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LOCH LEVEN PHOSPHORUS LOADING

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SUMMARY

1. The total loading of phosphorus (P) to Loch Leven, and the various external contributions have been assessed: of the results relating to run-off, sewage and industrial effluents, rain and wildfowl, only those on rain are seriously in question - as over estimates.

2. The history of enrichment of the loch is reviewed using literature on algal remains in the sediments, on plankton records and on rooted aquatic vegetation. Eutrophication trends are outlined with special reference to algal blooms, reduction in clarity and shifts in chemical quality of the water; these developments have had deleterious effects on the loch as a source of water for domestic and industrial use, on its trout fishery, its amenity (tourism) value and its international conservation status. Concern over these issues led to the present study.

3. Full details are given in the report on (i) field and analytical methods, (ii) data handling procedures and (iii) the calculation of flows, concentrations and loadings. An attempt is made, throughout, to demonstrate that the data make sense and justify the conclusions drawn from them. Reasons for discarding the spurious results on rain P are given.

4. The total external loading of phosphorus in 1985 was 20.5 tonnes (as total P - TP); run-off contributed 39.6%, treated sewage 25.9%, industry (woollen mill) 30.6%, rain falling on the loch surface 2.0% and roosting geese <u>ca</u> 1.8%. Equivalent figures for the soluble reactive fraction (SRP) available for algal growth are: total 12.3 tonnes to which run-off contributed 28.7%, sewage 31.4%, industry 36.8%, rain 1.6% and wildfowl <u>ca</u> 1.5%.

   Of the loading originating at the sewage treatment works (5323 kg TP ann^{-1} including 3847 kg SRP ann^{-1}), Kinross North contributed 38.7% of TP (and 44.8% of SRP), Kinross South 26.2% of TP (17.2% of SRP), Milnathort 25.9% TP (28.9% of SRP) and Kinnesswood 9.2% of TP (9.1% of SRP). The TP loads are equivalent to 1.77 - 2.42 g person^{-1} d^{-1}.

5. The mean, volume-weighted, concentration of P in water entering the loch is 2.4 times the mean in-loch concentration of 63 µg TP l^{-1}; this suggests that internal loading of P from the sediments is relatively unimportant over the year as a whole. Indeed, the loch retains phosphorus: the amounts leaving <u>via</u> the outflow are equivalent to only 40% and 12% of the inputs of TP and SRP respectively.

6. As a result of high rainfall in 1985 (1250 mm), flushing was high (> 2.5 loch volumes). The total input of water was 134.4 x $10^6 m^3$ (124.5 x $10^6 m^3$ in run-off and 9.9 x $10^6 m^3$ in direct rainfall); this compares well with the estimated export of 130.8 x $10^6 m^3$.

7. Relationships between the 1985 estimates of P loading (1540 mg m^{-2}) and P (63 µg l^{-1}) and chlorophyll (21 µg l^{-1}) in the loch fit closely to those predicted by certain eutrophication models, which also highlight the importance of flushing rate. Information extending back to 1937 on flushing rate has been used (i) to forecast its likely range and frequency, and (ii) to assess its effect on predicted P loadings and on P and chlorophyll concentrations, assuming a range of P reduction situations. This analysis indicates 2 major points. The first assumes that no control is exercised over

P supplies, and point-sources remain at present levels of ca 12 tonnes ann^{-1}. Total loading would then be reduced purely as a result of the expected natural variation in flushing rate; at the most, however, the reduction would be only one-tenth in, on average, one out of 4 years. The second point assumes removal of 80% of P in mill effluent; this would reduce annual mean chlorophyll in the loch in, on average, 9 out of 10 years to levels such that water clarity would increase. As a consequence, the spread of rooted macrophytic vegetation and the development of associated invertebrate populations would be enhanced. Following a P control strategy, concentrations in the loch would fall to within 5% of predicted decreases in 0.6-1.6 y at the long-term mean annual flushing rate of 1.88 y.

8. It is recommended that:

(a) immediate attention be given to preventing P-rich industrial effluent entering the loch; a consent recently set by the FRPB on this discharge, is intended to reduce loadings from this source by 80% from 1 May 1987.

(b) a programme of P loading and in-loch nutrient and chlorophyll surveillance be started as soon as possible; models predict that the reduction referred to above, should have a significant effect on these factors and, through expected increases in water clarity, on macrophyte distribution.

(c) strict control on increases in P in treated sewage be maintained; the necessity for removal of P from this source, however, would be determined by the monitoring programme - in any event, P stripping would not be logistically feasible before 1990.

1 INTRODUCTION

1.1 Aims and scope of this report

The main aim of this report is to present the findings of a 2-year study of the inputs of phosphorus (P) to the shallow eutrophic Loch Leven. A number of external sources were investigated in order to assess (i) the total input, and, (ii) the contribution from each source. These are, in decreasing order of relative importance according to earlier work (Holden & Caines 1974; Bailey-Watts 1983): point-sources of treated sewage effluent and industrial waste; non-point (diffuse) sources in run-off (mainly stream-borne water), in rain falling directly on the loch surface, and in the droppings of wildfowl. An internal supply of P - *via* recycling from the sediments of the loch itself - was not investigated, but its general significance was judged by comparing P masses in inflow, loch and sediment interstitial waters. In addition to demonstrating how the work was done, and the main features of the data collected, the information is interpreted, where appropriate, with special reference to the feasibility of controlling the eutrophication of the loch by reducing the P supply. For the 1985 data, relationships between P loading, P and chlorophyll in the loch, and factors such as flushing rate, fit very closely to those predicted by certain eutrophication models. The models highlight the important influence of flushing rate and data extending back to 1937 on this factor are used (i) to forecast its likely range and frequency, and (ii) to assess its effect on predicted P loadings and on P and chlorophyll concentrations in the loch, assuming a range of P reduction situations. Some aspects of the history of research on Loch Leven are reviewed first, placing into context the focus on nutrient loadings, concern over needless enrichment, and the rationale for the present study. While the report contains numerous illustrations of seasonal variation in water discharges, P concentrations and loadings, the main digest of information refers to annual figures. Further analyses of the data will focus on seasonality. Phosphorus data are exhibited with reference mainly to TP and SRP figures, although neither PP nor SURP information is completely ignored.

The nutrient enrichment (eutrophication) of freshwaters has interested aquatic ecologists for many years (Hasler 1947; Vollenwider 1968; Milway 1970). There was already an enormous literature on the subject some 15 years ago (Lund 1972), reflecting the major ecological and economic significance of the enrichment process - in particular those undesirable effects resulting from enhanced industrial (including agricultural) activities of man (Lund 1980; Greene & Hayes 1981 and, for Scotland, Greene in press).

In spite of the widespread appreciation of the importance of nutrient supplies to the functioning of aquatic systems (Jorgensen 1969; Le Cren & Lowe-McConnell 1980), the majority of works on eutrophication focus on lake chemistry and biology, eg on Loch Leven chemistry, Holden and Caines (1974) and Bailey-Watts (1986) and its phytoplankton, Bailey-Watts (1974, 1978, 1982, 1986). The actual process of enrichment and the rates of supply of nutrients to the receiving basins have been assessed for rather few waters. Notable examples include Lake Mendota, Wisconsin, U.S.A., (Brock 1985) where studies of this type have been attempted since the 1940s, and Lake Dillon, Colorado (Lewis *et al* 1984) where nutrient inputs have been assessed for the first time, 20 years after the formation of the lake by impoundment in 1963. There are good examples too from Europe. Yet, even in Switzerland, the country with arguably the most experience in this field, there are few thorough investigations - with short interval sampling and closely integrated flow gauging and chemical programmes - and, of these, the majority focuses on events following P reduction (Ambuhl 1981; Marsden 1986 and Mr. M. Marsden pers. comm.). In this respect, and taking the UK as an example, the new loading study on Loch Leven is paralleled only by investigations on Lough Neagh (Smith & Stewart 1977) and the Norfolk Broads (Osborne 1981).

Loadings of nitrogen and phosphorus to Loch Leven were estimated earlier (Holden & Caines 1974; Holden 1976); the results provide a background to the recent work but are based on a less intensive sampling programme than the one adopted now. In any event, in the 14 years that have elapsed since that assessment, inputs of nutrients have altered considerably: the character and intensity of agriculture have changed (Cuttle 1982), production of P-rich effluent by a loch-side industry has fluctuated (Bailey-Watts 1983) and the P loading in treated sewage effluent has increased as the number of people residing in the catchment has grown.

1.2 The history of Loch Leven research with special reference to eutrophication and the role and significance of phosphorus (P)

Loch Leven became more eutrophic, as do all lakes, with gradual in-filling from the time of its formation. Bearing in mind the important influence of lake morphometry on eutrophy (Rawson 1960), it is probable that the early geologically-mediated reduction in water depth and the later man-made lowering of water level by 1.5 m in between 1830 and 1832 (Kirby 1974) enhanced the effects of the enrichment process. Shallow lakes, as a consequence of their more favourable light regime, are generally more productive per unit (mass) of nutrients supplied than deeper waters (see Sakamoto 1966; Hutchinson 1967).

It is the later stage of the enrichment process, associated with human cultural activities, that is the concern of this study. Evidence of this phase of eutrophication over more recent times comes from a number of studies. First, the stratigraphy of diatoms in the sediments shows changes, becoming more like the current eutrophic assemblages (Haworth 1972). Second, shifts in the population densities of other microscopic algae and of rooted macrophytes, including large algae (Charales), are documented. Bailey-Watts (1974) reviews key papers extending back to the turn of the century illustrating an increase in phytoplankton productivity (Wesenberg-Lund 1905; Bachmann 1906; Rosenberg 1938; Brook 1958, 1965). Jupp, Spence and Britton (1974) similarly cover information on macrophytes which have declined in abundance. During the approximately 25-year period for which nutrient data are available, concentrations of nitrate-N in the loch and its inflows also reflect the eutrophication trend (Holden & Caines 1974; Bailey-Watts & Maitland 1984). The situation as regards P is not so clear; this may reflect the contrasting positions of N and P in respect of the likelihood of their limiting phytoplankton growth (see below). Bailey-Watts and Maitland (1984) illustrate a relatively unchanging pattern of total P concentration in the loch (using data for 1971 to 1976, 1982 and 1983) in spite of marked fluctuations in the concentrations of P in one of its largest and heaviest P-laden inflows.

1.3 Major events leading to a desk study (1983), its main findings and the rationale for a new assessment of phosphorus inputs and the feasibility of controlling the loading

In common with studies on nutrient loadings elsewhere, the current work on Loch Leven was largely initiated by concern about undesirable changes in water quality associated with eutrophication. Similarly, interests in a freshwater fishery provided the stimulus for the studies on Lough Neagh (Smith 1983), and concern about effects of algal blooms on macrophyte growth stimulated work on the Norfolk Broads (Moss 1977). The nature and density of phytoplankton at Loch Leven led to its being considered unsuitable on economic grounds as a supply of potable water (Johnson, Farley & Youngman 1974); the serious consequences of eutrophication for other drinking water reservoirs elsewhere are widely reported (Greene & Hayes 1981). Furthermore, the chemical and biological quality of Loch Leven water has given rise to concern about the effects on tourism in the area and on the requirement for quality water by downstream paper mills.

As a result of the developments at this National Nature Reserve, the Nature Conservancy Council commissioned a desk study (Bailey-Watts 1983) to review the situation with regard to P inputs: had the total loading increased since the early 70s, where was the P coming from, and would control of the inputs, if feasible, be effective? The study concluded that the majority of P entering the loch from outside did so from point sources, the main significance of this suggestion being that point effluents are controllable, and a considerable reduction in the % of P loading could be achieved. If, as is thought to be the case with N, the greater part of the loading (mainly in the form of nitrate) had been considered to occur via diffuse run-off and land-drains (Holden 1976; Cuttle 1982), such effective control could not be so easily envisaged. The review also suggested that if a large percentage of P could be prevented from entering the loch, the incidence of algal blooms would be appreciably reduced. This is because the population maxima of the densest crops of phytoplankton appeared to be already controlled, (that is, limited from proliferating further) by the present P levels. A partial reduction in industrial P inputs to the loch in the 1970s had been paralleled by a decrease in algal biomass, indicated by falling mean annual concentrations of chlorophyll (Bailey-Watts 1978). In contrast, although N appears to fall to algal growth-limiting levels in summer, it is present well in excess of requirements throughout most of the rest of the year (Bailey-Watts 1986). Furthermore, some of the particularly unsightly algal aggregations consist of certain species of blue-green 'algae' - cyanobacteria - which are not solely dependent on nitrate as a source of inorganic N (Carr & Whitton 1982). As a consequence, even if a large reduction in the N loading could be achieved, it would probably not effect a corresponding decrease in algal biomass. As these views rested on essentially 'old' loading data, the desk study also emphasised the need for up-to-date quantitative information on P loading - hence the current programme.

