classified into two main types. One of these includes the pressures bearing on the water resource directly; these relate to the needs to satisfy the demands for water-borne recreation and for potable water. Pressures of the other type stem from activities in loch and stream catchments; here, the concern is over water quality, and with aspects of urbanisation and rural development - with the latter of particular significance in the case of Loch Eye.

Many forms of pollution can result from such developments. However, eutrophication is invariably a major consequence, and we need to consider its direct influence on the levels of nitrogen (N) and/or phosphorus (P) *per se* , and the indirect effects on higher plants and other freshwater biota. Then, a number of complex community and trophic interactions that are likely to be affected can be identified. For example, moderate increases in nutrient concentrations may lead to increased biomass of rooted plants and their attached flora (see below), but perhaps little discernible changes in species diversity. However, in most situations nutrients do not just 'increase'; there are additional changes which are likely to affect the 'natural' balance of species. The levels, relative importance, annual rates of supply, and seasonal patterns of availability of the different nutrients will be altered, depending on the type of eutrophication, i.e. whether it stems from animal husbandry or plant agriculture, or from domestic sewage (Bailey-Watts and Kirika, 1987; Bailey-Watts *et al*, 1987; Bailey-Watts 1990).

Increases in inputs (loadings) of nutrients need not necessarily be manifested in elevated N and P concentrations in the receiving waters. In the case of runoff from land, the inputs are controlled primarily by rainfall. Depending on the schedule of agricultural activities and general availability of nutrients, and on the type and fraction of nutrient involved (Bailey-Watts *et al*, 1989), there will be periods when loadings are on the increase, but, due to heavy dilution by rainwater, the concentrations may actually decline. Biological uptake can also mask the effects of increased loadings on in-lake nutrient concentrations. Plants of various types are able to sequester the extra nutrients, and the relative success of the planktonic algae, their attached epiphytic and periphytic cousins, and the macrophytes themselves, depends on a number of factors. However, the outcome of the competition between these plants usually determines much of what is perceived as the response of a lake system to enrichment.

The plant responses may then affect the ecology of animals. For example, the greater the morphological diversity of macrophytes, the more varied the associated assemblages of invertebrates and the quality of fish food. Contrastingly, where phytoplankton dominate, their fixed energy may be utilised by fish *via* one of two main routes, depending on time of year, and the species composition and size structure of the algal assemblages. One pathway to fish may be through herbivorous zooplankton in the water column, while the other involves invertebrates dwelling in the bottom deposits. In this connection, it should be remembered that fish have to be extremely adaptable in their feeding behaviour, and while switches to different foods may affect their growth rate, fatalities are only likely where eutrophication is especially advanced and severe depletion of oxygen results.

The nature and extent of the eutrophication are not the only factors that determine how the chemistry and biology of a lake will respond; the availability of energy and nutrient resources other than those being added by the eutrophication, the rates of throughput of water ('flushing'), and the basic morphology of the lake basin are of paramount importance. Indeed, if carbon, iron, or other elements required for growth are present in very low quantities, enrichment with N and P may have a rather minor impact. Equally, additions of N may effect little response if prevailing P levels are low, although the situation is not quite the same with regard to additions of P when N is low. While enrichment of some lake waters with P may bring about little response, this appears to be a summer feature (Bailey-Watts 1990); indeed, the growth elicited by experimental additions of inorganic P (alone) to a range of Scottish reservoir waters over a number of seasons, indicates that planktonic algae at least, are not commonly limited by shortages of nitrogen here (Bailey-Watts *et al*, 1988). In addition, some planktonic organisms including classic bloom-forming, blue-green algae ('cyanobacteria'), can be unaffected, since they can 'fix' the dissolved, gaseous form of N (Horne and Cummins, 1987) and are not wholly dependent on nitrate or ammonia as sources of inorganic N.

In many situations, because of the contrasts in their ecology with respect to light, planktonic algae of one sort or another will out-compete species of rooted or attached plants. The planktonic forms are often distributed throughout the water column and are therefore not restricted to lighted sediment or other surfaces in the manner of the other plants. The depth distribution of species of rooted vegetation is largely determined by the clarity and the spectral quality of the w ater (Spence, 1982). Neither is the growth of phytoplankton as seasonally restricted by short day lengths as that of many macrophytes.

The availability of macrophyte surfaces influences the success of some attached algae, the growth of which in turn modifies the light environment of the host plant; indeed, Phillips *et al*, (1978) suggest that in some situations, these algae rather than phytoplankton alone initiate the demise of macrophytes. There is a large number of species that colonise the various surfaces, and they range in size from a few microns (1 $\mu m = 10^{-6}$ m) in the case of epiphytic diatoms and cyanobacteria, to centimetres and more in the case of filamentous green algae, e.g. *Cladophora* species. In extreme situations, these may literally festoon the submerged plant (Bailey-Watts, unpublished observations), so there are mechanical as well as light effects to consider.

Phytoplankton, with the more diverse array of species, and consisting of organisms that may have the higher growth rates, can commonly capitalise more rapidly than the larger plants on nutrient resources, and in doing so, 'shade' them out. However, the light climate in the water column is continually changing - not least, in many eutrophic waters, due to shifts in the population density, size structure and species composition of the algal plankton itself. The light climate of the suspended cells is additionally influenced by vertical mixing patterns and movements into and out of the euphotic zone - the lower boundary of which is defined as the depth where gains of energy through photosynthesis are just sufficient to balance respiratory losses (Kirk, 1983). Light is thus another resource for which the different plants may compete. The production of all of these communities is ultimately determined by light availability, even when nutrients are super-abundant, so it can also be viewed as one of the physical factors controlling the extent to which nutrient resources are utilised.

Taking these considerations into account, one can appreciate why phytoplankton blooms are so manifest in eutrophic waters, and why enhanced phytoplankton growth is often implicated in macrophyte die-back (Jupp and Spence, 1977). However, the success of phytoplankton relative to that of the attached and rooted plants generally decreases where the water residence time is low (that is, when flushing rate is high) with only the plankton susceptible to being washed out of the system. Indeed, the nature and rates of a wide variety of physical, chemical and biological processes in lakes appear to be influenced by variations in this factor (Bailey-Watts *et al*, 1990); hence, the general predominance of attached algae and higher plants over plankton in many flowing waters (see, however, Reynolds 1988 regarding river phytoplankton), and the enhanced growth of algae - especially the cyanobacteria - in a number of UK waters during recent dry summers (NRA, 1990; Bailey-Watts, 1990). In this connection, it should be realised that a particularly troublesome algal bloom can result from very subtle differences in the schedule of environmental change. Conditions favouring growth need only last for a few days longer than usual, to allow many planktonic algae to divide, and thus double in biomass once more than usual - if nutrients permit.

Lake morphology comprises another important determinant of the outcome of nutrient enrichment *vis à vis* the balance between macrophytes and plankton. Factors such as length-to-breadth and area-to-depth ratios determine the behaviour of a water mass as regards thermal structure; the degree of mixing or stratification in turn influences the position of cells within the water column (and hence the light environment they experience). Variations in temperature regime between different lakes will also influence the rates of respiration and photosynthesis of phytoplankton, and the grazing rates of herbivorous zooplankton. At the same time, the detailed bathymetry, the slope of the lake bottom, and the degree of exposure of shores to wave action, control macrophyte distribution (Jupp and Spence, 1977). Other conditions which are important and peculiar to individual loch systems are dealt with in the following section.

1.3 Why a Loch Eye study?

Evidently, limnologists have a good enough grasp of many aspects of the functioning of lake systems to be able to predict the general biological consequences of eutrophication. Certainly there is an enormous literature on this subject (Bailey-Watts, 1990, in press). It is, then, pertinent to address the question as to the need for an intensive, site-specific study.

One answer is that the vast majority of papers purporting to concern eutrophication do not deal with actual rates of inputs of nutrients, but the in-lake, biological manifestations of the nutrient enrichment process. As a consequence, data from even some of the best-studied systems have proved inadequate when attempting, for example, to predict the outcome of eutrophication control programmes (Sas, 1989). It is encouraging, however, that UK studies, including those on Lough Neagh, Northern Ireland (Gibson, 1986) and the Norfolk Broads in particular (Moss 1980), feature among the few exceptions.

Secondly, as indicated above, the good predictive ability extends to general aspects of the biological responses to changes in nutrient inputs. Present knowledge provides little of significant value to the manager of a specific water body, and the situation at Loch Eye prior to the present study provides a good example of this. The long association of anglers with the loch, and the more recent involvements of NCC and the Highland River Purification Board have provided a useful body of information. However, as these agencies point out, there are few quantitative data on eutrophication due to farming and wildfowl and these are referred to elsewhere in this report (4.4.1, 4.4.4). Hence the main foci of the present study.

Also, while the loch certainly appears to have deteriorated with 'losses' of macrophyte species, such as *Baldellia ranunculoides*, *Potamogeton filiformis* and *P. praelongus* (Charter, 1988), it is debatable whether overall plant cover has decreased. One would expect some increases in plant biomass with eutrophication, and recorded instances of weed-clearing suggest this has been the case, but even this may reflect a re-distribution of plant material, rather than a change in total biomass. Charter's review also suggests that Loch Eye supports species such as *Isoetes lacustris* and *Lobelia dortmanna* which would suggest oligotrophic conditions, as well as *Chara* and *Potamogeton* species indicative of eutrophic conditions.

Similarly equivocal information concerns the organisms at the other end of the food chain; the deterioration in trout fishing has been variously attributed to (a) migration upstream to another water body (b) loss of shelter and spawning habitat, (c) predation by eels, (d) predation by birds and (e) poaching. It is likely that each of these factors is involved, but to what extent has not been quantified. In addition, it might be worth noting that the Loch Leven experience shows that much of the variation in annual trout catches can be explained by variation in fishing effort (Bailey-Watts and Maitland 1984). As indicated above, the situation is complicated. Increases in fish productivity in line with eutrophication trends may be masked by effects of reduced food quality due to the loss of macrophyte beds and associated invertebrates.

Thus, different waters can be categorised, but only on the basis of broadest aspects of their physical, chemical and biological 'behaviour'. As our understanding advances, it is realised not only that lake systems function in an extremely individual manner, but they do so as a result of differences in gross physical and chemical features. Thus, the following information on the Loch Eye system, appears to fully justify the programme - on the grounds of adding significant scientific knowledge and enhancing our freshwater conservation and management capabilities:

Shallowness: the present work adds materially to our understanding of the functioning of shallow systems with Loch Eye having a mean depth ($\bar{z}$) of only 1.2 m (**Figure 1a**). Apart from the UK eutrophication studies mentioned above, and the outstanding work on Lough Neagh, N.I., the vast majority of research on the dynamics of nutrients and phytoplankton concerns large, deep and regularly stratifying waters. The work in the English Lake District (eg Lund and Reynolds, 1982; Reynolds 1987) is an example. These waters appear to behave quite differently from the shallower systems, with the former tending to exhibit more regular, seasonal patterns of plankton abundance and nutrient concentrations (see also Round 1971, Bailey-Watts *et al*, 1990). The factors controlling the fluxes of nutrients between sediments and water in shallow lakes also differ from those in deep water bodies (Mortimer, 1941, 1942, 1971; Drake and Heaney, 1989; Marsden, 1989).

The nature of the eutrophication: the Loch Eye work is unusual in the context of eutrophication studies in focusing on a system receiving rather little nutrient-rich waste by way of point sources. As the population in the catchment numbers only a few dozens, domestic sewage is of minor importance. Runoff from agricultural land is thus apparently the major, stream-borne, nutrient source.

The over-wintering goose populations comprise another special aspect of study, as they can be considered as potentially exacerbating the presumed, agriculture-driven eutrophication; in many winters, the loch supports a maximum of *ca* 30,000 Greylag Geese. In one of the very few previous studies on the subject, Hancock (1982) found that the faeces from Greylag and Pink-footed geese could account for a considerable proportion of the total P input to the Loch of Strathbeg, Aberdeenshire. Rutschke and Schiele (1978/1979) also point to the importance of geese in this regard at Lake Gulpe in eastern Germany. This is likely to be the case in any remote, but particularly coastal areas of shallow water. A similar situation exists in some of the Norfolk Broads, but gulls rather than geese are the important agency (Moss and Leah, 1980).

International conservation status: The concern over the status of macrophytes in Loch Eye is particularly acute, as the lake is a Site of Special Scientific Interest included in the Nature Conservation Review (Ratcliffe, 1977) and is in the Ramsar List of Wetlands of International Importance. It is also a European Special Protection Area for Wild Birds (Stroud *et al*, 1990) - not least on account of its goose populations!

Encouragingly, the NCC has recognised the points about the individuality of lake catchments, and this is reflected in its funding of a number of separate eutrophication studies. Indeed, following closely on the present report, will be one (also by IFE) summarising the findings of the studies which have covered a range of lake types over the UK - Loch Leven and Loch Eye in Scotland, Bosherston Lake in Wales, and Malham Tarn, Esthwaite Water and a number of the systems in the Norfolk Broadland, England.

1.4 Scope of the report

Graphs and Tables are used extensively to support the analyses and interpretations of the data obtained. Chapter 2 discusses general features of the catchment and the loch, to give a preliminary idea on the extent to which the ecology of the loch might be influenced by its surroundings. Investigative methods used in the field and the laboratory, and the procedures used for data analysis are covered in Chapter 3, with reference mainly to previously-published accounts. The results follow in Chapter 4, initially with reference to the water balance (Section 4.1) and secondly, with information on temperature and light penetration (4.2). In keeping with the focus on eutrophication, three sections (4.3, 4.4 and 4.5) are devoted to water chemistry; while spot measurements of pH, conductivity and dissolved oxygen are reported, the attention is mainly on nutrients, i.e. nitrate (NO_3.N), particulate and dissolved fractions of phosphorus (P), and dissolved silica (SiO_2). Section 4.3 looks at fluctuations in the concentrations of each nutrient, and discusses the results from the streams and the loch together; this identifies similarities and contrasts between the standing and running waters, and at the same time, emphasises the integral nature of the drainage area and the loch. Section 4.4 concerns nutrient loadings and their relationship to the standing stocks of material in the loch itself, and our attempts at an input-output budget for phosphorus. The P and SiO_2 status of the sediments and their potential influence on the overlying water are also discussed. The final section of results (4.5) concerns species composition, biomass fluctuations and size structure of the phytoplankton. The nature of the zooplankton is briefly discussed in relation to the nutrient status and the phytoplankton of the loch. The main results are discussed as they are reported, but a separate chapter (5) is reserved for a concluding discussion on the ecology of the loch, and on what the study has achieved. Chapter 6 presents recommendations for future research and management of the loch. Chapters 7, 8 and 9 cover Acknowledgements, References and the Figures respectively.

2. THE ENVIRONMENT OF LOCH EYE

2.1 The catchment: general physical features and land-use

The catchment of Loch Eye (**Figure 1b**) is fairly low lying with a maximum altitude of only 120 m at the top of the western drainage area. Ordnance Survey topographical maps suggest that the area (including the loch itself) is 13.8 km^2; this compares with the value of 13.7 km^2 quoted by Charter (1988) and derived from figures given by Murray and Pullar (1910). The region is of Old Red Sandstone partly covered by glacial till. Information from Smith (1981) shows that some 75% of the catchment land area had been developed for cereal production on the glacial till, compared to *ca* 54% some 10 years previously. Heather and woodland, including conifer plantations which are largely confined to the Old Red Sandstone area, covered a further 24% of the area. Tree planting has increased in the intervening years, and Fountain Forestry has recently proposed the planting of some 0.8 km^2 (including 0.5 km^2 in the Loch Eye catchment) of mainly mixed hardwoods, but also Sitka spruce, Scots pine and Japanese larch. Such developments are of relevance to considerations about the trophic status of the loch; if the planting involves the application of fertilisers to the young trees, runoff of P-enriched water into the loch is likely (Bailey-Watts *et al*, 1988). The human population within the catchment is low, with dwelling houses being confined to relatively few farmsteads, although there is some new housing with septic tanks situated in the northern part of the catchment.

Water is transported from the land to the loch mainly by three streams, draining 10.42 km^2, i.e. 88% of the total area. Proceeding in a clockwise direction and starting from the west, the largest sub-catchment is that of the Garrick. It covers 6.18 km^2, some 30% of which is planted forest, and the rest mixed scrub and agricultural land. On the northern side of the loch, the Loinnbuie, which is the smallest of the main inflows, drains 0.74 km^2 with about 50% consisting of cereal agriculture and 50% of scrubland and heath. Surrounding the eastern border of the loch is the drainage zone of the Erracht (3.50 km^2) consisting primarily of agricultural land put over mainly to cereals, but some improved grassland for livestock. A more detailed analysis of the catchment with respect to land use is needed, and the potential of GIS techniques in this connection has been considered (see 6.1).

2.2 **Rainfall**

Towards gaining one of a number of indications about the amounts of water entering the loch, the flushing rate and seasonal variation in the throughput of water (see below), rainfall records from the two Meteorological Office stations nearest to Loch Eye have been examined. The Geanies recorder is situated at approximately 68 m a.s.l. some 5 km to the east of the eastern edge of the catchment. The other site - Morangie - lies to the north west of the loch some 6 km beyond the catchment boundary at 42 m a.s.l. **Figure 2** displays monthly values and shows that rainfall patterns over the 3-year period 1986 to 1988 were similar at the two sites, with Morangie appearing some 10% wetter; annual precipitation totals were, for Geanies and Morangie respectively, 572 mm and 646 mm for 1986, 600 and 659 for 1987, and 708 and 761 for 1988. These figures are in keeping with the general impression of an equable climate in the coastal area, and a situation in the rain-shadow of the mountains to the north-west. The data in Figure 2 illustrate the marked month-to-month variation in the amounts of rain, and the lack of a strong seasonal pattern of precipitation; this appears to be a feature of Northern Britain. Note, for example, rainfall peaks of >60 mm per month at Morangie; these were recorded in January, May, August, and December 1986, but March, June, July, September and October in 1987, and in each month except February, June, September and November of 1988.

2.3 The potential influence of the catchment on the ecology of the loch

The relative sizes of the loch and its drainage area indicate the degree to which the catchment influences the ecology of the loch. On the basis of their area ratio - the catchment being only *ca* 6 times that of the loch surface - a moderate influence would be predicted. By contrast, the catchment-to-loch area ratio for Loch Leven is 11:1. However, because Loch Eye is shallow, the ratio of the catchment area to the loch volume is correspondingly high; on this basis, the effect of the catchment on the functioning of the loch could be considerable. A third factor needs to be taken into account, however, and this is rainfall. It is important in the general context of eutrophication because of its control of flushing rate, but it is especially important in a situation like Loch Eye because the great majority of the supplies of materials to the loch are thought to enter in runoff from the land, i.e. they are rain- or flushing-dependent. In Easter Ross, however, evaporation is high, i.e. *ca* 400 mm, and this amounts to 55 to 65% of rainfall. The net annual influx of water to Loch Eye is equivalent to approximately only 1.25 loch volumes from the catchment, plus an amount equivalent to *ca* 15% of the loch volume in rain falling directly on the loch surface. Taking each of these factors into account, the potential influence of the Loch Eye catchment would appear minor in comparison to many other sites which have higher catchment-to-loch area ratios, and which gain more rain water, and more nutrients from flushing-independent point-sources.

3. INVESTIGATIVE METHODS

3.1 General scope

Field and laboratory studies concentrated on factors of relevance to the measurement of eutrophication, the physical and chemical factors affecting the concentrations and loadings of nutrients, and the responses of the loch with special reference to the phytoplankton.

3.2 Fieldwork

The inflows (at IG, IL and IE in **Figure 1b**), the loch edge (C) and the outflow (OL and OR) were sampled at 3-weekly intervals from September 1986 to June 1988, for physical (except flows - see below) and chemical information. The two points on the outflow - OL just a few metres from the loch, and OR some 250 m further downstream and near a road, differed considerably in nutrient content, with much higher concentrations often being found at OR (**Table 1**). This appears to be due to the discharges of field drains which join the outflow by the time it reaches the road. While this is an interesting observation, it is not a main concern of the present study, and it will not be referred to again. It should be noted, however, that this site was the only one sampled on the outflow by Charter (1988), and is also the site regularly monitored by the Highland River Purification Board.

Plankton was also collected at the loch edge at 3-weekly intervals. In addition, on one in every three of these visits (i.e. at 9-weekly intervals) the open water areas of the loch were sampled (A and B in Figure 1b). On nine of the occasions of open water work (September 1986;

Table 1. Nutrient levels at the 'road' sampling site on the outflow from L. Eye 250 m from the loch itself, expressed as percentages of those measured at a site within a few metres of the loch.

Nutrient	Mean values	Maximum Values
Nitrate	303	215
Total phosphorus	149	126
Total soluble phosphorus	139	251
Particulate phosphorus	157	122
Soluble reactive phosphorus	258	456
Dissolved silica	555	543

February, April, June, August and October 1987; February, April and June 1988), sediment cores were taken for total phosphorus (TP) content, and for the levels of soluble reactive phosphorus (SRP), dissolved organic phosphorus (DOP) and silica (SiO_2) in the interstitial water.

Arrangements were made for local personnel to record, on as many occasions as possible (daily, ideally), the water levels of the streams, the loch and the outflow, from August 1987 by which time staff gauges had been installed, to the end of the contract ie June 1988 (321 days),. In the event, however, a number of readings made in 1987 for the Garrick and the outflow were unfortunately lost. The outflow was monitored at more or less daily intervals until the end of 1988. Furthermore, it was not until the end of the study period, that it became evident that some of the recorders had tended not to visit the gauges as regularly, when they assumed levels were low and not varying. As a result, while the 3-weekly readings taken on the occasions of our nutrient work do not form the ideal basis for flow and loading estimates, it is likely that the more frequent readings over-estimate these factors. Nevertheless, comparisons between the 3-weekly records and the fuller data are made for periods when the recording bias was not so marked, in order to extend our knowledge on the likely flow and loading regime. The records were converted to discharges (stream flows - Q, in $l\ s^{-1}$) using equations (rating curves) relating levels to flows measured by detailed current metering at various points over a section of each stream on approximately 15 occasions over the period August 1987 to June 1988 - referred to below as the loading period. The work provided estimates of flushing rate that could be compared with those based on rainfall.

Water temperatures were always measured with a mercury-in-glass thermometer, but in addition, on the 9-weekly visits for open water sampling a probe for measuring temperature and pH was used; the probe was dipped directly into the streams and the loch - not into water transferred to a collecting bottle.

Duplicate dip samples were taken for chemical analyses of stream and loch edge waters. The samples were placed in cool boxes as soon as possible after collection.

Water samples for the analysis of phytoplankton and rotifers were taken only at the loch sites. Bearing in mind the shallowness of Loch Eye the procedure was to plunge wide-mouth, 10-litre polyethylene containers below the surface. Sub-samples for chlorophyll analysis (to measure total algal biomass) were kept in the dark and as cool as possible until further treatment e.g. filtration, in the laboratory set up in Balintore - some 3 miles from the loch. Samples for the determination of algal species, their sizes and population densities were fixed with Lugol's Iodine - a saturated solution of iodine in a saturated aqueous solution of potassium iodide (2 ml per 1000 ml of water sample). The collections set aside for rotifer analysis were fixed immediately with procaine hydrochloride (1 g per litre of water).

To determine the euphotic depth (z_{eu}, the depth - in metres - at which photosynthetic gains by an algal cell will just balance its respiratory losses), the spectral quality of the underwater light field in open water was assessed according to the procedures of Bindloss (1976). These use a submersible photo-cell and duplicate surface cell combination, with diffusing opal and interchangeable Schott glass colour filters - blue (with an optical mid-point of 460 nm), green (540 nm), orange (590 nm) and red (630 nm). The instrumentation developed by Benham and George (1981) was used for measuring the intensity of light at depth as a percentage of the surface value. A further measure of water clarity was obtained using an 8-cm diameter, black-and-white, quartered Secchi disc. Surface water temperature was measured to the nearest 0.5 of a Celsius degree with a mercury-in-glass thermometer mounted in a Ruttner water bottle. Vertical profiles of temperature recorded to the nearest 0.01 of a degree were made with a thermistor incorporated in the Benham and George module. The same module was used for the vertical profiling of conductivity and dissolved oxygen.

On most of the occasions of open water sampling, material for assessing the percentage species composition of the crustacean zooplankton was also taken; a 180-mesh nylon net was towed through the water, and the concentrated material was transferred to a small polythene jar, and fixed immediately with 4% formaldehyde.

For information on the chemistry of the sediments, duplicate cores of up to *ca* 20 cm depth were taken with a Jenkin Surface Mud Sampler. On return to shore, the overlying water was siphoned off, with subsamples of that 10 cm above, and immediately above the sediment surface being retained for nutrient analysis. The mud itself was then extruded from the core tubes a centimetre at a time, sliced and transferred into polythene bags. One-centimetre slices were taken throughout the top 10 cm of core, and a further 1-cm slice was taken from 15 cm below the sediment surface when the core depth allowed. In this field situation, it was not possible to prevent oxidation of the sediment material, and this is known to lead to adsorption of e.g. phosphate ions that might otherwise be present in the interstitial water (Bostrom *et al.*, 1982). The results on pore water P must therefore be considered as underestimates of the real values.

3.3 Laboratory analyses

Full details of most of the procedures used for analysing the water, sediment and biological materials have been described: thus, Bailey-Watts *et al.*, (1987) for P fractions and methods for handling the sediments; Bailey-Watts (1976a), Bailey-Watts, *et al.*, (1989) for dissolved SiO_2; Bailey-Watts and Duncan (1981) for NO_3.N; and Bailey-Watts (1986, 1987, 1988) for recent references to literature consulted in the determination of phytoplankton species, and the estimation of their population densities and sizes. In view of the importance of P in eutrophication studies, it is worth summarizing what was done regarding the different fractions of this element. The total amounts of P (TP) and the total soluble component (TSP) were determined respectively on un-filtered water and water passed through a Whatman grade C, glass-fibre disc. The organic P - particulate P (PP) and dissolved organic P (DOP) - in these fractions, was acid-digested to convert it to the soluble reactive form (SRP). Any SRP present in the original sample was then determined on an aliquot of filtrate without acid digestion; this is an important fraction, in that it represents a pool of P most immediately available to algae. From the results obtained for TP, TSP and SRP, the levels of PP and DOP were calculated from:

$$PP = TP - TSP$$

$$\text{and,} \quad DOP = TSP - SRP$$

3.4 Data analysis

The field and laboratory data generated by this study are lodged in various files on the micro-Vax computer housed in the Bush Laboratory of ITE to which the IFE Edinburgh Laboratory is attached. The 'MINITAB' package was used for handling and sorting the original data, for summary statistical analyses, and for the rapid preparation of simple graphs. The graphs selected for inclusion in this report were then prepared using the 'SAS' system.

4. RESULTS

4.1 Inputs and Outputs of Water: Estimates of Flushing Rate

4.1.1 Flows

The data derived from the gauge heights and flow meterings of the feeder streams are shown in **Figure 3a**. Arrows indicate the occasions of field sampling and thus the flow values which are paired later with the nutrient concentrations to give loadings. The mean discharges calculated from the 17 3-weekly readings are 75 l s^{-1} for the Garrick, 25 l s^{-1} for the Erracht and 15 l s^{-1} for the Loinnbuie. Linear plots are displayed in preference to logarithmic graphs because, although the latter would enable values of the whole ranges to be identified more readily, the present figures demonstrate better the seasonal variation in discharge and the important episodes of high flow. The streams appear to behave somewhat similarly in these respects, but perhaps this is to be expected for a small catchment varying little in altitude.

The irregular changes in flow are in keeping with those exhibited by rainfall, particularly if month-to-month variation in evaporation is taken into account. Individual flows varied considerably - from less than 1 l s^{-1} to *ca* 600 l s^{-1} depending on the stream. Monthly mean discharges calculated from these figures ranged from < 1 to *ca* 60 l s^{-1} in the Loinnbuie, to *ca* 150 l s^{-1} in the Erracht and to *ca* 450 l s^{-1} in the Garrick. The maxima rank in the same order as the areas of the stream catchments.

However, neither these, the mean monthly discharges, nor the means calculated for the loading period, bear as close a relationship to the areas as might be expected from work at Loch Leven, for example (Bailey-Watts *et al* 1987). This is undoubtedly due to at least two factors. The first relates to difficulties of measuring low flows; there are limitations to current metering in very small channels, and yet low flows are likely to occur for most of the time. For example, the mean discharges in August 1987, of the Erracht and the Loinnbuie were estimated at 0.4 and 0.9 l s^{-1} respectively, yet the drainage area of the Erracht is approximately 5 times that of the Loinnbuie. The second factor contributing to the apparent lack of a sensible relationship between stream size and flow regime, concerns the probable bias in the recording of stream heights, discussed under 3.2.

Although (i) the high flow periods are of much greater importance as regards annual discharges, and (ii) even if many of the low flows were over- or under-estimated by a factor of 5, their effect on the results for annual loads and flushing would be relatively minor, knowledge about the time and duration of low flows is important.

Taking flow values based on the more frequent gauge readings, but remembering that assumed low flows tended to be ignored, the 3-weekly gaugings appear to under-estimate the discharges to a degree related to stream size, ie by 28% for the Garrick, by 22% in the case of the Erracht, and by 8% for the Loinnbuie. If these factors are taken into account, the estimated mean discharges are raised from 75 up to 104 l s^{-1}, 25 up to 32 l s^{-1}, and 15 up to 16 l s^{-1} for the Garrick, Erracht and Loinnbuie respectively. By the same token, the total of the mean discharges is raised from 115 l s^{-1} to 152 l s^{-1}.

Figure 3b shows the discharge fluctuations at the outlet during 1988. This watercourse is less 'flashy' than the feeder streams, reflecting how the loch dampens short-term variation in flow. Indeed, in contrast to the situation with the feeder waters, the 3-weekly record appears to slightly over-estimate the outflow, ie by *ca* 4%. This comparison is based on the period January to June 1988 over which the outflow was monitored daily with few gaps. The major peaks here, follow those of the Garrick (the only stream likely to affect the outflow to any great extent on its own), but decreases in discharge are the more gradual at the outflow. Nevertheless, variation in discharge over the year as a whole, is considerable, and the Figure illustrates well the contrasts between seasons; the mean, monthly discharge ranged from 139 to 293 l s^{-1} over the first 5 months and accounted for 77% of the annual discharge, 7 to 21 l s^{-1} over the period June to September (equivalent to only 3%) and 66 to 151 l s^{-1} for the last quarter of the year (making up the remaining 20%). The patterns of flow into and out of the loch thus relate in a generally predictable manner, but the inputs and outputs rarely balance - even after accounting for changes in loch volume due to shifts in level, for rain falling directly on the loch surface and for the 12% of the catchment not drained by these streams. Nevertheless, the major inputs of water in January 1988 easily exceed the outflow and do correspond to a period of rising loch levels (**Figure 3c**).

4.1.2 **Flushing rates**

Flushing rates (**p** - expressed as loch volumes per unit time) have been estimated by three methods. **Table 2** presents examples of the calculations based on (i) stream flows plus rain falling directly on the loch, (ii) the discharge at the outflow, and (iii) rainfall alone. The figures refer to the period August 1987 to June 1988 when loadings were estimated, but the calculations include a correction to produce annual values.

Table 2. **Three methods used to calculate flushing rate (p)**

Method I uses daily inflow measurements (i) and rainfall data (ii):-
(i) the sums of the mean total instantaneous discharges from the 3 major inflows is 115 or 152 $l\ s^{-1}$ depending on whether the 3-weekly figures are considered to under-estimate the discharges (see Section 4.1.1). When adjusted for the 12% of the catchment not drained by these streams, these values are equivalent to annual inputs (V_{inflow}) of $4.13 \times 10^6\ m^3\ y^{-1}$.

(ii) rainfall records from the Geanies and Morangie sites for August 1987 to June 1988 give a mean annual value of 709 mm. Evapo-transpiration in the Easter Ross area, is *ca* 400 mm which, multiplied by a factor of 1.2 to correct for the greater losses from the water surface gives 480 mm.

The net input of water in rain (V_{rain}) is thus (709-480) ie 229 mm which is equivalent to a total volume of $0.45 \times 10^6\ m^3$ over the loch surface ($A_1 = 1.95 \times 10^6\ m^2$)

The total input of water (V_{in}) is obtained from:

$$V_{in} = V_{inflow} + V_{rain}$$

Where V_{inflow} is $4.13 \times 10^6\ m^3$, V_{in} is $4.58 \times 10^6\ m^3$, and where V_{inflow} is $5.46 \times 10^6\ m^3$, V_{in} is $5.91 \times 10^6\ m^3$

If, by this method of calculation

$p = V_{in}/(A_1.z)$, where z is the mean depth (1.2 m)

$p = 1.96$ or 2.53 loch volumes y^{-1}

Method II: uses the daily mean instantaneous discharges at the outflow - 124 $l\ s^{-1}$, equivalent to $3.91 \times 10^6\ m^3\ y^{-1}$. If this value is V_{out},

$$p = V_{out}/(A_1.z), \text{ with } A_1 \text{ and } z \text{ as defined above, so}$$

$$p = \frac{3.91 \times 10^6}{1.95 \times 10^6 \times 1.2} = 1.67 \text{ loch volumes } y^{-1}$$

Method III uses rainfall data alone ie (i) rain falling over the drainage area (A_d - $11.85 \times 10^3\ m^2$), and (ii) rain falling on the loch surface as calculated in *Method II*(ii)

(i) The net input of water in rain over the land area (R) is given by:

$$R = (P-E).A_d$$

where P and E are the annual values for precipitation and evapo-transpiration (in m). Thus,

$$R = (0.709-0.400).\ 11.85 \times 10^6 = 3.66 \times 10^6\ m^3$$

(ii) the input in rain over the loch surface (V_{rain}) is $0.450 \times 10^6\ m^3$ calculated by *Method I;* then by this method of calculation:

$$p = (R + V_{rain})/(A_1.z) = \frac{(3.66 + 0.45) \times 10^6}{1.95 \times 10^6 \times 1.2} = 1.76 \text{ loch volumes } y^{-1}$$

In view of the general points raised in Section 2.3, about the likely influence of the catchment on the loch *via* its feeder waters, both of the streamflow-derived flushing rates (1.96 and 2.53 loch volumes y^{-1}) are considered to be too high. The similarity between the estimates of 1.67 (outflow), and 1.76 (rainfall) supports this view. Plainly, more information is needed on the water balance (see Section 6.1), but a figure of 1.7 seems appropriate, for use in later considerations about nutrient loadings and budgets. However, the possibility should be borne in mind, that the actual flushing rate over the loading period is higher than 1.7 but not as high as 1.96, and certainly not as high as 2.53.

Table 3 lists a number of **p** values estimated for a variety of periods and using the different methods of calculation. On the basis of rainfall records, the flushing rate was considerably greater in 1988 (**p** = 1.91) than in either of the previous two years. 1988 also contrasts with the other years, in that flushing over the first six months was greater than that estimated for the second six months. The mean annual value for the 3-year period is 1.46 which is very close to the initial 'guestimate' given in Section 2.3.

None of the estimates take account of the possibility of 'short-circuiting' of water; in Loch Eye this might occur between the Erracht and the outflow (Figure 1a). By the same token, water from the other streams is likely to reside in the loch for longer than suggested by the estimates of flushing rate.

Table 3. Flushing rates (loch volumes per unit time) at Loch Eye, estimated by the methods explained in table 2, for, or corrected for (*) various 6-month (6) or 12-month (12) periods

period of calculation	method of calculation	flushing rates
January to June 1986	rainfall	0.48 (6)
July to December 1986	"	0.71 (6)
January to December 1986	"	1.19 (12)
January to June 1987	"	0.51 (6)
July to December 1987	"	0.60 (6)
August 1987 to January 1988	inflows + direct rain rainfall	1.08 (6) 0.77 (6)
January to December 1987	rainfall	1.29 (12)
August 1987 to June 1988	inflow + direct rain (3 weekly values unadjusted)	1.96 (12)*
"	inflow + direct rain (3 weekly values adjusted in relation to daily records)	2.53 (12)*
"	outflow (3 weekly values)	1.67 (12)*
"	rainfall	1.76 (12)
January to June 1988	outflow (3 weekly values)	1.88 (6)
"	outflow ('daily' values)	1.32 (6)
"	rainfall	1.04 (6)
February to June 1988	inflows + direct rain	0.73 (6)*
February to July 1988	rainfall	0.89 (6)
July to December 1988	outflow ('daily' values)	0.35 (6)
"	rainfall	0.75 (6)
January to December 1988	outflow ('daily' values)	1.67 (12)
"	rainfall	1.91 (12)

4.2 Temperature and underwater light

4.2.1 Stream and loch temperatures

Fluctuations in the temperatures of the three main inflows show similar, marked seasonal patterns. The Garrick Water exhibited the widest range of values (**Figure 4a**) with the lowest winter figure being below 1°C, and the highest summer value being 14°C. Otherwise, there is no clear relationship between the temperature regimes of the three water courses: no stream is consistently warmer or cooler than another. Surface water temperatures of the loch itself (**Figure 4b**) exibited a similar seasonal pattern, but because the water resides in the lake basin for longer than it stays in the stream channels, the temperature ranges are greater in the loch, i.e. from around 0°C (January 1987) to 17°C in August 1987, and 20°C in June 1988. The extreme temperatures were recorded at the shallow sites. As in Loch Leven, Kinross-shire, the temperature fluctuations at Loch Eye can be divided into four phases (Smith 1974): one of rapid warming (February to May), one of fluctuating high temperatures (May to August), one of rapid cooling (September to December) and a fourth period of varying, but generally low temperatures over the turn of the year.

4.2.2 Light penetration and water clarity in the loch

Only 7 Secchi disc readings were taken, but these varied widely - from 0.72 m in June 1987 to high values of 2.2 m (October 1987) and 2.5 m (June 1988). The high values are indicative of very clear water, such that on occasions, the sediment surface over much of the loch area is in the lighted zone.

In lake waters, light exhibits a logarithmic decrease with increasing depth - especially in well-mixed water bodies where organisms are unable to accumulate as surface or mid-water maxima, for example. The rate of attenuation, however, differs with wavelength and in Loch Eye, where the column is often laden with particles and water contains dissolved humic material ('Gelbstoff'), the shorter wavelength colours, e.g. blue, are absorbed the most rapidly. The data in **Figure 5a** refer to August 1987 and are typical in showing the contrast between the rates of attenuation of light in different parts of the spectrum; blue light is reduced to *ca* 1% of the surface intensities within 1.0 m of the surface, while the slopes of attenuation in the green, red and orange parts of the spectrum, are such that the intensities at 1 m are still 20-30% of the surface values. These slopes can be expressed as extinction coefficients (in ln units m^{-1}); values ranged from *ca* 0.6 to 2.0 for the three longer wavelengths, and from 2.0 to 5.7 for the shorter wavelengths (**Figure 5b**). In the absence of measurements of the vertical distribution of phytoplankton, the minimum extinction coefficient, which in Loch Eye is that measured with the red filter, has been used to calculate z_{eu}. This varies inversely with extinction coefficients and in Loch Eye ranged from approximately 2.0 to 6.5 m (also in Figure 5b). To illustrate how these findings relate to the attenuation of light in the red part of the spectrum, **Figure 5c** shows the range of slopes obtained on 7 occasions spanning the study period.

4.3 Stream and loch chemistry

4.3.1 Dissolved oxygen, pH and conductivity

Dissolved oxygen was normally at 85-95% saturation. However, some values as low as 70% were recorded in the winter even when the water appeared to be well-mixed. Instrumental faults were discounted. The overall range of pH measured in the inflows was 5.9 to 7.9 units (**Figure 6a**). Results from the four occasions on which all three inflows were measured on the same day, suggest that the streams behave similarly, and that none is consistently more alkaline or more acid than the others. On any particular sampling occasion, the loch was usually fairly uniform with regard to pH levels, although in June 1988, readings ranged over 1.5 units with the outflow sample giving a low outlying value (**Figure 6b**). Over the study period, however, pH varied from 6.4 to 9.5. It is likely that in Loch Eye the higher pH values result from the photosynthetic activity of the submerged macrophytes and attached microflora rather than that of its relatively sparse phytoplankton (see 4.5.1). The loch appears not especially remarkable in terms of conductivity, but the values (standardised for 25°C) ranging from 224 to 255 µS cm^{-1} probably reflect the proximity to the coast.

4.3.2 Nutrients

a) *nitrate*

The graph of stream nitrate values is dominated by the high, sharply-fluctuating levels in the Erracht (**Figure 7a**) which reflects the prominence of cereal production in its drainage area. Generally lower concentrations exhibiting summer minima and winter maxima were recorded in the other streams. However, because the nitrate falls to below 0.1 mg N l^{-1} in these more dilute waters, the relative variation is some 20- to 50-fold, whereas in the richer, though 'flashier' Erracht, the variation is only 4- or 5-fold.

Fluctuations in nitrate in the loch parallel those in the Garrick and Loinnbuie. Apart from the peak of some 6 mg N l^{-1} recorded in the Loinnbuie in February 1988, the concentration in the loch and these two streams is also similar (**Figure 7b**). The similarity in results from the different loch stations suggests that the water mass is reasonably well-mixed as regards nitrate. The winter peaks, which are of the same order as those recorded in Loch Leven in the 1970s (Bailey-Watts *et al.*, 1990), almost certainly correspond to periods of reduced N uptake by organisms, and the summer minima to enhanced activity of de-nitrifying bacteria in particular (e.g. Bailey-Watts, 1988; Bailey-Watts *et al.*, 1990).

The disappearance of *ca* 1.5 mg N l^{-1} over the period April to June 1987, and an apparent loss of 2 mg l^{-1} during much of the first half of 1988, are unlikely to be due to algal growth alone. They indicate that sediment-water fluxes may be very important - in this case, with fluxes of nitrogen to the deposits. The observations emphasise that even nitrate-rich lakes can exhibit phases of very low N. It is under these regimes that N compounds are reduced to levels limiting the growth of many plants, so that certain cyanobacteria have the competitive edge, in being able to draw on (i.e. 'fix') dissolved, gaseous elemental N as a source of this nutrient.

(b) *phosphorus*

Total phosphorus in the inflows varied overall some 15-fold from *ca* 10 µg P l^{-1} (**Figure 8a**), but values of >50 µg l^{-1} were rare. Such peaks might be expected to represent increased loadings of particulate material associated with episodes of high rainfall, but reference to **Figures 8b** and **8c** reveals that as many TP peaks coincided with increases in TSP as with those of the PP. In general, the majority of the P in these streams is in soluble form with average values of 84% TSP/TP for the Garrick, 81% for the Loinnbuie and 74% for the Erracht. This may reflect the small size of the streams, and the gentle slopes of the land which they traverse - features probably precluding very marked scouring of the stream beds and heavy sediment transport.

The stream values show few trends, but the apparent lack of any seasonal cycles is known from other Scottish studies (see Bailey-Watts and Kirika, 1987, who also illustrate TP levels in Loch Leven agricultural drains, similar to those found in Loch Eye streams). There is more evidence of seasonal trends in the fluctuations of TP and TSP in the loch (**Figures 9a** and **9b**); the build-up of soluble P in summer may indicate releases from the sediment. While there is reasonable evidence of good mixing (or similar dynamics) throughout the loch, the high values are not always recorded at all of the sampling sites. The phosphorus in the loch contains a smaller proportion of soluble material than the stream P, i.e. around 50% as against 75-85%. At times, this is due to planktonic algae, and especially during windy weather, detrital material which increase the particulate fraction (note the PP maxima in **Figure 9c** recorded mainly at the loch edge and outflow sites). This view is supported by examination of material under the microscope. In spite of the presence of material disturbed from the sediments, the TP concentrations here - averaging over the study period *ca* 30-40 µg l^{-1} depending on site - are considerably lower than the 60 µg l^{-1} recorded in Loch Leven in the wet year of 1985.

Much of the P entering the loch is potentially available to algae, especially in the Loch Eye situation with the high proportion of the runoff P being in soluble form. However, the concentrations of SRP, being the component most immediately available to algae, provide the best indication of the production potential of the water at any one time. In the streams, the percentages of SRP/TSP are high - averaging 79 in the Garrick and the Loinnbuie, and 57 in the Erracht. SRP levels in these waters show few trends (**Figure 10a**), with irregular fluctuations ranging over approximately one order of magnitude. However, the pattern of changes found in the Erracht was somewhat different to that recorded in the other streams. In the context of concerns over the eutrophication of Loch Eye, it is notable that the levels of SRP in the inflows are very moderate; they remain below 20 µg P l^{-1} for most of the year in the Garrick and the Loinnbuie, and commonly lie between 5 and 10 µg P l^{-1} in the Erracht - the latter stream having exhibited the highest nitrate concentrations.

Soluble reactive P also exhibited few trends in the loch, but many peaks and troughs from different stations coincide, thus indicating reasonably good mixing (**Figure 10b**). This need not mean that the whole water mass behaves uniformly; P uptake rates within macrophyte stands may differ from those outside these patches, and fluxes of P over the sediment surface will vary between different types of deposit. Shifts of between 5 and 10 µg SRP l^{-1} between consecutive sampling occasions (3-week periods) are common and there is evidence of considerable inter-annual variation. The two relatively short-lived pulses of SRP recorded in spring and summer 1987 may well be due to sediment release. On the basis of the data described so far, however, the peak in autumn 1987 is not so easily explained, since, although October was wet, it was not especially so. The comment made above with regard to the P content of the streams in the context of eutrophication, applies also to the loch situation. Averaging *ca* 2 to 6 µg SRP l^{-1} depending on site, the open water concentrations are very moderate, although they represent, at the instant of sampling, the reservoir of nutrient not incorporated into organisms.

(c) *silica*

Much of the overall variation in the concentration of dissolved SiO_2 in the streams is covered by the range 3 to 10 mg l^{-1} (**Figure 11a**). This is small compared to the 10-fold range exhibited by SRP, but SiO_2 concentrations are characteristically less variable, unless (seasonal) uptake by e.g. diatoms is important (Bailey-Watts *et al.*, 1989). As elsewhere too, the Loch Eye inflows showed marked, irregular fluctuations. When the concentrations are plotted against the corresponding flow figures, a variety of distributions of points are obtained, with even log-log graphs showing few consistent relationships. However, as Bailey-Watts *et al.*, (1989) found, when the chronology of the concentration/flow relationship is examined, some patterns are often revealed; (arrows in **Figure 11b** which illustrates an example with Garrick data). No stream is consistently richer in SiO_2 than another, but the Garrick Water is generally the most dilute, with the mean value of 4.7 mg l^{-1} compared with 6.4 mg l^{-1} for the Loinnbuie and 7.4 mg l^{-1} for the Erracht.

Fluctuations in dissolved SiO_2 in the loch (**Figure 11c**) suggest that the dynamics of the nutrient here contrast considerably with those in the streams. Firstly, the concentrations are much lower, with even the maxima (of 1.5 and 2.0 mg l^{-1}) being less than nearly all of the concentrations recorded in the streams. This could be due in part to dilution with rain water which is likely to be poor in dissolved silica, but the more probable cause is uptake by organisms. The presence of planktonic diatoms and chrysoflagellates, epiphytic diatoms and macrophytes has been noted already. Other diatom communities are likely to be important in Loch Eye where for much of the year most of the bottom is in the euphotic zone. These are benthic associations often rich in species characteristic of sandy deposits, and others just as diverse occurring in muddy sediments; they deserve more attention (see 6.2), but with the help of Mr John Carter (Denholm, Hawick), a preliminary examination of some of this material revealed a host of different types.

Complex physico-chemical (ie abiotic) processes, such as adsorption of dissolved silica onto siliceous mineral deposits - studied primarily in running waters - could also explain some of the decreases in silica here (see Bailey-Watts, 1976b and Bailey-Watts *et al.*, 1989 for references). In spite of the generally low levels of SiO_2, the range is some 20-fold, ie higher than in the streams. This is mainly a result of the concentrations being occasionally reduced to less than 0.1 mg l^{-1}. The second marked contrast with the streams, is the marked seasonality associated with this. Since the major depletions occur in late winter, diatoms and chrysoflagellates which are common at this time of the year in many temperate lakes are likely to be a major cause. A more puzzling aspect concerns the timing of the increases in concentration to the maxima. If rain-driven, stream inputs were the major source of SiO_2 to the water column, a much less regular timing of the increases would have been expected. Also, while release of SiO_2 from the sediments could explain these changes, we have no quantitative data on this (see, however, 4.4.5). Unless aerobic release during windy conditions and disturbance of the sediments is important (see eg Drake and Heaney, 1989 for P), sediment release would probably not be important at the cool times of the year, which is when the increases are observed. In the absence of other information, the influence of the autumnal die-back of macrophytes might be considered; in addition to the epiphytic diatoms, many higher plants contain SiO_2, and the increases in dissolved SiO_2 may result as much from reduced uptake of the nutrient, as from an increased loading from the streams.

4.4 Nutrient loadings and phosphorus budget

4.4.1 Inputs in runoff

An initial indication of the loading of each nutrient (L, in e.g. mg s^{-1}) is given by the products of the annual mean flow (Q_i, in l s^{-1}) for each stream, and the annual mean concentration (C_i, in e.g. mg l^{-1}) according to the equation of Verhoff, Melfi and Yaksich (1979):

$$L = k \sum_{i=1}^{n} \left[C_i/n \right] \cdot \sum_{i=1}^{n} \left[Q_i/n \right]$$

where k is a factor taking acount of the period of record - and n is the number of samples; here, 16 samples were taken over the 321-day period August 1987 to June 1988. The results obtained have been adjusted to give annual values (**Table 4**).

Plainly, the nutrient concentrations used in the loading calculations refer only to the 321-day period to which the flow data correspond. It should be noted, however, that average levels of some of the nutrients over that period differ markedly from those described above (Section 4.3.2) which refer to the whole study period (see **Table 5**).

Table 4. Annual nutrient loadings to, and losses from the catchment of Loch Eye; nitrate-nitrogen (N), total phosphorus (TP), soluble reactive phosphorus (SRP) and dissolved silica (SiO_2), calculated using the equation of Verhoff *et al* (1979).

Stream	Catchment area (ha)	Annual Loadings (kg)				Annual loss rates (kg ha^{-1})			
		N	TP	SRP	SiO_2	N	TP	SRP	SiO_2
Garrick	618	1419	91.0	48.7	12582	2.3	0.15	0.08	20.4
Loinnbuie	74	680	13.7	8.6	3263	9.2	0.19	0.12	44.1
Erracht	350	4730	21.2	10.0	5763	13.5	0.06	0.03	16.5

Table 5. Comparisons of mean concentrations (in µg l^{-1} for P, mg l^{-1} for N and SiO_2) of nutrients for (i) the whole study period, September 1986 to June 1988 and (ii) the period August 1987 June 1988 when stream flows, the outflow, and loadings were estimated.

Site (and sampling interval in weeks) Nutrient / Period	TP		SRP		PP		NO_3N		SiO_2	
	i	ii	i	ii	i	ii	i	ii	i	ii
Inflows:										
Garrick (3)	37	39	19	21	6.8	6.3	0.61	0.63	4.7	5.3
Loinnbuie (3)	42	29	20	18	18	2.3	1.3	1.4	6.4	6.9
Erracht (3)	23	27	9.4	13	6.3	6.8	5.9	6.0	7.1	7.3
Outflow: (3)	39	37	5.3	5.5	21	20	0.77	0.87	0.40	0.39
Loch:										
a (9)	30	23	2.2	2.3	17	11	0.92	0.99	0.23	0.26
b (9)	37	25	2.5	2.1	25	13	0.86	0.93	0.41	0.22
c (3)	42	29	6.1	5.5	25	12	0.57	0.66	0.46	0.41

These values are only approximate because the chemical sampling interval is quite long. Improved estimates are obtained using the equation of Rodda and Jones (1981), because, by incorporating the mean of the products of C_i and Q_i, as opposed to the product of their means, it retains variation before calculating the loading whereas Verhoff *et al* remove variation by calculating means first.

$$L = k \sum_{i=1}^{n} \left[\frac{C_i . Q_i}{n} \right]$$

with all the terms defined as above. The results from these calculations - adjusted as before - are shown in **Table 6**. Later discussions about nutrient budgets incorporate a flushing rate figure of 1.7 loch volumes y^{-1} (as explained in Section 4.1.2); loadings in Tables 4 and 6, however, are based on the flows measured at the instants of chemical sampling, giving a flushing rate estimate of 1.96 loch volumes y^{-1} (as in Table 2). Strictly speaking, an adjustment should be made for this, but, assuming nutrient losses vary *pro rata* with total water discharge (see e.g. Bailey-Watts *et al.*, 1987) the loadings would be less than 15% lower than shown, and this would make little difference to the conclusions drawn below.

The P losses per unit area of land are close to values published for other lowland, agricultural, reasonably base-rich catchments (Stevens and Stewart, 1981; Foy *et al.*, 1982; Jordan and Smith, 1985; Bailey-Watts and Kirika, 1987; Bailey-Watts *et al.*, 1987). The range of SiO_2 losses overlaps with values obtained by Bailey-Watts *et al.*, (1989) and with those quoted by them from the literature. Nitrogen losses are also similar to those calculated by Cuttle (1989), and measured by Bailey-Watts and Kirika (unpublished data) for Loch Leven.

Table 6. **Annual nutrient loadings in runoff to Loch Eye, and loss rates per unit area of the catchment: nitrate-nitrogen (N), total phosphorus (TP), particulate phosphorus (PP), soluble unreactive phosphorus (SURP), soluble reactive phosphorus (SRP) and dissolved silica (SiO_2), calculated using the equation of Rodda and Jones (1981)**

Stream	Catchment area (ha)	Annual loadings (kg)						Annual loss rates ($kg\ ha^{-1}$)					
		N	TP	PP	SRP	SURP	SiO_2	N	TP	PP	SRP	SURP	SiO_2
Garrick	618	2179	112.4	26.0	61.0	22.9	13075	3.5	0.18	0.04	0.10	0.04	20.0
Loinnbuie	74	1132	16.1	2.6	9.1	4.0	2191	15.3	0.22	0.04	0.12	0.05	29.6
Erracht	350	4219	71.0	16.2	37.8	17.5	5683	12.0	0.20	0.04	0.11	0.05	16.2
Total loads assuming *pro rata* losses for the 12% of the catchment not drained by these streams:		8578	227	51	123	51	20949						

Losses of TP and of each P fraction per unit area of land are also remarkably similar for each stream. The specific areal loading of TP to the loch calculated from these values is 0.12 g TP m^{-2} y^{-1}, including 0.06 g SRP m^{-2} y^{-1}. The value for TP is only one-thirteenth of that measured for Loch Leven (1.54 g m^{-2} y^{-1}) which has, however, a mean depth some 3 times that of Loch Eye. In contrast to the situation with P, the exports of nitrate-N differ with the Garrick catchment losing *ca* one-third of the losses from the other drainage areas. More detailed information on land-use is necessary for a more definitive statement on the causes of this difference (see 6.1), but it is notable that the Garrick drains land with relatively more forest, and less cereal agriculture than the other sub-catchments. The very small losses are not greatly in excess of the values reported for annual wet deposition in the Northern Scotland, ie. *ca* 2 g N m^{-2} although the UK precipitation chemistry monitoring site nearest to Loch Eye is to the west of Straithvaich Dam where annual rainfall is *ca* 1250 mm (United Kingdom Review Group on Acid Rain, 1990).

4.4.2 **Runoff and in-loch stocks compared**

Comparisons of the stream-borne loadings with observed 'standing stocks' in the loch, indicate the extent to which the supplies of the different nutrients are utilised. On the basis of a **p** value of 1.7, the 8.6 t $NO_3.N$ estimated to enter Loch Eye annually is equivalent to 2.5 times the average in-lake standing stock; equivalent figures for the other nutrients are 2.1 for TP, 1.0 for PP, 8.5 for SRP, and 20.4 for SiO_2. Silica thus decreased to the greatest extent relative to SRP, with nitrate being reduced by the smallest proportion. PP levels show virtually no change. In other words, the flow-weighted stream concentration (13 µg l^{-1}) is little different from the level recorded in the loch (see Table 4). However, the PP in the running waters is different to the plankton in the loch, in being largely organic or detrital - and probably settling onto the deposits in the loch.

4.4.3 A first attempt at a phosphorus budget

The budget refers to the period August 1987 to July 1988. This section examines first, the export of P from the loch *via* the outflow. By comparing the export with the runoff supplies, it assesses whether, on an annual basis, the loch is accumulating or losing P. In order to assess whether the data make sense, use is made of the models developed by Dillon and Rigler (1974) and Kirchner and Dillon (1975) which together aim to predict, from aspects of the water balance, how much P delivered to a lake is retained (ie the P retention coefficient - *R*), and also how much P should be expected, from the loadings obtained. **Table 6** sets out 2 calculations of the export of P, giving values of 145 and 215 kg y^{-1}.

Table 6. **Two estimates of the annual export of total phosphorus (TP) from Loch Eye *via* its outflow**

Estimate I: this is based on the equation of Verhoff *et al* (1979) and takes the product of the mean TP concentration (37 $\mu g\ l^{-1}$ - see Table 4) and the mean water discharge rate (124 $l\ s^{-1}$ - see Table 2) from the 16 paired values over the period of record:-

$= 124 = 4588\ \mu g\ s^{-1}$

$= 396 \times 10^6\ \mu g\ d^{-1}$

$= 145 \times 10^9\ \mu g\ y^{-1}$

$= 145\ kg\ y^{-1}$

Estimate II: this is based on the equation of Rodda and Jones (1981) and takes the mean of the 16 products of concentration and instantaneous flow rates, ie 6183 $\mu g\ s^{-1}$

$= 589 \times 10^6\ \mu g\ d^{-1}$

$= 215 \times 10^9\ \mu g\ y^{-1}$

$= 215\ kg\ y^{-1}$

As with the calculations of the loadings to the loch, the second estimate has to be taken as the more reliable. This compares with the 227 kg entering the loch over the same time (Table 6), which gives an R value of 0.05. [It is possible that the mean P concentration measured at the outflow (37 $\mu g\ l^{-1}$) over-estimates the output - since the P values for the other sampling sites are lower (see Table 4); however, even if one takes the average of the mean values for the four sites (29 $\mu g\ l^{-1}$) R is raised to only 0.07; if inputs are some 15% lower than calculated (for reasons outlined in Section 4.4.1), the situation becomes even more unlikely with the export almost certainly exceeding the input.] However, a system with a net retention of only 5% would be remarkable and there are good reasons for suspecting that Loch Eye retains a high proportion of the P supplied to it. Firstly, the bottom sediments of the loch appear to have accumulated phosphorus (see 4.4.5) and rooted and attached vegetation are abundant. The second reason for doubting this value, relates to Kirchner and Dillon's empirical model, developed from observations on lakes varying widely in area and depth.

Its central term is q_s, the 'areal water loading', in units of m y^{-1}; this is the volume of water entering a lake, V_{in} ($m^3\ y^{-1}$) divided by the surface area of the lake, A_1 (in units of m^2). V_{in} has already been assessed in relation to the calculation of **p**, and equates to an exchange of 3.98×10^6 $m^3\ y^{-1}$ resulting in a q_s value of 2.04 m y^{-1}. The model then predicts R from q_s according to:

$$R = 0.426e^{(-0.271q_s)} + 0.574e^{(-0.00949q_s)}$$

so, where q_s is 2.04, R is 0.81.

Although the errors involved in arriving at this value are high, the equation supports the original notion that Loch Eye retains the large majority of its external, stream-borne supply of P.

The large discrepancy - at least equivalent to the presently estimated stream-derived P load - requires further exploration, because it very much affects the development of a P budget. For example, knowledge of R is crucial for predicting mean in-lake P concentrations ($[TP]_1$) from the specific areal loading of P (L, in mg m^{-2}). One of the main models of this relationship is that of Dillon and Rigler (1974):

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

where all the terms are as defined already. The effect of R can be illustrated by taking the 'measured' and 'predicted' values (0.05 and 0.81 respectively), using 1.2 m for z, 1.7 for $\mathbf{p}$, and 116 mg m^{-2} for L (= 227 kg TP y^{-1} from Table 6). With the lower R value a mean in-loch TP concentration of 54 µg l^{-1} is predicted, and where R is 0.81, 18 µg TP l^{-1} is predicted. These concentrations can be compared with the measured value of *ca* 29 µg l^{-1} (Table 3).

In summary, Loch Eye appears to be retaining a much smaller proportion of its P supply than would appear likely. Also, under a more probable P retention regime, the mean TP levels in the loch are higher than those presently predicted. It is unlikely that the nature and functioning of Loch Eye is unusual enough to invalidate the use of these models; the annual P losses ha^{-1} are similar to those reported elsewhere, and the rates of conversion of P to plant biomass are likely to be within the range predicted by such models. The situation thus points to a large source of P not yet accounted for, yet the observations suggest that the extra loading does not reach the loch *via* the feeder waters.

Two assumptions can help in assessing what amounts of P might be involved. The first assumes that R is 0.81. Then, the 215 kg P estimated to have passed out of the loch is 19% of the 'true' external load, so this load is 1075 kg of which we can already account for the 215 kg exported, and 227 entering in stream waters. This leaves 633 kg unaccounted for. Another assumption is that the value of 29 µg l^{-1} for $[TP]_l$ is correct. If this is inserted into the Dillon and Rigler equation, along with the values of 0.81 for R, 1.2 for z and 1.7 for $\mathbf{p}$, the solution for L is 311 mg TP m^{-2}; this is equivalent to 607 kg TP y^{-1} but of this only the stream input of 227 kg is accounted for. In any event, a source of many hundreds of kilogrammes of P is still to be identified. Some of the possible sources are discussed in the next section.

4.4.4 **Geese as additional transporters of P**

Phosphorus can enter the water column of a lake basin in a number of ways other than by feeder streams, and involve external sources and recycling. External supplies include (a) the rain falling directly on the loch surface, (b) point-sources e.g. pipes (that may have been overlooked), (c) diffuse runoff or seepage of P, and (d) the faeces of waterfowl. For a variety of reasons the focus is on roosting birds - although release of P from the sediments is also considered.

Firstly, while we have very few data on nutrients in rain, a low concentration of *ca* 20 µg P l^{-1} is indicated. If this is taken as a mean value representative of the approximately 660 mm of rain falling on the loch surface per annum, i.e. 1.3 x 10^6 m^3 of water, an input of 26 kg from this source is calculated. This is only 11% of the stream-borne input, and a very small proportion of the anticipated shortfall.

Over the 2½-year period of the study, eutrophication issues were often discussed with local personnel, but there was no suggestion of a major source of P in the form of either septic tank discharges or other point-sources. In any event, some point-sources and diffuse seepage of nutrient-rich water into the loch will already have been accounted for in (i) the stream loads, and/or (ii) the areas not drained by the major streams assuming that their P loss rates (per hectare) are the same as the stream catchments.

As to recycling of P within the loch, we have no quantitative information on release of P from the sediments. Experience suggests, however, that in lakes as shallow as Loch Eye, aerobic release from disturbed sediments could be important (Marsden, 1989). Anaerobic release could also be important during prolonged spells of hot, calm weather, and when residence time is low (see Bailey-Watts *et al.*, 1990). Two summer spells of sharp increases in SRP have already been noted. These are equivalent to an influx of some 10 $\mu g\ l^{-1}$ (which amount also disappeared rapidly - see Figure 10b); it is thus unlikely to be anything other than a minor proportion of the shortfall. It therefore remains to explore whether geese could contribute the outstanding amount, feasibly approaching 0.5 t y^{-1}.

The calculations performed to assess the impact of these birds use the results of detailed work by Hancock (1982) on the inputs of nutrients in goose faeces to the Loch of Strathbeg, Aberdeenshire. However, earlier, exploratory analyses by Cooke (1976) give similar values. By incorporating the records of the numbers of Greylag Geese overwintering and roosting on the loch, **Table 7** shows that the goose populations could be introducing P in amounts sufficient to account for much of the considerable discrepancy in the preliminary budget.

The example shown uses figures that probably err on the low side, yet an annual value of 326 kg is obtained. This is equivalent to nearly 1½ times the stream loading. A large impact of the geese is also suggested by the high incidence of droppings noticed on the loch floor, and the considerable amount of cereal tissue found in the sediments. Even bearing in mind that not all of this P would necessarily appear in the water column, it is likely that geese could account for much of the discrepancy in the budget.

Table 7: **An assessment of the likely impact of Greylag goose populations on phosphorus inputs to Loch Eye.**

Let N equal the number of birds roosting on the loch and P_{out} the estimated daily rate of P output per goose. P_{out} values taken from Hancock (1982) assume that the birds roost on the water for 16 hours per day. P_{out} values differ depending on whether the geese are feeding primarily on cereal (barley in the Strathbeg study) or on grass, with the former giving the higher value, i.e. 332 mg P goose^{-1} d^{-1} as against 211 mg P goose^{-1} d^{-1}.

N varied considerably at Loch Eye, from the 54 birds counted in September 1986 to the 16500 estimated in late October 1987. Greylag geese are present in appreciable numbers for 6 months of the year. For present purposes it is assumed that, on average, 10000 geese are present for 4 months, i.e. 120 days. In the absence of detailed accounts of the feeding regimes, it is assumed that the population is divided equally between those feeding on cereals and those on grass.

The total P_{out} from this source is:

$$(5000 \times 120 \times 332) + (5000 \times 120 \times 211) \text{ mg}$$

$$= 326 \times 10 \text{ mg}^6$$

$$= 326 \text{ kg}$$

4.4.5 Phosphorus and silica status of the sediments and their potential influence on water chemistry.

This aspect aimed to assess the 'standing stocks' of P and SiO_2 in the sediment, in order to gauge to what extent the overlying column might be enriched assuming that all, or portions of these stocks were released into the overlying water. The likely extent of any internal loading can then be compared with the figures for the external loadings.

(a) *phosphorus*

Dissolved P in the mud interstitial water: the concentrations of both TSP and SRP in the interstitial water (W_i) increase with increasing depth, and the shifts in the concentrations parallel each other. However, considerable differences between the analytical results of the duplicate cores were occasionally obtained, and **Figure 12** shows the worst case. Although the water content decreases with increasing depth, the variation is small relative to that of the P concentrations. The standing stock of interstitial P is thus greater at depth (**Figure 13**). Most frequently, TSP outweighs SRP by about 5- to 10-fold, but in February 1988 the concentrations of TSP were only marginally higher than those of the soluble reactive component (**Figure 14**). Maximum concentrations of SRP - the fraction most readily available to algae - commonly occurred at *ca* 8 cm depth (as in Figure 14) but varied from 20 to 160 μg P l^{-1}. These concentrations are slightly lower than those measured in Loch Leven.

To what extent would the P concentration in the overlying water be affected by admixture with the interstitial pool? The likely maximum effect of such an event can be gauged by assuming that the total water content of the sediment is interstitial water. Water constitutes some 85 to 95% of the deposits by volume. Although this includes liquid driven off from the particulate matter during drying, the upper layers are likely to comprise *ca* 92% water. A layer 1 cm thick, covering 1 square metre can then be assumed to contain 0.0092 m^3 (i.e. 0.92/100) or 9.2 l interstitial water. If, on average, this water contains 20 μg SRP l^{-1}, (a reasonably representative value, and one well in excess of most of the values measured in the overlying loch water), the stock of SRP is 184 μg m^{-2} (ie 9.2 x 20). If this amount were released into the overlying column of 1.2 m, the average concentration there would be raised by only 1.5 $\mu g\ l^{-1}$ (i.e. 184/120). As in other lakes investigated, the pool of P in the interstitial water at any instant appears to be of minor significance in the sediment release process.

P as a percentage of sediment dry weight: analyses of dried sediment suggest that *P* constitutes *ca* 0.1 to 0.2% by weight. The range is typical of the slices of material taken from the uppermost 10 cm of sediment (**Figure 15**) and neatly encompasses the mean figure of 0.17% obtained by Bostrom *et al.*, (1982) working on a variety of North Temperate lakes. It also overlaps considerably with the range found for Loch Leven (authors, unpublished observations); this may be somewhat surprising in that Leven would probably be considered the much richer loch. Commonly, and in contrast to the situation with P in the interstitial water, the higher percentages of dry weight occur at the top of the sediment, although sub-surface maxima have been recorded below 5 or 6 cm into the mud. Variability between analyses on duplicate cores may mask some of the depth trends in P content, but the contrasting profiles shown in **Figure 15** appear to represent real differences. More extensive surveys of the distribution of this nutrient in Loch Eye would help to distinguish temporal trends from spatial variation.

These data allow the weights of P per unit volume of sediment to be calculated, and some idea of the extent of the sediment store of P to be gained. By taking 92 and 8 as typical values for the percentages by volume, of water and dry matter respectively, and assuming the specific gravity of the water is 1.0 g cm^{-3} and that of the dry particulate matter 1.1 g cm^{-3}, it is calculated that a cubic metre of sediment contains 0.088

metric tonnes dry matter. A layer 1 cm thick and covering 1 square metre, therefore contains 0.88 kg dry matter, and, at 0.15% P of dry weight, 1.32 g P. If all of this were rendered soluble, and released instantaneously into the overlying column, the concentration of P there would increase by 1.32/1.2 or 1.1g m^{-3} (= mg l^{-1}). The eventuality of such high concentrations is unlikely, but even if only 1/50th of such an exchange took place, the concentrations would be increased by *ca* 22 µg l^{-1}. Bearing in mind the range of SRP concentrations recorded (see **Figure 10b**), this would be very noticeable. Such considerations illustrate (i) how potentially important the sediments are as a source of P, and (ii) that the flux of P to the pore waters, by means of desorption from particulate matter, is considerably more important than the interstitial P measured at any instant (see Bostrom *et al.*, 1982).

(b) *silica*

Vertical profiles of soluble reactive SiO_2 were somewhat similar throughout the period of study, with virtually linear decreases in concentration with increasing depth. **Figure 16** displays data for three occasions contrasting in the overlying water SiO_2 concentration. Variation in results from each of the pairs of cores was usually much less than that found with the SRP. Maximum concentrations - at the deepest levels sampled - also varied rather less than with SRP; a range of 10.0 to 12.5 mg SiO_2 l^{-1} spans most of the values. In common with the situation found with interstitial P, however, concentrations in excess of those of the overlying water column were found even in the near-surface layers of the sediments. The only occasion when a marked surface gradient was not found, was when the loch SiO_2 concentrations were high - as in February 1987.

Since the profiles of interstitial SiO_2 changed rather little, there are few obvious relationships between these and events in the overlying water. However, there was a net increase in concentrations some time between October 1987 and February 1988 corresponding to a decrease in SiO_2 in the overlying water; the latter is likely to have been accompanied by a collapse of silica-containing organisms, and diatoms were relatively abundant prior to, and following the turn of the year. There was also a decrease in SiO_2 in the interstitial water in the top 5 cm of sediment between August and October 1987 (**Figure 17**). However, this had no visible effect on the loch concentrations. A question is whether the observed loss of about 1.5 mg l^{-1} from this layer would be detected - assuming that it was not rapidly utilised by planktonic diatoms, for example. A similar calculation to that performed on the data on interstitial SRP is thus instructive. Even if one assumes that the interstitial water comprised the whole sediment, it is plain that the (instantaneous) mixing of a 5-cm slice of it, with the overlying column of 24 times this depth, would have but a minor impact: a 24-fold dilution would reduce a solution of *ca* 1.5 mg l^{-1} to *ca* 62.5 µg l^{-1}.

4.5 Phytoplankton

4.5.1 General considerations and estimates of total algal abundance

While, by definition, a study of eutrophication must pay attention primarily to nutrient regimes, the planktonic algal assemblages are of considerable interest. This is because problems resulting from eutrophication are not often due to the elevated concentrations of nutrients *per se*, but to the resultant changes in the algal populations.

Taking the concentration of chlorophyll *a* as a measure of total phytoplankton abundance, the levels found in Loch Eye are very moderate. The mean concentrations for the whole study period and the 12 months covered by the loading estimates, were *ca* 17 $\mu g\ l^{-1}$ and 13 $\mu g\ l^{-1}$ respectively at edge site C. On most of the occasions when more than one station was sampled, a fair degree of uniformity over the loch was found (**Figure 18**). While maxima exceeding 30 μg chlorophyll *a* l^{-1} place this loch in the mesotrophic to eutrophic category, concentrations in excess of 100 $\mu g\ l^{-1}$ are more typical of the classic eutrophic sites e.g. Loch Leven, Scotland; the Loosdrecht lakes in the Netherlands, and the Norfolk Broads, England. Fluctuations in chlorophyll in Loch Eye show few annually-repeating trends, and relatively major increases and decreases in crop density occurred throughout the period. This is in keeping with the highly variable environment in terms of weather and chemistry. Not surprisingly, the sequences of phytoplankton species were also complex.