2 THE STUDY SITE

2.1 Loch Leven and its catchment

As the site of a UK main contribution to the International Biological Programme from 1966 to 1972 (Golterman 1975; Le Cren 1976; Le Cren & Lowe-McConnell 1980), many features of Loch Leven are well-documented. Of special relevance to the present study are papers on the geomorphology of the loch (Kirby 1974), its physical environment, structure and internal physical conditions (Smith 1974), the distribution of its bottom sediments (Calvert 1974) and its nutrients (Holden & Caines 1974).

The loch ($56^{\circ}10'$N, $3^{\circ}30'$W) was formed 16000 years ago by the advance of an ice-sheet into a natural bedrock basin overlying Old Red Sandstone. Kirby (1974) discusses the formation in detail. Being now of large surface area (1330 ha, Table 1, Fig. 1), small mean depth (3.9 m) and situated in an exposed maritime zone on the eastern edge of the Atlantic Ocean, the loch is usually well-mixed and unstratified (Smith 1974).

Values for the average annual inflow and mean renewal time in Table 1 (120.7 x 10^6 m^3 and 5.2 months respectively) are for the period 1959-1969 and based on rainfall records for 1916-1950, information on changes in loch storage for 1959-1969, and evaporation data for 1950-1964. The throughput value will be compared below to that estimated from the present study. On the basis of the long-term records, Smith (1974) and Bown and Shipley (1982) commented on the more or less even distribution of rainfall throughout the year. Monthly values 1968 to 1983, however, reveal considerable variation between eg 24 mm in June 1968 and 197 mm in October 1968, and 15 mm in August 1976 and 190 mm in October 1976. Over the period 1968 to 1983, the annual rainfall ranged from 640 mm (1975) to 1190 mm (1968) and precipitation in any one month varied considerably; eg for April, 5 mm (1980) to 136 mm (1973) and, for December, 22 mm (1975) to 176 mm (1978). Marked seasonal and annual differences in temperature have also been recorded. Since the loch comprises a broad expanse of shallow water, its temperature closely parallels that of the air (Smith 1974).

Figure 2 shows the loch in relation to its catchment; for the purposes of analysing water discharge, P concentration and P loading data (see below), the watershed is sub-divided as shown. The total catchment area is 145 km^2, consisting largely of land used for crops and grass (61%) and rough grazing (21%).

The main drainage courses comprise 4 systems: North Queich, South Queich, Pow Burn and the Gairney water. As will be demonstrated later, these together drain 84% of the total catchment. Nevertheless, minor water courses draining about ½ of the remaining 16% of the watershed were also gauged.

2.2 Sampling sites

Sites chosen for the gauging of water flows and collection of samples for P analysis are included in Figure 1.

2.2.1 Rivers and streams

By adopting the sampling coverage shown above, an attempt was made to account for all the water entering the loch; in addition to the major stream systems, a number of smaller water courses was investigated, including the Camel Burn (Ca) and drainage channels (Ba and Da) which pass through agricultural land. Phosphorus losses from sectors of land abutting the loch but apparently not drained by well-defined water courses, were calculated, assuming P loss/catchment area relationships obtained from adjacent (gauged) areas (see section 3.1.2). Insofar as such sectors are areas of particle deposition rather than erosion - because there is no water course of high velocity and, therefore, with substantial carrying capacity - the method of calculation will lead to overestimated P losses. However, the areas amount to only 6% of the total catchment.

2.2.2 Point sources of phosphorus

Earlier studies (Holden & Caines 1974; Bailey-Watts 1983) suggested that, taken together, the point supplies of P comprise a much greater source than diffuse river-borne inputs of the nutrient to Loch Leven. In view of their likely continued significance, and bearing in mind that they could be controlled, a considerable amount of time has been spent investigating them. In this area, sources of P-rich liquor arise from 4 sewage treatment works and from one industrial concern; the positions of their discharges are indicated in Figure 1.

a. Effluent from sewage treatment works (STWs)

A total of 4 works serve the towns of Kinross and Milnathort and the village of Kinnesswood. These are as follows:

(i) Kinross North STW is a modern works utilising an oxidation ditch and clarifier. It serves an estimated population of 3,200 and discharges directly to the loch at a point on its west side.

(ii) Kinross South septic tank provides only rudimentary sewage treatment. It serves an estimated population of 2,100 and also discharges directly to the loch - just south of the South Queich.

(iii) Milnathort STW is an older, percolating filter, works serving an estimated population of 1,950. The effluent from here discharges to the North Queich between sampling sites Nb and Nc on that river.

(iv) Kinnesswood STW is also a percolating filter works and serves an estimated population of 555. It discharges to the loch <u>via</u> a small ditch on the N.E. shore.

All these works serve combined sewerage systems. This means that foul sewage and surface water are carried by one sewer. The population figures quoted above were supplied by the Water Services Department of Tayside Regional Council.

b. Industrial effluent

Phosphorus-rich effluent from a woollen mill enters the South Queich near its confluence with the loch, ie between sampling points Sa and Sc on this stream.

2.2.3 Other sources of phosphorus

a. Rain falling directly on the loch surface

Because Loch Leven is broad and shallow, the volume of rain falling directly on its surface can be quite significant in terms of the (mean) loch volume. For example, over a mean depth of 3.9 m, even a moderate annual rainfall of 1,000 mm (= 1 m) represents a gross input of 25%. While corrections for evaporative losses reduce the net input of water, they do not affect the amounts of P introduced. It was considered important, therefore, to assess the concentrations of P in rainwater; if these proved to be high, the background P loading from this source could be appreciable.

b. Wildfowl

The large water surface, islands for nesting, extensive shallows and rich adjoining farmland make Loch Leven a very suitable site for wildfowl (Allison & Newton 1974). It often supports the largest concentration of breeding ducks in Britain, including tufted duck, mallard, gadwall, wigeon, shelduck, shoveller and teal. More important, in the context of eutrophication and P inputs to the loch, are the roosting in winter of many thousands of Greylag (Anser anser (L.)) and Pinkfooted (A. brachyrhynchus Baillon) geese. As a result, waterfowl numbers may exceed 20,000 in autumn.

It is largely on the strength of these wildfowl populations that the loch was given national and international conservation status. In March 1964 it was declared a National Nature Reserve by the Nature Conservancy (now Nature Conservancy Council) under an agreement with the owner. It was also included in the first list of internationally important sites drawn up at the International Conference on the Conservation of Wetlands and Wildfowl held at Ramsar, Iran in 1971 (Morgan 1974).

No new research was done to estimate the inputs of P to the loch in wildfowl faeces. Instead, the loadings were calculated from seasonal counts of geese and published figures for outputs of P by geese (see sections 3.2.4, 4.4.3 and 5.3.3).

c. Sediments

There is constant exchange of P (and other nutrients) between the water mass and the bottom sediments. Of special importance in determining the dynamics of P, is knowledge about recycling. Phosphorus may be released into the water column from sedimented particulate material originating outside the loch or forming (eg as plankton) within the loch. The measurement of the processes involved and the many, complex factors controlling them, are beyond the remit of the study. Some attention had to be given to this aspect of the loch's ecology however, as there is general concern that the internal loading of P from the sediments would not only continue, but possibly increase, after external sources were reduced.

On the practical side, the magnitude of the 'reservoir' of soluble reactive P in the surface sediments has been assessed from preliminary analyses of interstitial water (sections 5.2.5 and 5.3.5). A guide to the importance of internal loading relative to external supplies on an annual basis, is gained by comparing the mean in-loch concentration of P with the mean P concentration of water entering the loch (section 6.7). After the completion of this report, a detailed, seasonal budget of P inputs to, and outputs from Loch Leven will be attempted from the data now available. Until that analysis is completed, no internal loading can be calculated. However, previously published work on Loch Leven and elsewhere will be discussed (also section 6.7) in order to identify the conditions associated with P release and consider how important the process is and whether it is likely to change, in the event of a reduction in external loading.

2.2.4 The loch and its outflow

In order to maintain observations to the extent indicated by the distribution of sampling sites in Figure 1, work on the loch itself was kept to a minimum. In this respect, it was considered unnecessary to devote valuable time and resources to sampling the open water area. Previous work shows that nutrient concentrations and algal population densities usually reflect minor differences between different areas of the loch. The outflow site (sluices) is particularly representative of open water (Bailey-Watts 1974, 1978, 1982); measurements there, together with discharge records, also provide information on the volume of water and the amounts of chemical and biological material leaving the loch. For these reasons, station L in Figure 1 was used.

3 FIELD METHODS

3.1 Hydrology

3.1.1 Rivers and streams

The Forth River Purification Board (FRPB) operates 3 gauging stations in the Loch Leven catchment: on the South Queich at Kinross (N.G.R. NO 118 016), on the North Queich at Lathro (N.G.R. NO 114 042) and on the Greens (Pow) Burn at Killyford Bridge (N.G.R. NO 115 053). These gauges (included in Figure 1) are all velocity area stations in which a calibration is derived by current metering over the range of river level and used to convert a level record into discharge. This process is described in section 4.1. River level is recorded at half-hour intervals at these stations and each recording is converted to discharge before processing to produce daily means, flow extremes and so on. The calibration is checked by current metering at regular intervals, usually monthly, and corrections applied as necessary.

Most of the sites used in the study were at locations away from permanent gauges so temporary stations were required. Gauge boards were installed at most of the river chemical sampling points (Figure 1), with the exception of the small ditches on the N.E. shore of the loch (see below), and calibrations obtained for the site by current metering. These gauge boards were read at the time a sample was taken and the level reading converted into discharge using the derived calibration. The system worked well for all sites except the North Queich at Burgher Bridge, where a few recordings were affected by a high level in the loch.

Four small ditches running through agricultural land on the N.E. shore of the loch had to be dealt with in a different way; they were often only a few millimetres deep, and their mobile sandy beds made the use of calibrated gauge boards unreliable. Flow at these points was therefore measured by current meter on 2 occasions, near the time of peak flows in many of the other (routinely gauged) rivers. They were sampled for chemical analysis from April 1985, not January as in the case of the other sites. To put them in perspective, it should be noted that they drain <4% of the catchment.

3.1.2 Point-source discharges

The point-source discharges gauged were the woollen mill (site Sm in Figure 1) and the sewage discharges at Kinross South septic tank, and the Kinross North, Milnathort and Kinnesswood STWs. At each of these a flow gauging structure was installed and continuous level records obtained. These were converted to discharge using the known calibration of the structure. Whilst a permanent level recorder was installed at the woollen mill, recorders were installed only during the intensive sampling periods at the STWs.

The discharge at the woollen mill was gauged after the final mixing tank, immediately before discharge to the South Queich. A glass-fibre trapezoidal flume was installed along with a wet well and recorder house to enable accurate measurement of the turbulent flow. This level was recorded on a weekly chart, which was replaced by a daily chart for a short period to allow a better resolution of the rapid fluctuation in the process water.

At all STWs except Kinross North, 'V' notch weirs were installed and levels were recorded by ultrasonic detectors recording on a daily chart. At Kinross South and Kinnesswood the recorder was installed on the outflow from the works, so any storm overflow was not included. At Kinross North the outflow channel was not hydraulically suitable so the inflow was measured; it was possible however to measure storm overflow drainage here too. In contrast to the other works, gauging was done with float level recorders at rectangular sections.

The situation at Milnathort is more complicated in that 2 of the 3 filterbeds discharge to one drain and the third to a separate drain. 'V' notches were placed on both these channels and the flows recorded simultaneously. Because this STW is very low lying, the outflow from the drains frequently backed up into the works; however, valid data were obtained for 2 weeks here.

3.1.3 Rain falling on the loch surface

An estimate of the volume of rain falling on the loch surface was obtained from 4 rain gauges deployed in the Loch Leven catchment (see Figure 1). Other indications of the volume of rain water entering the loch in this way were derived from the water collectors referred to in section 3.2.3.

3.1.4 Loch volume

A depth/storage relationship was derived for the loch, from 2 published sources of widely differing age. A report to the Loch Leven Trustees by John Sang (Sang 1872) gives details of the acreage of the loch at spillway level, shortly after completion of the construction work at the existing sluice gates. A recent bathymetrical study (Kirby 1971) allowed the area of the loch to be calculated at different depths. These 2 surveys showed excellent agreement despite the long period between them.

The volume of the loch at different levels was calculated from the area at 50 mm increments of loch height, assuming a uniform slope between each increment. The depth/storage curve so derived is very smooth, lending confidence to the assumption of uniform slope over small increments. Smith (1974) also calculated the volume of the loch using an equation derived from bathymetric data published by Kirby (1974) and from additional, unpublished morphometric data supplied by Kirby. This equation relates loch level (H, in metres A.O.D) and volume (V, in millions of cubic metres) as follows:

$$V = 12.12\,H - 1.2429$$

(Mr I R Smith <u>pers. comm.</u>). Figure 5 (the volume-depth curve) of Smith (1974) shows that changes in volume are linearly related to changes in depth, particularly at the high levels.

3.1.5 The outflow

The volume of water leaving the loch via its outflow was calculated from daily mean discharges derived from sluice height and gauge readings recorded continuously ($\frac{1}{2}$-hourly intervals).