Algal population densities were commonly between 10^4 and 10^5 individuals ml^{-1}. A total list of some 200 species has been collated, but this ignores a number of forms, such as small chrysoflagellates which were not determined to species level; they are notoriously difficult to identify, even for a reasonably experienced observer. The major hallmarks of the Loch Eye phytoplankton are thus the preponderance of tiny species, the highly diverse nature of the assemblages, and the considerable contrast between crops at different times.

4.5.2 **Algal size distributions**

The size frequency distributions of the assemblages were invariably skewed to the right. Even where relatively large forms were recorded, these were rare in numerical terms. **Figure 19** shows results from a crop consisting principally of organisms less than 15 μm in greatest dimension, and for one exhibiting a much greater total range of size - from *ca* 4 to 160 μm. Two approaches are adopted for assessing the size structures. One of these samples the population on the basis of the relative numerical abundance of each type of organism, while the other selects the individuals according to their size, i.e. 'areal cover'. In most cases, in arrays of 50 individuals measured, little difference between the two sets of results was found. But, where relatively larger forms are (relatively) more common, the contrast between the two arrays is evident (see **Figure 20**). Occasionally, very complicated size distributions were revealed, as in the sample of September 1986 (**Figure 21**) distinct groups of different-sized organisms are evident, even though the total variation in size is less than 12 μm.

As the types of size frequency distribution vary over time, the arrays of data cannot all be normalised by the same transformation, and no single statistic is ideal for comparing the data from different seasons. Also, owing to the skewness, the means are commonly meaningless! However, **Figure 22** plots these with the median values. The median highlights the predominance of small species over much of the year, while the mean perhaps reflects better the increased importance of larger algae in summer. Examination of the phytoplankton of a number of lakes in the Temperate Zone suggests that this seasonal trend is fairly common, regardless of detailed species composition. Bailey-Watts and Kirika (1981) and Bailey-Watts (1986, 1987) have discussed the numerous factors affecting phytoplankton performance, and how the seasonally-shifting influence of these appears to be reflected in the algal size characteristics. Fluctuations in the

Table 8: A comparison of the phytoplankton species composition, the forms of algae and their percentage frequencies recorded in random samples of 50 individuals taken on three occasions in 1987. Each assemblage contained approximately the same total number of species. The forms are coded as follows: BGcol and BGfil, colonial and filamentous blue-green algae/cyanobacteria; Cr, cryptophycean flagellates; Dc and Dp, centric and pennate diatoms; Des, desmids; Gcoen, Gcol, Gflag and Guni, coenobial, colonial, flagellate and unicellular green algae respectively; Xcol and Xuni, colonial and unicellular xanthophytes; YGF, chrysophycean monads.

Month: Species	Form	July %	September %	November %
Anabaena flos-aquae f. flos-aquae	BGfil	6	6	4
Aphanothece clathrata	BGcol	8	4	4
Coenochloris ovalis	Gcol	2	-	-
Cryptomonas sp. nr. erosa	Cr	6	2	4
Gomphosphaeria lacustris	BGcol	2	-	4
Monoraphidium arcuatum	Guni	2	-	-
M. contortum	Guni	4	4	-
M. minutum	Guni	6	-	-
Oscillatoria sp. nr. agardhii	BGfil	8	10	2
Oocystis lacustris	Gcol	4	2	2
Oocystis sp.	Gcol	2	2	-
Pinnularia sp.*	Dp	2	-	-
Pediastrum duplex	Gcoen	2	-	-
?Radiococcus sp.	Gcol	2	-	-
Rhodomonas sp. *nr. lacustris*	Cr	8	-	14
Schroederia setigera	Guni	4	-	-
Sphaerocystis schroeteri	Gcoen	10	4	-
Scenedesmus ecornis	Gcoen	2	-	-
S. quadricauda	Gcoen	8	2	8
S. '2-cell'	Gcoen	2	2	8
Stephanodiscus sp.	Dc	2	2	-
unidentified green unicell (7 mm diam.)	Guni	2	6	-
unidentified chrysoflagellate	YGF	2	-	6
unidentified unicellular xanthophyte	Xuni	4	-	-

Species recorded in September that were not encountered in July:	Form	%
Coelastrum microporum	Cgol	2
Eucapsis sp (nr. alpina)	Bcol	2
Kephyrion sp.	YGF	6
Microcystis aeruginosa	BGcol	30**
Oscillatoria sp. (nr. limnetica)	BGfil	2
Pediastrum boryanum	Gcoen	2
Pseudanabaena catenata	BGfil	2
Scenedesmus serratus	Gcoen	2
Staurastrum sp. (nr. planctonicum)	Des	2
Tetraedron minimum	Guni	2
chloroflagellate	Gflag	2

Species recorded in November that were not encountered in either July or September:	Form	%
Asterionella formosa	Dep	10
Closteriopsis acicularis	Guni	2
?Botryochloris sp	Xcol	2
Chroomonas sp.	Cr	2
Crucigenia quadrata	Gcoen?	4
Elakatothrix gelatinosa	Gfil	2
Nephrodiella sp.	Xuni	2
Navicula sp.	Dp	4
Pediastrum tetras	Gcoen	2
Raphidonema sp.	Gfil	2
unidentified chrysoflagellates	YGF	10***

Footnotes: * these diatoms are not primarily planktonic, but they are often resuspended from the sediments during windy weather.
** approximately one-half of this percentage is due to single cells
*** at least 3 different species seemed to be present on this occasion.

intensity of grazing by micro-Crustacea - particularly the filter-feeding daphniids - appear to have an especially marked influence. Certainly, Loch Leven data spanning a number of years show that peaks in the relative abundance of larger algae are invariably associated with *Daphnia* population maxima. The same species of *Daphnia (D. hyalina* (Sars)) is present in Loch Eye, but a smaller cladoceran - *Bosmina coregoni*, var. *obtusirostris* (Sars) is even more prevalent. However, the population densities of these animals have not been estimated.

4.5.3 Species diversity

It is difficult to judge whether the total of approximately 150 species recorded during the formal, random 'sampling' of individuals for measuring, is of note. As few workers record the number of species in this quantitative manner, there is little opportunity for comparing species lists. However, in comparison to Loch Leven, where the work has been carried out in the same way, Loch Eye appears to be very rich. In the array of 30 individuals referred to in Figure 21, 9 different types of algae were recorded, and this again ignores the possibility of chrysoflagellates comprising more than one species. In many of the other samples - all of 50 individuals - the number of species/types commonly exceeded 20; size arrays of three assemblages of this type are illustrated in **Figure 23**.

4.5.4 Fluctuations in the qualitative composition of the phytoplankton crop

In addition to the changes in overall algal abundance and size structure, marked changes in species composition occur. Good examples concern the components of the three assemblages referred to in Figure 23. Only 17 of the 24 species present in July 1987, were also recorded in September, and in the November sample, only 18 of the species recorded in either of the two previous samples were detected (**Table 8**).

A display of the changing temporal abundances of all the species recorded would not be very instructive. However, a further aspect of change in the phytoplankton of Loch Eye is demonstrated in **Table 9**. This lists the major species in the assemblages corresponding to a number of chlorophyll maxima and minima (in Figure 18).

Table 9: **Algae dominating the phytoplankton at chlorophyll biomass maxima and minima**

Maxima	**Species**
27 November 1986	*Anabaena flos-aquae*, *Aphanothece clathrata* and chrysoflagellates
19 May 1987	chrysoflagellates
1 July 1987	*Aphanothece clathrata*, unicellular and coenobial green algae, and chrysoflagellates
21 September 1987	chrysoflagellates, *Aphanothece clathrata* and *Oscillatoria* sp.
26 January 1988	*Asterionella formosa*, *Scenedesmus* spp. and *Rhodomonas*
1 June 1988	*Anabaena flos-aquae* and *Aphanothece clathrata*
Minima	
7 September 1987	*Microcystis aeruginosa* and *Aphanothece clathrata*
13 October 1987	chrysoflagellates
29 March 1988	disintegrating colonial cyanobacteria

5. RESEARCH ACHIEVEMENTS AND CONCLUDING DISCUSSION ON THE ECOLOGY OF LOCH EYE

5.1 Research achievements in relation to the objectives of the study

It seems pertinent at this point, to consider firstly how the findings of the study relate to the objectives of the work, and assess whether the aims have been achieved.

To assess the nutrient status of the loch:

This has been satisfactorily achieved, particularly bearing in mind the remoteness of the site. The objective has been addressed with special reference to the actual nutrient levels, but at the same time, features of the phytoplankton community have been examined in order to see if it reflects the nutrient status. Algal populations have been found to be good indicators of trophic status elsewhere, and are commonly the focus of major concern in the context of eutrophication. During the Loch Eye study, a considerable amount of time has also been spent on assessing the nutrient status of the sediments.

To provide base-line data and methods for monitoring water quality, bird numbers and land-use:

Much of the work done in relation to this objective has concerned water quality, and most of the attention has been given to the influence of birds and land-use on nutrient chemistry. It is accepted, however, that a more detailed, closer-time interval record of goose numbers is still needed, and a more than the generally cursory examination of land-use carried out so far should also be undertaken (see Chapter 6). NCC staff are as well-placed and qualified as ourselves for collating these data, but the present authors would be interested in being involved in the interpretation of such information.

To consider, if relevant, means of ameliorating the nutrient enrichment of the loch:
As problems concerning the macrophyte communities and the trout fishery of the loch exist, and these undoubtedly relate in part to increasing eutrophication of the system, the consideration of 'means of ameliorating the nutrient enrichment of the loch' must be highly relevant. Certainly, this philosophy is reflected in the large proportion of time devoted to assessing the nutrient loadings, the fate of the nutrients in the loch, and the nutrient balances. Without information on the likely sources of nutrients and their relative contributions to the burden on the loch, the causes of observed nutrient dynamics and nutrient-organism interactions could not be judged. Indeed, the following discussion and conclusions relate mainly to these aspects, and the recommendations for future research on, and the management of Loch Eye focus on stemming eutrophication trends.

5.2 General aspects of the ecology of Loch Eye in relation to its nutrient status

The estimated losses of nutrients from the land surrounding Loch Eye are similar to the values published for (the few) other well-studied agricultural catchments in the UK. Together with the additional inputs of P from geese and direct rain, plus the amounts of the nutrient released from the sediment, the total P loading could exceed 500 kg yr^{-1}. This is equivalent to a specific areal load of 0.26 g m^{-1}, which is only one-sixth of the burden to Loch Leven. However, if the differences in the mean depths of these water bodies are taken into account - Leven being 3.25 times as deep as Eye - the P burdens are not so dissimilar.

Microscopic examination of the suspended particulate matter often reveals high proportions of organic detritus. This is not surprising considering the exposed position and shallowness of this loch, in an area characterised by windy weather. The PP determinations include this component, and the analysis does not distinguish between detritus and living material such as algae and zooplankton. The concentrations of PP attributable to phytoplankton are thus lower than the 'total PP' records suggest. In the absence of measures of actual algal P, but with PP and chlorophyll concentrations often being fairly similar (i.e. with a P to chlorophyll ratio of 1:1), there is no evidence of extreme P limitation as far as the phytoplankton is concerned. On the other hand, a number of algae including *Aphanothece* and *Oscillatoria* species are characteristically low in chlorophyll content. OECD (1982) models predicting mean chlorophyll levels from flushing-weighted P concentrations in the inflows, were developed from observations on less extreme lakes, ie ones somewhat deeper than Loch Eye, and in which macrophyte communities are not so prominent. Nevertheless, the other models used in this report are similar in this respect. It is therefore worth exploring whether the TP-chlorophyll model based on data from all of the lakes covered by the OECD analysis, describes the Loch Eye situation adequately. The equation is as follows:

$$\overline{chl_a} = 0.43\left[[\overline{TP}]_i/(1 + T_w)\right]^{0.88}$$

where $[\overline{chl_a}]$ is the predicted mean annual concentration of chlorophyll *a* (in µg l^{-1}), T_w is the water residence time (0.59 y, ie the reciprocal of **p**), and $[TP]_i$ is the mean, flow-weighted concentration of P entering the lake (also in µg l^{-1}).

Let chlorophyll values for two situations as regards $[TP]_i$ be predicted. If the value of 57 µg P l^{-1} - relating to the measured stream loading of 227 kg - is used, a figure of 9.2 µg chlorophyll l^{-1} is obtained. If a value of 146 µg P l^{-1} is used - this being calculated from the predicted loading of 553 kg (stream runoff plus geese) - a figure of 20.4 µg chlorophyll l^{-1} results.

The mean chlorophyll concentration from the measurements carried out from August 1987 to June 1988 is 13.1 ug l^{-1}. From the OECD model, this would suggest that the true loading of P to Loch Eye is between 227 and 553 kg y^{-1} and nearer to the lower end of that range. However, not only is phytoplankton in Loch Eye a minor component of its plant biomass, it is probable that much of the P in goose faeces is not readily available to phytoplankton.

The quality of the phytoplankton gives clues to the nature of its environment and insights into what population losses may be taking place. Such losses could otherwise be determined only by measuring turnover rates of the algae and the rate of grazing by e.g. zooplankton, of flushing out of the loch, and of sinking on to the deposits - each of these processes representing 'unseen' algal production, and emphasising that observed biomass is not necessarily a good index of production rate. The predominance of small algae over most of the year suggests that grazing by filter-feeding cladocera is relatively unimportant. By the same token, the summer increase in the relative importance of larger algae indicates some grazing pressure.

Alternating periods of calm and windy weather are likely to affect temperature structure, water movements and mixing rapidly here, and the complex sequences of phytoplankton species and the irregular fluctuations in biomass may well reflect this (see Bailey-Watts *et al.*, 1990 for Loch Leven findings on this subject). Many of the green algae commonly recorded are traditionally associated as much with surfaces and deposits as with the pelagic environment. Also the blue-green *Aphanothece clathrata*, which is prominent in Loch Eye, is of major significance in some of the Norfolk Broads where its persistence is attributed to its gelatinous consistency which confers protection from grazing, and reduces its sinking rate during calm weather (Moss and Leah, 1980). Planktonic diatoms are relatively rare; with some 50% of their dry weight consisting of opaline silica (specific gravity of 2.2 g cm^{-3}), their populations would be liable to considerable losses through sinking (Bailey-Watts 1976a, 1976b and Sommer 1988). At Loch Eye, no portion of the water column is particularly far from these deposits! In numerical terms, small chrysoflagellates commonly dominate the plankton of Loch Eye. This probably reflects their ability to maintain station in a still column, and survive in a mixed environment. Their preponderance is also indicative of organically acid waters, and some of the inflowing water to Loch Eye is heavily stained with humic material.

Even if no information were available on the P content of the outflowing water, which indicates that more is leaving the loch than entering it *via* the streams, a number of anomalies point to the existence of a source of P that reaches Loch Eye by means other than the streams. Not least of the clues, are those relating to the preponderance of rooted plants in the loch, to the abundant attached algal populations and to the P-rich surficial sediments. In spite of the competition from macrophytes and algal periphyton, the phytoplankton biomass observed fits generally with model predictions. Yet, it can be calculated from the data on sediment P, that if only 10% of the P in the uppermost 1 cm consisted of settled algae, it would equate to an average concentration of 83 μg chl *a* l^{-1} throughout the overlying water column. As it is, while the phytoplankton comprises one of the main components of P exported from the loch, i.e. washed down the outflow, it contributes a minor percentage of the total plant biomass.

As the aims of the study indicate, the role of geese in eutrophication of the loch was considered important enough to merit some attention. This opinion has proved well-justified. By assuming that what is unlikely to be an excessively large population of geese is associated with the loch and its surrounding feeding areas for only 4 months of the year, the likely P input due to these birds is equivalent to *ca* one-and-a-half times that attributed to runoff. If reported peak numbers of 30,000 geese are representative of flocks present for only 1 month of the year, the total P input from geese to this small loch could be well in excess of 0.5 t. A diagrammatic summary of what the study has established so far about fluxes and standing stocks of material is shown in **Figure 24** which uses data on total phosphorus. This is a very interesting situation, as geese constitute a diffuse source of nutrients, but unlike traditional diffuse supplies, their inputs are flushing-independent. What could amount to high burdens of P etc., are likely to be introduced to the loch by these birds, whether the winter is wet or dry.

6. RECOMMENDATIONS FOR FUTURE WORK ON LOCH EYE: RESEARCH AND MANAGEMENT

6.1 Work aimed at improving existing aspects

A first priority is to improve the estimates of water inputs and outputs. Together with reasonably frequent chemical analyses of the inflows and the outflow and fuller records of the numbers, feeding and roosting schedules of geese, the improved water budget would provide information on eutrophication trends. Experiments to assess the solubility and availability of sediment P, and assays to determine its algal growth-promoting potential are needed.

In spite of the relatively small contribution by *phytoplankton* to the total plant biomass in the loch, its quality and abundance should be monitored. As has been demonstrated, this information gives clues to a number of physical and chemical, especially nutrient, aspects of the loch. The structure of the algal assemblages also reflects features of the other biota. *Rotifers* (preliminary work on which has been done at Loch Eye, but not reported here) should also be monitored as they too, are useful indicators of environmental condition. A pair of 2-litre dip samples taken offshore would facilitate chemical, algal and rotifer analyses. For the chemical and phytoplankton determinations in particular, samples should be taken at intervals of no longer than one month. If the method devised by May (1985) for determining rotifer species composition were used, however, a single winter-time collection of bottom sediment would suffice for monitoring these animals.

Macrophyte performance in the way of species composition and total cover/

distribution, should be checked by means of surveys every 3 to 5 years.

A new record of land-use should be compiled and updated annually; this should include precise and accurate estimates of housing densities.

Techniques adopted for managing the Loch Eye trout fishery should be recorded very carefully. The following diverse activities should be monitored with a view to quantifying their effects:

- the cutting and/or clearing of aquatic weeds
- removal of trout predators, e.g. eels
- scaring of piscivorous birds
- stocking with fish fry or fingerlings
- boat- and shore-based fishing (expressed in angler-hours, for example) and the rod catches.

6.2 New studies

The present work has highlighted a number of gaps in our knowledge of the Loch Eye system. The following are considered a priority in that they will improve not only the understanding of how this particular site functions, but how to manage it to the advantage of botanical and zoological conservationists, anglers and ornithologists alike.

The present study suggests that if eutrophication is to be stemmed, something should be done about the external sources of nutrients i.e. from roosting wildfowl (geese) and runoff from the land. The Garrick catchment is the source of most of the stream-borne P entering the loch, but the smaller, and thus probably more manageable, Erracht drainage area supplies about one-third of the total stream-borne P loading, and just over one-half of the N. As the prospect of limiting the access of geese to the loch is somewhat daunting (although guarding certain areas could form the basis of interesting experiments to test the effects of shutting off this nutrient supply), perhaps some thought should be given to controlling the inputs from all of the streams.

Each of these watercourses is small, so techniques being developed elsewhere and involving, for example, bales of straw to form a substrate for microflora to 'strip' water of nutrients, might be well worth trying. Present trials focus on the use of straw as a potential algicide, but the mode of straw action in reducing algal growth, and the optimum application regimes is still little understood (Barrett *et al.*, 1990; Welch *et al.*, 1990). Evidently, bacterial microflora and their exudates are important, as nutrients are not removed (Gibson *et al.*, 1990).

Another option for Loch Eye, is to widen the mouths of the incoming streams, in order to create broad, shallow substrates (of perhaps stones or artificial materials - even glass slides on a large scale) to which algae would attach and build up a nutrient-removal unit. This idea stems from knowledge of literature on the use of glass slides for assessing water quality on the basis of the colonising algal species - diatoms in particular (see e.g. Tippett, 1970). As far as we are aware, however, the techniques have not been put into practice on the scale envisaged here. As with straw bales, these substrates would need to be cleaned or 're-vitalised' in some way at appropriate intervals.

A third possibility would capitalise on the probable (but as yet untested) nutrient sequestering activities of existing macrophytes. Some re-planting may be necessary, but the potential use of macrophyte beds in this context at the 'estuaries' of the inflowing streams, should be explored. A large literature (particularly on tropical situations, e.g. Denny, 1985) points to the efficiency of macrophyte beds in removing nutrients from water passing through them. Emergent hydrophytes have also been used for sewage treatment for more than a century, and more recently, various forms of root-zone and gravel-bed hydroponic systems have been developed (e.g. Butler *et al.*, 1990); it is not suggested that these would be definitely appropriate for the Loch Eye situation, however.

Research by means of field experiments is necessary to investigate the feasibility and efficacy of these management proposals: management-orientated research is being carried out at the Norfolk Broads, for example, but while these sites are shallow like Loch Eye, and birds feature strongly in their eutrophication, the gross morphology of the systems and the species of birds involved differ considerably.

There are other studies that would break new ground as far as Loch Eye is concerned, and these might look closely at the population dynamics and production of organisms at different trophic levels, e.g. attached algae, macrophytes, zooplankton, zoobenthos and fish. However, work on the spatial and temporal distribution of attached algae would appear to be the area warranted most; this should include especially the communities likely to exist on, or near the surface of the sediments, and those comprising the epiphytic cover of macrophytes. Plainly, resources sufficient to execute all these programmes are unlikely to be available, but the studies might be tackled piece-meal.

7. ACKNOWLEDGEMENTS

While some valuable data on loch levels and stream heights were lost, the assistance of Dr R W Graham and Mr A Prickett was invaluable. Mr J A Douglas-Menzies, Mr G Ross, Mrs S Ross, Mr J Ross, Mrs D MacDonald and Mrs E U Cummings allowed us access to the loch across their land. We are especially grateful to Dr Gillian Kendrick for taking water samples between our major field visits. We much appreciate the kind hospitality received from all of these people and from the proprietors of the Balintore Hotel, Mr and Mrs Kennedy, and Mr and Mrs Redworth who also provided space for our laboratory work. Mr T Inglis and Mr D Brown of the Highland River Purification Board directed the installation of the staff gauges on the streams, the metering of flows, and the derivation of the rating curves. Dr Kendrick and Mr D L Howell (now NCC) and Mr A A Lyle (IFE) also helped with the major field programmes. We thank Dr Linda May (IFE) for drawing Figure 1, and Mr T Furnass (IFE, Windermere) for preparing the excellent Figure 24. We are most grateful to Mrs Elma Lawrie and Mrs Marjorie Ferguson for typing the report - their patience was severely tested! Finally, we acknowledge the continued interest in the work shown by NCC staff, particularly Dr A J Watson and Mr P Wortham (Inverness) and Mrs Mary Gibson and Dr P Boon (Peterborough), and Mr D L Howell (Edinburgh).

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F I G U R E S

Figure 1(a)
Loch Eye sampling locations, main inflows and approximate mean discharges; the position of the loch in Scotland is also shown.

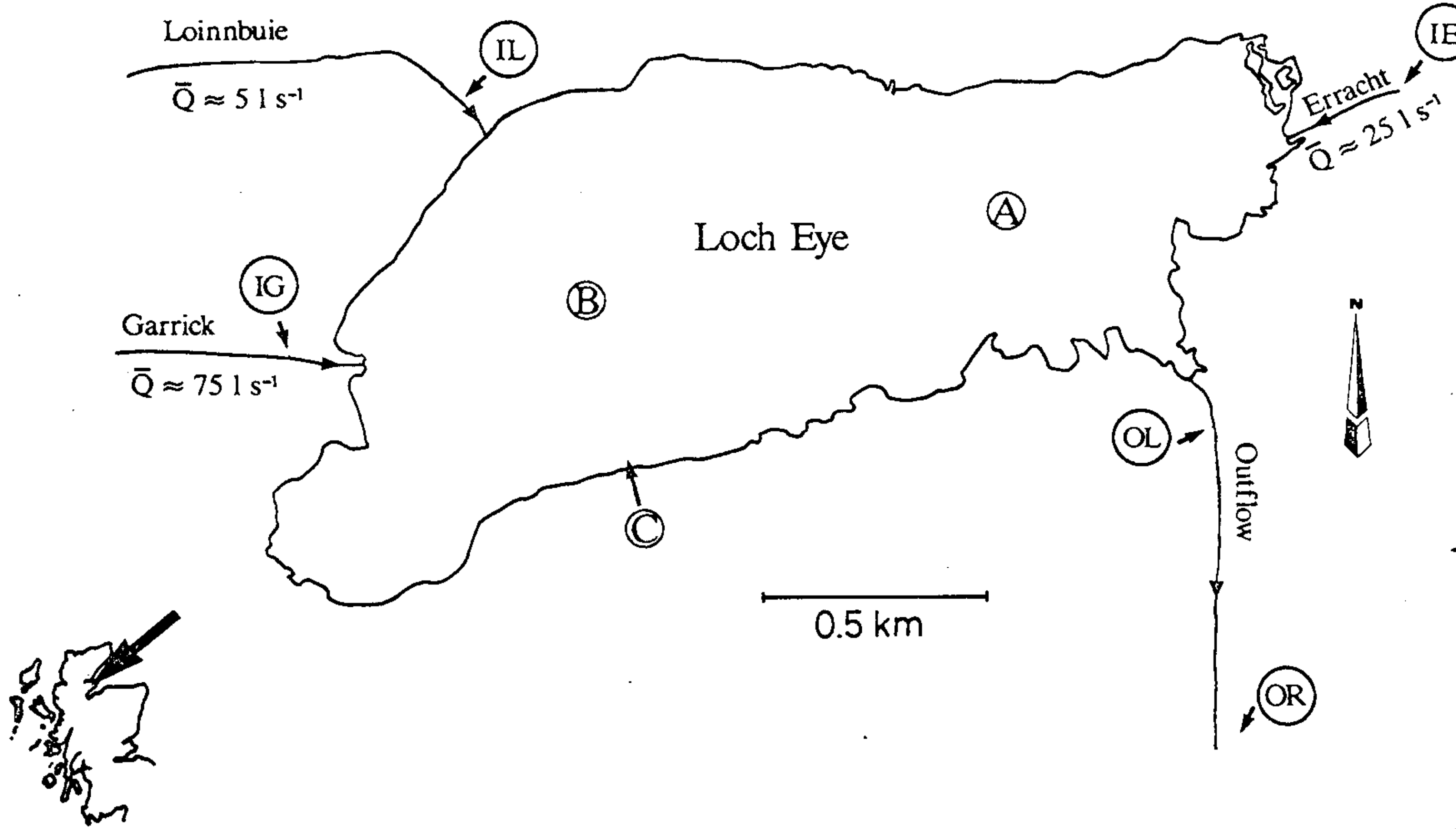
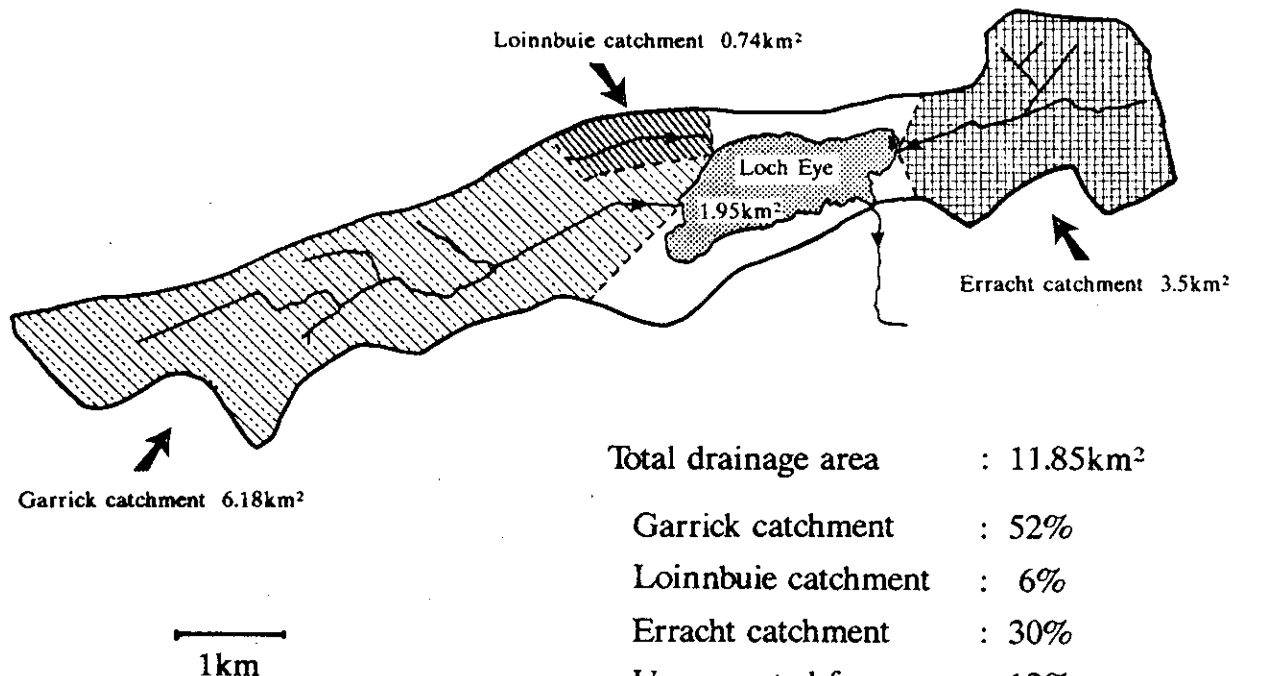

Loinnbuie
Q̄ ≈ 5 l s⁻¹
IL
IE
Erracht
Q̄ ≈ 25 l s⁻¹
A
Loch Eye
B
IG
Garrick
Q̄ ≈ 75 l s⁻¹
N
OL
Outflow
C
0.5 km
OR

Figure 1(b)
Some features of the Loch Eye catchment



Total drainage area	: 11.85km²
Garrick catchment	: 52%
Loinnbuie catchment	: 6%
Erracht catchment	: 30%
Unaccounted for	: 12%

Ratio of total drainage area to loch area 6.1:1

Figure 2(a)

Rainfall near L.Eye 1986 to 1988

Geanies met. station

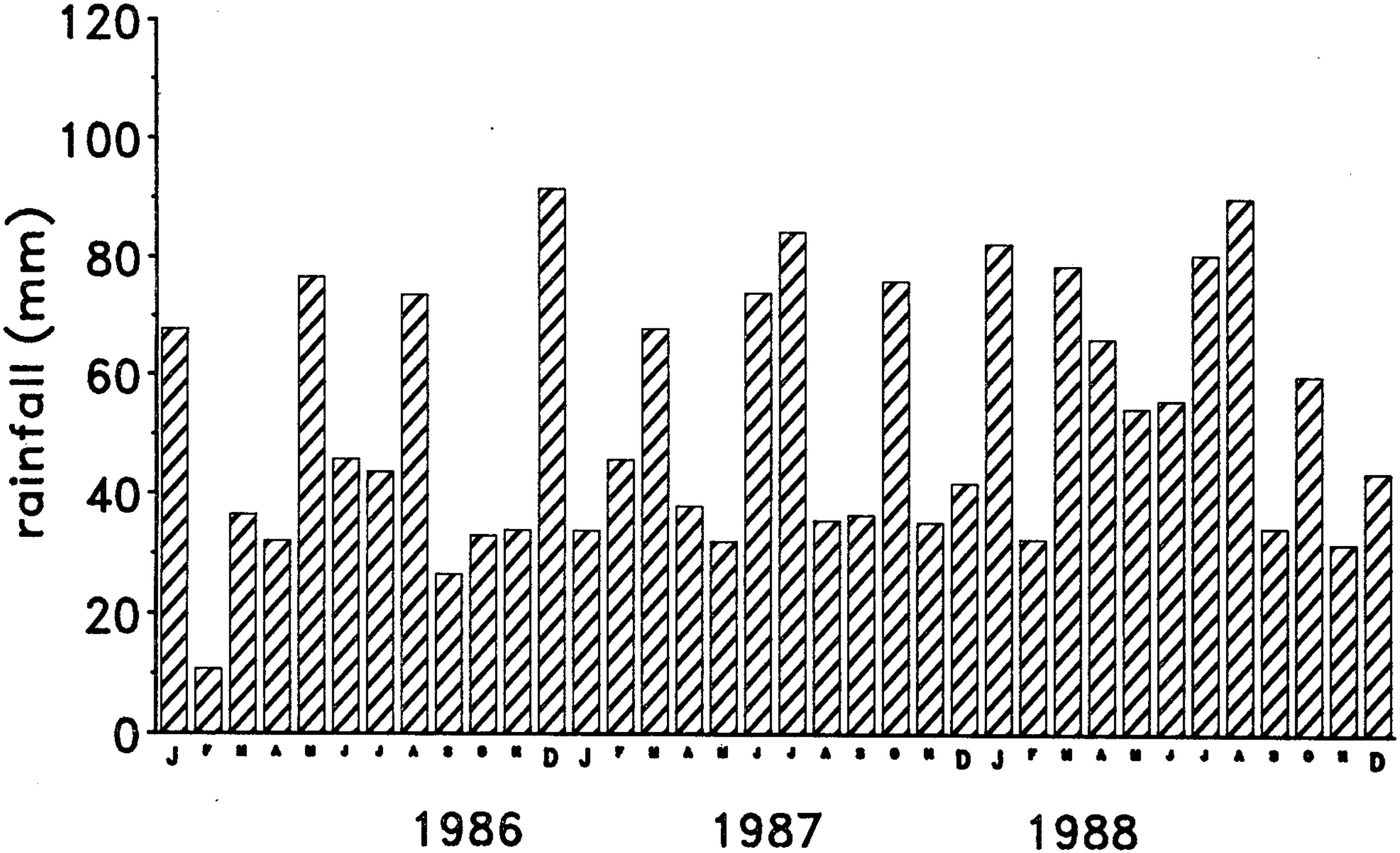


Figure 2(b)

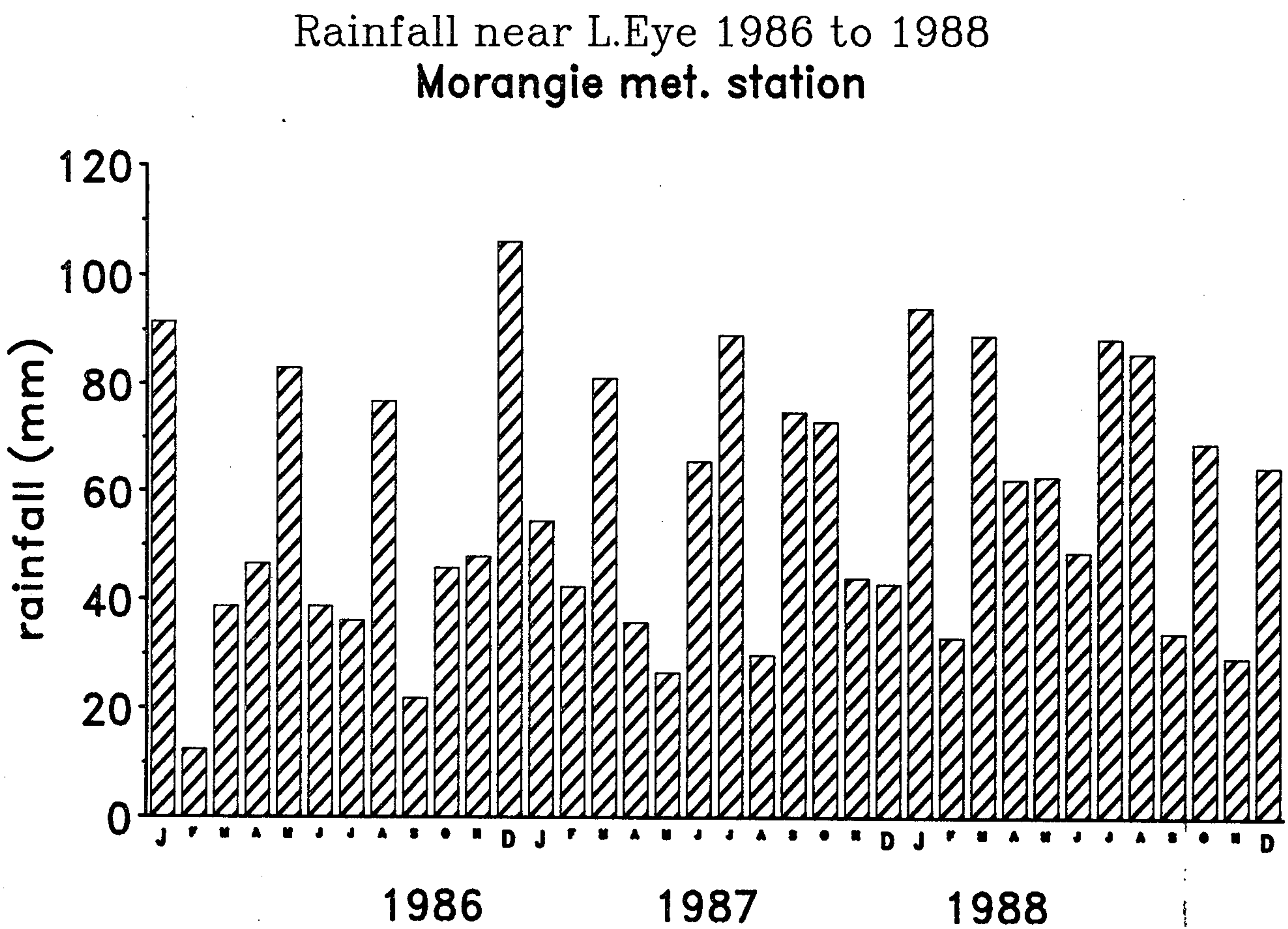
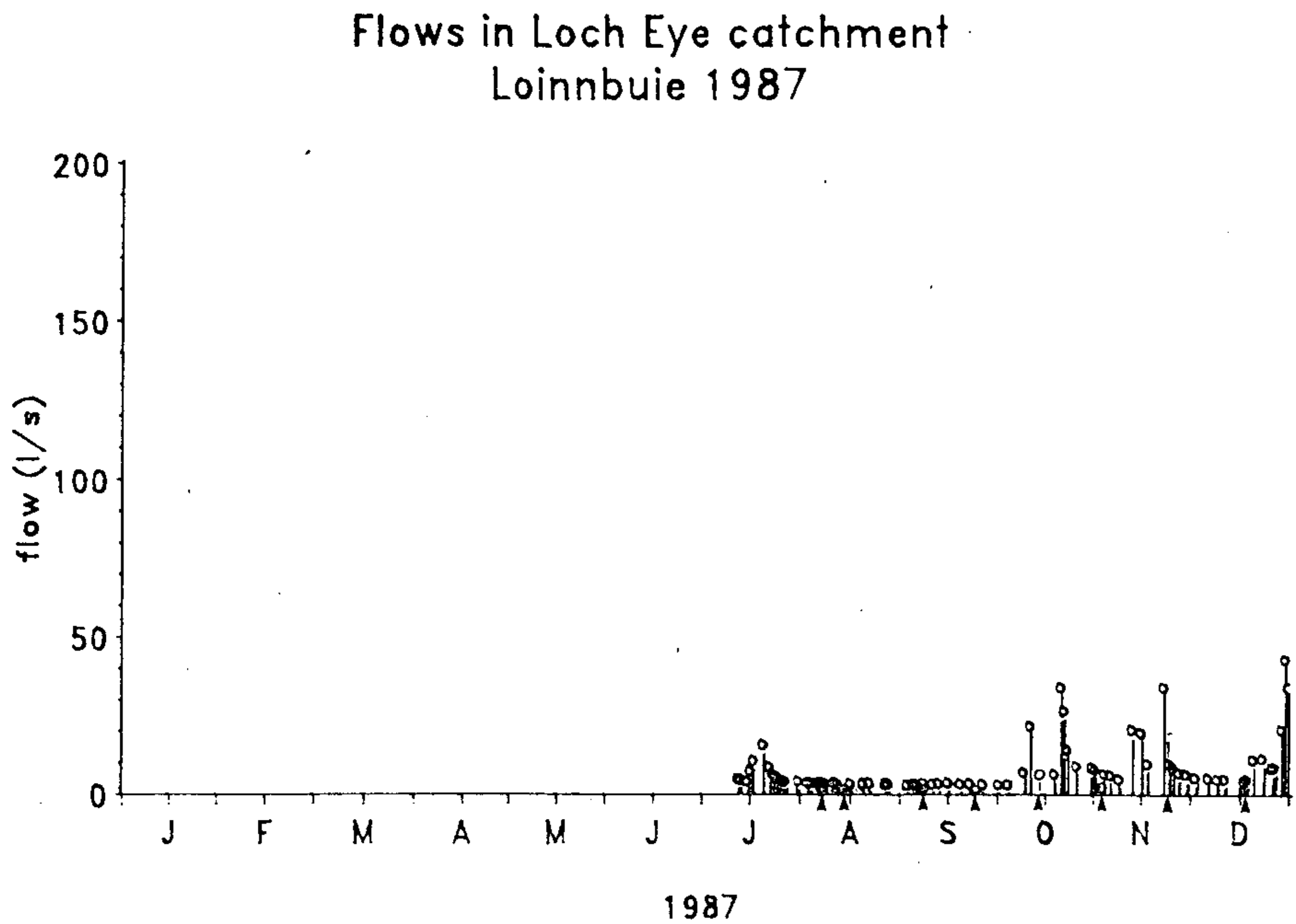
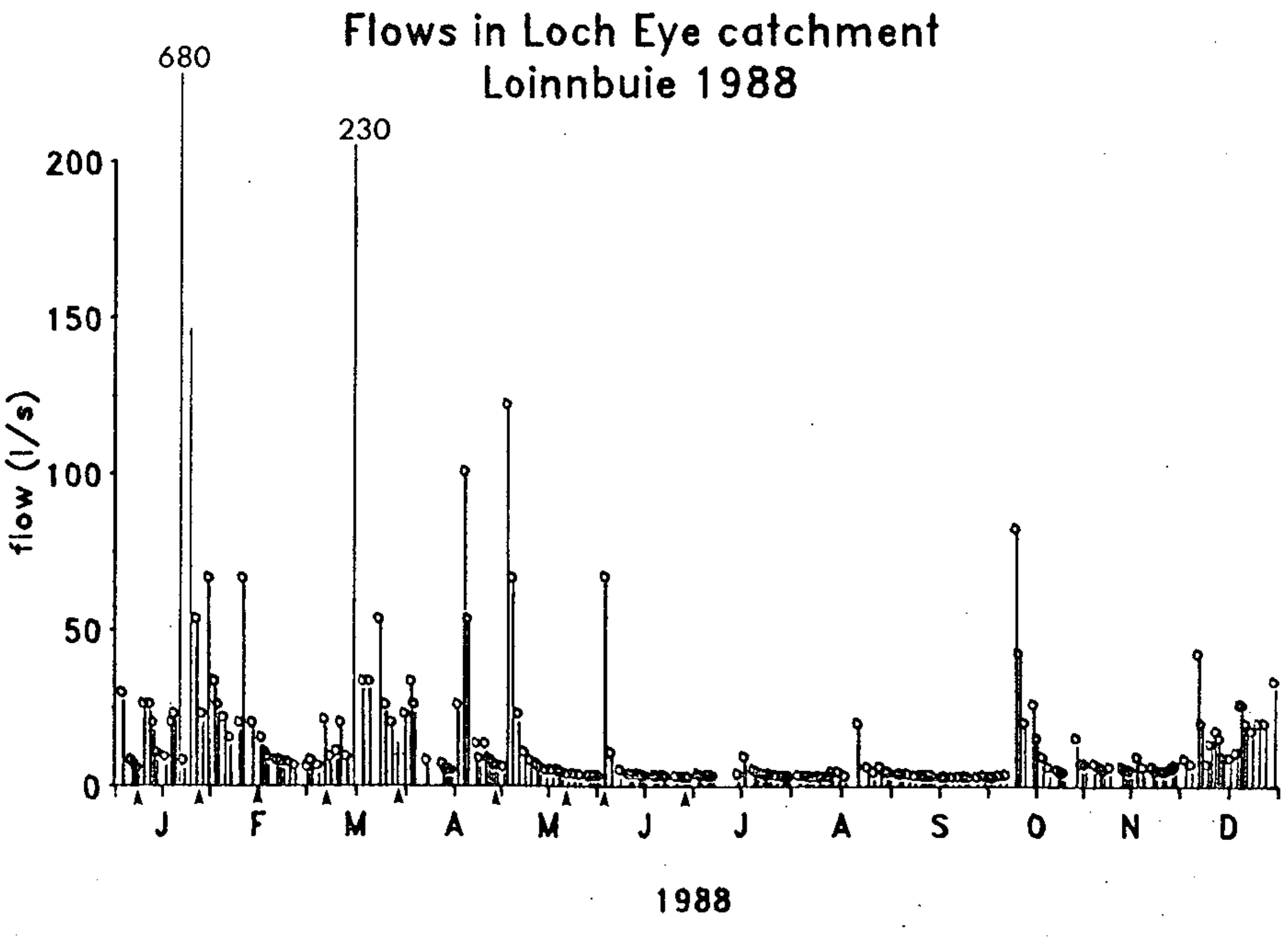

Rainfall near L.Eye 1986 to 1988
Morangie met. station
rainfall (mm)
0
20
40
60
80
100
120
J
D
J
D
J
D
1986
1987
1988

Figure 3(a)
Part 1
Arrow points indicate occasions of
chemical and biological sampling

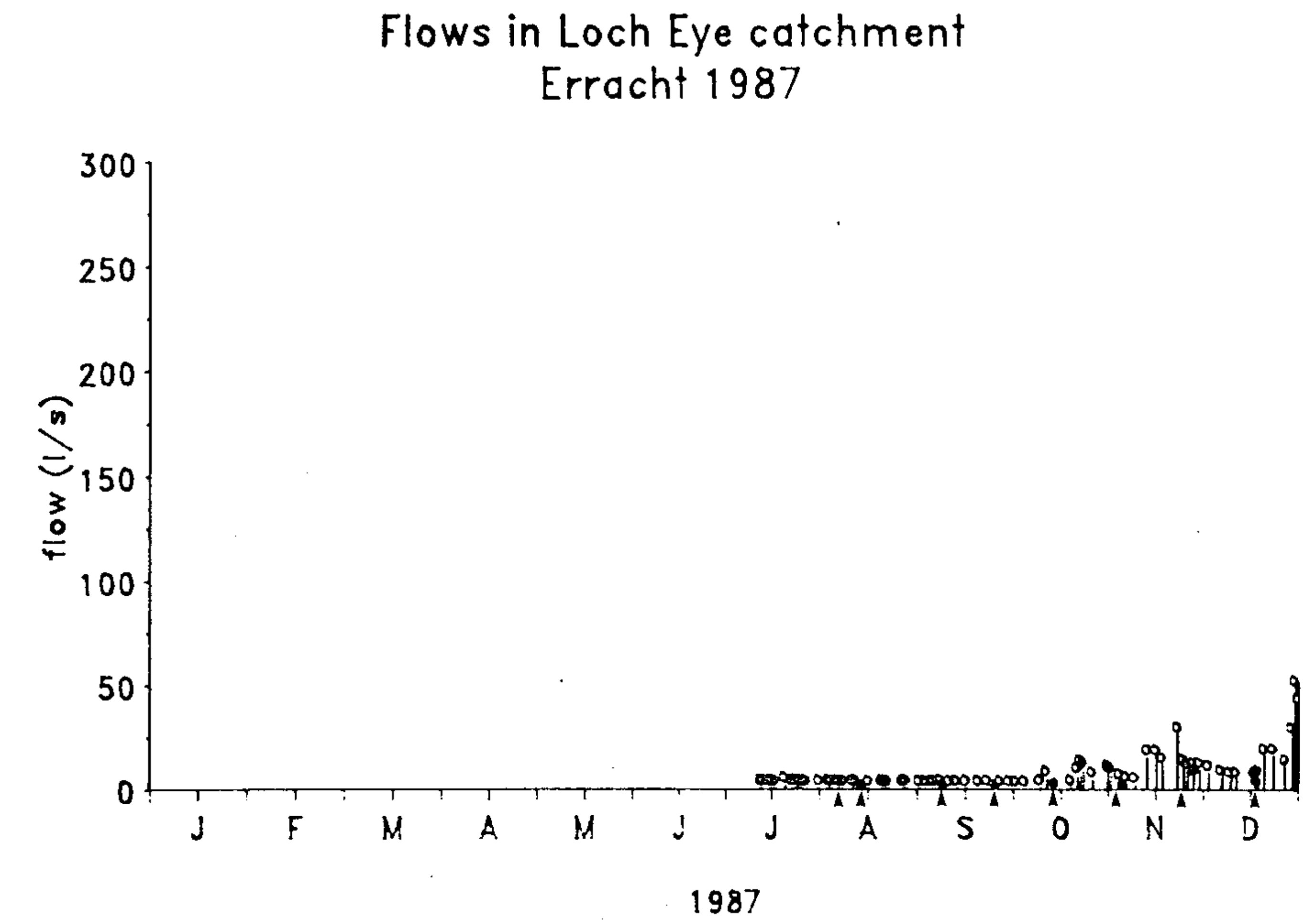


Flows in Loch Eye catchment
Loinnbuie 1987
200
150
100
50
0
flow (l/s)
J F M A M J J A S O N D
1987



Flows in Loch Eye catchment
Loinnbuie 1988
680
230
200
150
100
50
0
flow (l/s)
J F M A M J J A S O N D
1988

Figure 3(a)
Part 2
Arrow points indicate occasions of
chemical and biological sampling


Flows in Loch Eye catchment
Erracht 1987
flow (l/s)
300
250
200
150
100
50
0
J
F
M
A
M
J
J
A
S
O
N
D
1987

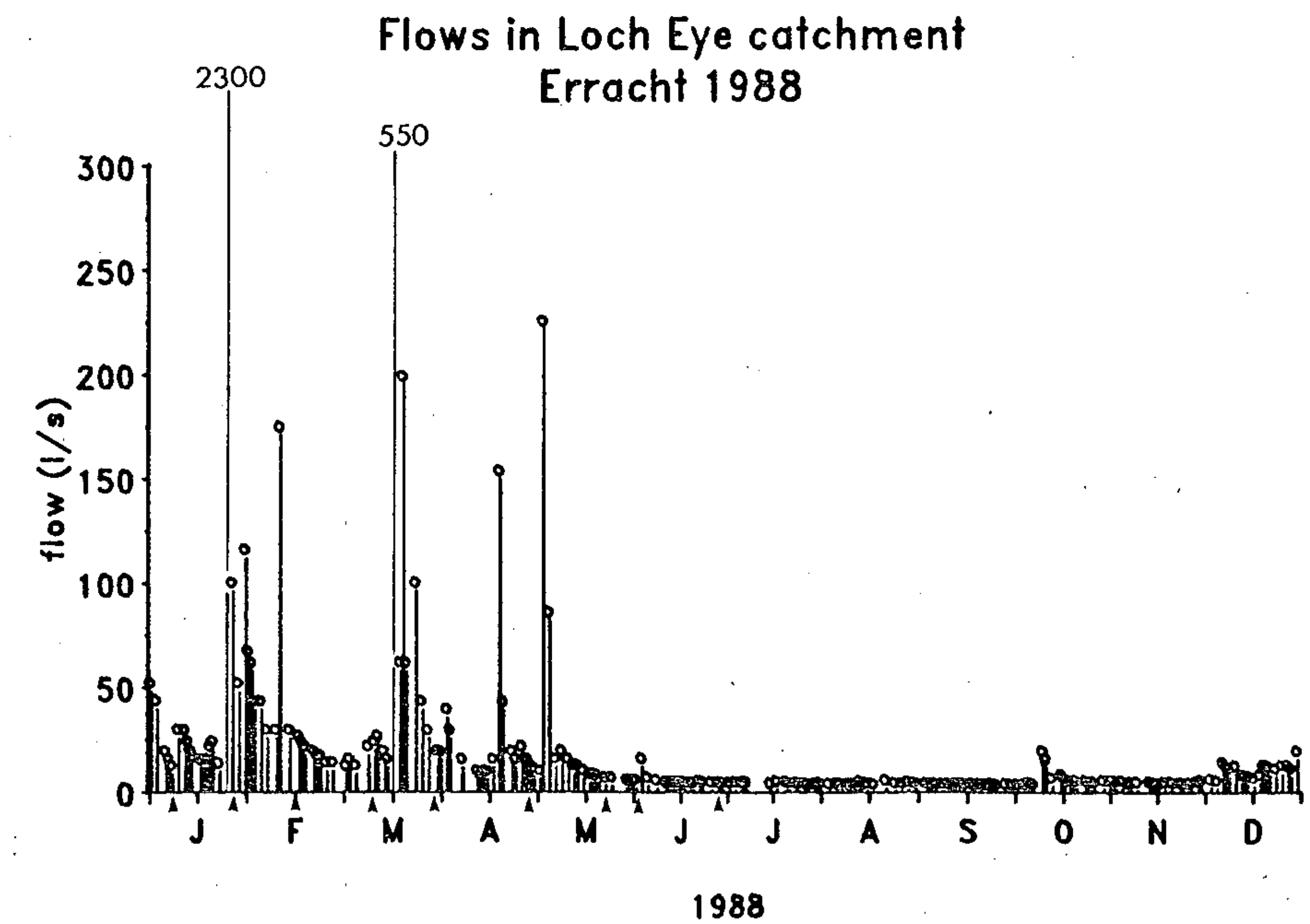
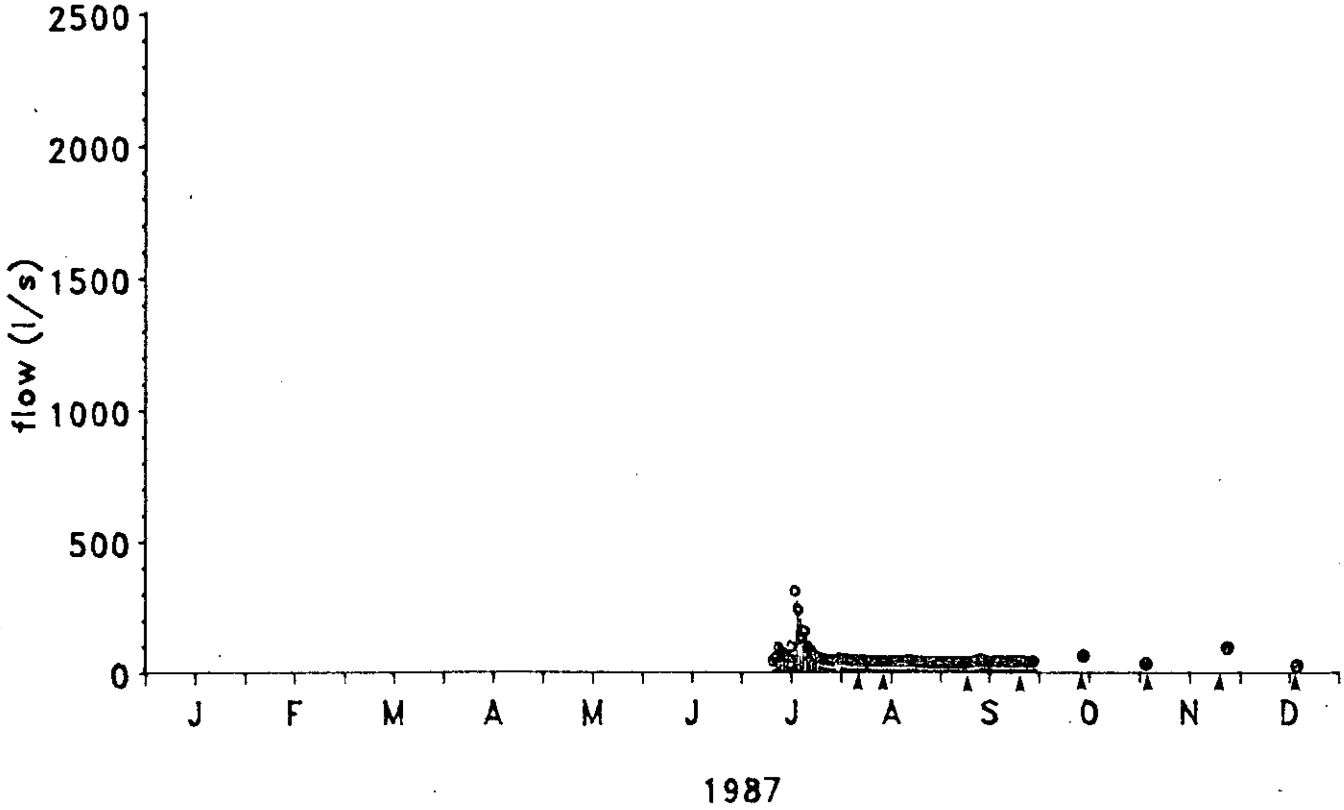

Flows in Loch Eye catchment
Erracht 1988
2300
550
flow (l/s)
300
250
200
150
100
50
0
J
F
M
A
M
J
J
A
S
O
N
D
1988

Figure 3(a)
Part 3
Arrow points indicate occasions of
chemical and biological sampling

Flows in Loch Eye catchment
Garrick 1987



Flows in Loch Eye catchment
Garrick 1988

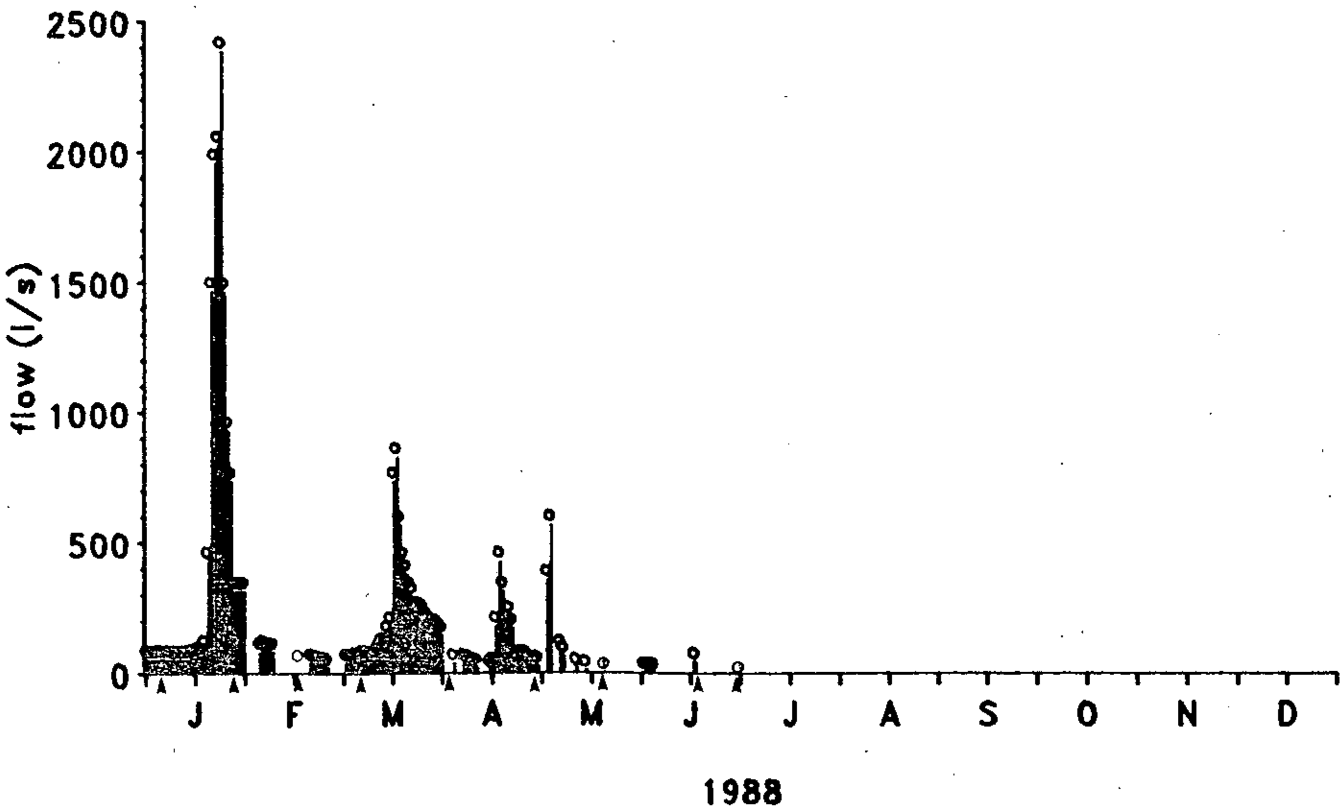


Figure 3(b)
Arrows indicate occasions of
chemical and biological sampling

Flows in Loch Eye catchment
Outflow 1988

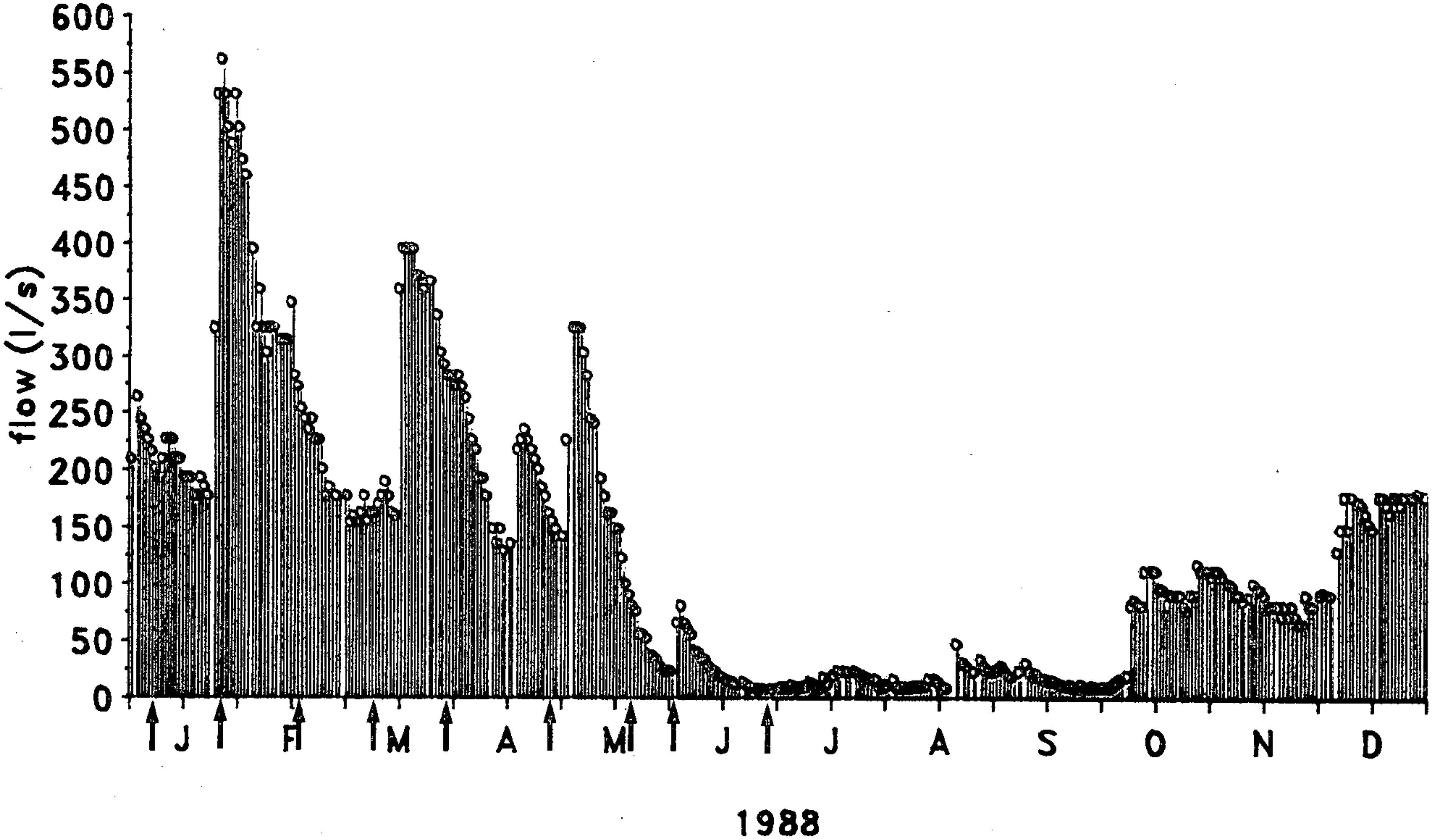


Figure 3(c)

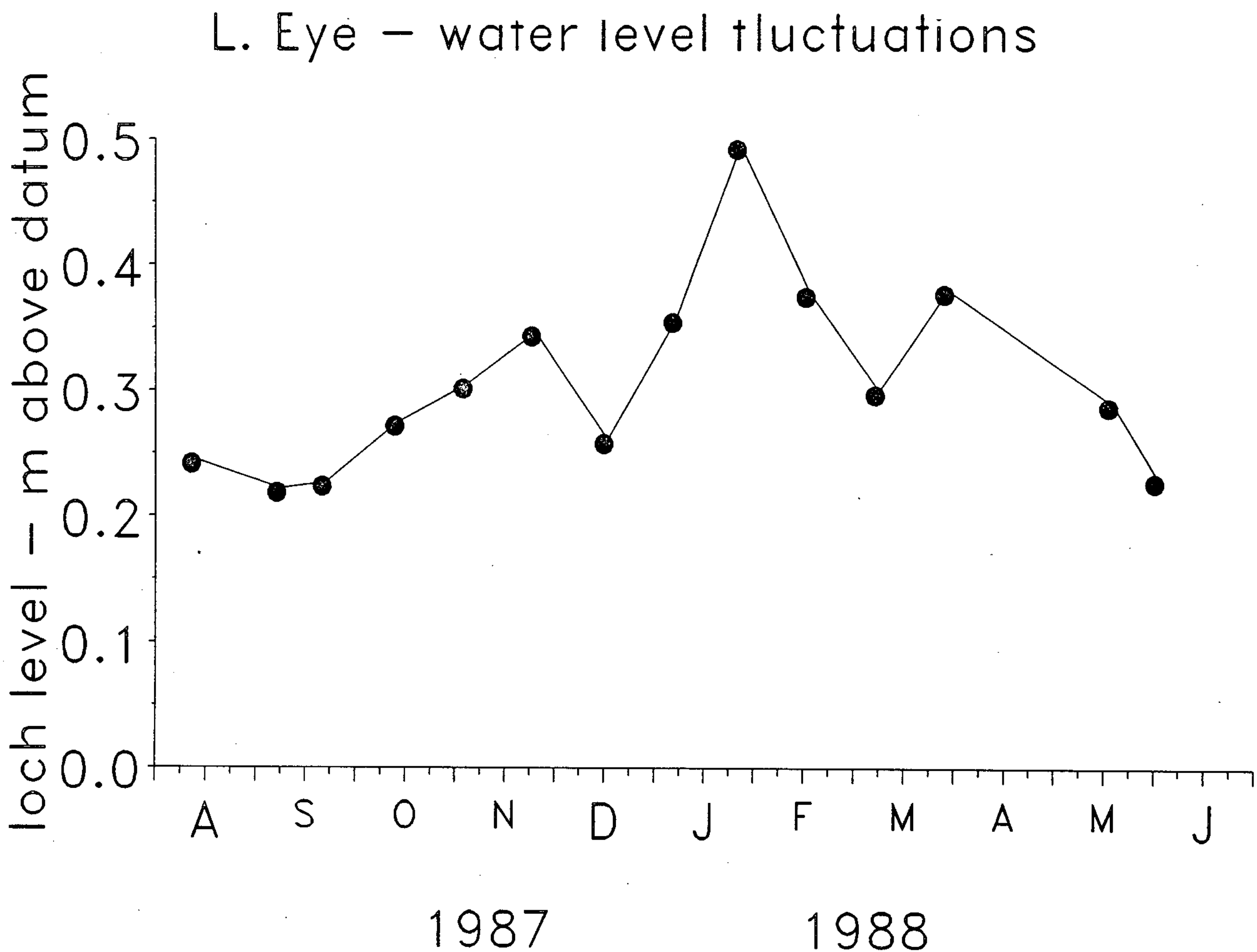

L. Eye – water level fluctuations
loch level – m above datum
0.5
0.4
0.3
0.2
0.1
0.0
A
S
O
N
D
J
F
M
A
M
J
1987
1988

Figure 4(a)

Surface water temperatures at Loch Eye
Garrick ( ), Loinnbuie (°) and Erracht (▲)

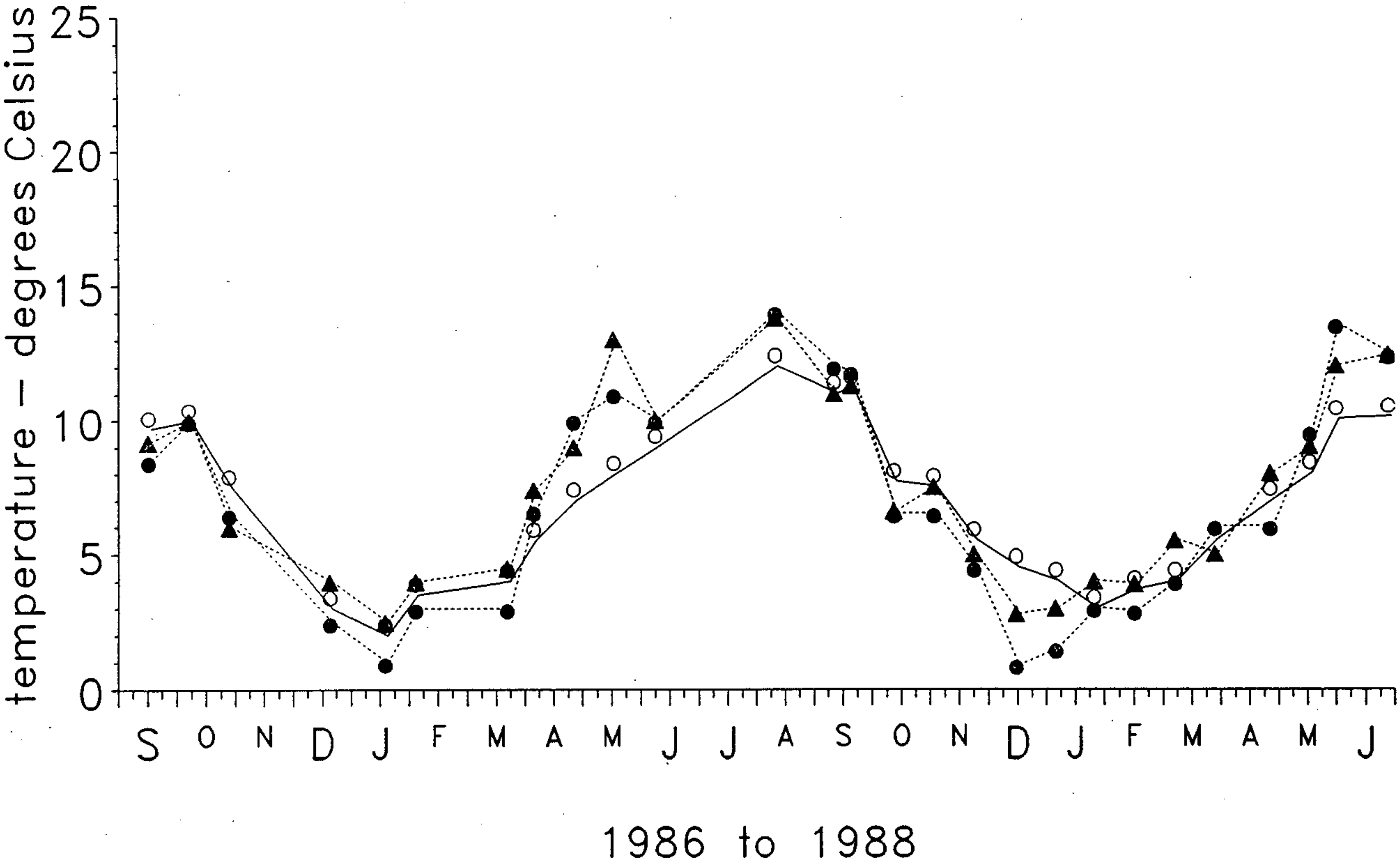


Figure 4(b)

Surface water temperatures at Loch Eye loch sites A (•), B (°), C (▲) and outflow (★).