3.2 Chemistry

3.2.1 River and stream waters

Water samples for chemical analysis were taken at 8-day intervals in duplicate at each site (Figure 1). This regime was adopted because the chemical quality of certain water courses - especially those receiving industrial or sewage effluent - was expected (and indeed was later shown) to vary according to the day of the week. Half-litre polyethylene sample bottles were first rinsed in the waters; the samples were then collected by dipping the bottles ca 20 cm below the water surface, with their inlets facing upstream. Water temperature was recorded at the time of sampling by means of a mercury-in-glass thermometer calibrated to 0.1 Celsius degrees.

3.2.2 Point-source discharges (treated sewage and industrial effluents)

The sampling programme was designed to take account of the anticipated variations in the P loading in each discharge. In addition to random variations, the following cyclic variations would be expected:
DAILY - Likely to give the greatest variation.
WEEKLY - Showing the influence of a Monday washing day etc.
ANNUAL - Showing the effect of eg seasonal population increase with tourism, seasonal variation in rainfall, and seasonal effects on the sewage treatment process.

It was thus decided to mount 7-day monitoring runs using 24-hr automatic samplers. Experience from a previous exercise suggested a sampling interval of not more than approx. 30 minutes. It was originally hoped to carry out 4 runs on each works covering spring, summer, autumn and winter, but in the event this did not prove possible.

Samplers were loaned, principally by Tayside Regional Council:
Warren-Jones samplers; these were set to take 1 sample every 15 minutes, combining 4 samples in 1 bottle, ie 1 bottle per hour for each 24 hour period. Samples were taken virtually instantaneously by a small submerged centrifugal pump.
Epic 1010 samplers; these held 12 bottles and were set to take a sample every 30 minutes and 4 samples per bottle ie 1 bottle every 2 hours. Samples were taken virtually instantaneously but by a peristaltic pump.

For both samplers, high density polyethylene bottles were used with no preservative added. Samples were returned to the laboratory as soon as possible after the completion of each 24-hr sampling run.

For the majority of runs, flow-related composite samples were prepared to cover each 24-hr period. They were prepared by firstly calculating a figure proportional to the average flow for the duration of sampling of each bottle in the automatic sampler, then taking a volume from each bottle in proportion to this figure to prepare the composite. In order to estimate diurnal variations in loading, individual samples were analysed for some runs on Milnathort and Kinnesswood works.

The information obtained from this part of the study is sufficient for assessing the likely export of P from STWs, and, therefore, their importance relative to the other supplies. However, considerable problems were occasionally experienced. These were due to, for example, exceptionally high flows and breakdown of automatic samplers.

Routine samples of the industrial effluent were taken on the 8-day schedule in the manner described in section 3.2.1 for the 'natural' streams. In addition, an intensive study was carried out over a 24-hour period on 2-3 December 1985 in an attempt to assess short-term variation in the export of P from this source. Samples of the effluent issuing from the pipe were collected hourly. The time of collection was recorded and levels of P concentration were related later to the corresponding instantaneous flow values read from the chart recorder referred to above.

3.2.3 Rainfall

Collections of rain for chemical analysis were made in 1986, there being little time for this in 1985, whilst laboratory and field work on the inflows was carried out. Four collectors were deployed - 2 near the N.W. shore of the loch and Kinross (Kennel Cottage and Factor's Pier) and 2 near the S.W. shore and the outflow ie at Larch Cottage and Vane Farm (see Figure 1).

Each collector consisted of a 20-cm diameter polyethylene funnel, with a 1-litre bottle attached to its stem. The bottle was supported in a 50 cm length of spirally-reinforced tubing, itself mounted on a wooden post so that the top of the funnel was 2 m above the ground. In an attempt to dissuade birds from perching on the funnels, a network of wire spikes was clamped outside the funnel; the spikes protruded some 15 cm beyond the upper (sharpened) edge of the funnel, but, in following the contours of the funnel, did not occlude its opening.

Two types of problem were encountered in the collection of rain. One related to the swarming of midges over the collectors and insects dropping into the water. The other concerned the behaviour of the collectors at the sites chosen, ie how suitable they were as recorders of 'open-water' precipitation chemistry and volume, and how they compared to standard Meteorological Office rain gauges. These issues will be considered later (section 5.2.4).

3.2.4 Wildfowl inputs

Numbers of wildfowl associated with the loch and its adjacent fields were counted at monthly intervals September to March. For the purposes of the present study, numbers of Pinkfooted and Greylag geese for the periods January to March and September to December 1985 have been used; these were extracted from standard forms issued by the Wildfowl Trust (Slimbridge, Gloucester) listing the figures obtained by the NCC Senior Warden's team at Loch Leven.

3.2.5 Loch sediments

A Jenkin sampler was used to obtain undisturbed cores of muddy sediment and overlying water - at 4-m and 10-m sites - on various dates between 24 April and 30 June 1986. On shore, water within 20 cm of the bottom deposits was withdrawn by siphoning into plastic tubs and stored in a cool-box until later preparation and chemical analysis. One- or two-centimetre slices of mud were also withdrawn from the core from the sediment surface to between 5 and 10 cm depth into the deposits, and at 15 cm depth. The material from each slice was transferred to a plastic Sterilin tube and then stored in a cool-box for laboratory analysis.

3.2.6 The loch and its outflow

Loch water was collected near the outflow (see section 2.2.4) in the same manner as described in section 3.2.1 for river and stream waters.

3.3 Phytoplankton

Samples of water for phytoplankton studies were taken in parallel to those collected from station L (the loch near its outflow) and used for chemical analysis (see section 3.1.2).

4 ANALYTICAL METHODS

4.1 Derivation of river height-discharge relationships (rating curves)

All the river sites relied upon the calibration of channel sections to derive a height-discharge relationship. These were produced by combining current meter gaugings over a range of flows by means of an iterative best-fit algorithm. This assumes a relationship between discharge (Q) and water level (H) of the following type for each calibration:

$$Q = b\,(H + a)^c$$

The program assumes a value for a and calculates b and c for a best fit line. The standard error of the mean relationship is calculated for the line produced and another value of a is chosen and the process repeated. The new standard error is compared to the previous one and the next value of a is chosen so as to optimise the standard error. When the standard error is at a minimum the final equation is output.

The value of the optimised standard error is used as a guide to the need for more current meterings at a particular site. In general the standard error of the mean relationship is accepted if it is less than 1.0; permanent river gauges are usually calibrated to a standard well above this. If the error is greater than 1.0, more gaugings are required to produce a useable calibration.

4.2 Chemical analytical methods

4.2.1 Analytical Quality Control (AQC), precision and bias testing

(a) Design of programme

The Loch Leven project involves the analysis of large numbers of samples by 2 laboratories, so it is necessary to check comparability of the results and their accuracy. The AQC programme which has been implemented, fulfils both of these functions and is based on the protocol used extensively by the Water Research Centre (Wilson 1979). The various stages of the programme are as follows:

- (i) Establish working group, to plan and co-ordinate all subsequent activities.
- (ii) Define determinands and required accuracy, to ensure clear specification of analytical requirements.
- (iii) Choose analytical methods capable of the required accuracy.
- (iv) Ensure unambiguous description of methods, to ensure that the chosen methods are properly followed.
- (v) Estimate within-laboratory precision, to ensure that each laboratory achieves adequate precision.
- (vi) Ensure accuracy of standard solutions, to eliminate this source of bias in each laboratory.
- (vii) Set up quality-control charts to maintain continuing check of precision in each laboratory.
- (viii) Check between-laboratory bias, to ensure that each laboratory achieves adequately small bias: tests to be repeated at regular intervals to maintain a continuing check on bias.

Three laboratories took part in this exercise: those of the Freshwater Biological Investigation Unit (Co. Antrim, N. Ireland), the Forth River Purification Board (Stirling, Scotland) and the Institute of Terrestrial Ecology (Edinburgh, Scotland). The determinands examined were total phosphorus (TP) and soluble reactive phosphorus (SRP) and all 3 laboratories used very similar analytical methods.

Targets for precision and bias were set in terms of both percentage and an absolute target.

Ranges: River waters (ITE and FBIU) 0 - 500 μg $P.l^{-1}$
Sewage effluents (FRPB) 0 - 5000 μg $P.l^{-1}$
Targets for maximum total error (95% confidence level):
20% or 10 μg P.l (whichever was the greater)

The total error target was divided equally between precision and bias. Within-laboratory precision testing was done first. This resulted in the following targets: precision, standard deviation of 5% or 2.5 μg P l , bias, 10% or 5.0 μg P l . The next stage (comparison of standard solutions) was omitted, as it was unlikely to cause errors. The final stage was the between-laboratory bias test.

(b) Tests of precision and bias
Two replicate determinations were made, in random order, on each of the following solutions, on each of 10 days:-

(i) A blank solution, ie the appropriate volume of water from the batch used to prepare the standard solutions (ii) and (iii), which were subjected to the same analytical procedures as samples and standards.
(ii) Standard solution of high concentration
(iii) Standard solution of low concentration
(iv) A natural water sample of similar composition and TP concentration to those encountered in the Loch Leven work.
(v) The natural waters samples in (iv) above, spiked with a known amount of P, added as a known volume of a standard solution.

The volumes of each of these solutions taken for analysis were the same as the volume taken in the method or, if the method allows for a variation in the sample volume taken, the volume used should be that applicable to the lowest concentration range of the method.

The standard solutions (ii) and (iii) were prepared prior to the batch of analysis in the same way as normally used in preparing calibration standards. The low concentration was approximately one-tenth of the upper limit of the range used and the high concentration approximately nine-tenths of this limit.

The natural water sample was of similar composition and determinand concentration as those of routine interest. The spiked natural water was prepared freshly for each batch of analyses using an aliquot of natural water taken from the sub-sample also analysed in that batch. The concentration to which the water was spiked was approximately 0.5 of the upper limit of the range used.

For bias testing 3 samples were distributed to each laboratory: a standard solution, and high and low concentration real samples. Each sample was analysed 5 times, in order to provide a check on precision.

(c) Calculations

The precision tests enabled the separate measurement of within- and between-batch standard deviations. Between-batch standard deviations were not calculated for the real sample and spiked real sample because of actual variations in concentrations between batches in some of the tests. A figure obtained from the standard solutions was used in calculating the total standard deviation. The results of the tests were entered on a worksheet which was then returned to the FRPB (see section 5.2.1).

4.2.2 Routine phosphorus analyses

Phosphorus determinations on all stream waters, loch (outflow) water and mill effluent, commenced within one hour of collecting samples at the last site on the sampling tour (total time 4½-5 hours). Bailey-Watts (1986) and Bailey-Watts et al. (1987) describe the analytical methods based on the procedures described by Murphy and Riley (1962). Total amounts of P (TP), and the total soluble fraction (TSP) were each analysed in addition to the soluble reactive fraction (SRP). Unfiltered subsamples were used for TP and filtered subsamples for TSP and SRP. Concentrations of particulate phosphorus (PP) and soluble unreactive ("dissolved organic") P (SURP) were calculated according to:

$$PP = TP - TSP \text{ and}$$
$$SURP = TSP - SRP$$

The relationship between the different P fractions and how they are analysed, is outlined further in Appendix I.

Soluble P concentrations in water overlying mud cores, and in the interstitial water of the core sections of muddy sediment, were analysed as above. The interstitial (pore) water was obtained by centrifuging the wet mud, siphoning off the supernatant and filtering this through a Whatman GF/C pad. Especially in the case of the deeper mud sections, the amounts of water so obtained were often very small, eg 1 - 2 ml. However, this water was commonly very rich in dissolved P, so dilution with distilled water prior to adding chemical reagents for analysis was necessary; sufficient volume of sample was thus produced. In order to assess the 'standing stock' of dissolved P in the surface muds, aliquots of fresh mud were weighed, and then dried to constant weight at 80°C; this enabled the pore-water volume per unit volume of sediment to be calculated. The product of pore-water volume and P concentration is the mass of dissolved P.

Analytical methods used for determining P levels in sewage effluents were similar to those used for other waters. These are as indicated in the previous section.

4.3 Phytoplankton species composition and abundance

Plankton, concentrated by sedimentation after fixing with Lugol's iodine, has been examined and counted in order to estimate the population densities of the different species. Bailey-Watts (1974, 1978) and Bailey-Watts et al. (1987) describe the procedures in detail, and cite taxonomic literature consulted for identification of the algae. For the present study, densities of only the dominant forms were assessed. A measure of species diversity, including rarer forms, was obtained, however, during routine monitoring of the size distribution of the assemblages. For this purpose, the longest axis of normally 30 or 50 randomly chosen phytoplankton individuals was measured; Bailey-Watts and Kirika (1981) and Bailey-Watts (1986) give details of the methods used.

Chlorophyll$_a$ concentration was used as an index of total algal biomass. For this purpose, the pigment was extracted with 90% methanol from phytoplankton concentrated by filtration onto Whatman GF/C glass-fibre discs. The equation proposed by Talling and Driver (1963) was used to convert spectrophotometric absorbances to concentrations of chlorophyll$_a$. No corrections were made for the presence of breakdown products of this pigment - eg pheophytin$_a$.

4.4 The calculation of loadings

4.4.1 River and stream systems

The seasonal information developed from the 8-day records of stream discharges (Q) and P concentrations ([P]) as described in sections 3.1.1, 3.1.5, 3.2.1 and 3.2.6, allow instantaneous loadings of P in run-off (PL_{ri}) to be calculated, ie:

$$Q \times [P] = PL_{ri}$$

A mean loading over, for example, a year, is the average of the instantaneous products (46 in the case of a year); whilst this can still be expressed in units of µg (or mg) P s^{-1} (because flows are commonly expressed in l s^{-1} or m^3 s^{-1}) the values are normally converted to units of kg ann^{-1}.

As will be demonstrated later, loading data of this type were obtained for water courses draining 88% of the total catchment area. Loadings based on regular (8-day) chemical analyses, but on only 2 flow measurements (the means of which have been applied to the whole year's results) were also obtained for the small drainage channels referred to in section 3.1.1 and draining a further 4% of the catchment. The remaining 8% of the watershed comprises: (i) sectors of land not drained by well-defined water courses ('Cavelstone', 'Vane Farm' and an area near sites Ca and Wa shown in Figure 2); (ii) the area (also shown in Figure 2) drained by the Clash Burn (Ha in Figure 1) and the Wood Burn (Wa) which, though sampled regularly for P analysis, were not gauged. Phosphorus loadings from these zones were computed, using the relationships between drainage area, flow and P levels (and thus P losses) found for adjacent (gauged) areas.

Mathematical relationships between flows and measured concentrations, and thus loadings, of P were used to generate loadings for times at which only flow data were available. For the inflow sites Nd, Pc and Sc (in Figure 1) for which flows were recorded every day at ½-hourly intervals, 'continuous' P loadings were generated in this way. The areas drained by these 3 systems together constitute 48% of the total catchment. Similar, daily, loading values were derived for 3 other sites - Gb, Nh and Ua in Figure 1. As flows, measured at the instant of sampling every 8 days at these sites, were highly correlated with those measured at one or other of the sites for which continuous flows were available, daily flows were generated. Daily P loadings were derived as before. The 3 additional sites together drain 28% of the total watershed. Thus, 'continuous' and 'instantaneous' loading information is available for water courses draining some 76% of the Loch Leven catchment. In calculating the annual P loadings based on the derived daily values, the corrections proposed by Ferguson (1986) were made. The continuous values probably provide the better indications of P loading, as the 8-day figures are likely to underestimate the inputs, by missing important, short-lived, episodes of high flows (see eg Smith & Stewart 1977).

4.4.2 Point-sources

Loadings (L, in kg P ie the products of concentration C, in mg P l^{-1} and mean flow Q, in $l\ s^{-1}$) from the STWs were calculated for each period covered by a sample, as follows:

24-hour composite samples	L = C x Q x 0.0864
samples covering 2 hours	L = C x Q x 0.0072
samples covering 1 hour	L = C x Q x 0.0036

Missing data were estimated only where it was likely to yield useful information without significantly lowering the accuracy of the final results. Occasions when sewage was discharged via storm water overflows were also taken into account at Kinross North works, although it was not possible to estimate the contribution from this source for the other 3 works. From the individual sample period results, mean daily, weekly and annual loadings and per capita loadings were calculated. The export of P from the woollen mill was also estimated from short-time interval (eg hourly) measurements over 24-hr periods, giving information on diurnal variation. Another estimate was made from measurements incorporated into the routine 8-day schedule operated for the streams - for information on seasonal variation. As results from these exercises showed little evidence of regular diurnal or seasonal patterns of discharge, P concentration or loading, a third approach is under consideration; this will take account of the production schedules operated by the mill, and the records of its purchases of the P-containing chemicals.

There was little opportunity for generating 'continuous' loading figures for these point-sources; this is because, in contrast to the stream systems, there was little relationship between loading and flow from which to derive the continuous loading figures - even in the case of the industrial discharge for which continuous flow measurements were available.

4.4.3 Rain falling on the loch surface

The amounts of P introduced in precipitation were calculated from the products of nutrient concentration and the volume of rain falling during the various sampling intervals. The values included dry deposited P. As the main aim of this part of the work was to assess whether rain P was comparable to the other sources of the nutrient, a range of likely inputs is presented which use Meteorological Office rainfall figures (see section 3.1.4) and P concentrations commensurate with those measured.

4.4.4 Wildfowl

Loadings from wildfowl were calculated by taking the products of geese numbers (obtained as outlined in section 3.2.4) and figures published by Cooke (1976) and Hancock (1982) on P outputs in geese droppings.

4.4.5 Contributions from all sources - the total loading

The P contributions from all sources were summed to give an estimate of the total loading. Of greater significance, however, is how the contributions compared: only after this is established, can sensible decisions be made on their control. For this purpose, in addition to using the loading results obtained from the present study, it has been thought instructive to consider other possible situations, especially for inputs that are likely to be strongly influenced by, for example, the weather conditions prevailing in a particular year. These thoughts are developed further in later sections.

4.5 The calculation of losses of phosphorus *via* the outflow

Instantaneous values for the rate of export of P from loch *via* its outflow, were estimated from the product of the P concentration and the daily mean discharge on the day of chemical sampling. As there was little relationship between P concentration or loading and flow, 'continuous' export estimates of the type discussed in relation to the inflows were not possible. If changes in P concentration were assumed to be regular between each sampling day, then daily export values could be derived. For present purposes, however, it is thought that the instantaneous values (n = 46) suffice.

5 RESULTS

5.1 Water volumes

5.1.1 River and stream inputs

The annual mean flows of the feeder waters studied ranged over nearly 2 orders of magnitude, ie ca. 0.02 - 1.50 cumecs (Figure 3). Variation in instantaneous flows of at least the larger water courses (Q of > 0.1 cumecs) was also considerable (see below). Four systems come into this category, and, as Table 2 illustrates, they comprise 86% of the total estimated input of river-borne and run-off water to the loch; the table also shows that this percentage is matched almost exactly by that based on the proportion of the Loch Leven watershed drained by these rivers (cf. Figure 2). It is not surprising, therefore, that flows exhibit a high positive correlation with their corresponding drainage areas (Figure 4). This observation is gratifying, but may relate to the fact that the annual mean discharges calculated from the 8-day instantaneous flow values are - in the cases of the 3 sites for which the extra data are available - very similar to the values based on daily flows. Table 3 compares - for stations Nd on the North Queich, Pc on the Pow Burn and Sc on the South Queich - annual mean flows estimated by the 2 methods referred to above; it also gives figures derived by a third method, ie on the basis of the daily mean flows calculated from the continuous (½-hourly) records for the sampling days (which are at 8-day intervals). The close correspondence between the different sets of results does not imply that continuous flow gauging operations can be dispensed with. Indeed, the agreement in the data is a chance one; preliminary calculations of mean flows based on other 8-day sampling schedules, commencing on eg 4 January, 5 January and so on, instead of 3 January as in the case of this study, produced quite different values, which do not match as well with those based on the (365) daily records. The convenient conformity of the present set of instantaneous flow records to the continuous discharge data, also relates to their similar frequency distributions. Figure 5a shows that, in common with general findings on river flows, the flows in the Loch Leven feeder waters approximated to log normal. These plots are similar to those of daily values for the sites referred to above; Figure 5b compares the 2 data sets for site Sc on the South Queich as an example. In other words, the basic field data reflected well the spread of flow values and the relative occurrence of low and high flows. This is an extremely important finding when interpreting information on continuous loadings derived from the measured (instantaneous) estimates. One of the main concerns in studies of this type, is that sampling at regular intervals can lead to an underestimation of the (important) influence of high flow episodes.

Each inflow exhibited similar seasonal patterns of variation in water discharge. Figure 6 illustrates this with information on a river of relatively large proportions, one of small dimensions and one of intermediate size. In relation to these findings, and although the authors are not generally in favour of presenting correlation matrices without the graphs of the source data, Table 4 illustrates 2 main points. First, it can be seen that most of the (linear) correlation coefficients using $\log_{10}Q$ data - are high; for streams referred to in Figure 6, Table 4 indicates that $\geq$80% of variation in flow in one stream was associated with variation in flow in the other two. As a result, it has been considered reasonable to derive daily flows, and, in turn, daily P loadings for some of the sites not gauged continuously (see section 5.3.1(c)). Second, the table illustrates the contrastingly weaker associations involving discharges that (a) did not exhibit 'natural' seasonality in flow and/or (b) were of significant size relative to the receiving water, eg sites on the Kinnesswood Burn (Ka, Kb) which received an intermittent piped discharge (Kp) which, on occasions, reached 100% of the 'natural' stream volume.

The sum of the flows estimated from the 8-day staff gauge readings (see Table 2) gives an annual input to the loch of $110.5 \times 10^6 m^3$. The estimated discharge from the small drainage channels on the N.E. shore raises the input to $114.6 \times 10^6 m^3$. If the remaining ungauged area of the catchment (7.9% of the total) lost the same amount of water per unit area, the estimated total run-off would be $124.5 \times 10^6 m^3$ - similar to the figure of $120.7 \times 10^6 m^3$ derived by Smith (1974) and listed in Table 1.

5.1.2 Point-source effluents

The volumes of point-source effluents were small in comparison to the 'natural' inflows. Typical rates of discharge from, for example, the mill and Kinross North and Kinross South STWs lay between 10 and 20 $l\ s^{-1}$.

5.1.3 Rain falling on the loch surface

One millimetre of water falling over a square metre is equivalent to 1 litre. So, an annual rainfall of 1250 mm over the loch surface of 13.3 km^2 (ie $13.3 \times 10^6 m^2$) gives a volume of $16625 \times 10^6 l$ or $16.63 \times 10^6 m^3$. This input, not corrected for losses in evaporation, was equivalent to nearly one-eighth of the total run-off (a net input) and one-third of the mean loch volume (see below).

5.1.4 The loch

The volume of the loch at the modal water level was taken as $52.4 \times 10^6 m^3$ - as calculated by Smith (1974 - see Table 1).

5.1.5 The outflow

The mean rate of export of water from the loch via the outflow, estimated from the mean daily flows on each of the 46 sampling days, was 4.1478 cumecs. This is equivalent to $130.81 \times 10^6 m^3\ ann^{-1}$, a value only slightly less than the annual input (I) of $134.40 \times 10^6 m^3$ calculated from

$$I = R + (P_s - E_s)$$

where R is the annual run-off ($124.47 \times 10^6 m^3$ - see section 5.1.1), P_s is annual rainfall over the loch surface ($16.63 \times 10^6 m^3$ - see section 5.1.3) and E_s is annual evaporation over the loch surface (taken as 1.2 times the evaporation rate of 420 mm given by Smith 1974); this results in a value of $6.70 \times 10^6 m^3$; so $I = [124.47 + (16.63 - 6.70)].10^6 m^3 = 134.40 \times 10^6 m^3$.

5.2 Water chemistry

5.2.1 Analytical Quality Control (AQC)

The results of the AQC precision tests on our procedures for determining TP and SRP concentrations are shown in Table 5. It can be seen that ITE were well within targets for all solutions for both SRP and TP. FBIU were similarly well within all targets except for the spiking recovery for TP which was on the limit of the acceptable range. For FRPB, testing was carried out on higher concentration samples, of the type expected in the sewage works monitoring programme. Results were well within targets for all solutions.

The results from the bias tests are given in Table 6. All results were well within the targets with the exception of FBIU for TP. The fact that these were so far out probably indicates a simple error rather than a consistent bias of this magnitude in routine analytical results. Contrastingly, the ITE and FRPB results indicate that more stringent total error targets of 10% or 5 $\mu g\ P\ l^{-1}$ would probably have been met.

The control of analytical errors and comparability of analytical results between ITE and FRBP has thus been satisfactorily demonstrated.

5.2.2 Inflows

Concentrations of TP at feeder water sites above industrial or treated sewage discharges ranged overall from <10 $\mu g\ P.l^{-1}$ to ca 4 mg $P.l^{-1}$ (Figure 7). The lower values were recorded in upper reaches of the larger rivers whilst the highest values were more characteristic of the small 'natural' streams (eg Ury Burn, Camel Burn) and the agriculturally-influenced drainage courses (eg sites Ea, Ba); comparatively invariable levels of TP in the loch itself are also well illustrated here. Figure 8 highlights this loch/inflow distinction and shows the considerable differences in TP concentrations, in spite of the general similarity in seasonal patterns of stream flow (cf. Figure 6). Firstly, the timing of annual maxima and minima varied; note, as examples, the late maxima in the Pow and Ury Burns and the early maximum in the Gairney Water, and contrast the April minimum in the South Queich with that for June in the Ury Burn. Differences in frequency distributions of TP concentrations in these waters are reflected in Figure 8; for instance, the flat 'base' to the plot for the Pow Burn contrasts with the trough for that of the Ury Burn. Such differences are more plainly demonstrated however in Figure 9, which also highlights departures from log normality in the inflow values and, again, the relatively narrow range of values recorded in the loch. In addition to different degrees of skewness, a number of modes is indicated.