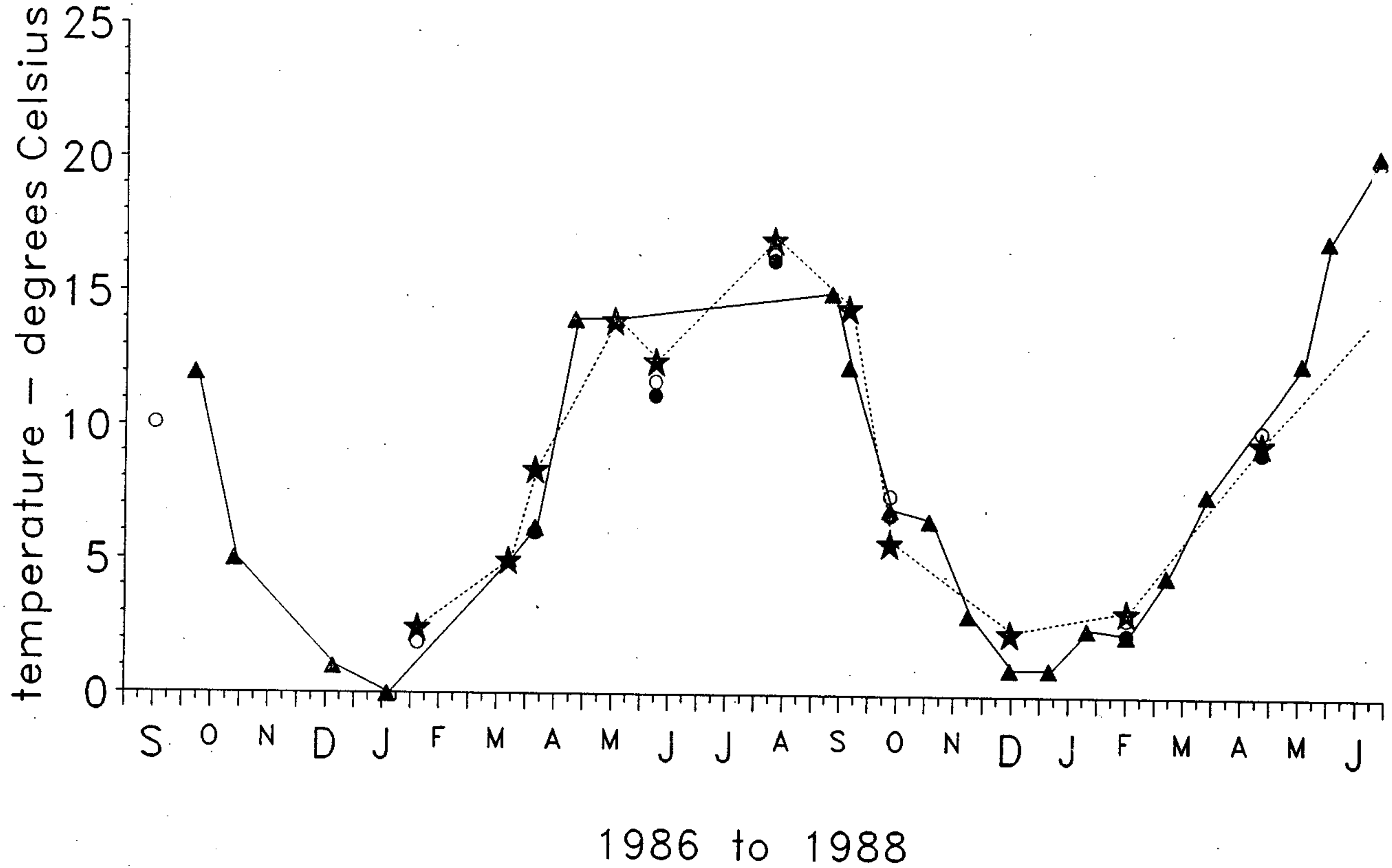


Figure 5(a)

Attenuation of light at Loch Eye
orange (•), red (°), green (▲) and blue (★)

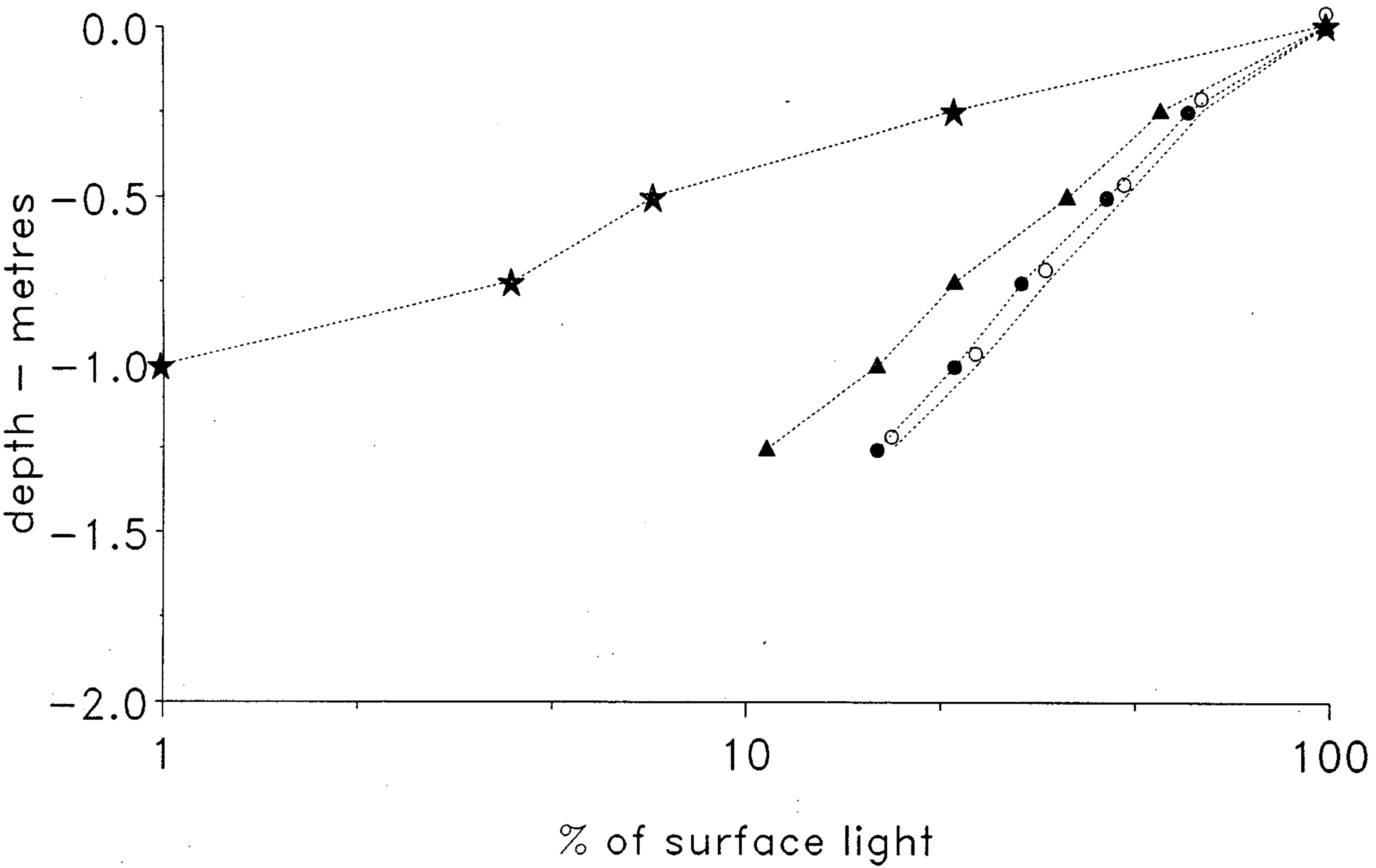


Figure 5(b)

Light attenuation coefficients in Loch Eye

orange (•), red (°), green (▲) and blue (★); euphotic depth (☆).

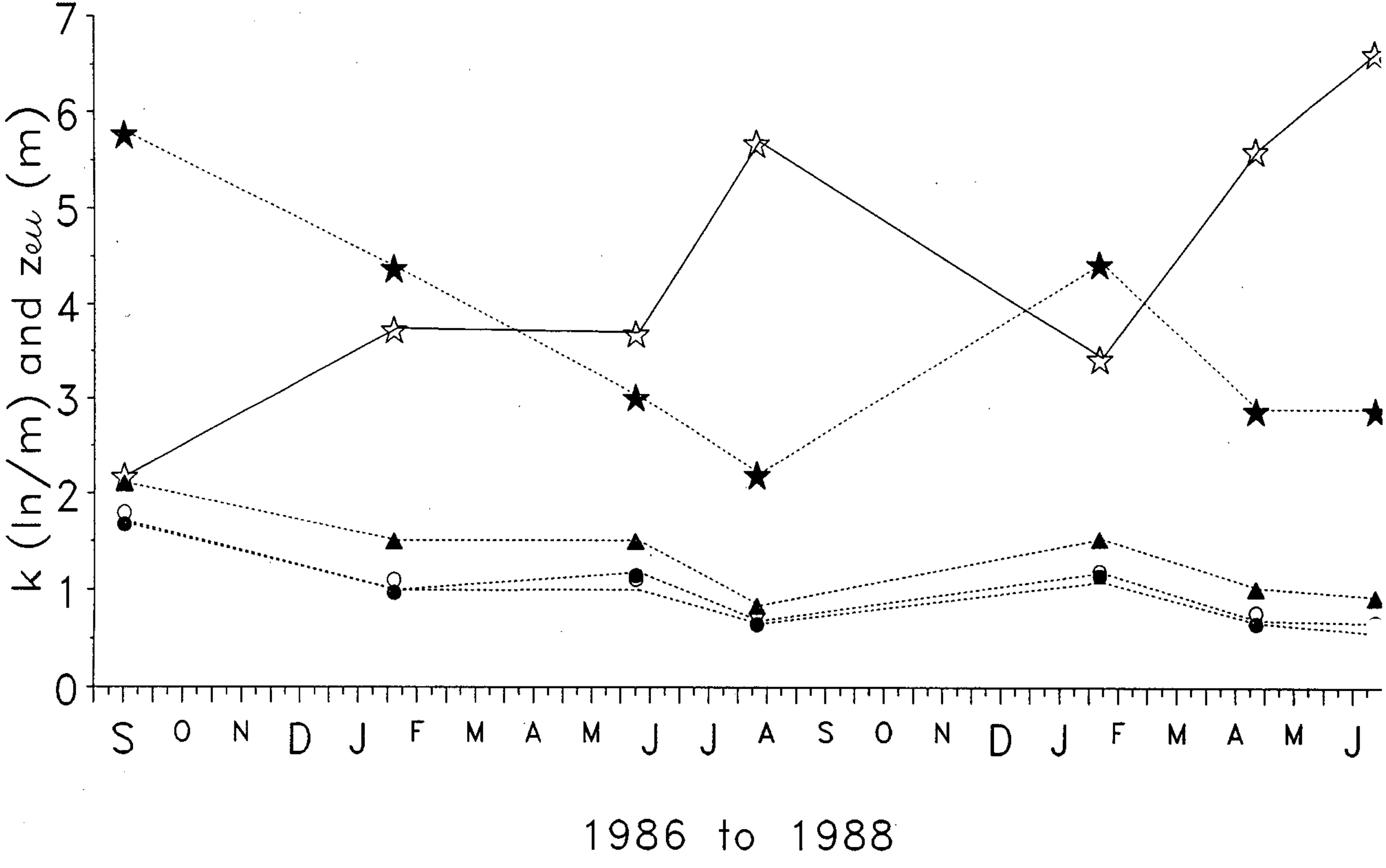


Figure 5(c)

Attenuation of red light (k *min*) at Loch Eye

9/86 (•), 4/87 (°), 6/87 (▲), 8/87 (★), 2/88 (☆), 4/88 (▪) and 6/88 (*)

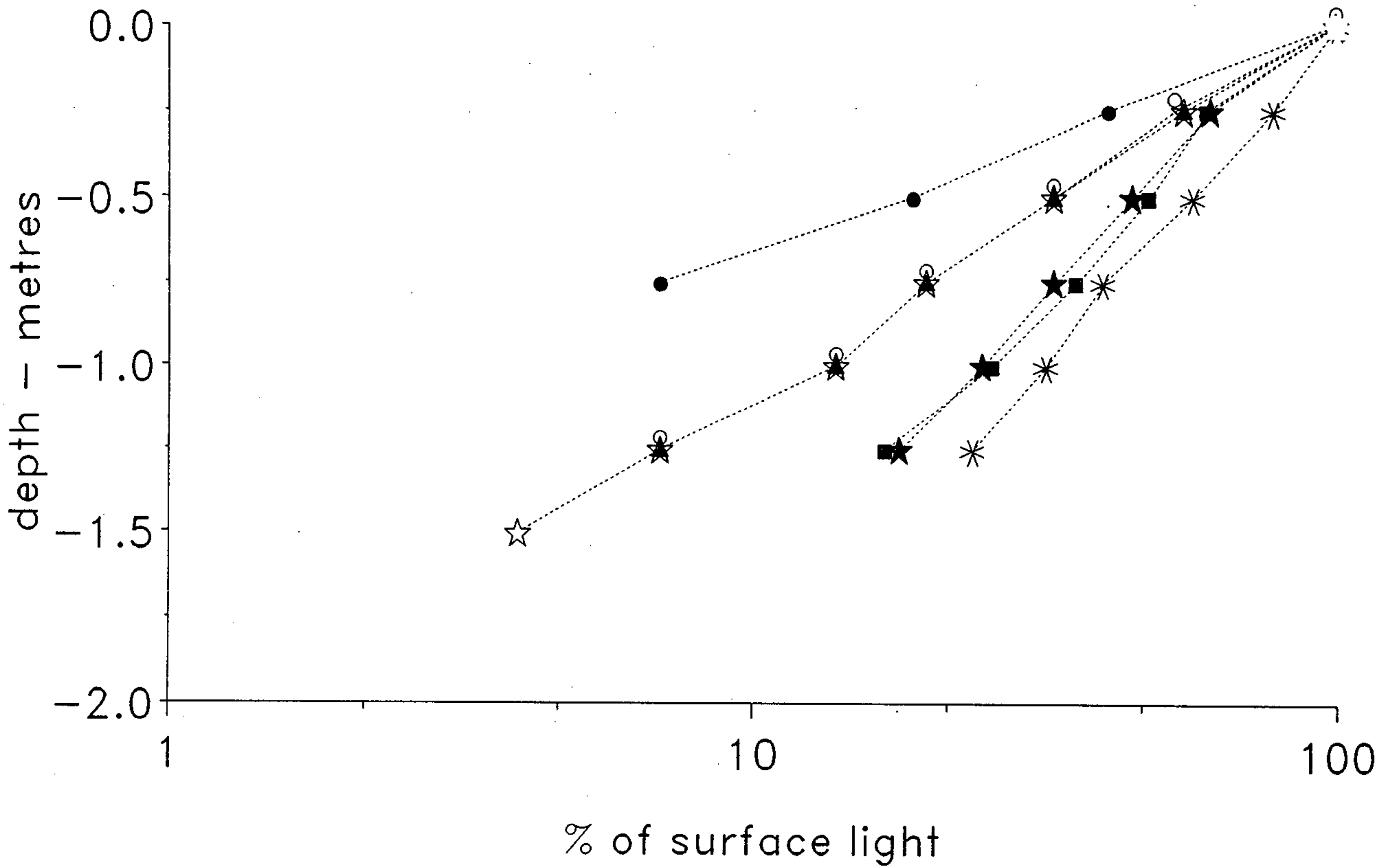


Figure 6(a)

pH of Loch Eye inflows
Garrick ( ), Loinnbuie (°) and Erracht (▲)

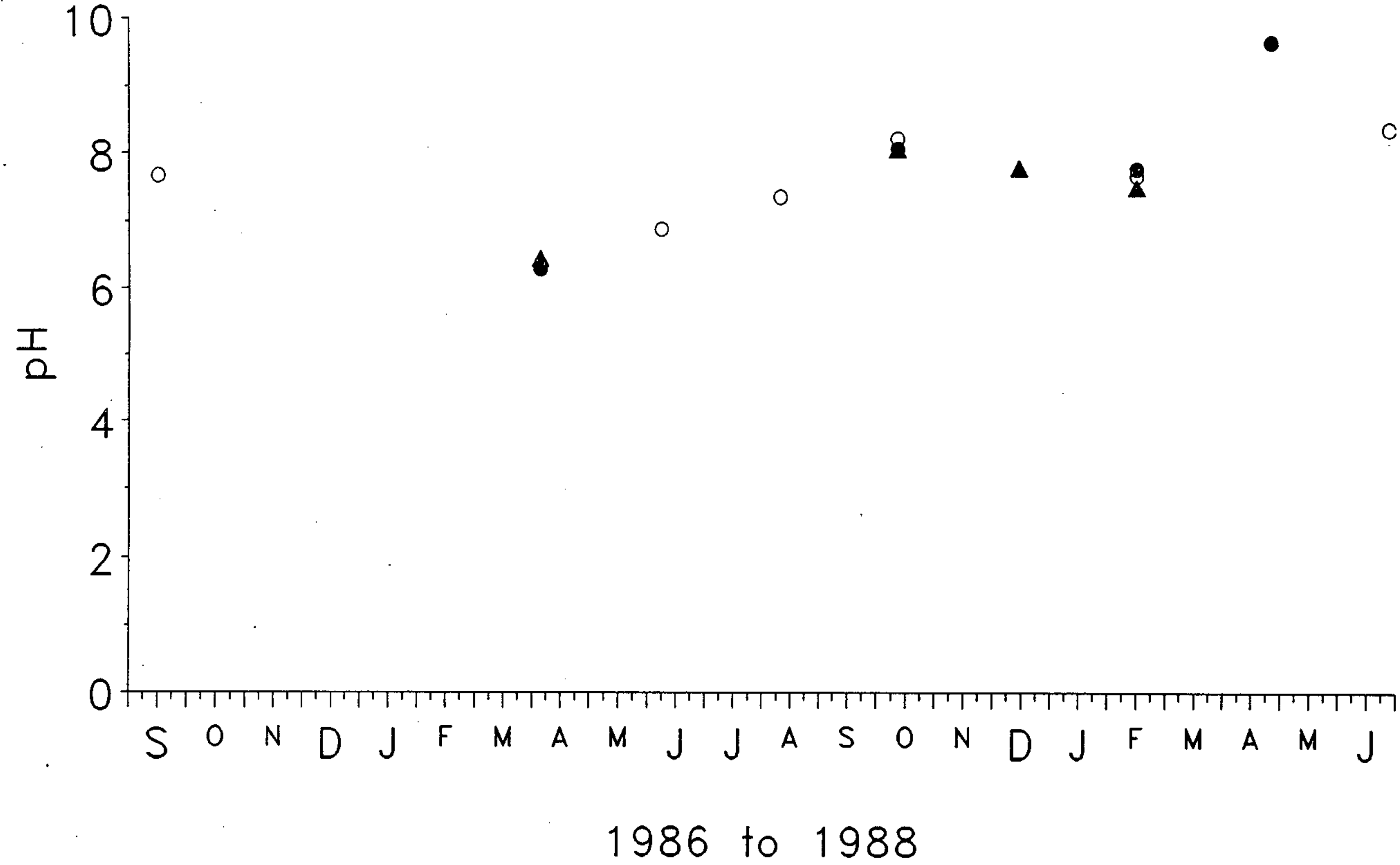


Figure 6(b)

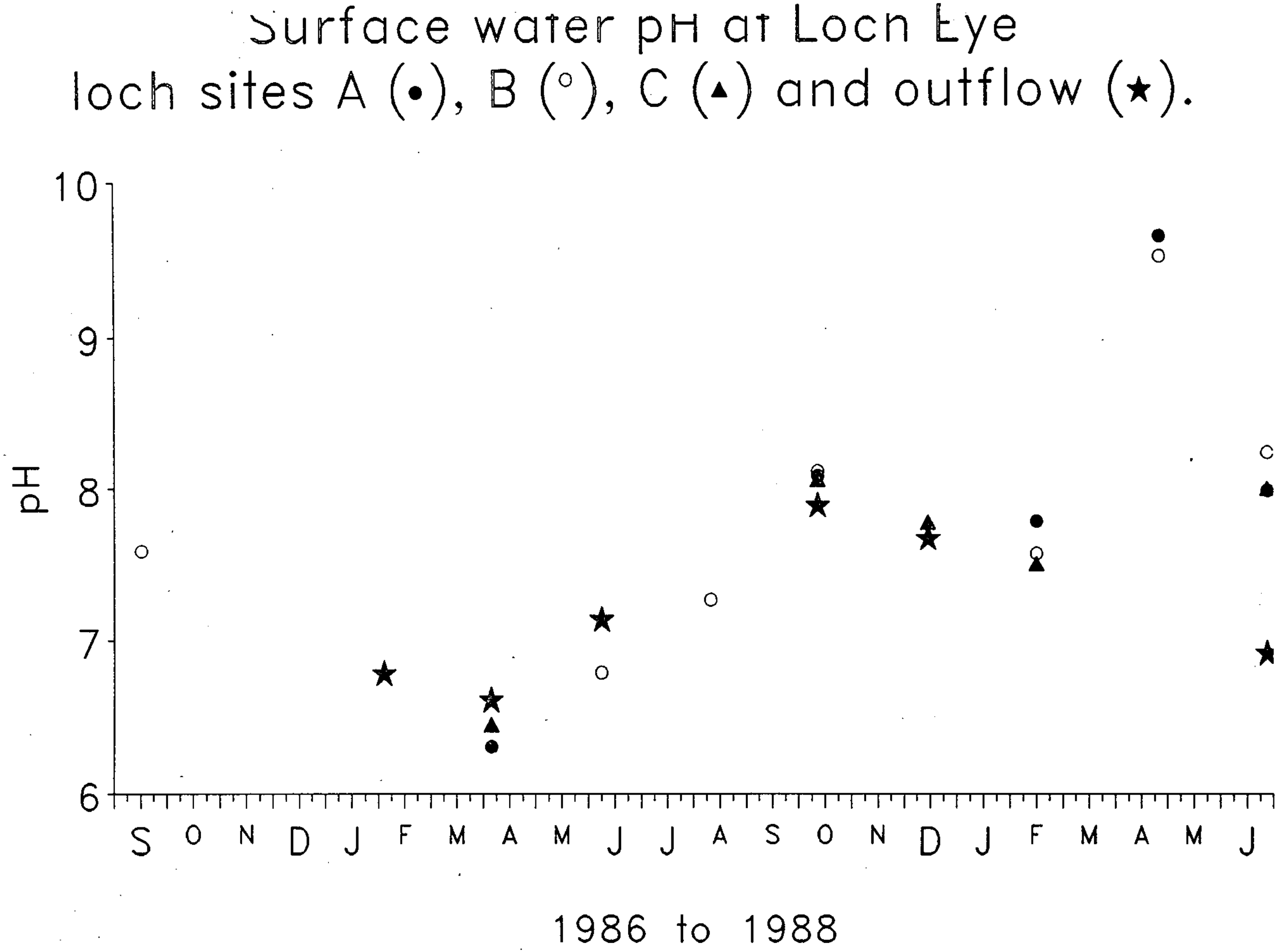

Surface water pH at Loch Eye
loch sites A (•), B (°), C (▲) and outflow (★).
pH
10
9
8
7
6
S O N D J F M A M J J A S O N D J F M A M J
1986 to 1988

Figure 7(a)

Feeder stream nitrate levels at Loch Eye
Garrick ( ), Loinnbuie (°) and Erracht (▲)

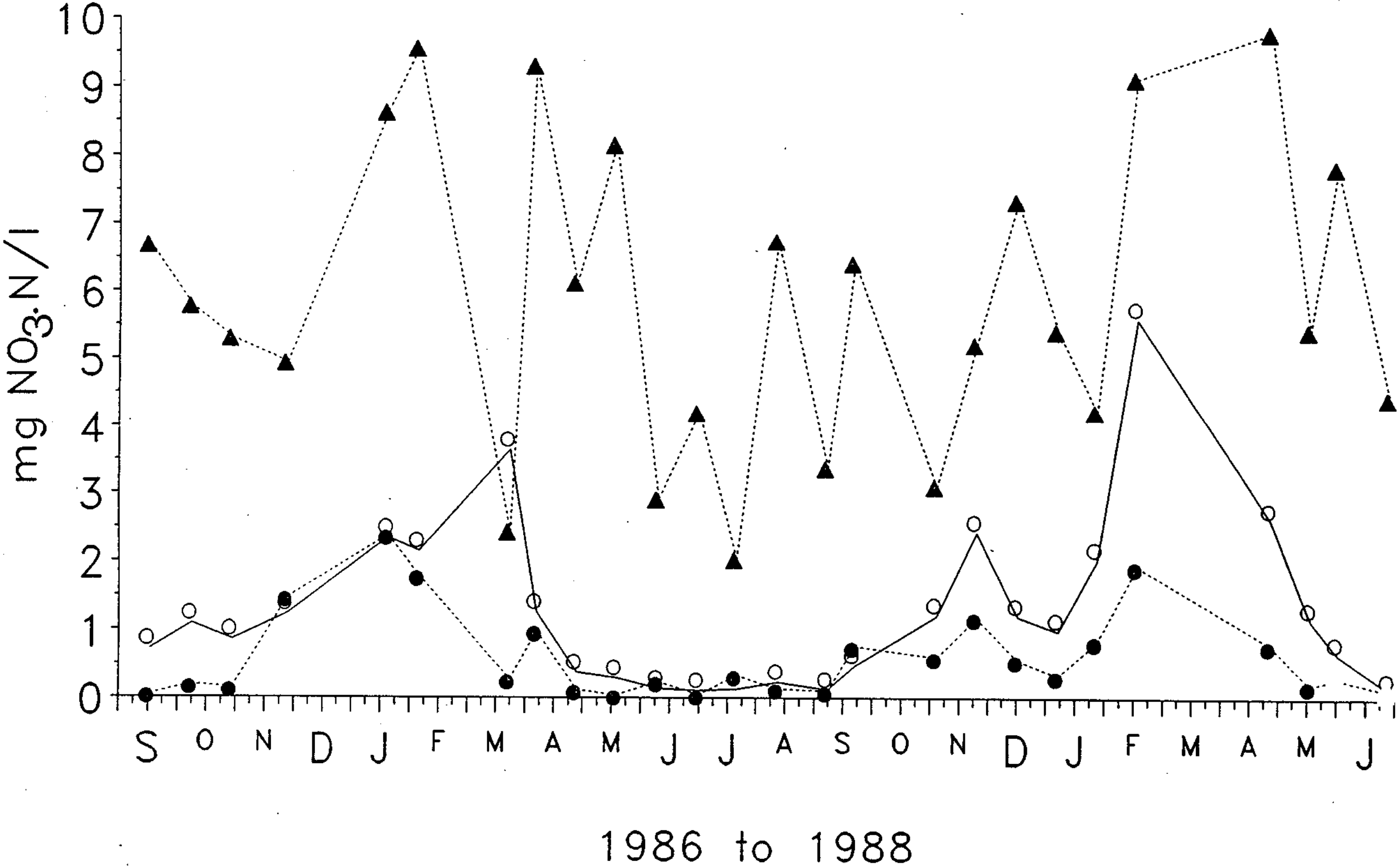


Figure 7(b)

Nitrate concentrations in Loch Eye
A (•), B (°), C (▲) and outflow (★)

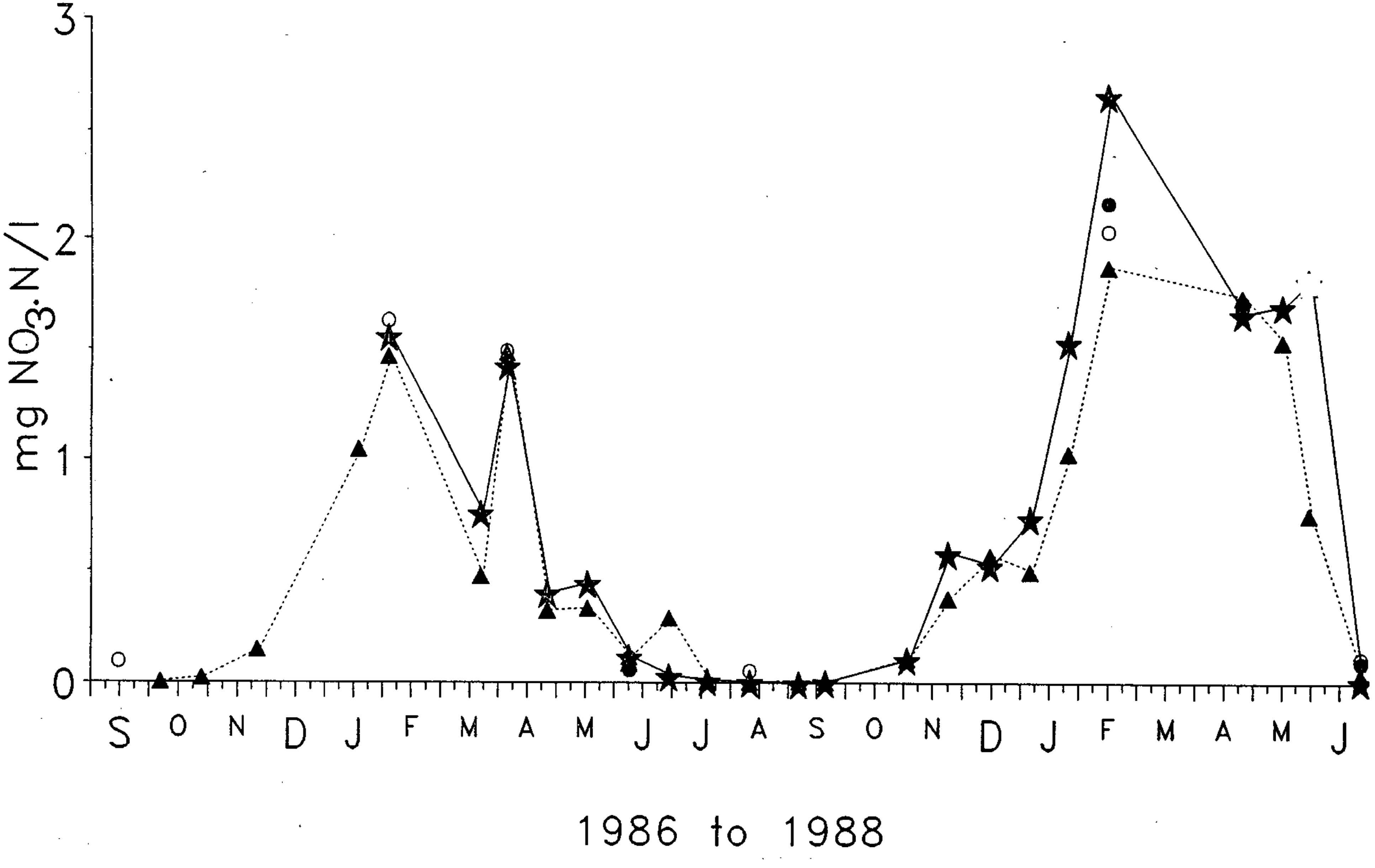
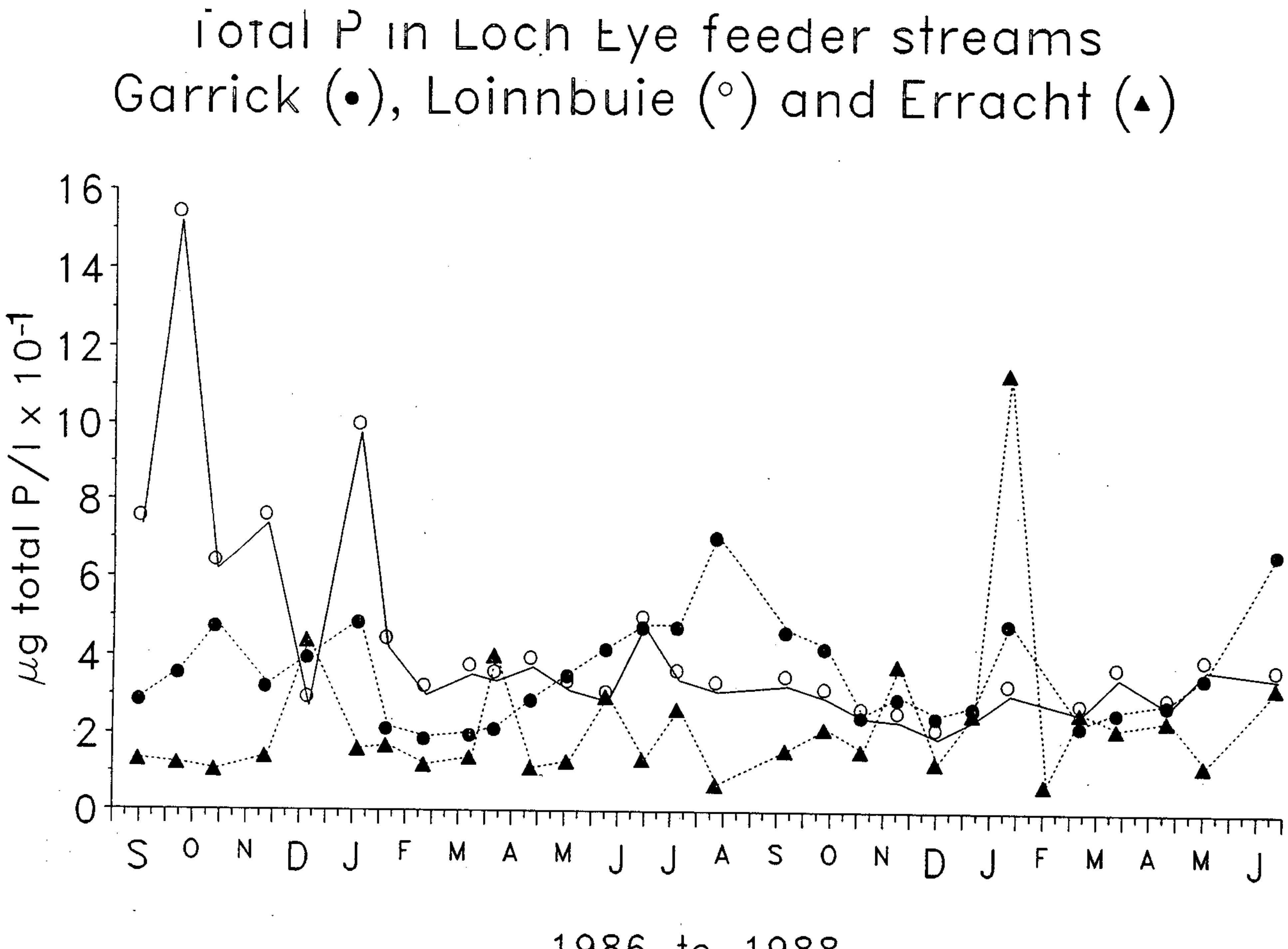


Figure 8(a)



Total P in Loch Eye feeder streams
Garrick (•), Loinnbuie (°) and Erracht (▲)
μg total P/l x 10^{-1}
16
14
12
10
8
6
4
2
0
S O N D J F M A M J J A S O N D J F M A M J
1986 to 1988

Figure 8(b)

Total soluble P in Loch Eye inflows
Garrick (•), Loinnbuie (°) and Erracht (▲)