It is important to consider the composition of the TP ie the proportions of soluble and particulate components. This is because, while virtually all fractions of P may eventually be utilised by algae, SRP is considered to represent the nutrient pool most immediately available for uptake (Kuhl 1974, and Jordan, in Smith 1983 for general considerations, and Stewart & Alexander 1971, and the present authors' unpublished observations for work with Loch Leven phytoplankton). Knowledge of the composition of the P is also important, because the fractions differ in (i) concentrations and their seasonal changes, (ii) loadings, (iii) the relationships between concentrations, loadings and flow, and (iv) susceptibility to run-off from land. These issues will be considered in detail elsewhere, but for present purposes it is sufficient to demonstrate that feeder waters varied in their composition. The proportions of the different fractions are shown in Figure 10, where points near the apices of the triangles reflect a high proportion of particulate P (PP), and those near the bases indicate high proportions of the soluble fractions - SRP if they lie in the left-hand corner and SURP if they fall in the right-hand corner. It is evident that the composition fluctuated seasonally within each river system, but the different waters varied also in the general balance between PP and SRP; SURP contributions were comparatively invariable and generally low (in contrast to the effluent referred to in section 5.2.3). On this basis, the triangular plots have been arranged in Figure 10 to give examples of sites where PP > SRP are at the top (Camel Burn and upper S. Queich) and those in which PP < SRP are at the bottom (Ury and Pow Burns).

The concentrations of TP in some rivers increased with increasing flow (Figure 11). However, linear correlation coefficients between the 2 parameters were low; the highest r^2 values of 0.56 and 0.57 were obtained for the Ury Burn and upper South Queich sites respectively; TP levels and discharge of the Pow Burn exhibited even less association.

5.2.3 Point-source effluents

Phosphorus concentrations in the point-source effluents entering the Loch Leven system, were extremely high, being more commonly expressed in mg l^{-1} (cf µg l^{-1} in run-off waters). As examples, the annual mean levels of TP in the woollen mill effluent, was 12.85 mg l^{-1} (from the 8-day routine sampling - see Figure 12 which also demonstrates day-of-the-week variation in P concentration) and 16.78 mg l^{-1} from the 24-hour study 2-3 December 1985; Milnathort STW effluent contains typically 5-10 mg TP.l^{-1}. As a consequence, P concentrations were considerably elevated in the waters receiving these wastes. This is even the case where the river is quite large, as for example, the North Queich, which carried the effluent from Milnathort STW (Figure 13a), and the South Queich, which took mill effluent (Figure 13b).

In addition to their being of high P content, the effluents were often especially rich in soluble P. In the case of treated sewage, this is manifest in the SRP results; Figure 14 illustrates the distinct frequency distributions of SRP concentrations, between reaches above and below the input of sewage effluent, and Figure 15a the increase in the % SRP/TP (shift of points to the lower left hand corner of the triangular plot of the lower reach data). Figure 15b illustrates the relative proportions of the different P fractions from the sewage works. It shows the typically high proportion of SRP from works with biological treatment and the more equal proportion of all 3 fractions from those without biological treatments. The occasional points outside the main groupings were obtained during high flows. The mill effluent is unusual, with its high % TSP/TP; the annual mean concentration from the 8-day sampling schedule is 8.69 mg l^{-1} (cf 12.85 mg TP l^{-1} above) and 12.62 mg l^{-1} (cf 16.78 mg TP l^{-1}). It is also unusual in that the major part of the TSP was the soluble organic fraction (SURP), and not the inorganic fraction (SRP) - see Figure 16. However, earlier analyses (Bailey-Watts 1983) indicated that much of this SURP is likely to be rapidly transformed to SRP in the receiving waters, and indeed levels of the inorganic fraction in the South Queich below the outfall were generally much higher than those above it (Figure 17).

One of the effects of the ingress of rich, point-sources of P to the natural stream systems, was on the relationship between nutrient concentration and flow. For example, upriver of the outfall of Milnathort STW effluent on the North Queich, TP concentration was positively correlated with flow (r^2 for log-log plot is 0.626), whereas downstream of the input the (commonly higher) concentrations of TP generally declined (r^2 = 0.444) with increasing flow (Figure 18). This may have been due to the higher % SRP/TP in STW effluent (cf. run-off water - Figure 15); Stevens and Smith (1978) found that log SRP concentration in the River Main, Co. Antrim, showed a statistically significant negative correlation with log flow (although r was only -0.73).

5.2.4 Rain water

Bailey-Watts (1983) considered that, in contrast to values quoted from widely different geographical sources, the earlier (DAFS) estimates of P entering Loch Leven in rain falling directly over its surface were remarkably high. An earlier estimate of the annual mean concentration of TP in rain collections (including dry deposition) was 355 µg l^{-1} - in contrast to preliminary analyses of freshly fallen rain, giving values of ca 25 µg l^{-1}. Yet, values (not volume-weighted) from the 1986 study are also high, ie 312 µg l^{-1} (Vane Farm site), 341 µg l^{-1} (Larch Cottage), 351 µg l^{-1} (Tarr Hill) and, at the site least affected by trees, 174 µg l^{-1} (Private Pier). One explanation for what are still considered misleadingly high values, relates to interference by midges (see section 3.2.3) and the siting of 3 of the collecting funnels too near trees. Figure 19 shows the high P concentration in mid-May to mid-June and in October corresponding to the occurrence of midges in the samples. If these values are excluded from the analyses, the mean concentration of TP and SRP from all collections, are 174 µg l^{-1} and 94 µg l^{-1} respectively.

Even these figures appear to be inordinately high. It is hoped that more work can be done on this aspect of the study: in the absence of expensive and sophisticated rain collection apparatus, analyses will be confined to recently fallen rain. (NB: Analyses of this type done in 1987 give typical concentrations of 25µg TP l^{-1} and 10 µg SRP l^{-1}).

5.2.5 Sediment pore water

Dissolved P concentrations in the interstitial waters (Figure 20 uses data for the 4-m sampling site) were often > 20 times those of the overlying water, even in the top centimetre slice. Greater concentrations were found at greater depths. The maximum concentration could occur at any of the sampled depths below the sediment-water interface, ie anywhere between 1 cm and 10 cm, or at 15 cm.

Maxima were commonly hundreds of micrograms of P per litre April to May but, in June especially, concentrations an order of magnitude greater than this were found. However, there was no clear relationship between the overall amounts of P in solution and the temperature of the overlying loch water; note (Figure 20) the contrasting P profiles of 19 May and 5 June when prevailing temperatures were the same, and the average concentrations of P over the uppermost 10 cm of deposits on 5 June (11.5°C) which were marginally greater than those on 30 June (20.5°C). This preliminary investigation also detected no consistently greater P concentrations in mud below 10 m of water.

5.2.6 The loch and its outflow

Phosphorus levels in the loch (and thus the water passing through the sluice gates) during part of 1986 have been indicated in the preceding section which refers to water above the sediment sampled at that time. These are generally low. So too are the concentrations that prevailed in 1985 (Figure 21). The TP levels varied little more than 4-fold (minimum 29 μg l^{-1}, maximum 134 μg l^{-1}) with a mean for the year of 63 μg l^{-1}; contrastingly, SRP varied much more (0.5 to 37.9 μg l^{-1}) around a mean of 9.2 μg l^{-1} - the range being largely accounted for by the shift associated with phytoplankton growth in late-winter to early-spring.

5.3 Phosphorus loadings (L) from individual sources

5.3.1 River systems (run-off loadings)

(a) General features

Seasonal fluctuations in TPL varied between the rivers (Figure 22), but there were broadly similar patterns of change in water courses of comparable size (discharge). Thus, the data for the two Queichs (Figure 22a) and the Gairney Water (Figure 22b) each show peaks in March, July, September and December. Similarly-timed pulses occurred in the Pow Burn (Figure 22b), although the summer peaks there represented considerably smaller loads than the winter ones. Two of the smaller streams - the Camel and Ury Burns (Figure 22c) - exhibited other types of seasonality, the former showing a very restricted range of loadings and the latter a series of values ranging over 3 orders of magnitude.

Whilst this report concentrates mainly on the annual loadings of P, a point on seasonality in loading is of interest here. Data also plotted in Figure 22(d-f) show the rates at which the loadings accumulated through the year. Contrasts in seasonality between streams, and the influence of episodes of increased loading are now evident.

In the case of the Gairney Water, 3 records (in August, September and December) accounted for ca 45% of the annual input and ca 60% of the annual supply from the Pow Burn entered the loch between 14 and 20 December 1985.

A generally close relationship between instantaneous loadings and flow was found (Figure 23), with some small streams (eg Ury Burn) behaving similarly to some larger rivers (eg North Queich); Table 7 gives statistics from linear correlation and regression analyses of these data.

(b) Run-off loadings estimated from instantaneous measurements of flow and P concentration

Annual loadings from the products of instantaneous flows and instantaneous concentrations of TP are shown in Table 8, with the calculated rates of loss of TP from the catchments. In general, the water courses draining the larger areas contributed the greater loads, but losses per unit area of catchment varied. This is especially noticeable in the case of the small channels on the north-east border of the loch; the high annual loss rates for 'B', 'D', and 'E' (1.67 kg TP ha^{-1}, 1.20 kg TP ha^{-1} and 2.12 kg TP ha^{-1}) reflect the influence of agricultural activity, and contrast with site 'F' losing only 0.39 kg TP ha^{-1}.

It is worth commenting on the approximate contributions of particulate and soluble fractions of P to the inputs. Although the mean values for the contributions of these fractions (Table 9) provide only a general impression of the situation in the different water courses, some contrasts are evident. The Pow Burn and 3 of the 4 small channels to the east of it were relatively the most particle-laden. The small channel ('B'), along with the Ury Burn and tributary Gt of the Gairney Water, was characterised by higher proportions of soluble P. In the Camel Burn and the 3 main feeder waters however (the North Queich at Nd and Nh, the South Queich at Sc and the Gairney Water at Gb), the mean proportions of particulate and dissolved material were approximately the same. Consequently, the TSP loading from the sites listed in Table 9 (together draining 84% of the total catchment) represented approximately one-half of their TP loading, ie 46%. Whilst it is likely that the majority of this soluble fraction was potentially utilisable by algae for growth, the measured soluble reactive (inorganic) P component, which was immediately available, represented only 32% of the TP loading.

There are 3 ungauged streams which, together with sectors of land lacking an identifiable water course, represent 18% of the total watershed. However, nearly one-half of this % relates to the Fochy Burn - a tributary of the North Queich - for which the annual TP loading has been estimated by subtracting from the loading measured at the mouth of the Queich, the sum of the loads associated with the Hatton Burn, the upper reaches of the North Queich itself and Milnathort STW. This, however, has resulted in a suspiciously high loss rate of 1.607 kg TP ha^{-1}. On the basis of P concentration data in the 3 ungauged waters, the proportions of PP, TSP and SRP were, respectively, 0.66, 0.34 and 0.25 (Ha), 0.38, 0.62 and 0.48 (Nf) and 0.31, 0.69 and 0.62 (Wa).

(c) Run-off loadings estimated from continuous flow data and derived figures for continuous P loadings (L_{cr})

Table 10 lists, for sites not affected by important point-sources of pollution and which together account for 78% of the total catchment, annual loadings (TPL_{cr} and $SRPL_{cr}$) calculated from 365 daily figures derived as described in section 4.4.1. As also outlined in that section, for sites which were not continuously gauged, continuous flow values had to be derived before the TPL_{cr} figures could be generated. The flow data on which the information for sites Gb, Nh, Pb and Ua in Table 10 are based, were obtained from the following equations of measured flows ($\log_{10}Q$) there, regressed on those at either Nd or Pc, for which continuous flows were also available:-

$$\log_{10}Q_{Gb} = 0.8661 \log_{10}Q_{Nd} + 0.2531 \quad (r^2 = 0.908)$$

$$\log_{10}Q_{Nh} = 1.2213 \log_{10}Q_{Nd} - 1.7904 \quad (r^2 = 0.967)$$

$$\log_{10}Q_{Pb} = 0.8121 \log_{10}Q_{Pc} + 0.6363 \quad (r^2 = 0.974)$$

$$\log_{10}Q_{Ua} = 0.8074 \log_{10}Q_{Nd} - 1.0025 \quad (r^2 = 0.802)$$

Continuous loading values corrected as proposed by Ferguson (1986) are also given in Table 10, along with instantaneous loading estimates for comparison.

Assuming that the 'continuous' data provide the better estimates of loadings, it appears that the information from the discrete samplings would over-estimate the inputs (cf views expressed in section 4.4.1). This is especially so in the cases of the Pow and Hatton Burns for TPL but only for the Hatton Burn is the over-estimation large for SRPL.