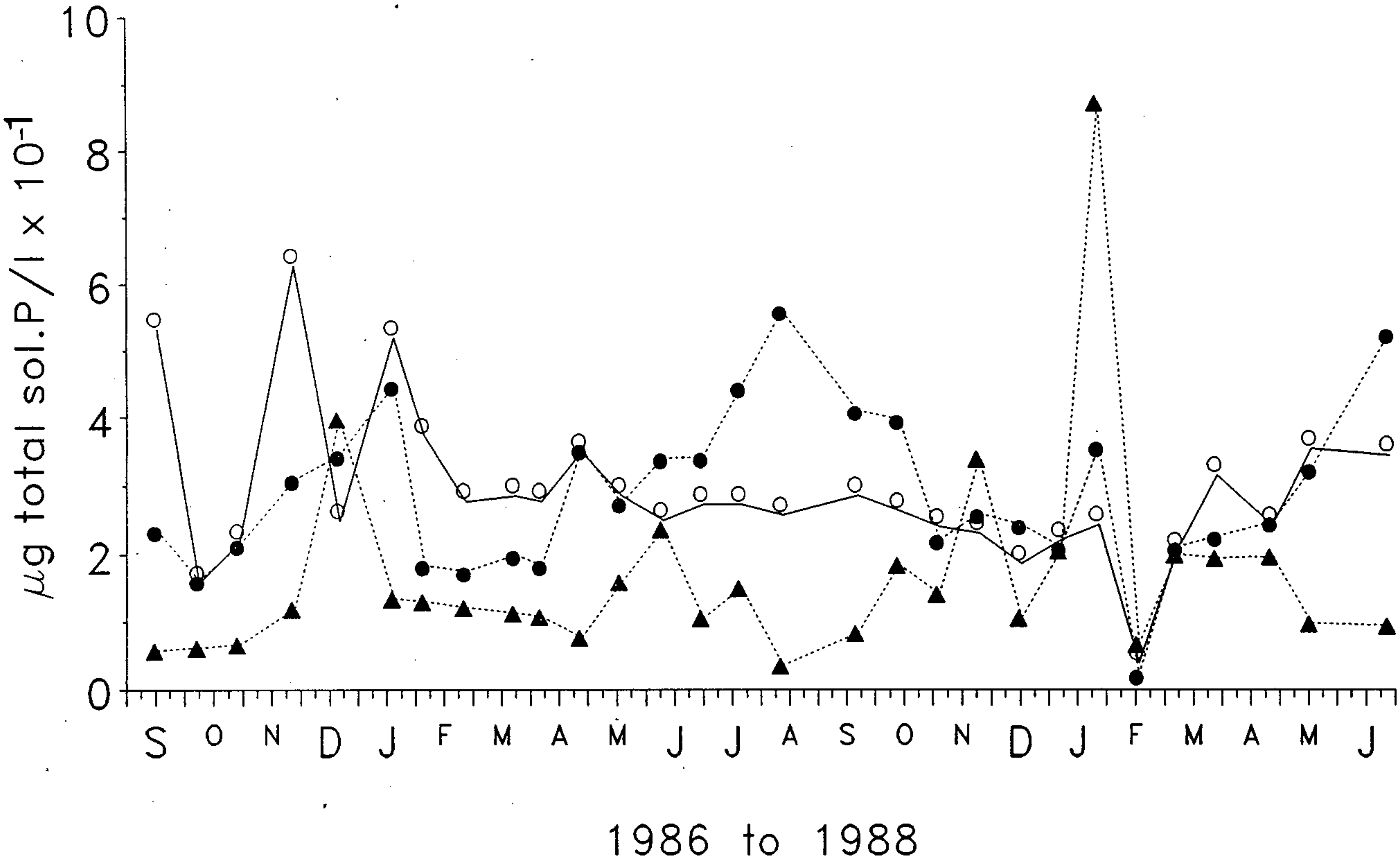


Figure 8(c)

Particulate P in Loch Eye inflows Garrick (•), Loinnbuie (°) and Erracht ( )

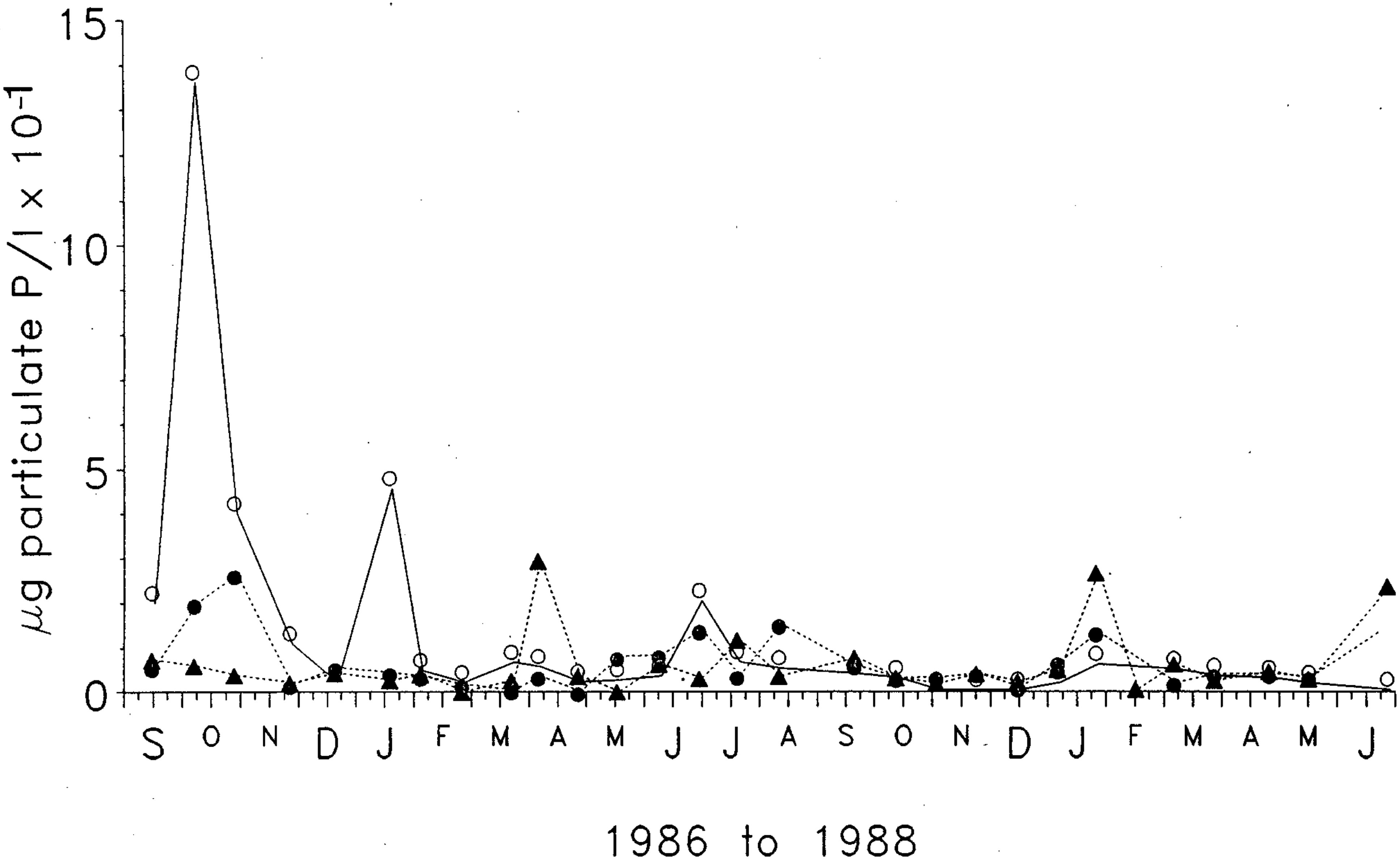


Figure 9(a)

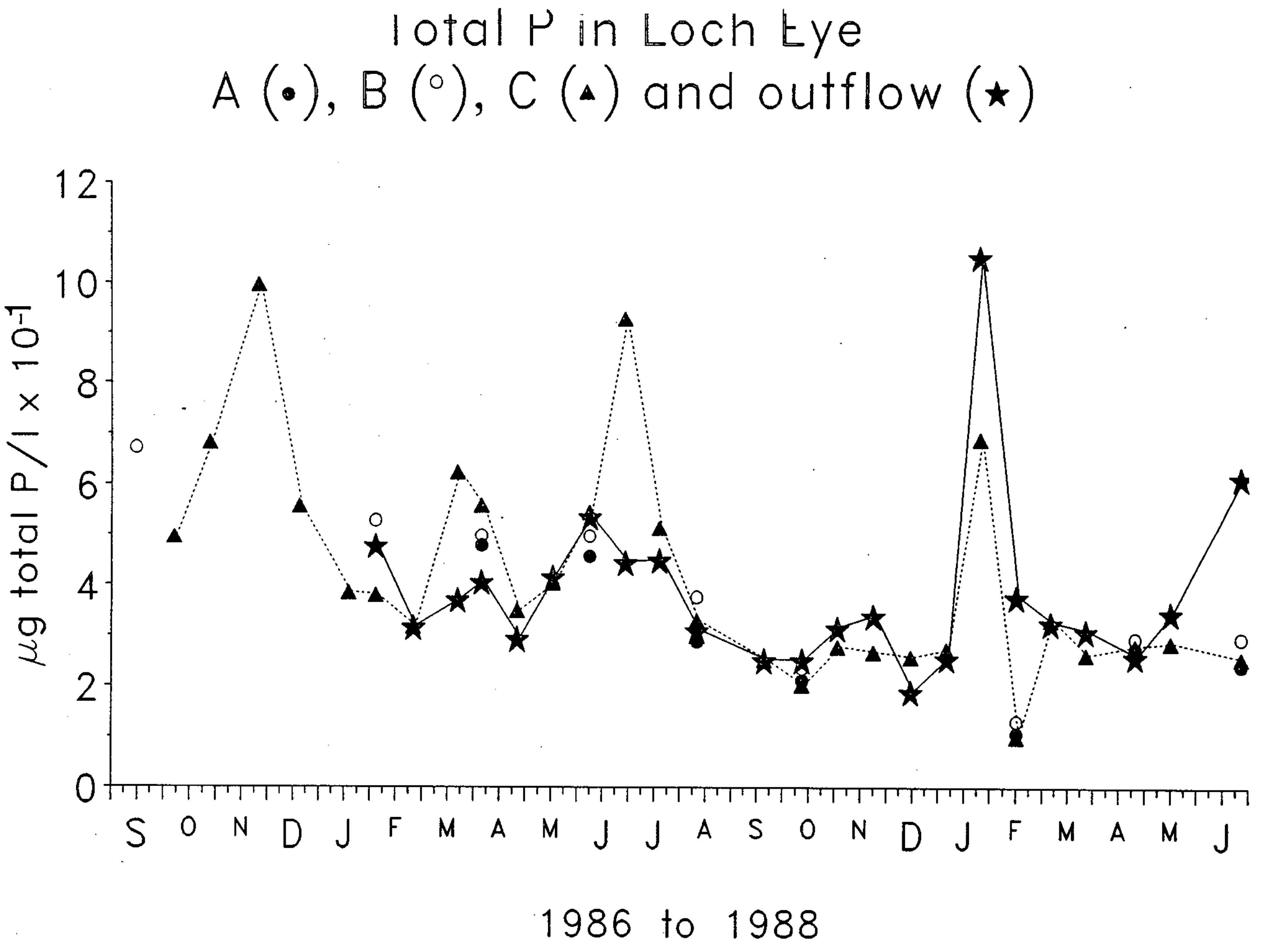

Total P in Loch Eye
A (•), B (°), C (▲) and outflow (★)
μg total P/l x 10-1
12
10
8
6
4
2
0
S O N D J F M A M J J A S O N D J F M A M J
1986 to 1988

Figure 9(b)

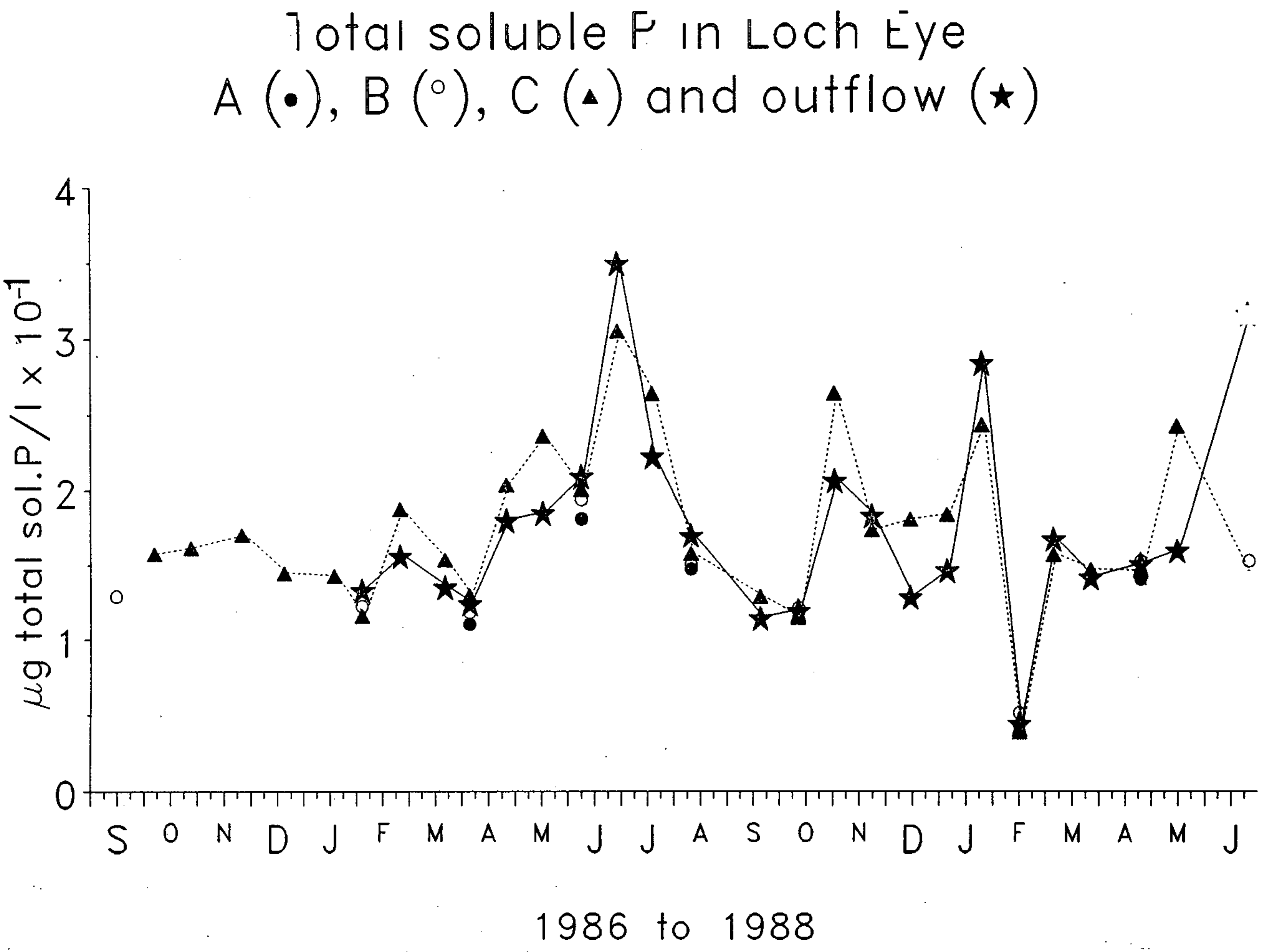

Total soluble P in Loch Eye
A (•), B (°), C (▲) and outflow (★)
μg total sol.P/l x 10-1
0
1
2
3
4
S
O
N
D
J
F
M
A
M
J
J
A
S
O
N
D
J
F
M
A
M
J
1986 to 1988

Figure 9(c)

Particulate P in Loch Eye
A (•), B (°), C (▲) and outflow ( )

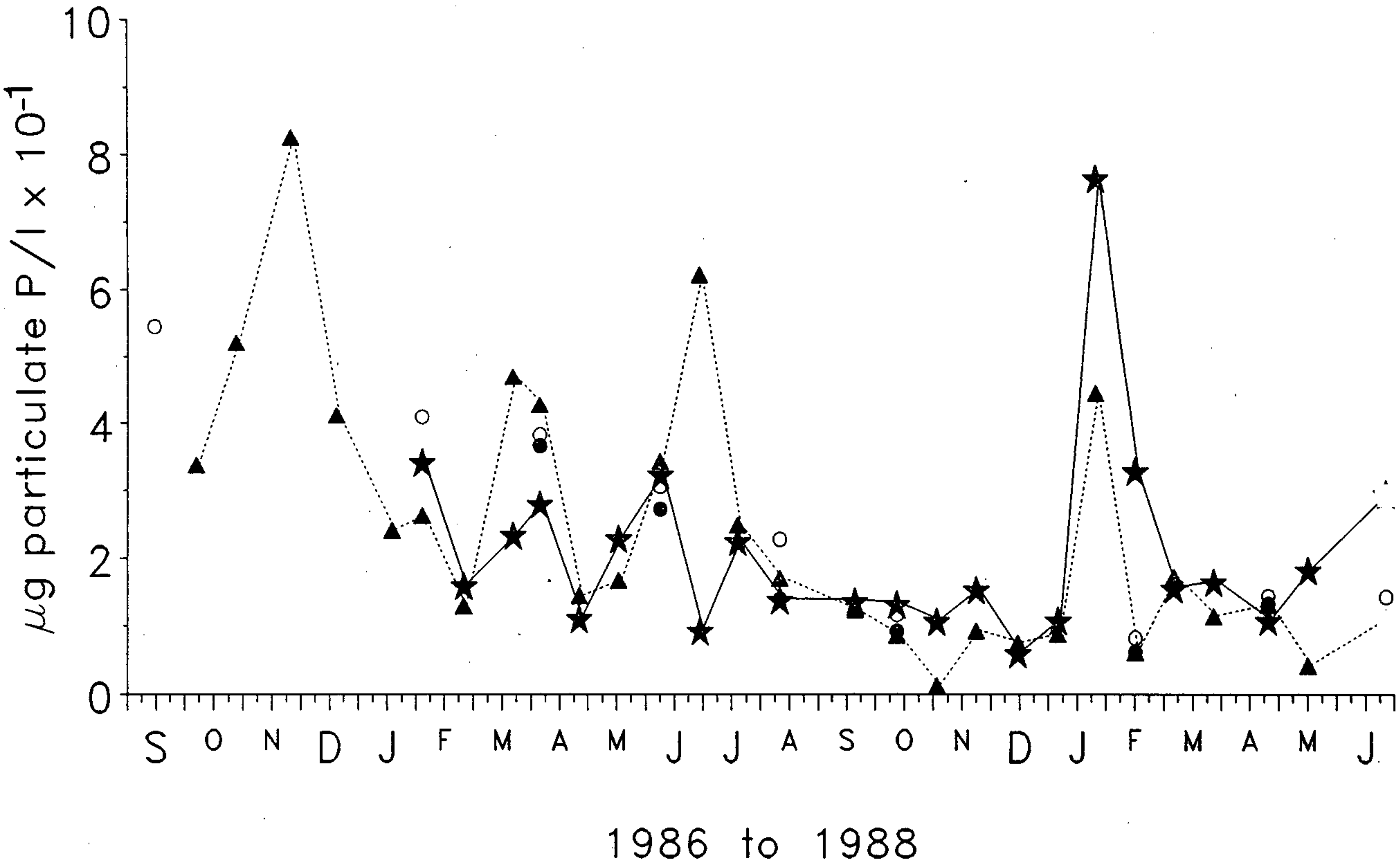
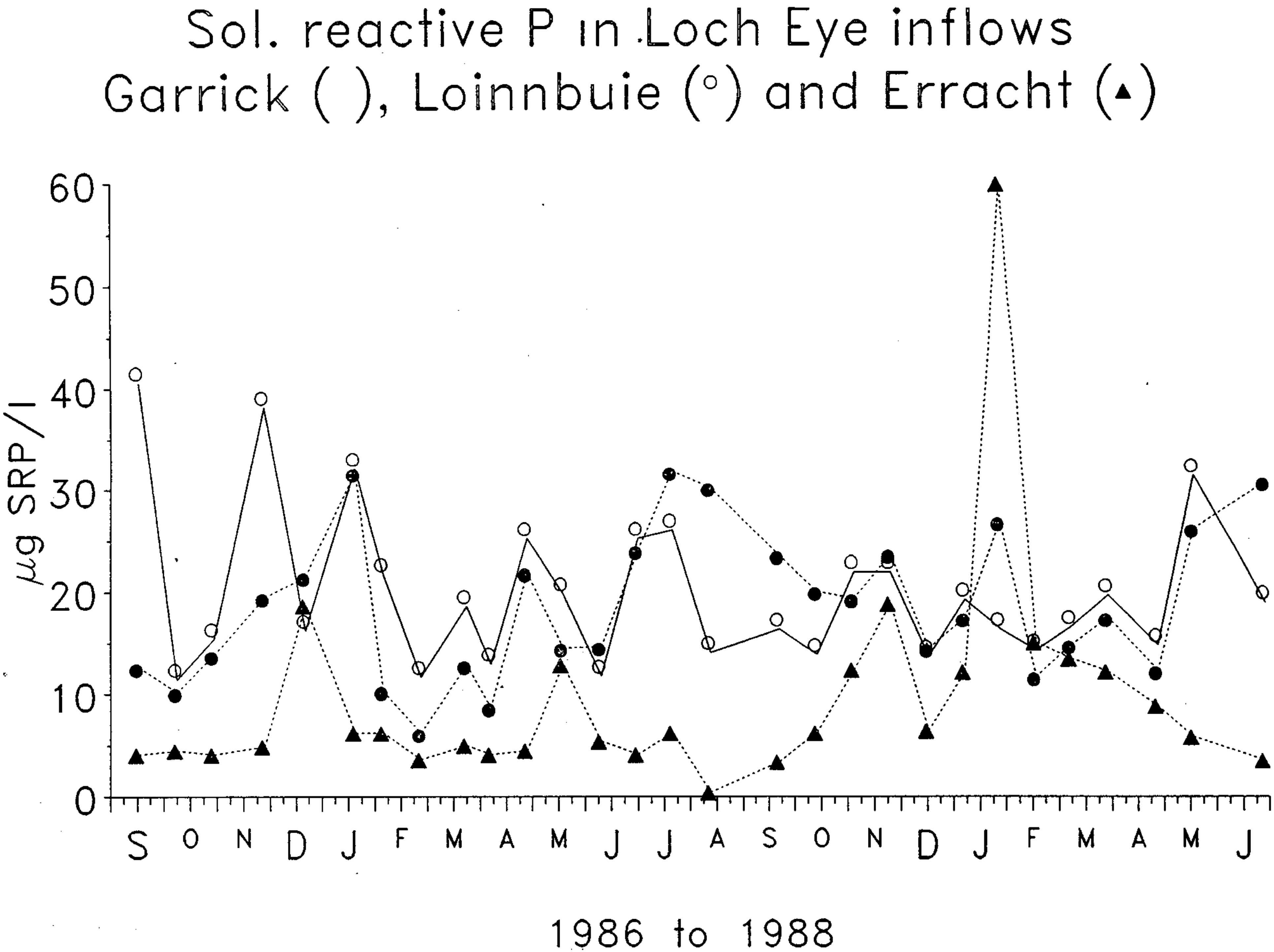


Figure 10(a)



Sol. reactive P in Loch Eye inflows
Garrick ( ), Loinnbuie (°) and Erracht (▲)
60
50
40
30
20
10
0
μg SRP/l
S O N D J F M A M J J A S O N D J F M A M J
1986 to 1988

Figure 10(b)

Sol. reactive P in Loch Eye
A (•), B (○), C (▲) and outflow (★)

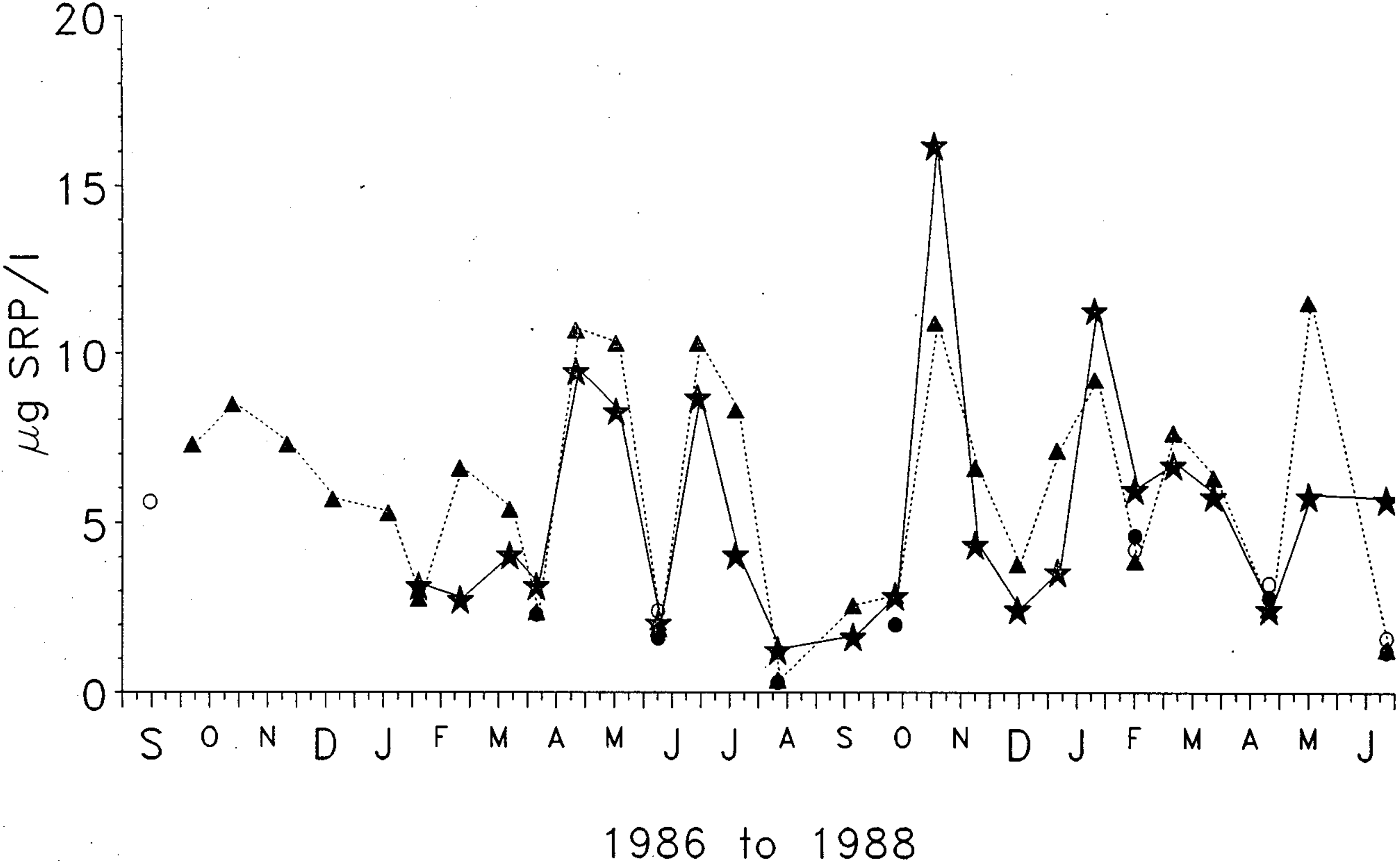


Figure 11(a)

Dissolved silica in Loch Eye inflows
Garrick (•), Loinnbuie (°), and Erracht (▲)

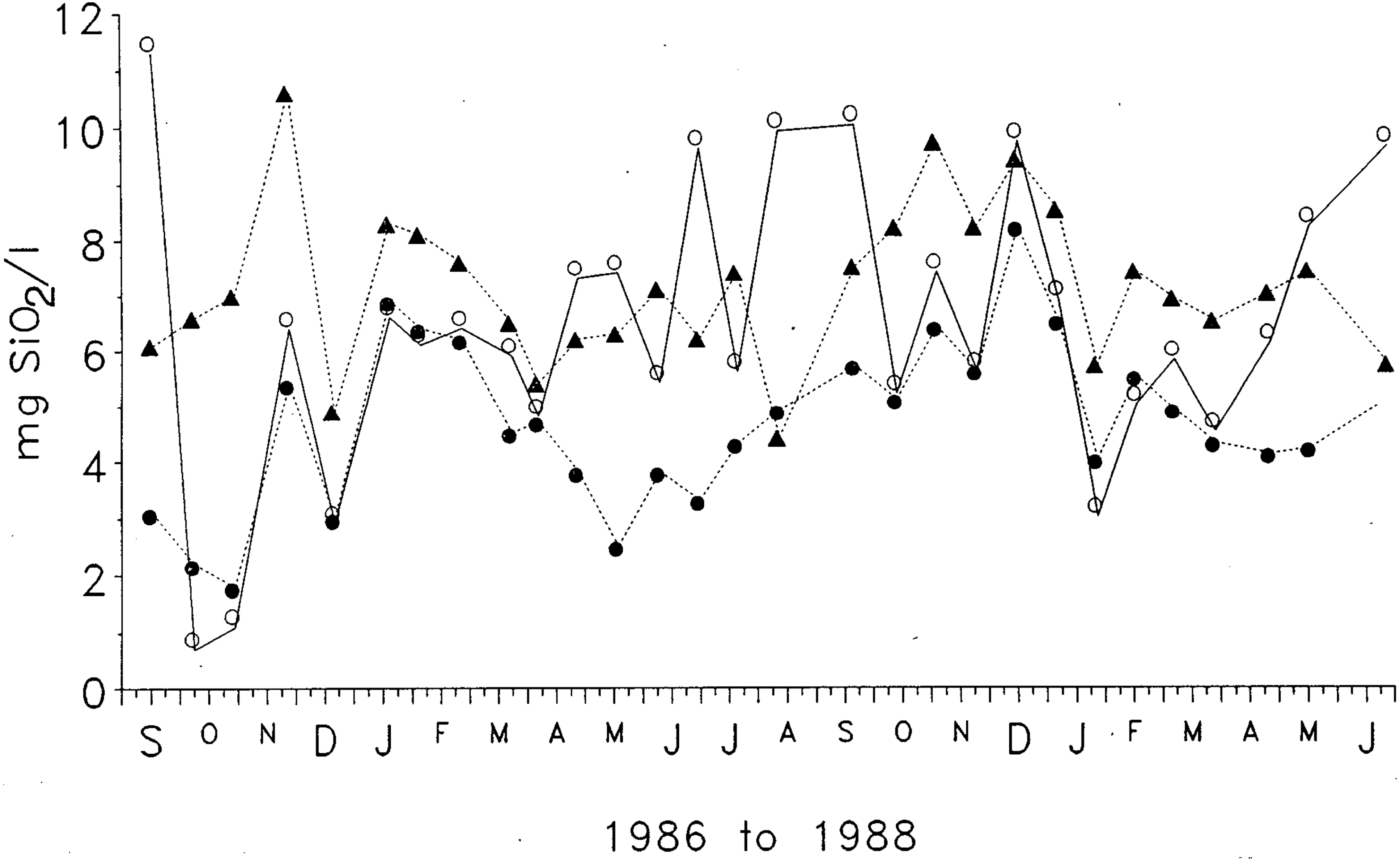


Figure 11(b)

Arrows indicate the shifts in the concentration-flow relationship from the start (S) to the end (E) of the sampling period August 1987 to June 1988

Dissolved silica in the Garrick:
the concentration–flow relationship

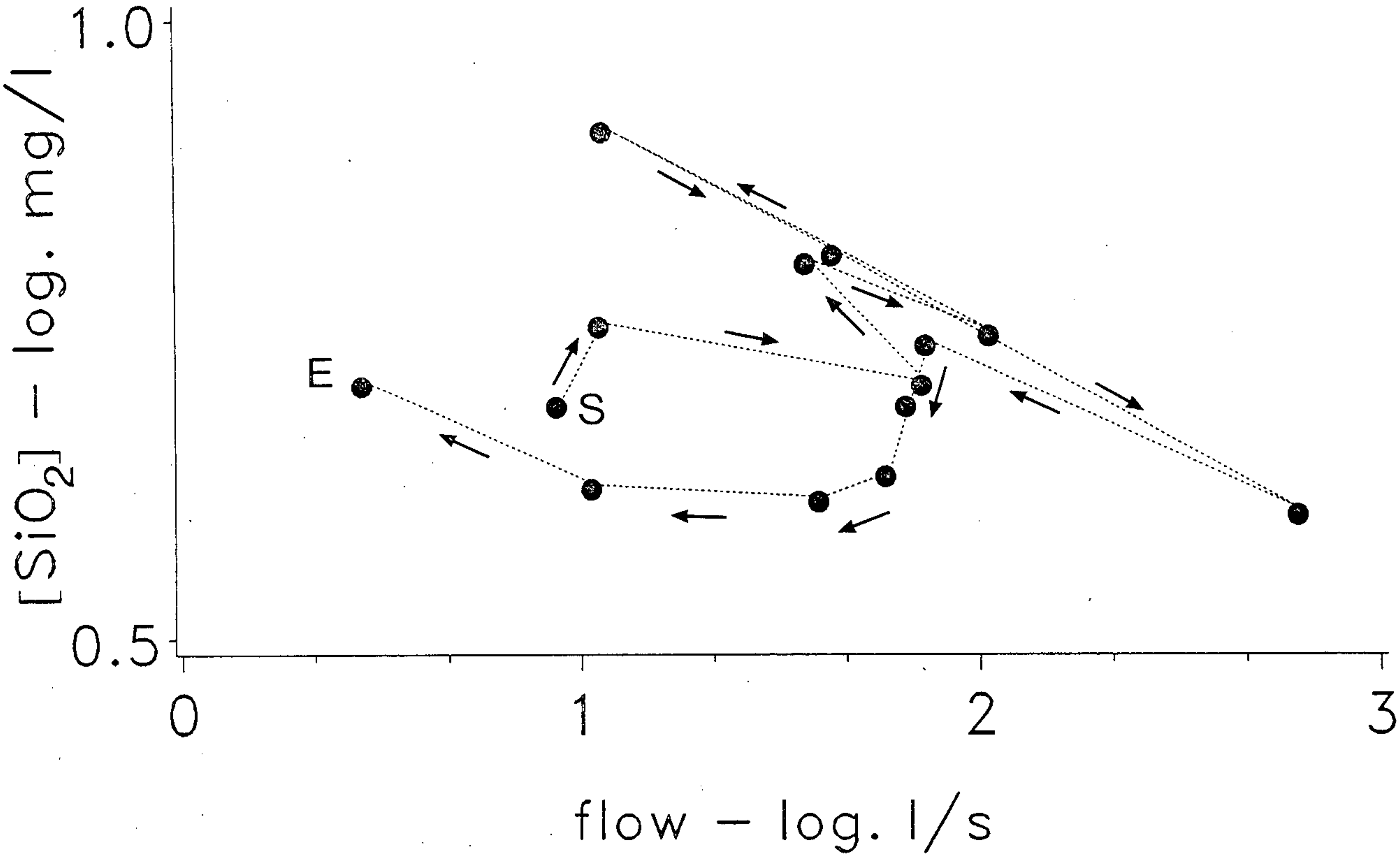


Figure 11(c)

Dissolved silica in Loch Eye
A (•), B (°), C (▲) and outflow (★)