Future analyses of the data will provide estimates of the loadings for other components of TP ie PP and SURP (in addition to SRP) and compare the sums of these with the direct estimates of TPL. Studies of the Lough Neagh feeder waters suggest that prediction of TP concentration may be more reliably achieved by the addition of TSP and PP concentrations predicted separately.

5.3.2 Point-sources

(a) STWs (L_{sew})

From the products of 24-hourly mean flows and P concentrations in samples collected during the intensive studies of the STWs, the following 7-day loads were obtained. For Kinross North, 36.89 kg TP of which 78% consisted of SRP, 92% TSP, and 42.28 kg TP (88% SRP, 96% TSP). The Kinross South Works produced 24.39 kg TP (47% SRP, 81% TSP) and 29.02 kg TP (48% SRP, 82% TSP). The Milnathort effluent contained 28.22 kg TP (82% SRP, 87% TSP) and 26.38 kg TP (81% SRP, 87% TSP). The loads measured at the smallest Works - Kinnesswood - were 8.15 kg TP (67% SRP, 89% TSP) and 10.68 kg TP (74% SRP, 82% TSP).

Extrapolation of the mean values for these sampling runs, results in yearly TP loadings of 2064 kg for Kinross North, 1392 kg for Kinross South, 1376 kg for Milnathort and 491 kg for Kinnesswood. Daily TP loads *per capita* of population served by these works are 1.77 g for Kinross North, 1.82 g for Kinross South, 1.93 g for Milnathort and 2.42 g for Kinnesswood.

Diurnal variation in P loading was estimated for Milnathort (2-hourly composite samples) and Kinnesswood (hourly composite samples). Typical results are plotted in Figure 24. Points to note are the marked difference in the degree of variation which probably reflects the difference in quantity of surface and ground water reaching the works; the timing of the peaks - late morning and mid-evening for Kinnesswood and early morning and mid-evening from Milnathort - and the consistency in the ratio of SRP to TP.

As indicated in section 3.1.2, figures were obtained for the loading of P entering the Kinross North works. An estimate of P removal is thus afforded.

Percentage removal varied considerably on a daily basis. This was probably due to the residence time of sewage through the works with treated sewage discharged being derived from influent sewage received in the works several hours previously. Coupled with fluctuations in daily loading this could produce the variation observed. It was not possible to take account of the residence time in calculation of the P removal as it would vary considerably with flow. It seems likely that this effect would average out over the weekly monitoring runs and therefore not reduce the accuracy of the estimate of P removal to any great extent.

Percentage removal of TP was 39.6% (average of 2 weekly runs). This figure is higher than would be expected for a typical sewage works and is likely to be due both to the design of the works and the fact that it is at present hydraulically underloaded. As a consequence, it cannot be assumed that this efficiency would be sustained if the loading on the works were increased - as will be the case when sewage presently treated by the South works is transferred to the North works.

(b) Industry (L_{ind})

As already indicated, estimates of P supplied to the loch in woollen mill waste vary according to the available data sets. An intensive study of the effluent over 24 hours suggests a week-day loading of 35 kg TP. On the basis of spot chemical analyses and flow readings taken every 8 days in conjunction with the stream samples, a figure of > 100 kg TP wk^{-1} is calculated. Indeed, TPL_{ir-Sa} and $TSPL_{ir-Sa}$ values of 7956 and 5303 kg ann^{-1} respectively were obtained. The site Sa is on the South Queich *ca* 50m below the outfall of industrial effluent and represents the input from all of that river system. By subtracting from it, the estimated load passing site Sc (data in Tables 10 and 11), a contribution from the mill of 6287 kg TP ann^{-1}, including 4519 kg TSP ann^{-1} (or 126 kg TP and 90 kg TSP per week) is obtained.

5.3.3 Rainfall (L_{rain})

It is suggested that the results of the rainwater P analyses are misleadingly high, even excluding the figures on samples visibly affected by midges.

As a consequence, very high loading estimates are obtained ie 2893 kg TP ann^{-1} and 1563 kg SRP ann^{-1}. That these figures make little sense is evident from the following considerations. Expressed as losses of P per unit area of the atmosphere (in this case 1330 ha - the surface area of the loch) the loadings give the loss rates of:-
2893/1330 or 2.18 kg TP ha^{-1} ann^{-1}
and
1563/1330 or 1.18 kg SRP ha^{-1} ann^{-1}.

These values are higher than the majority of those describing P loss rates from land (cf Table 8). As these data are so plainly out of order, they have been excluded from further consideration. Instead, concentrations of 25 µg TP l^{-1} and 12 µg SRP l^{-1} - as typically found in samples of freshly-fallen rainwater - have been used. On the assumption that these levels are representative of annual rainfall, (1250 mm in 1985) revised loadings of 416 kg TP ann^{-1} and 200 kg SRP ann^{-1} are calculated.

5.3.4 Wildfowl (L_{wild})

The numbers of the 2 dominant goose species that roost on Loch Leven (Greylag and Pinkfooted) varied considerably in abundance. Monthly counts for two, 7-month periods (September 1984 to March 1985 and September 1985 to March 1986) ranged from ca 50 to 300 Greylag and 600 to 12700 Pinkfooted. The average monthly counts of Greylag for the 2 periods were also very different - 220 in 1984/85 and 1740 in 1985/86, but those of Pinkfooted were closer, at 6330 in the earlier period and 5760 in the later one. As, at present, seasonality of potential loading from wildfowl is not being assessed, the following calculations use the mean of each of the pairs of values just quoted, ie 980 Greylag and 6045 Pinkfooted; while these represent the average size of roosting populations for only 7 months of the year, the estimated P inputs from them are also annual figures.

According to Cooke (1976) the daily output in droppings per goose is 70 g of which 0.43% is P, ie 301 mg P. As he also assumes that the geese roost on the water for 16 hours per day, the daily output is ca 200 mg P. On this basis, a total of 7025 geese (980 + 6045) would introduce:

$$7025 \times 200 = 1.405 \times 10^{6} \text{ mg P.d}^{-1}$$
$$= 1.405 \text{ kg P.d}^{-1}$$

which is equivalent to 295.05 kg P in 7 months (and is also the total for the year).

In a more recent study, Hancock (1982) found that, on the Loch of Strathbeg in Aberdeenshire, these 2 types of goose differed in their daily export of P. The production also varied according to whether they had previously fed on barley or on grass. His estimates for geese feeding on barley were 332 mg P d^{-1} (Greylag) and 234 mg P d^{-1} (Pinkfooted) and for birds feeding on grass, 211 mg P d^{-1} (Greylag) and

172 mg P d^{-1} (Pinkfooted). Assuming that the geese associated with Loch Leven fed mainly on barley (to give what is likely to be a liberal estimate of their contribution to the P loading) the input for Greylag is:

$$980 \times 332 = 0.33 \times 10^{6} \text{ mg P.d}^{-1}$$
$$= 0.33 \text{ kg P.d}^{-1}$$

which is equivalent to 69.30 kg P in the over-wintering period (and the year) and for Pinkfooted:

$$6045 \times 234 = 1.41 \times 10^{6} \text{ mg P.d}^{-1}$$
$$= 1.41 \text{ kg P.d}^{-1}$$

or 297.05 kg P in the year.

The estimated total annual 'loading' of P in goose droppings is thus 366.35 kg.

5.3.5 Sediments

Information on pore-water chemistry provides a basis for a number of questions on internal P loading to be addressed. For example, how did the amounts of P in the sediments and in the overlying water column compare? For the 2 amounts to be the same - and assuming a column depth of 4.0m - the concentration in the interstitial water integrated over sediment depth (in cm) needed to be 400 (number of centimetres in 4m) ÷ 0.9 (the fractional volume of sediment occupied by pore water) ie 444 times the concentration in the column above. In a few instances, a single centimetre slice of sediment contained the requisite amount, eg the 2-3 cm layer on 29 May (with 540 x the loch concentration of 4.1 µg l^{-1}) and the 1-2 cm layer on 10 June (445 x 2.5 µg l^{-1}). Usually, however, a greater depth of mud had to be incorporated before its interstitial P equalled the water column P. Indeed, during much of the period up to the end of May, SRP in the column exceeded that in the uppermost 10 cm of sediment, ie the average concentration in each of ten 1-cm slices was < 44.4 times that in the overlying water. In June, however, a general increase in pore water SRP concentrations was recorded. As the absolute maximum concentration also appeared to move gradually nearer the surface (ref. Figure 20), amounts of P equivalent to those in the loch column were found in the top 4 to 5 cm of the deposits. Unfortunately, the sediment sampling could not be continued further into the summer: the developments recorded in June - suggesting that desorption of P from sediment particles took place - probably continued throughout the summer. As a consequence of these changes, gradients in P concentration were established. These comprise pre-conditions for diffusion of SRP - from zones of higher concentration (sediment) to the more dilute water above. In spite of these changes, which probably occur every summer, SRP concentrations in the loch were low in that season of 1985 (the main year for the P loading investigations).

5.4 Losses of phosphorus *via* the outflow (L_{out})

Mean 8-daily rates of export of TP and SRP in the outflow during 1985 were 260.0 $mg.s^{-1}$ and 45.9 $mg.s^{-1}$ respectively, ie 8199.4 kg and 1447.5 kg annually.

5.5 Phytoplankton

Figure 25 uses chlorophyll a concentration to illustrate fluctuations in total phytoplankton abundance during 1985. There were 3 similar peaks in biomass. Two of these comprised assemblages dominated by unicellular centric diatoms, although the species common in March differed from those prominent in December. The other chlorophyll a maximum - recorded in August - corresponded to a far more diverse assemblage; the most prominent within a very mixed array of types were *Asterionella formosa* Hass., (a colonial pennate diatom), *Closterium strigosum* Bréb., *Staurastrum planktonicum* Teiling (both desmids) and *Anabaena flos-aquae* Born. *et* Flah., (a filamentous 'blue-green alga' or cyanobacterium).

The triple-peak pattern of biomass changes is similar to that recorded in at least some previous years. One example is 1979 (Bailey-Watts 1982), but even in that case, the chlorophyll a concentrations were considerably higher than those found in 1985: the mean value in the earlier year was 40.7 µg l^{-1}, nearly double the figure of 21.1 µg l^{-1} calculated from the 1985 data. In both years, however, unicellular Centrales dominated the late winter-early spring plankton, as they have done in most years, although the component species differed - Bailey-Watts 1987). Moreover, in 1979, as in the present year, *Anabaena* was prominent in late summer; however, in 1979 *A. spiroides* was important, and, in addition to *A. flos-aquae*, was especially abundant in late autumn as well. So, in terms of even the dominant forms, there are detailed differences in phytoplankton periodicity between two, otherwise generally similar, years. This feature of inter-annual variability appears to be a characteristic of phytoplankton development in Loch Leven (Bailey-Watts 1978, 1982, 1986). A certain degree of year to year similarity is revealed, however, when the population changes are expressed in terms of the sizes of organism present. Thus, as in 1979 to 1982, and as resolved by Bailey-Watts (1986), small algae predominated in the early months and in the late months of 1979, whilst larger species became relatively more abundant in summer. Such a pattern appears to reflect, in part, the seasonally shifting intensity of grazing by zooplankton, which prefer small algal cells.

6 LOADING OF PHOSPHORUS FROM EXTERNAL SOURCES, LOSSES VIA THE OUTFLOW AND SOME THOUGHTS ON INTERNAL LOADING - AN EVALUATION OF THE DATA

This section considers separately, first for each external source of P, and second for the total input, the validity of the raw data on which they are based. As part of the exercise, it also examines the results in relation to the literature; only in this way can the assessment of nutrient inputs be improved and the understanding of eutrophication processes be increased. However, comparisons must be treated with caution, because studies usually concern watersheds differing in geology, topography, land-use and thus, hydrology. In addition, and partly as a result of this variability, investigative methods will differ, eg in the intensity in time and space of sampling and recording, and in whether the studies refer to 'natural' or experimental (eg lysimeter) systems.

6.1 Run-off (L_{ir} and L_{cr})

From Table 8 (section 5.3.1) the estimated total TPL_{ir} is 9393 kg ann^{-1}. The sum of L_{ir} values corresponding to sites for which L_{cr} values have also been estimated, is 6226 kg ann^{-1}, whereas the sum of those L_{cr} values is 5389 kg ann^{-1} (and not $> L_{ir}$ as might have been expected - see sections 4.4.1 and 5.3.1(c)). A proportional reduction of the 9393 figure (ie by multiplying it by 5389/6226) gives an annual total TPL_{cr} of 8130 kg ann^{-1}. A similar handling of the data for SRP loads gives a total $SRPL_{ir}$ of 3561 kg ann^{-1} (38% of TPL_{ir}) and a total $SRPL_{cr}$ of 3523 kg ann^{-1} (43% of TPL_{cr}).

As far as can be judged at the time of writing, the information on which these run-off estimates are based is sound. For example, the inflow and outflow figures make sense: 8-daily measurements of water entering the loch in run-off (together with precipitation minus evaporation over the loch surface) gave an annual (1985) input which is only 2.7% greater than the estimated output. What is more, the run-off values for different sectors of the watershed related strongly to their corresponding sub-catchment areas. Similarly, the results of the AQC precision and bias tests suggest that the determinations of P concentrations are satisfactory. Relationships between P levels and flows and between P loads and flows were found, for the cases of many water courses sampled, to be very robust; indeed, they facilitated the derivation of 'continuous' flows and loadings for stream systems representing 77% of the Loch Leven watershed.

The current estimate (of TPL_{cr}) to Loch Leven is ca 3 times the figure of 2784 kg TP calculated by Bailey-Watts (1983), based on the following annual rates of loss of P from land: 0.25 kg TP ha^{-1} arable land (taken as representing ca 65% of the catchment) and 0.07 kg TP ha^{-1} non-arable land (35%) as found by Cooke and Williams (1973, see also Cooke 1976) in their studies in S. England. A value of 0.25 kg P ha^{-1} was obtained by Kolenbrander (1972) for land under arable crops in the Netherlands, and by Burton and Hook (1979) for heterogenous sandy soils in high rainfall areas near the Great Lakes. Much higher (annual) values have been estimated elsewhere, however, and these are comparable to measured values ranging between 0.42 and 1.10 kg TP ha^{-1} of major parts of the Loch Leven watershed; eg 0.41 kg TP ha^{-1} for a land drain in managed grassland, Antrim, Northern Ireland (Stevens & Stewart 1981; Jordan & Smith 1985), 0.83 - 1.05 kg TP ha^{-1} from the peaty catchment of Lough Leane (Casey et al 1981) and those generally within the range 0.1 to 3.0 kg TP ha^{-1} from agriculturally affected sandy loams in New Jersey (Ryden et al 1973).

Many authors (eg Sharpley & Syers 1979; Cooke 1976) comment on the importance of PP, eg soil particles in the P transported into running water and lake basins. In most of the water entering Loch Leven, the concentrations of the particulate fraction represented 50% of TP. As already discussed, it is the remaining soluble component that is of more significance - especially SRP. Losses of this portion in run-off (as shown by data in Tables 8 and 9) are more closely comparable to those quoted elsewhere. For example, estimates of annual losses from the major sub-catchments around Loch Leven are (as kg SRP ha^{-1}): 0.16 (South Queich at Sc), 0.15 (North Queich at Nd) and 0.12 (Gairney Water at Gb). These match closely the figures of 0.17 kg SRP ha^{-1} reported by Stevens and Stewart (1981) in the study quoted above, and 0.14 kg SRP ha^{-1} obtained from the y intercept in a P export/population multiple regression analysis of Lough Neagh data by Foy et al. (1982); they consider a value of 0.22 kg SRP ha^{-1} more applicable, however. Steenvoorden and Oosterom (1979) quote losses of 0.25 kg PO_4-P ha^{-1} from Netherlands cropland. The present study also produced a similar figure of 0.28 kg SRP ha^{-1} in the Pow Burn drainage area which appears to be the richest of the 4 'major' sub-catchments.

Nevertheless, higher loss rates are indicated for other, generally smaller, areas of the Loch Leven catchment. Examples reflecting localised intensive farming activity or septic tank discharges are 0.44, 0.66, 0.92 (kg SRP ha^{-1} ann^{-1}) for drainage sites 'Da', 'Ea' and 'Ba' respectively, and 0.54 for the Ury Burn (Ua). Jordan and Smith (1985) attribute the high losses of 0.45 and 0.70 kg SRP ha^{-1} of Lough Neagh sub-catchments (cf 0.17 of Stevens and Stewart (1981) quoted above) to floods in 1981 and 1982 respectively. Stream-borne inputs of SRP to Loch Leven are generally commensurate with the moderate levels found elsewhere. Detailed information on agricultural statistics for the watershed remains to be analysed. Preliminary findings based on figures alloted to only 4 sub-divisions of the area, the 2 Queichs, the Gairney Water and the Pow Burn, suggest that the Pow is the most heavily fertilised; it is also the area with the greatest % of land under arable crops and grass. Nevertheless, the estimated loss of TP over the whole area represents only 5% of the approximately 21 kg P_2O_5-P ha^{-1} applied to crops and grass per year. In this respect, many of our data support the often-expressed view that P losses from agricultural land are "trivial" or "minor" (Cooke & Williams 1979; Armitage 1974; Holden 1976; Hay 1981; Bailey-Watts 1983).

There is, however, one water course for which a suspiciously high TP loading, and thus loss rate, of 1.10 kg TP ha^{-1} ann^{-1}, was obtained. This is the Fochy Burn which, though draining 955 ha, and constituting the largest of the tributaries of the North Queich, was not flow gauged. As a consequence, the loading figures were derived from differences in measured loadings on the North Queich above and below the point at which it receives the Fochy Burn. This matter will be referred to again in connection with the inputs from STWs, because the result depended in part on the estimated loading obtained for the Milnathort works.

6.2 STWs (L_{sew})

Whilst more series of intensive sampling runs than the number actually achieved would have been ideal, the similarity in the results for L_{sew} from separate runs on individual works is encouraging. The sum of the estimates for all works gives annual inputs of 5323 kg TP of which 68% (3847 kg) is

SRP. Kinross North works supplied *ca* 39% of TPL_{sew}, and 45% of the $SRPL_{sew}$, whilst the Kinross South plant contributed 26% of the TP load and 17% of the SRP load. Milnathort works, like Kinross South, supplied 26% of the TP load but a proportionally much greater contribution of the SRP load - 29% - than Kinross South. Kinnesswood contributed approximately 9% of both TPL_{sew} and $SRPL_{sew}$.

The loads, expressed as a function of the size of populations contributing to them, are feasible (see eg Alexander & Stevens 1976): *per capita* figures - in $g.person^{-1} d^{-1}$ - are 1.77 for Kinross North, 1.82 for Kinross South, 1.93 for Milnathort and 2.42 for Kinnesswood. Indeed, a considerably higher value of 3.26 g $TP.person^{-1} d^{-1}$ was predicted for Milnathort works before the actual results were available. This estimate - L_{sew-M} - was obtained from measured TP loadings (TPL_{ir}) at sites Na, Nd, Nh and Nf on the North Queich system according to:

$$TPL_{sew-M} = TPL_{ir-Na} - (TPL_{ir-Nd} + TPL_{ir-Nh} + TPL_{ir-Nf})$$

where, in kg ann^{-1}, TPL_{ir-Na} is 4793, TPL_{ir-Nd} is 1395, and TPL_{ir-Nh} is 571; assuming a TP loss rate from the catchment (955 ha) drained by Nf (the Fochy Burn), equivalent to the average calculated for Nd (0.532 kg ha^{-1} ann^{-1}) and Nh (0.788 kg ha^{-1} ann^{-1}) ie 0.660 kg ha^{-1} ann^{-1}, TPL_{ir-Nf} is 508 kg TP ann^{-1}. So:

$$TPL_{sew-M} = 4793 - (1395 + 571 + 508) = 2319 \text{ kg ann}^{-1}$$

This output represents an average daily *per capita* contribution (from 1950 persons) of 3.26 g. For the present, the measured value of 1.93 g $person^{-1}$ d^{-1} is taken in favour of this, high, predicted figure. As a consequence, in the absence of measurements of P loadings in the Fochy Burn, a correspondingly high loss rate of ≥ 1 kg TP ha^{-1} ann^{-1} has to be accepted (see section 6.1). (NB: note added in press: the Fochy Burn receives effluent from the small treatment plant serving the Ochill Hills Hospital - Mr I Fozzard, pers. comm.).

6.3 Industry (L_{ind})

Until a sounder understanding of the schedules of processes affecting P losses from the mill is achieved, the estimates of 6287 kg TP ann^{-1} and 4519 kg TSP ann^{-1} have to be taken. In this connection, further investigations, by way of short-term intensive sampling over long periods must be considered. In the 24-hour study briefly referred to in section 5.3.2(b), the concentrations of TP and SRP in the effluent at 1800h on day 1 were, respectively, 37 times and 10 times greater than the concentrations at 1800h on day 2.

6.4 Rainfall (L_{rain})

Annual inputs of P in rain were likely to be high in 1985 because it was a generally wet year (see section 7). Reasons for disgarding the very high concentration estimates obtained in 1986 have been argued in previous sections. As a result, annual loading values of 416 kg TP and 200 kg SRP are used in subsequent sections of this report (see especially 6.6 and 7).

6.5 Wildfowl (L_{wild})

A liberal estimate of 366 kg TP was obtained for the annual input from the 2 most numerous types of goose.

6.6 The total input from outside the loch (L_{tot})

From the values obtained for each of the external sources of P reviewed above, L_{tot} is calculated as follows:

(i) for TPL_{tot}:-

$$TPL_{tot} = TPL_{cr} + TPL_{sew} + TPL_{ind} + TPL_{rain} + TPL_{wild}$$

that is:

$$\begin{aligned} TPL_{tot} &= 8130 + 5323 + 6287 + 416 + 366 \\ &= 20522 \text{ kg ann}^{-1} \\ &= 20.5 \text{ tonnes ann}^{-1} \end{aligned}$$

This represents an average weekly loading of 394.7 kg TP, and a specific areal loading of 1.54 g TP m^{-2} ann^{-1}.

(ii) for $SRPL_{tot}$:-

$$SRPL_{tot} = SRPL_{cr} + SRPL_{sew} + SRPL_{ind} + SRPL_{rain} + SRPL_{wild}$$

which, assuming that $SRPL_{ind}$ is the same as $TSPL_{ind}$, and $SRPL_{wild}$ is one-half of TPL_{wild}:-

$$\begin{aligned} SRPL_{tot} &= 3523 + 3847 + 4519 + 200 + 183 \\ &= 12272 \text{ kg ann}^{-1} \\ &= 12.3 \text{ tonnes ann}^{-1} \end{aligned}$$

This is equivalent to an average weekly loading of 236.0 kg SRP and a specific areal loading of 0.92 g SRP m^{-2} ann^{-1} (60% of the total P loading). Results on the various inputs of P are included in Figure 26.

6.7 Losses of phosphorus via the outflow and some thoughts on the influence of the sediments on the phosphorus budget

The estimated rates at which P is supplied to the loch (summarised in the previous section) are compared, in the form of a bar chart in Figure 26, to the losses of P via the outflow. From this diagram, it appears that there was a large annual net retention of P by the loch. The amount of TP passing down the outflow (L_{out}) was well below one-half of that entering the loch (L_{tot}) ie

$$TPL_{out} = 0.40\ TPL_{tot}$$

where TPL_{out} is 8199 kg (section 5.4) and TPL_{tot} is 20522 kg.

Contrastingly, the figure for $SRPL_{out}$ is a little more than one-tenth of $SRPL_{tot}$ ie

$$SRPL_{out} = 0.12\ SRPL_{tot}$$

where $SRPL_{out}$ is 1448 kg and $SRPL_{tot}$ is 12272 kg.

An alternative interpretation is based on the assumption that, as much of the PP passing out of the loch is in the form of phytoplankton, it represents an equivalent amount of P originally in the form of SRP coming into the loch. So, if $SRPL_{out}$ equals the measured TPL_{out}, the % $SRPL_{out}/SRPL_{tot}$ is increased to 67%, reducing the apparent retention of SRP by the loch from 88% to 33%.

These considerations allow the importance of sediments - an internal source of P to the loch - to be put into some perspective. The factor central to this exercise is the ratio of the mean in-lake concentration of, in this case, P ($[P]_l$) to the mean inflow concentration ($[P]_i$). OECD (1982) outlines reasons for this ratio being of considerable a priori value for assessing whether an internal loading is an important source of P. Under normal conditions, the ratio in question for a non-conservative substance is <1. The ratio can be >1 when internal loading is important, unless external loadings have been considerably underestimated and/or in-lake concentrations considerably overestimated.

From the data now available for the Loch Leven system, a low $[TP]_l:[TP]_i$ ratio of 63:153 or 0.41, suggests that sediment release is of relatively minor importance over the year as a whole; this has been calculated as follows: $[TP]_l$ from TPL_{out} divided by the total volume of water passing out of the loch ie 8199 kg TP/130.8 x 10^6 m^3 or 63 $\mu g\ l^{-1}$; $[TP]_i$ from TPL_{tot} (ie including fractions of TPL_{sew} that enter the loch directly) divided by the total volume of water entering the loch ie 20522 kg TP/134.4 x 10^6 m^3 or 153 $\mu g\ l^{-1}$. The manner in which outflow losses contribute to the P budget is also summarised in Figure 26.