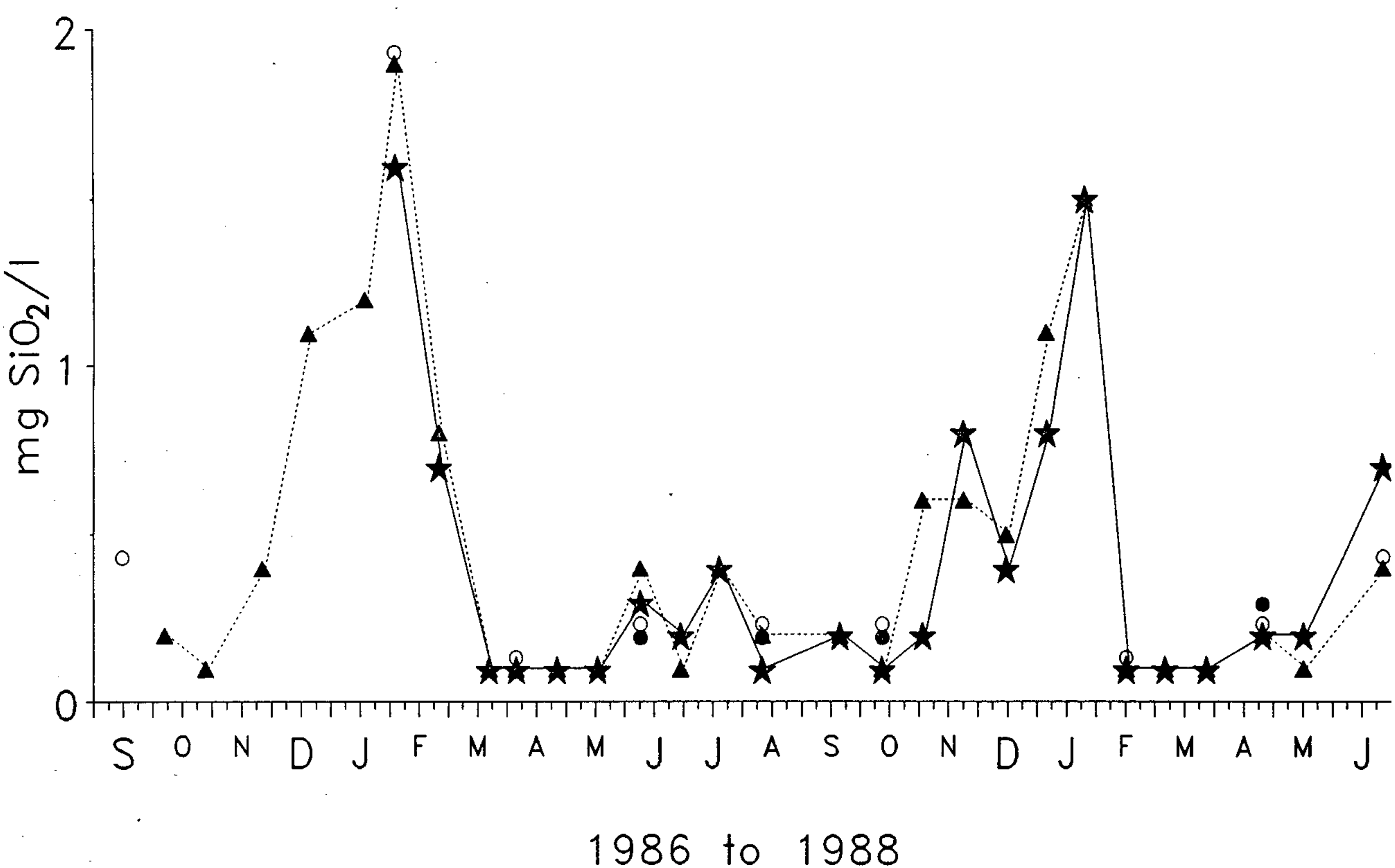


Figure 12

L.Eye sediment chemistry – Feb. 1987:
vertical profiles of sol.react. P

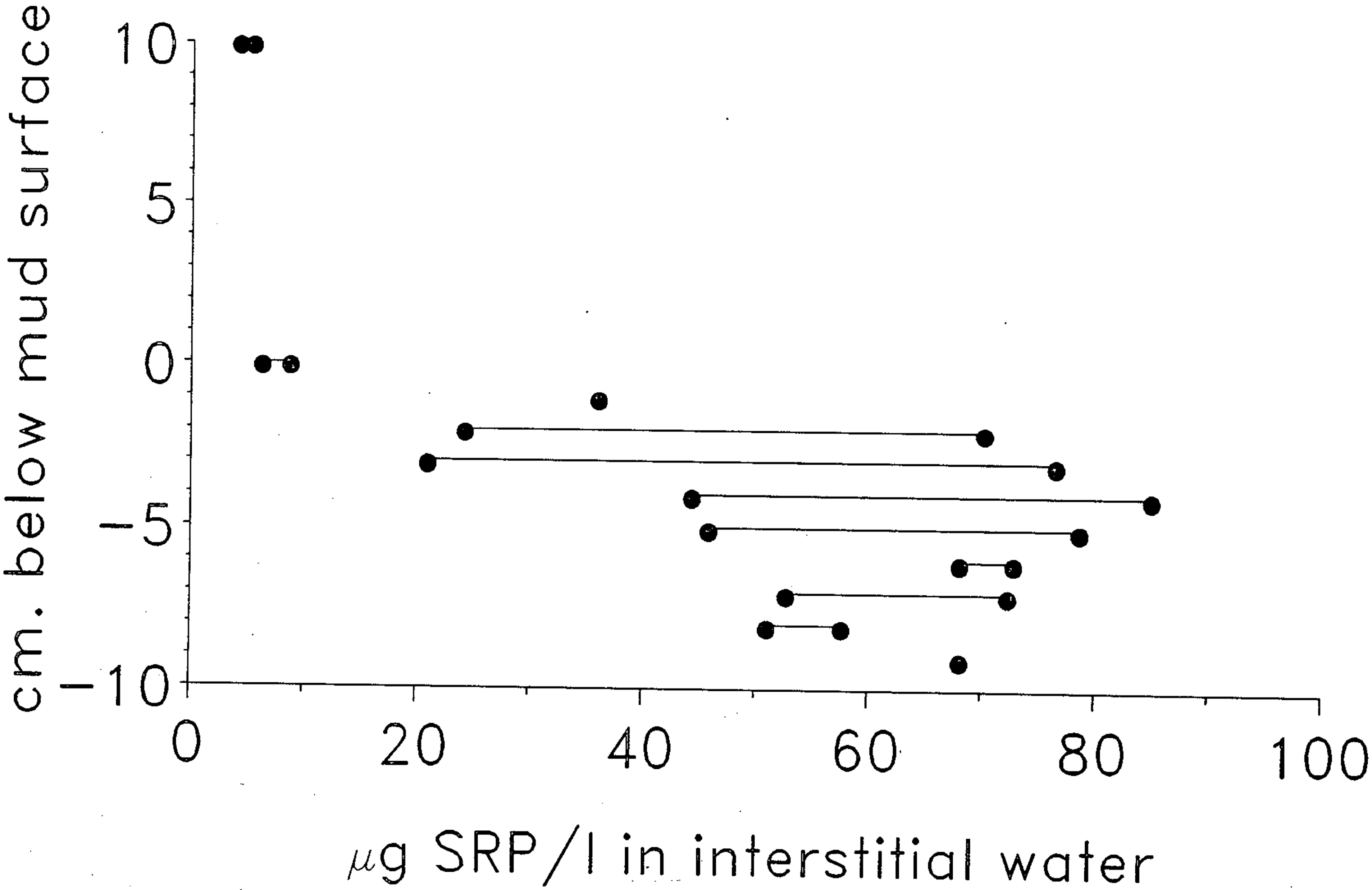


Figure 13

L.Eye sediment chemistry June 1987:
vertical profiles of sol.react. P

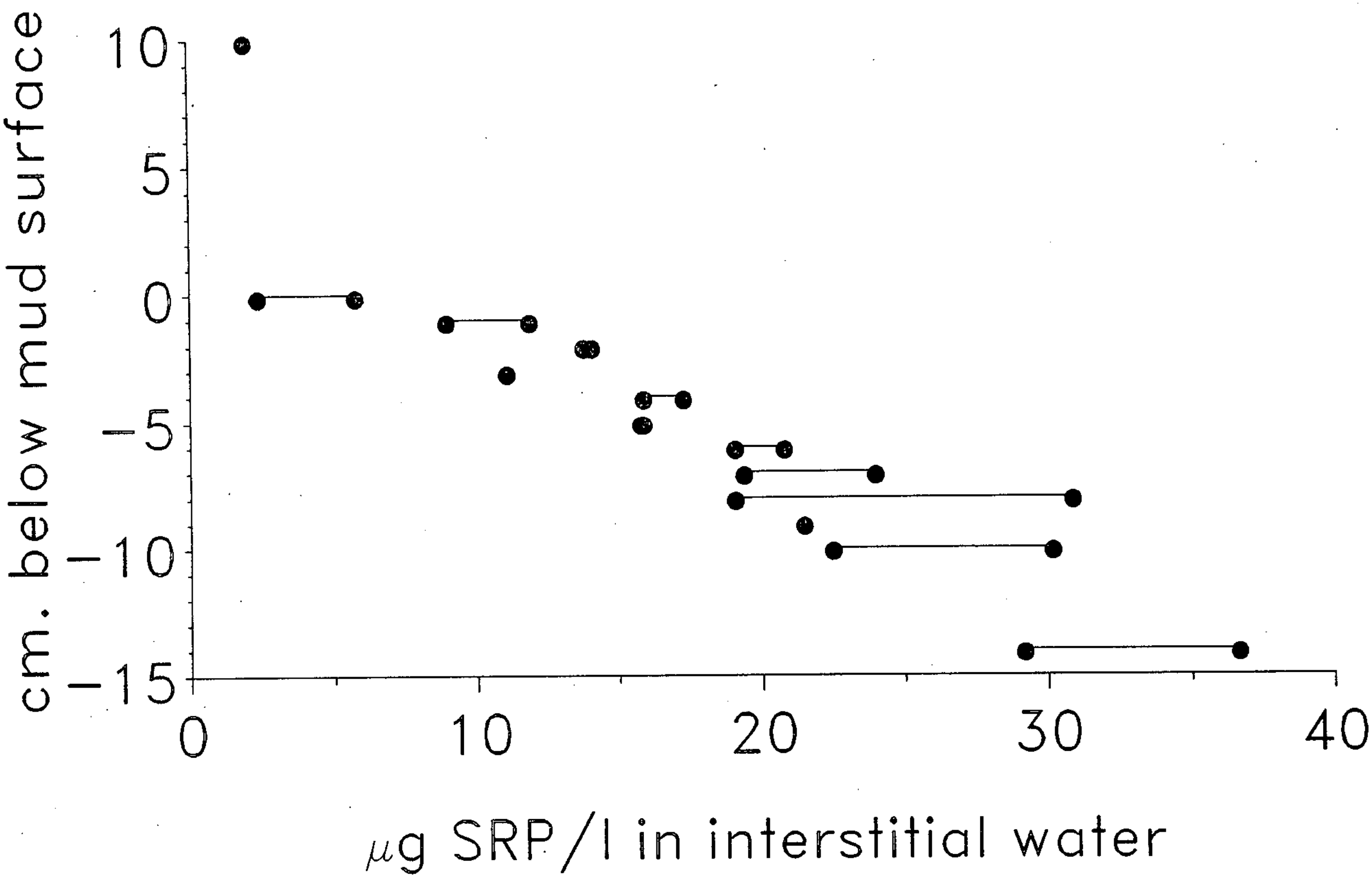


Figure 14

L.Eye sediment chemistry – Feb. 1980:
vertical profiles of TSP (°) and SRP ( )

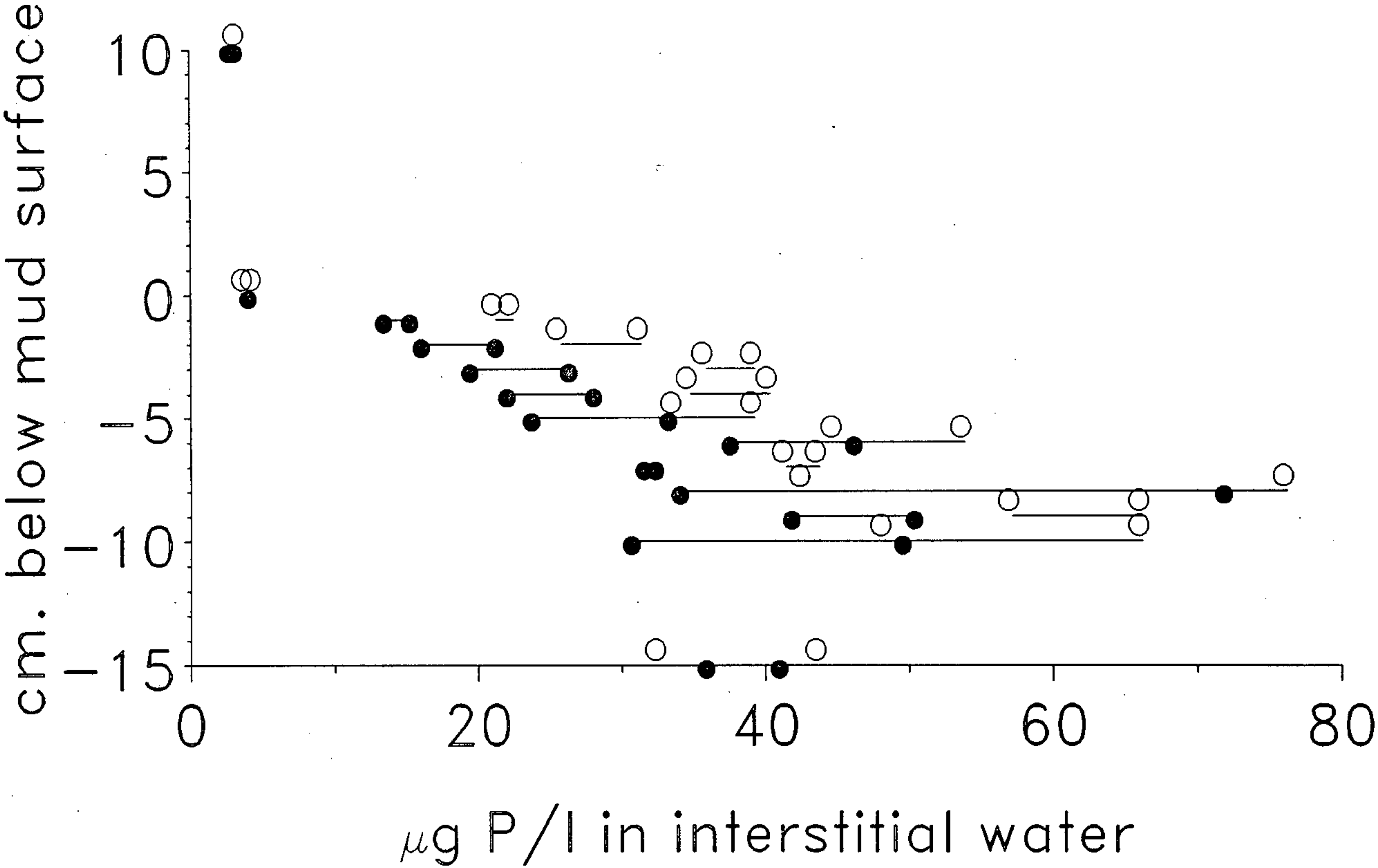


Figure 15(a)

L.Lye sediment chemistry – April 1987:
vertical profiles of P

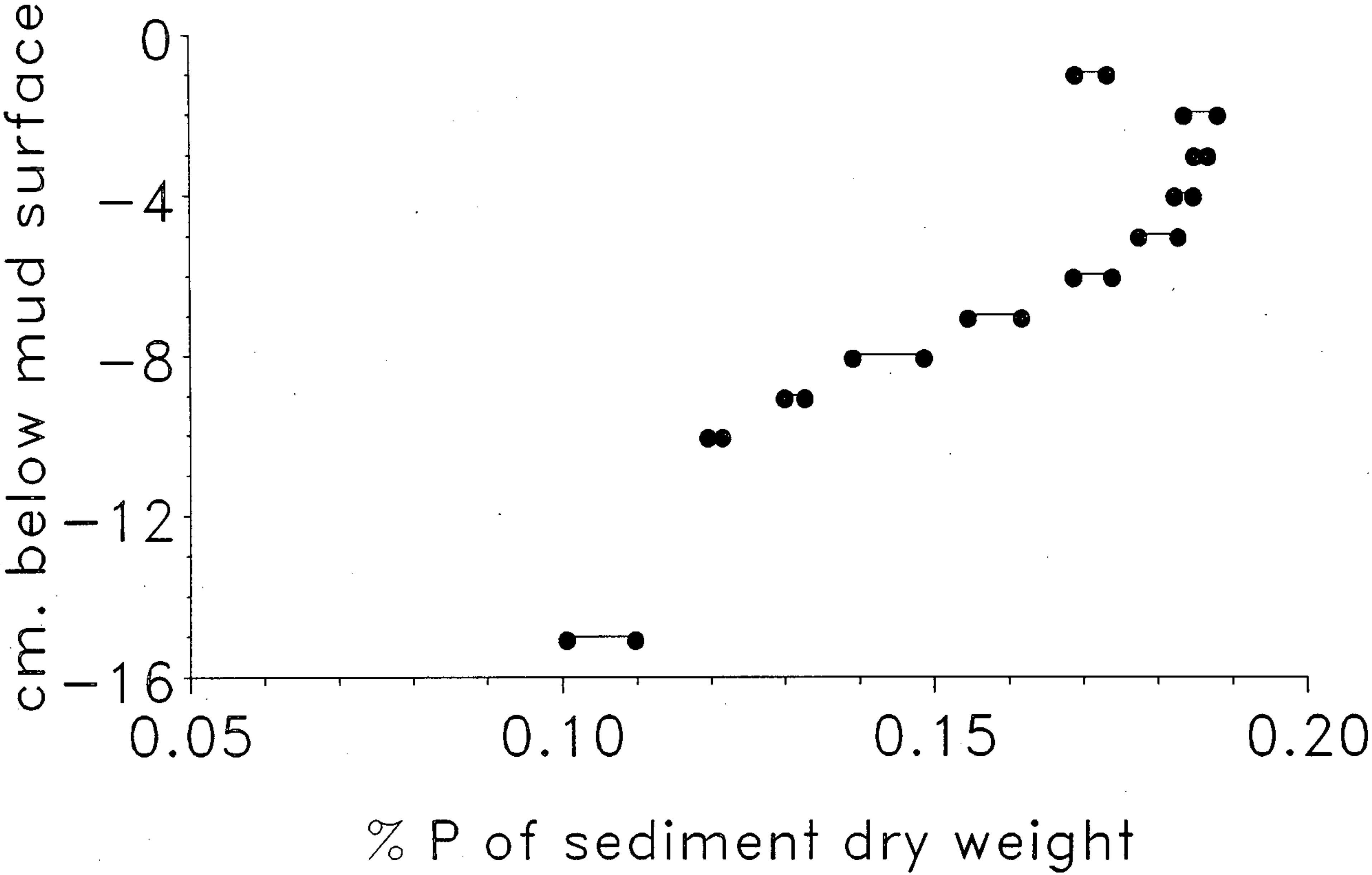


Figure 15(b)

L.Eye sediment chemistry – Oct. 1987:
vertical profiles of P

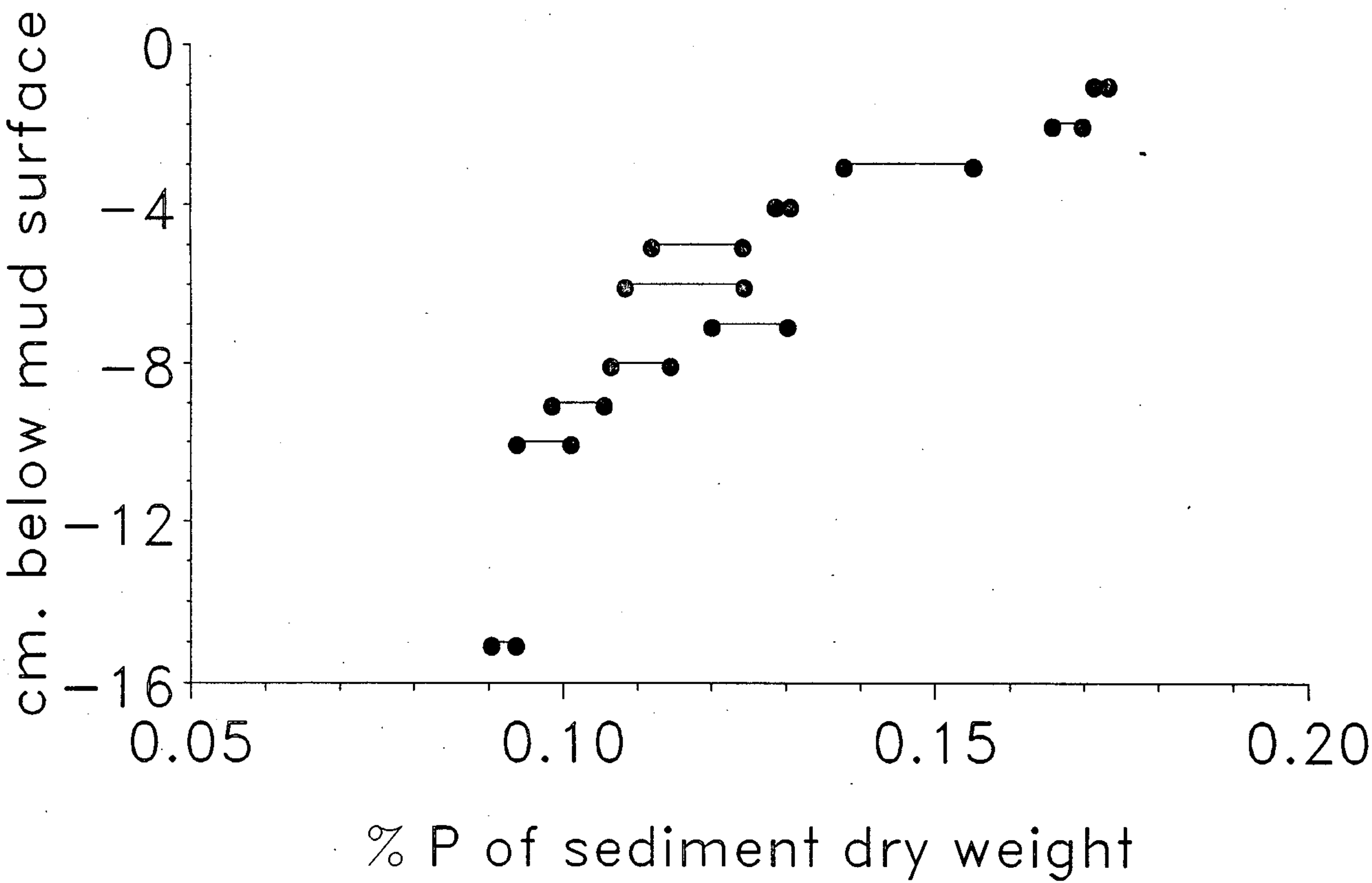


Figure 15(c)

L.Eye sediment chemistry – Feb. 1988:
vertical profiles of P

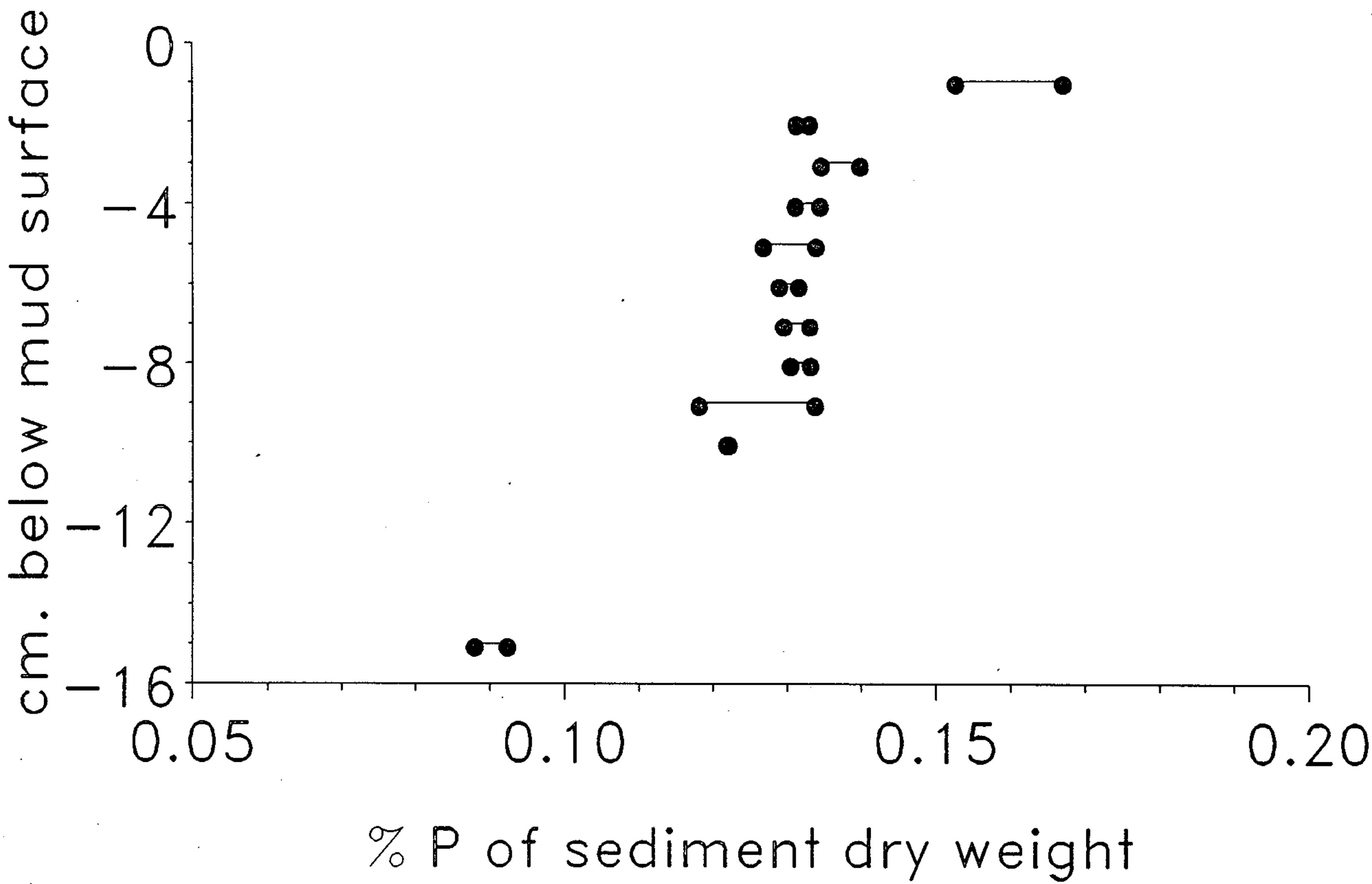


Figure 16(a)

L.Eye sediment chemistry – Feb. 1987:
vertical profiles of sol.react. SiO_2

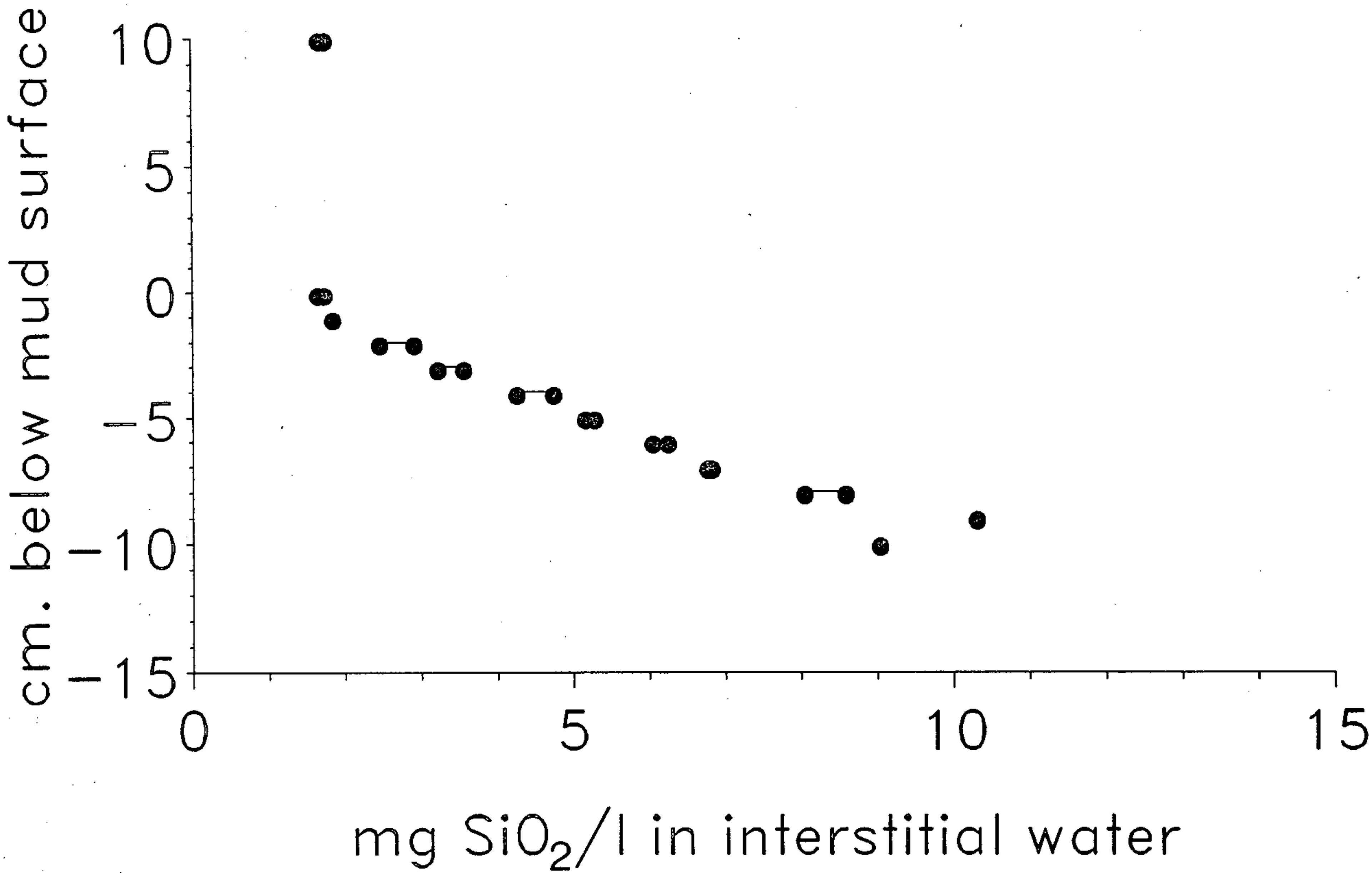


Figure 16(b)

L.Eye sediment chemistry – April 1987: vertical profiles of sol.react. SiO_2

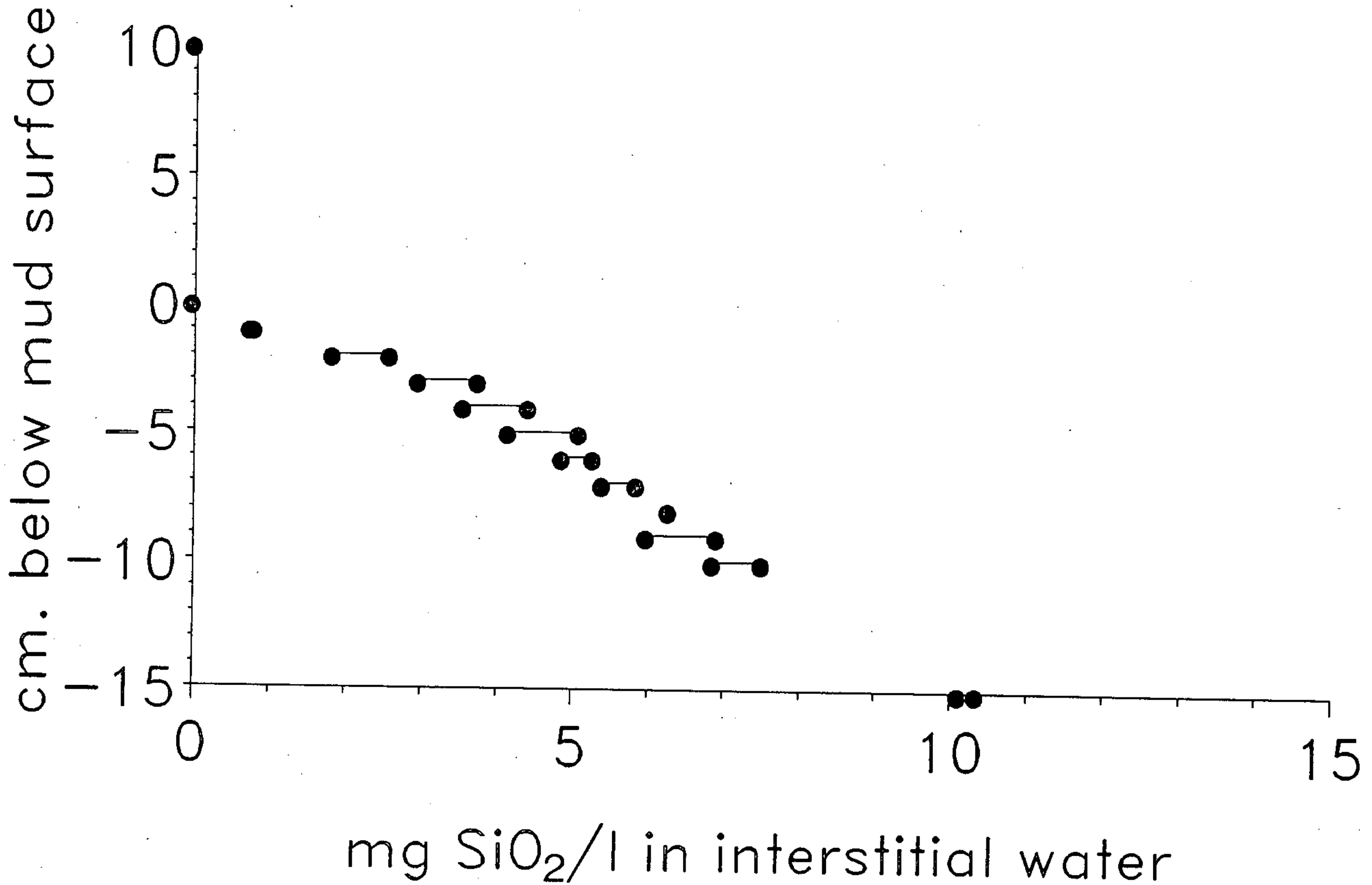


Figure 16(c)

L.Eye sediment chemistry – June 1988:
vertical profiles of sol.react. SiO_2

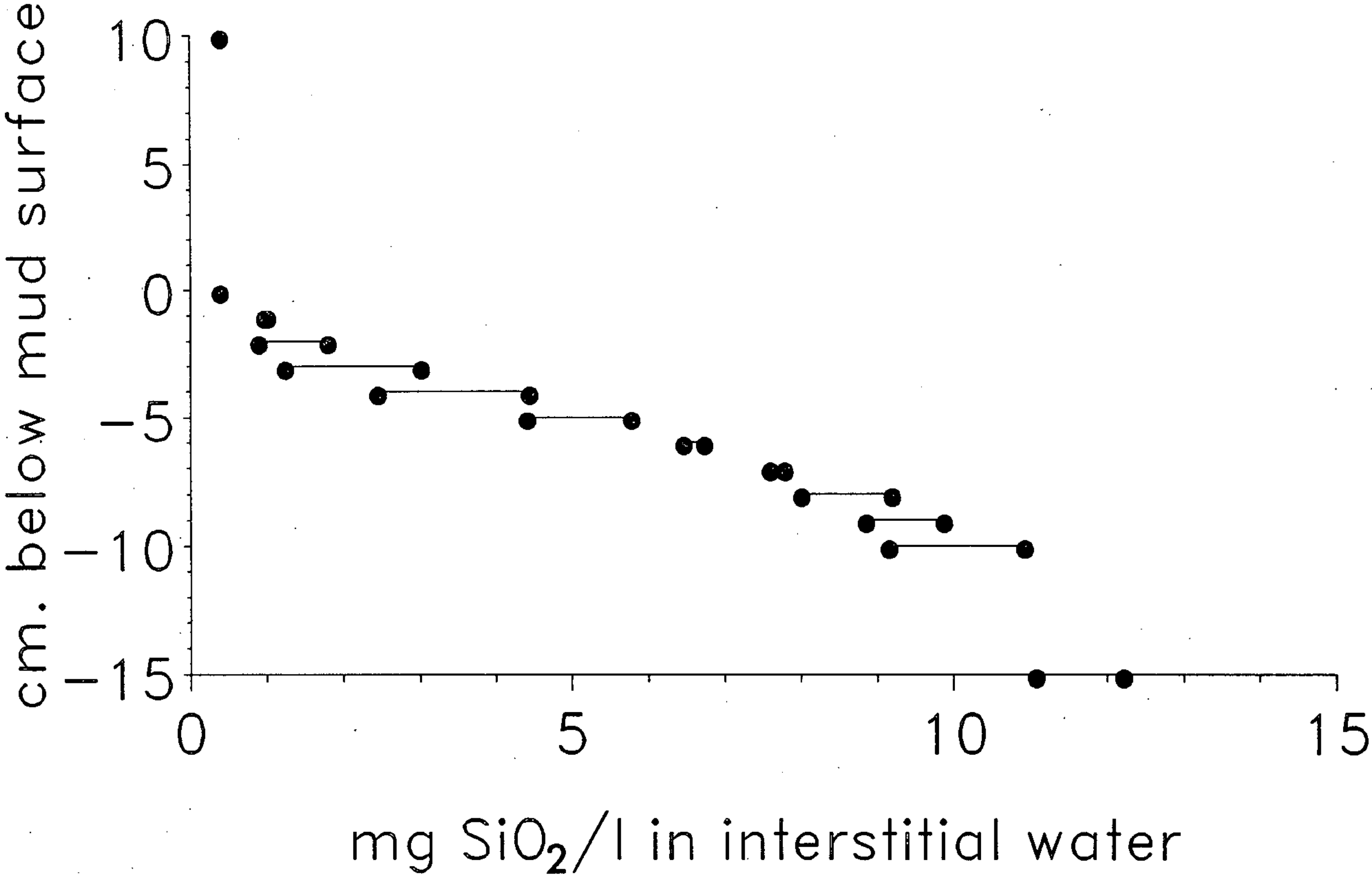


Figure 17(a)

L.Eye sediment chemistry – Aug. 1987:
vertical profiles of sol.react. SiO_2

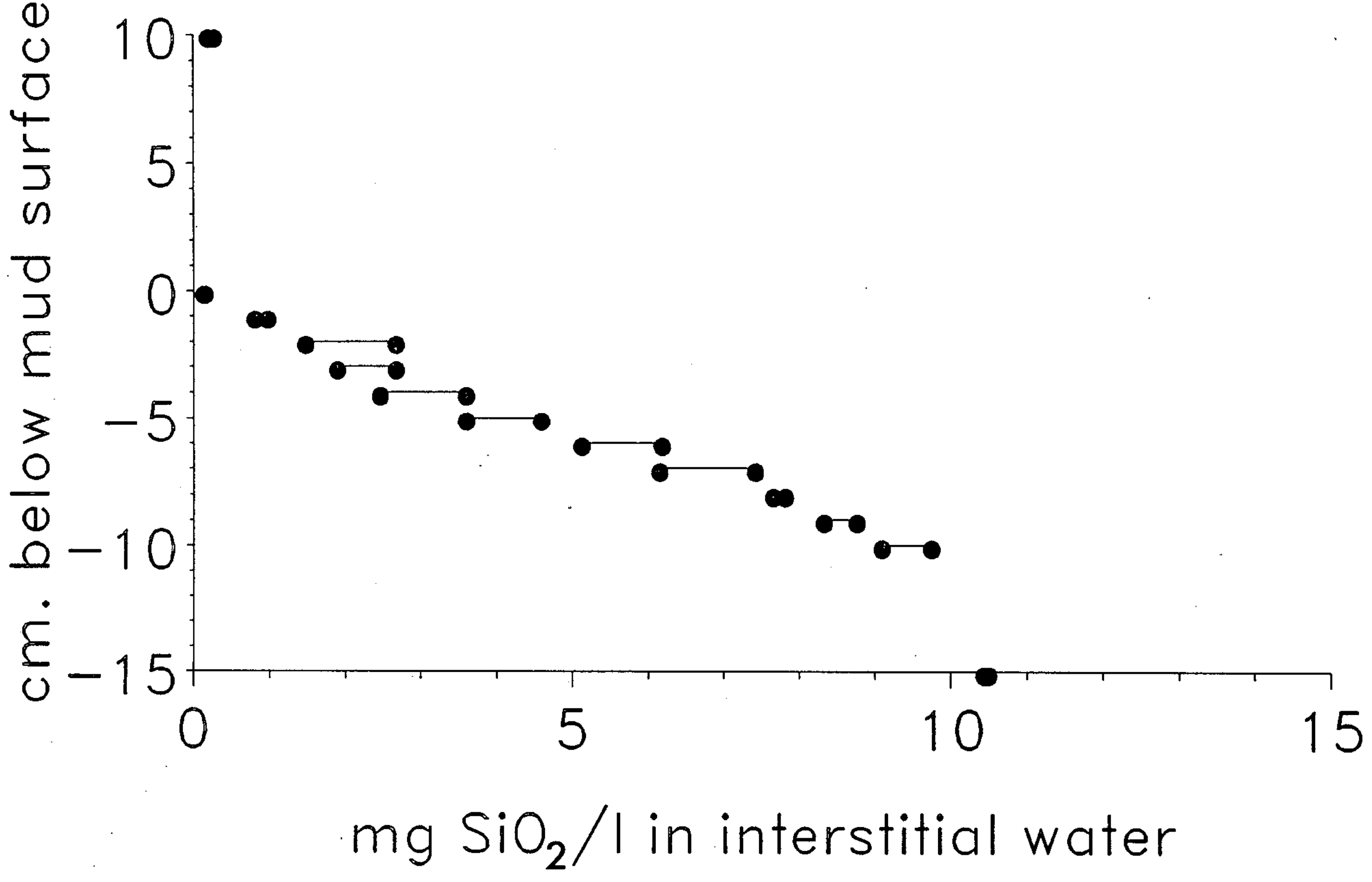


Figure 17(b)

L.Eye sediment chemistry — Oct. 1987: vertical profiles of sol.react. SiO_2

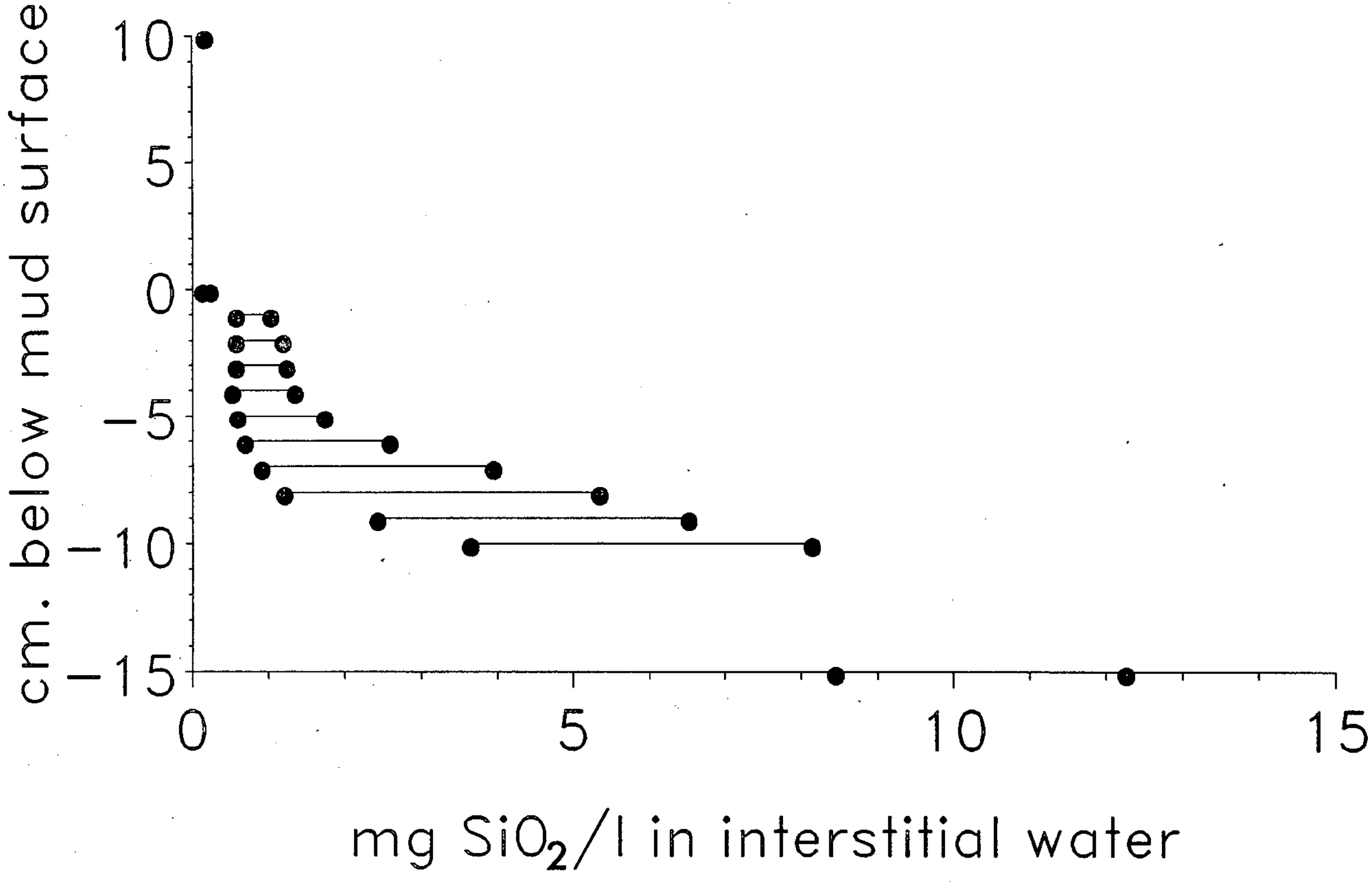


Figure 18

Phytoplankton chlorophyll in Loch Eye
open sites A (•), and B (°); edge sites C (▲) and outflow (★).

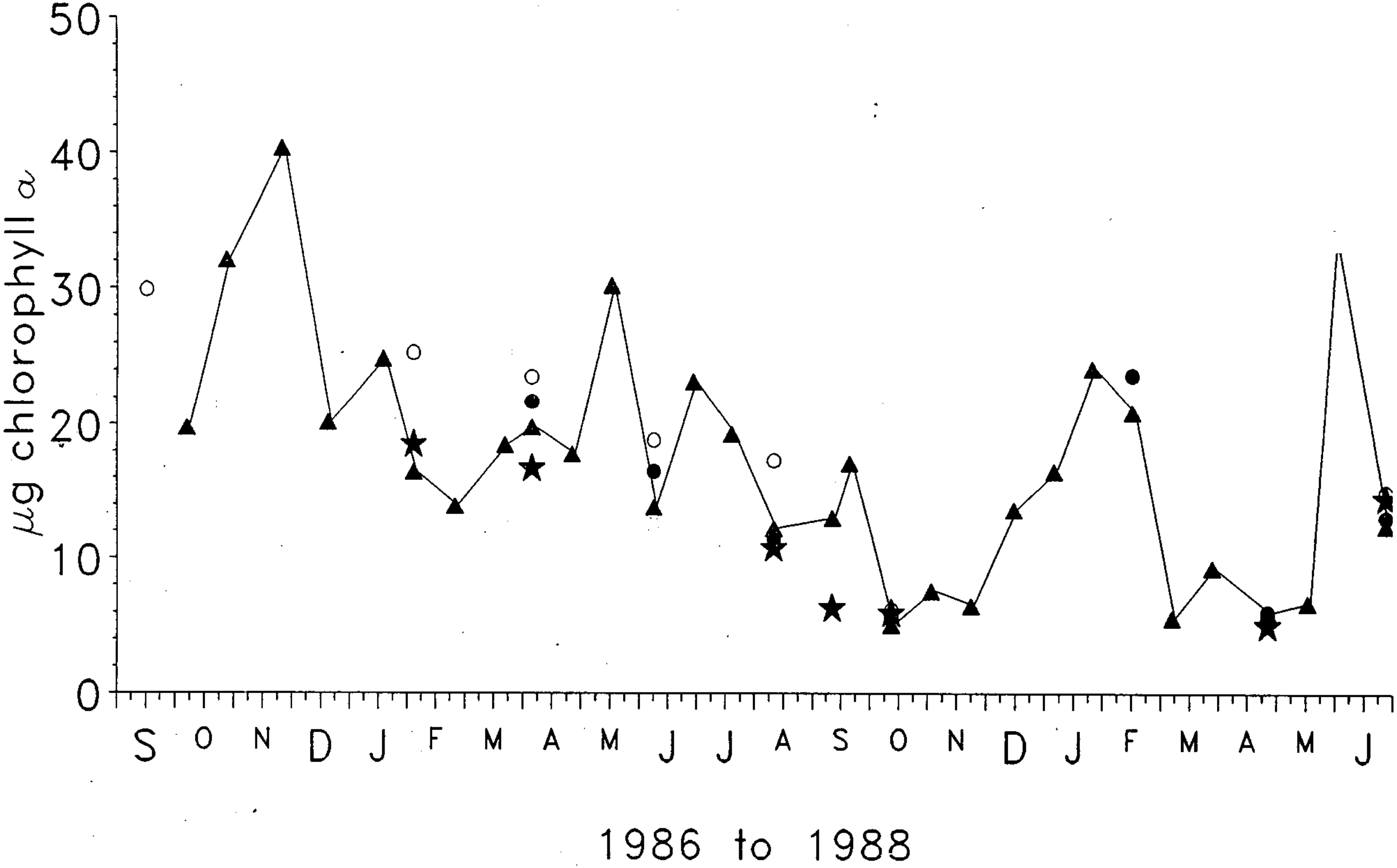


Figure 19

Size distributions of L.Eye phytoplankton:
16 Dec.1987 (•) and 1 June 1988 (°)

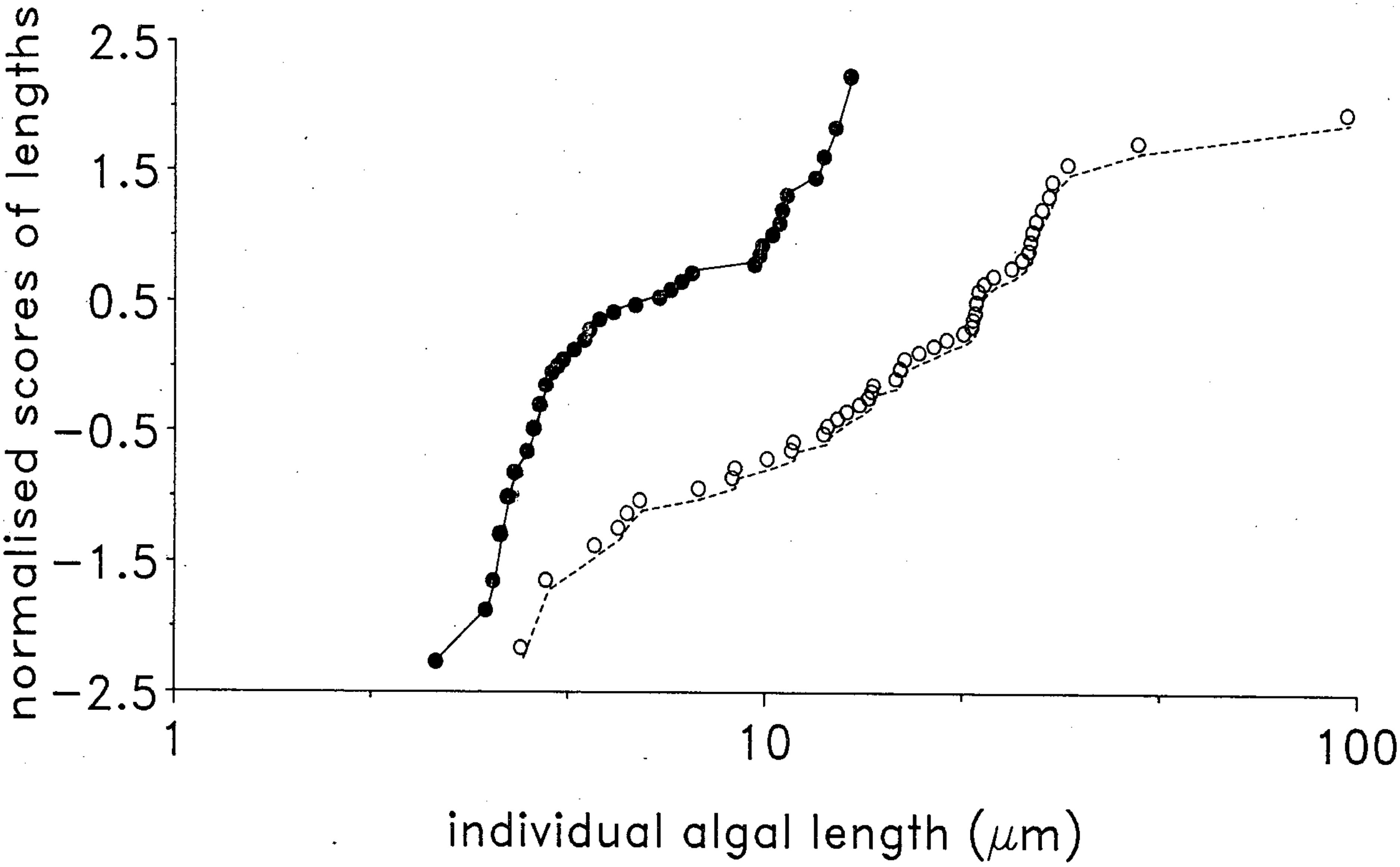


Figure 20

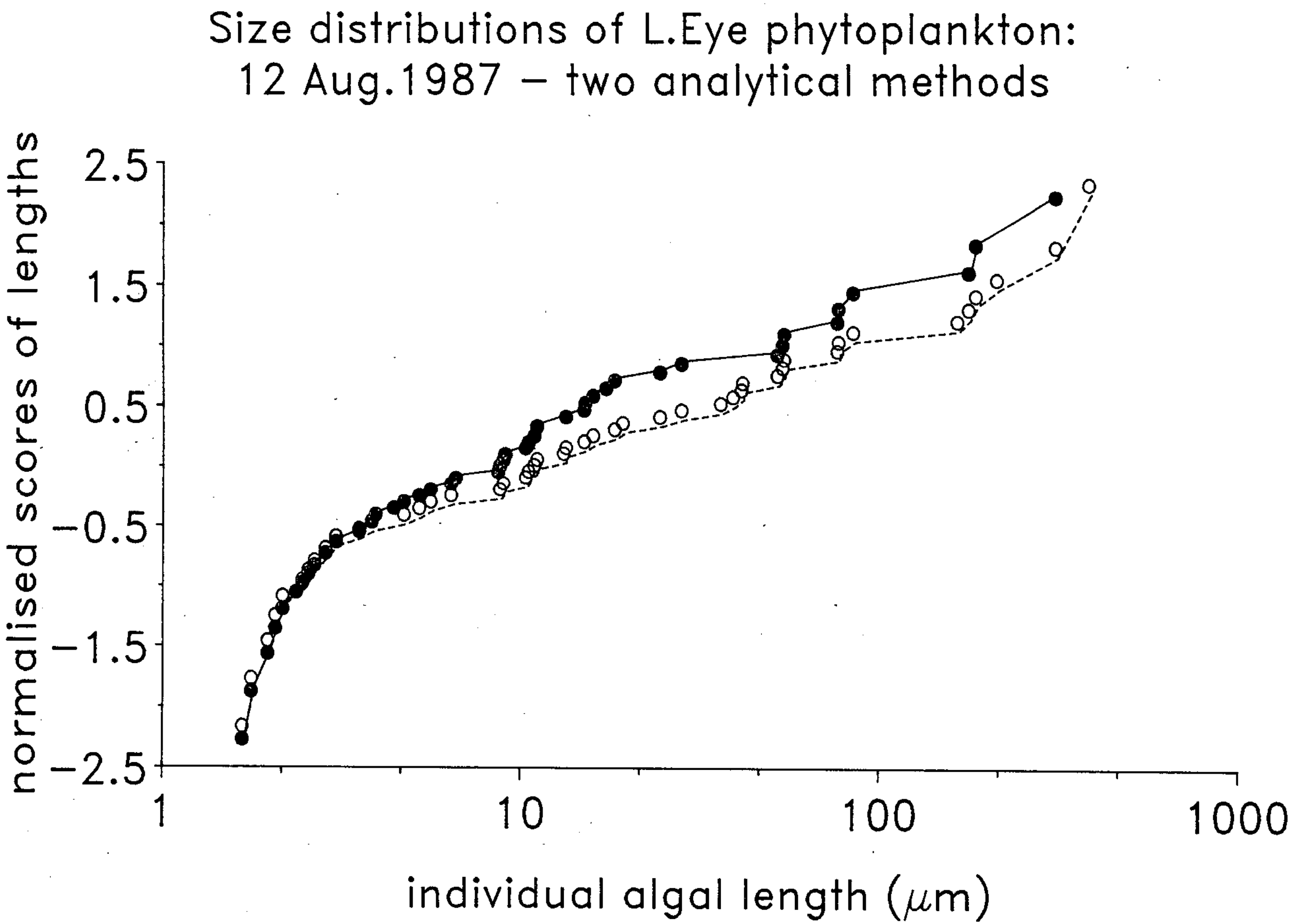

Size distributions of L.Eye phytoplankton:
12 Aug.1987 – two analytical methods
normalised scores of lengths
2.5
1.5
0.5
−0.5
−1.5
−2.5
1
10
100
1000
individual algal length (μm)

Figure 21

Size distribution of L.Eye phytoplankton:
an assemblage of small size range – Sept.1986

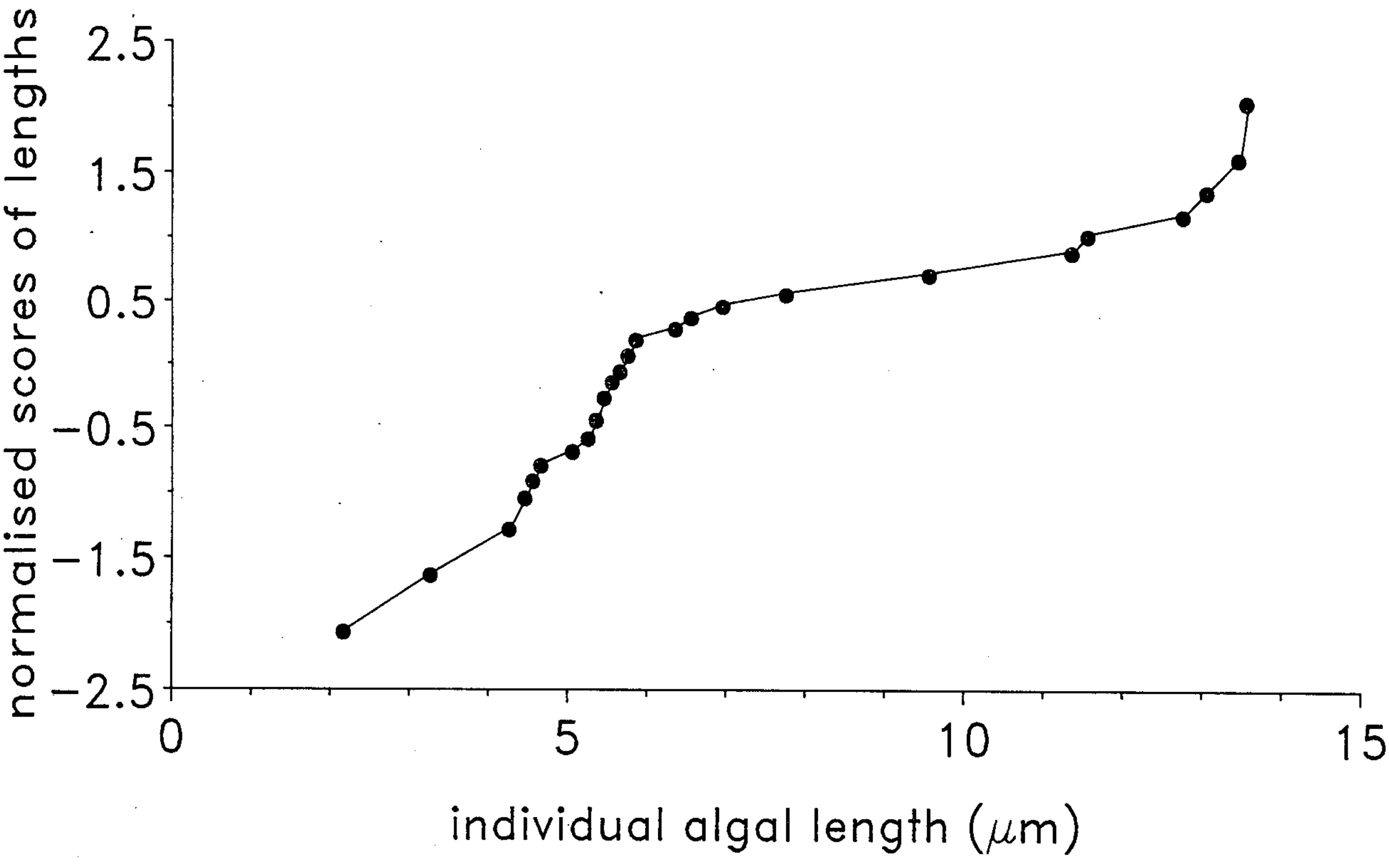
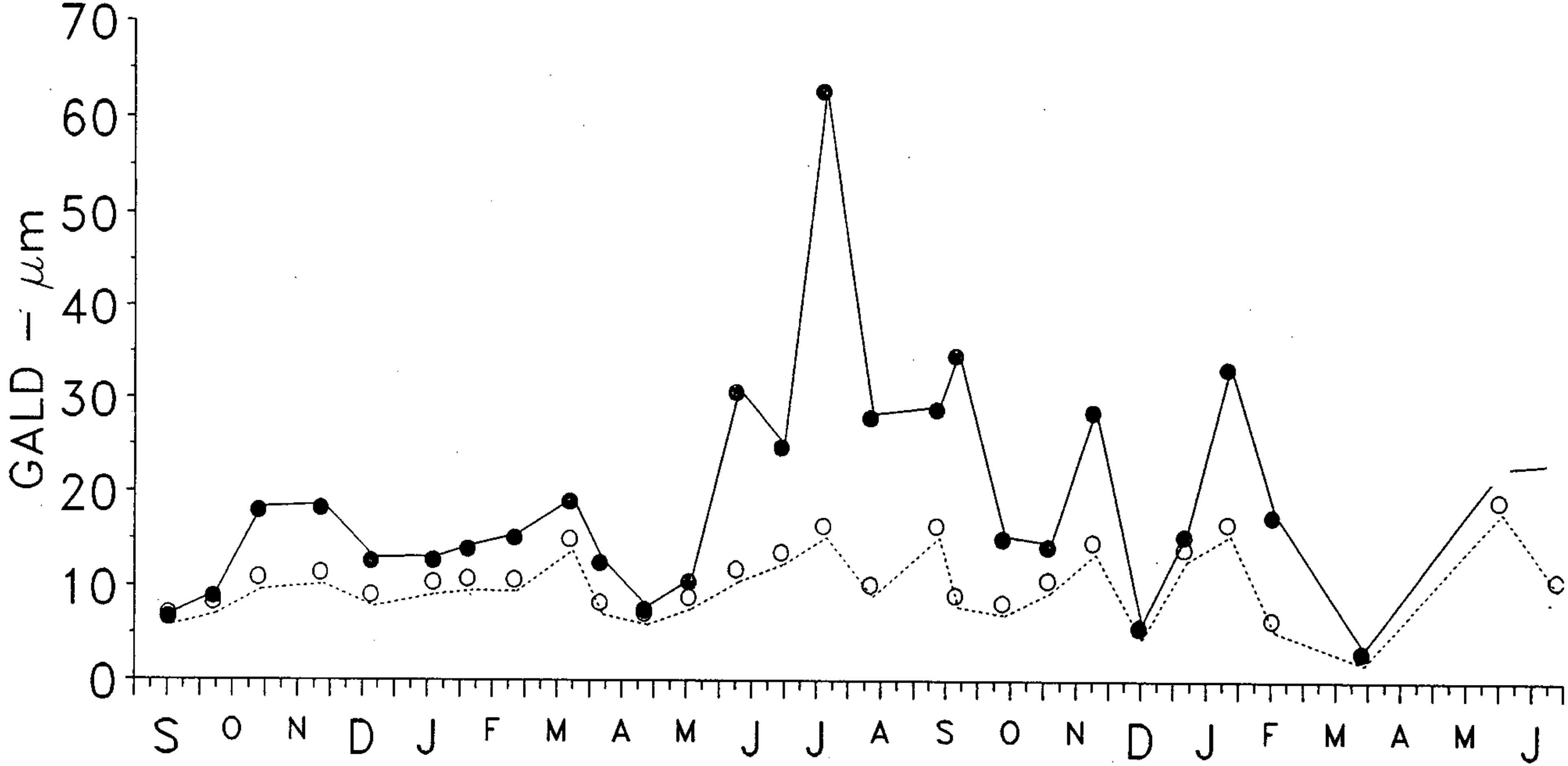


Figure 22

Loch Eye phytoplankton size:
mean ( ) and median (°) GALD" values



1986 to 1988

" greatest axial linear dimension

Figure 23

Size distributions of L.Eye phytoplankton:
21 July (•),7 Sept. (°) and 17 Dec. (▲)

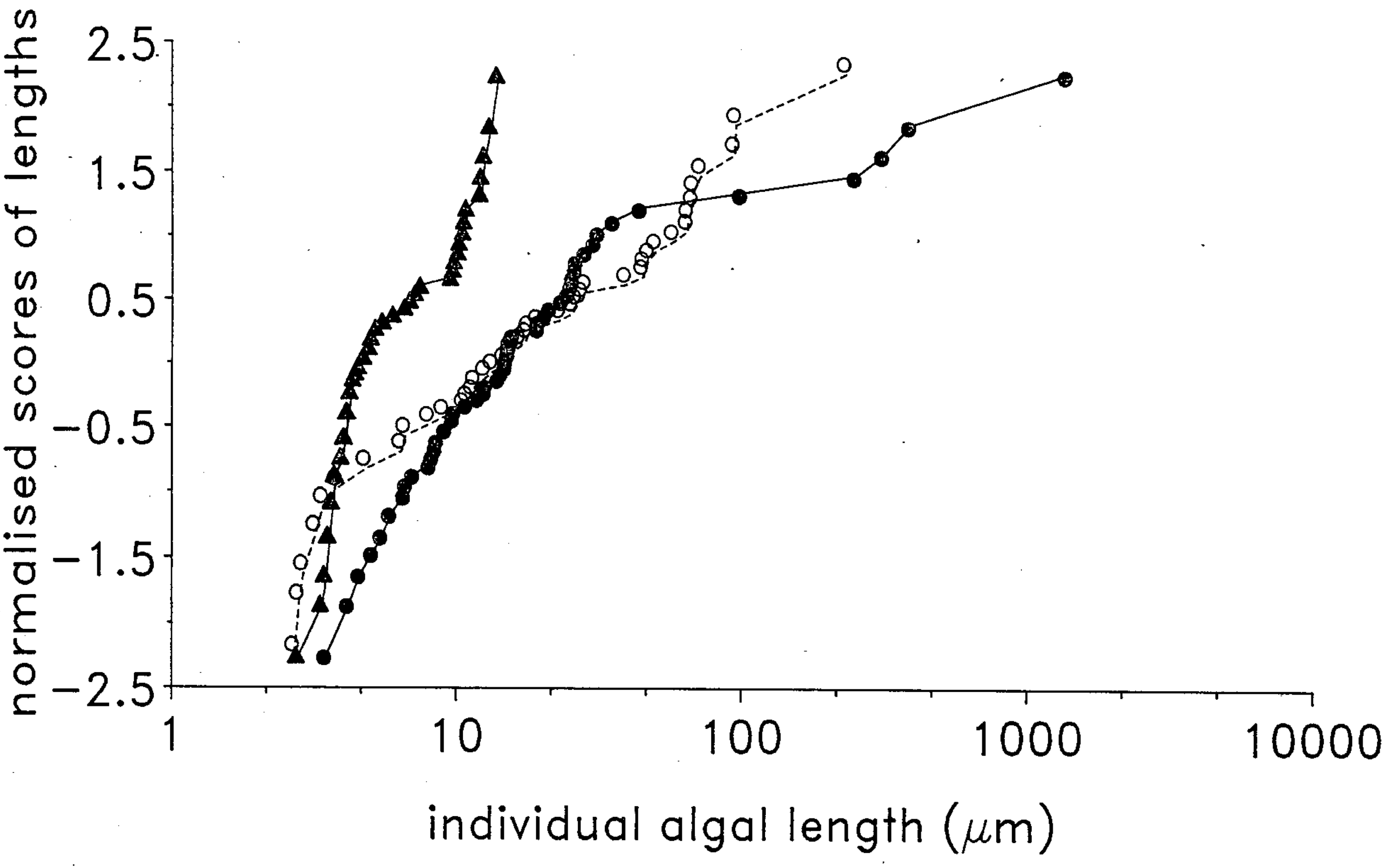
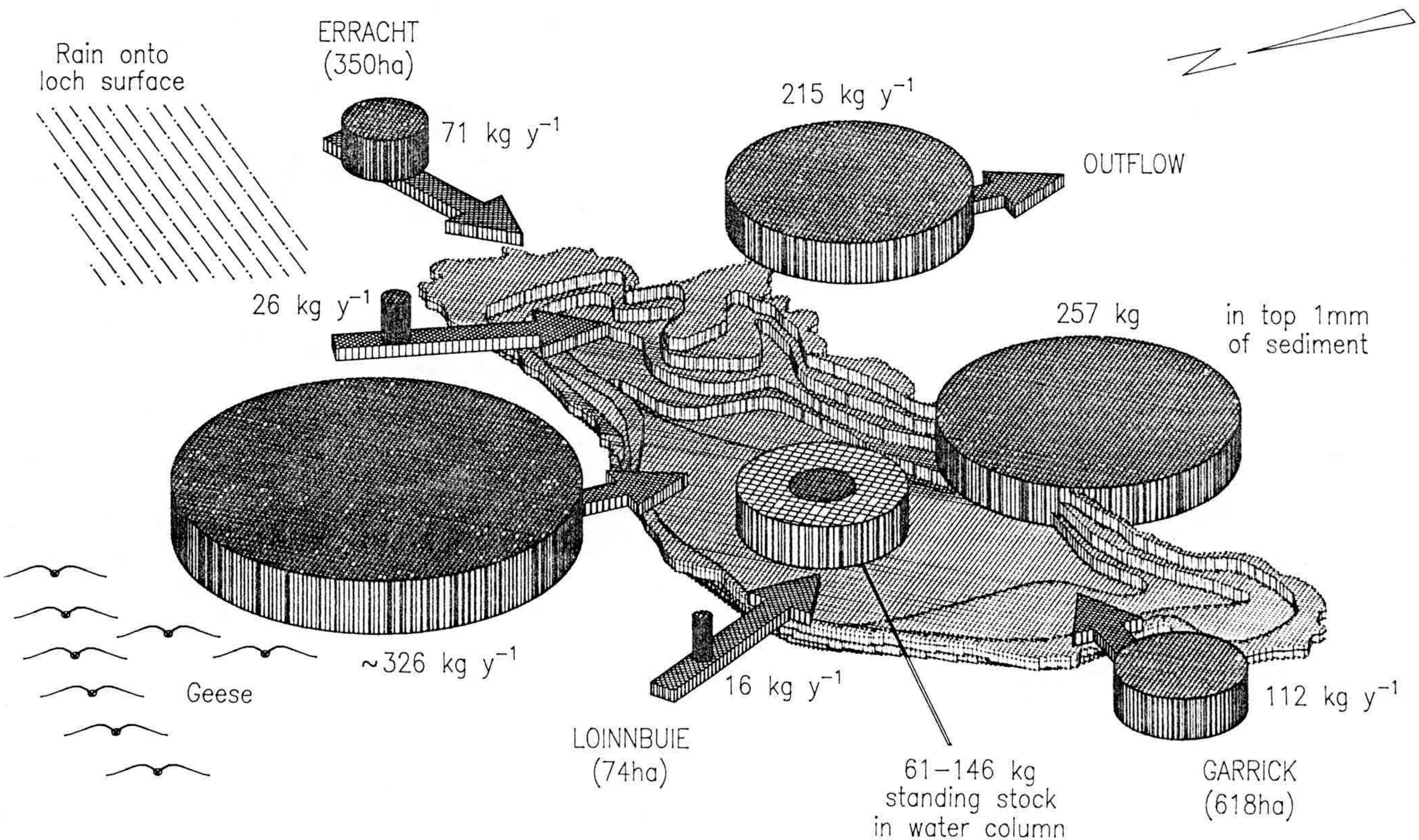


Figure 24

The values shown relate to the 12-month period August 1987 to July 1988 and a flushing rate of 1.70 loch volumes over that period.



LOCH EYE – gross fluxes of total phosphorus

Natural
Environment
Research
Council