There is no evidence yet to rule out the possibility that considerable amounts of P were released from the sediments; if correspondingly high fluxes of P back into the deposits occurred, no net gain of P by the water column would be detected. Long-term observations on physical and chemical factors at Loch Leven, however, suggest that release of the nutrient would not be important in 1985. Firstly, for rapid releases, high temperatures and low nitrate concentrations and calm conditions must prevail (Bailey-Watts 1986, Bailey-Watts, Wise & Kirika 1987). In 1985, the maximum temperature was only 15.9°C and on this and only one other occasion - in July - were temperatures of $\geq$15.0°C recorded. Secondly, in some hot summers, massive and rapid increases in SRP in the loch are recorded, but the nutrient subsequently decreases, mainly due to re-adsorption by the sediments. For example, no remarkably dense populations of phytoplankton were observed in 1980, following an internal loading of ca 2.4 tonnes SRP within a 2-week period in August.

If control of external inputs of P to Loch Leven is considered appropriate, and the supplies are reduced, subsequent internal loadings would naturally consititute a relatively greater factor in the P budget. Studies elsewhere, however, suggest that while waters differ in the time of response to reductions in external nutrient loadings, internal releases eventually subside (see eg Moss et al 1986, Welch et al 1986).

7 INTERPRETATION OF THE RESULTS: THE RELATIVE IMPORTANCE OF THE DIFFERENT SOURCES OF P, NATURAL VARIATION AND MANAGEMENT POSSIBILITIES

The purpose of this concluding section is to examine, firstly, the various inputs of P as percentages of the total. This will identify the major sources of the nutrient. Secondly, the relevance of the 1985 findings to the future is assessed. In this connection, relationships between P loading, and P concentrations in the loch in 1985 are compared with those predicted by eutrophication models selected from the literature. As the relationships between these variables are controlled to a considerable extent by flushing rate, historical data on this factor are used to illustrate how it is likely to vary from year to year, and thus modify the results of controlling the P loading, should this be considered appropriate. Indeed, without this interpretative exercise, the relative merits of alternative control measures and their effect on P and chlorophyll levels in the loch could not be established. Measures include that relating to a consent set in April 1987 by the FRPB on P loading from the mill.

7.1 The relative contributions of the different P sources to the total loading

Results on the various sources of P entering the loch are summarised in the form of a pie-chart in Figure 27. This shows the % contributions from the various diffuse and point-sources to the total loadings of both TP and SRP. Features of immediate relevance to the discussions that follow are (i) as a consequence of the wet 1985, the high contributions in run-off - <u>ca</u>. 42% of TP and 30% of SRP, and (ii), in spite of the high rain-related loadings, the major contribution in point-source effluents, ie a total of <u>ca</u>. 57% to TP and 68% to SRP.

7.2 Loading and in-lake P concentration - relationships in 1985 compared with predictions from eutrophication models

General ecological models of the type developed so far in the field of eutrophication must be interpreted with care, when viewed in relation to the functioning of a particular lake. Nevertheless, they provide a useful framework for examining relationships between different factors in the model equations; 2 models have been used for the purposes of handling the Loch Leven data, each attempting to predict the TP concentration in lakes ($[TP]_l$) from loadings. One of these was developed by Dillon and Rigler (1974) although it draws on concepts introduced earlier by Vollenweider (1969). Their equation is as follows:

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

where L is loading (1540 mg m^{-2} in 1985 at Loch Leven), R is the phosphorus retention coefficient or fraction of L retained by the loch (0.60), $\bar{z}$ is the mean depth (3.9 m) and p is the flushing rate or the annual volume of water passing through the loch expressed as number of loch volumes (2.57 if the inflow volume estimate of 134.4 x 10^6 m^3 is used, 2.50 if the estimated outflow discharge is used). This results in predicted $[TP]_l$ values of 61.5 and 63.3 mg m^{-3} (= µg l^{-1}) which are extremely close to the observed value of 62.7 µg l^{-1}. A similar value of 64.5 mg m^{-3} results if no estimate of the losses of P down the outflow is made, and R is calculated according to Kirchner and Dillon (1975); ie in terms of an areal water loading - q_s - which is the volume of water entering the lake divided by its surface area, ie 134.4 x $10^6 m^3$/13.3 x $10^6 m^2$ or 10.11 m ann^{-1} for Loch Leven in 1985. The value R is then given by:

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

and in this case, 0.58 which is close to the measured value of 0.60.

In a number of respects, therefore, the present data on loading and in-lake P fit this model. Strictly speaking, however, the Dillon and Rigler equation predicts lake P at spring overturn - ie within a few days of ice melt on the Canadian lakes sampled in formulating the model. Plainly, in the highly variable climate of Northern Britain, changes in temperature and stratification of the shallow, broad expanse of water in Loch Leven are not as regularly timed as those of lakes on the large continents. Indeed, in many winters, Loch Leven does not freeze over and so the phenomena of 'ice melt' and 'spring-overturn' do not constitute the cardinal points that control many annual events in other types of lake.

In spite of these reservations, the extremely good fit of the current data to the postulates of the model suggests that it could be instructive to use it in a predictive sense - see section 7.4. The same conclusion is drawn, on fitting the present data to models developed by OECD (1982). These take the general form:

$$[\overline{TP}]_l = a[[\overline{TP}]_i/(1 + \sqrt{T(w)}\ )]^b$$

where a and b are constants, T(w) is the water residence time in years - the reciprocal of p as defined above - and $[TP]_l$ and $[TP]_i$ are as defined in section 6.7. For example, where a is 1.55 and b is 0.82, as in the equation combining all data contributing to the OECD study, the model predicts a $[TP]_l$ value of 64.4 µg l^{-1}.

7.3 The relevance of 1985 findings to the prediction of Loch Leven P balances in the future - the influence of flushing rate

The highly variable climate controlling Loch Leven has already been mentioned. It results in considerable differences from year to year in the rates of environmental processes and in the succession and performance of the organisms. As a consequence, the findings of a one-year ecological study, such as this P loading investigation, are not likely to represent very closely situations obtaining in other years. One of the most highly variable factors of interest, in assessing the wider relevance of the 1985 results, is rainfall - through its effect on flushing rate, which, as p or its reciprocal, T(w), has been shown to feature prominently in loading models.

Apart from the obvious influence on the amounts of nutrients running off the land, flushing controls the time during which water entering the loch remains there (before passing down the outflow): it thus determines, for how long algae, and other biological manifestations of nutrient loadings, have the opportunity to utilise the nutrients.

In order to assess long-term variation in p or T(w), rainfall records would normally be used. Problems arise in doing this, as precipitation records corresponding to the region of interest may not describe adequately the rainfall regime of a particular lake watershed. Furthermore, blanket corrections for evaporation must be made before a run-off estimate is obtained. The authors are particularly fortunate, therefore, to be able to draw on a 150-year record of water discharge from Loch Leven. Dr D Ledger (see also Ledger & Sargent 1984) has kindly made these data available, and, for present purposes, information dating from as far back as 1937 has been consulted. From the original run-off figures, annual values (r, in mm) have been used to calculate flushing rate (p, as defined above) as follows:

$$p = \frac{r}{1000} \times \frac{A_c}{V}$$

where A_c is the combined area of the loch and its catchment ($159 \times 10^6 m^2$) and V is the loch volume ($52.4 \times 10^6 m^3$). From this equation p is 1 when r is 330 mm. For 1985, Ledger gives an r value of 802 mm (p = 2.43); this compares very favourably with values of 823 mm and 845 mm, corresponding respectively to p estimates of 2.50 and 2.57 discussed in section 7.2. However, as p is part of the denominator in the Dillon and Rigler equation, a lower estimate of it results in a higher estimate of $[\overline{TP}]_l$ (see below).

Figure 28 shows that values for p over the last 50 years have varied from 0.97 to 2.76 ($\overline{x}$ = 1.89). Whilst it is not the purpose here to discuss trends in annual run-off, the high position in the ranking of the 1985 datum, as recorded by Ledger, is of note and emphasises how extraordinarily wet that year was. Of greater relevance to the present exercise is that the values as a whole describe a frequency distribution approximating to normal. Assuming that, in the long-term, values will be similarly distributed, the probabilities of occurrence of different annual flushing rates can be readily estimated. These probabilities are indicated as % chances in Figure 28. As a reference point, it would appear that a year wet or wetter than 1985, has a 5 or 6% chance of occurrence.

Armed with this knowledge on likely variation in flushing rate, and using the P loading-P concentration relationships, which for 1985 correspond closely to those predicted by the models, a spectrum of predicted P loadings, q_s values (and thus R values) and in-lake P concentrations can be generated. Before presenting (in section 7.4) the results of this exercise, it is worth considering briefly the influence of flushing rate (p) on retention coefficient (R) and the combined effect of altering these factors on P loading. For present purposes it is assumed that rain-related P inputs, ie those introduced in run-off and in rain falling directly on the loch surface, vary with rainfall (and thus flushing rate) pro rata. The assumption is supported by a preliminary analysis of the relationship between monthly loadings and monthly mean flows calculated from the derived daily loading values and continuously measured daily flows at station Nd on the North Queich (Figure 29).

The nature of the relationship between R and p is illustrated in Figure 30. As rain-related P inputs are taken to vary pro rata with p (see above), predicted loadings of TP vary linearly with p. Figure 31 shows how the loading to Loch Leven is likely to vary, purely as a result of natural shifts in annual flushing rate, ie with no alteration in point-source inputs. Indeed, if the line described by the data plotted in Figure 31 is extended to a point at which p is 0 - representing a completely dry year - it would intercept the vertical axis at a value of 11776 kg TP ann^{-1} which is the total of the estimated inputs of P from industry (6287 kg), sewage (5323 kg) and wildfowl (366 kg). Incorporating the same data into the Dillon and Rigler equation, Figure 32 shows that the form of the relationship between predicted P concentrations in the loch, and p, is similar to that between R and p. Note that predicted [TP] corresponding to a p value of 2.57 (as measured) is 71 µg l^{-1}. Reasons for this being higher than the measured concentration (63 µg l^{-1}) have already been given.

7.4 Some options for the control of P inputs to Loch Leven: recommendations based on model predictions of the effects on loading and in-lake P and chlorophyll concentrations

Using the models referred to above, this section examines firstly, the effects on loading, of reducing the P content of point-source effluents to various degrees. That these effluents comprise <u>ca</u> two-thirds of the total loading (see section 7.1) is reason enough to consider controlling them; various policy issues relating to the feasibility of doing this at Loch Leven will be highlighted. The effects of different loading controls on in-loch P and chlorophyll concentrations will be assessed. Finally, the most appropriate courses of action will be identified and a broad schedule for a P reduction programme will be recommended.

There are numerous means by which the external supply of P to Loch Leven could be reduced. The effects of 6 chosen control options are considered here; in each case, the intention would be to prevent the stated amounts of P reaching the loch and these are labelled as follows in the corresponding Figures - in addition to 'A' for the current situation:-

B 80% of the P in sewage effluent from Kinross North and Kinross South works
C 50% of the P in mill effluent
D 80% of the P in mill effluent
E Both B and C above
F Both B and D above
G All P in effluent from all STWs plus 100% of P in mill effluent

From what has been discussed previously, the focus on controlling P in point-source discharges makes good sense. There are special reasons too for the emphasis on effluents from industry and the Kinross STWs.

The first reason concerns action already taken by the FRPB in setting a consent on P in mill effluent. As a result, from 1 March 1987 concentrations and loadings should be reduced so that a mean weekly loading of 25 kg P would be achieved. This represents an 80% reduction of the current input and corresponds to control option D above. It is thought that this situation, and the arbitrarily chosen value of 50% reduction - control option C - are relatively easily attainable, if due attention is paid to preventing spillage of P-rich powders and liquids; moreover, by 1990, the intention is that mill effluent will be routed through the Kinross North STWs.

The second reason for highlighting the importance of the Kinross STWs is that the North works, completed in 1980, have been designed with the spare capacity to accept the mill effluent referred to above, and the sewage currently received by the South works. The latter comprise an old septic tank which is grossly overloaded hydraulically. Thus, by the end of the decade, the majority of the sewage produced in the Loch Leven catchment (as far as present population numbers indicate) would be handled by a single, modern works. Then, if it were considered appropriate (see section 8), additional P-stripping facilities could be incorporated. Removal of 80% of P in the effluent corresponds to control situation B above. However, in view of the schedule for closure of the Kinross South septic tank, it is unlikely that any extension to the North works in regard to P removal, could be envisaged before 1990. It is thus more sensible to view option B (sewage control) in combination with option D (mill control) ie the situation referred to as F in Figures 33-35.

The predicted outcome of each of the reduction situations on annual loading of TP and of SRP is plotted in Figure 33; this includes, for comparison, a re-plot of the data in Figure 31 corresponding to the range of loading possibilities, assuming no future change in supplies that are independent of rainfall and river flows, ie point-source and wildfowl inputs. In this graph, the predicted loading values, following removal of all sewage and mill effluent (line G, achieved by eg diversion around the loch), represent P sources that cannot be readily controlled, ie in run-off, in rain falling directly on the loch surface and in wildfowl droppings. The marked, predicted, effect of controlling the mill effluent alone (line D) is of note vis à vis considerations about logistics and costs of P removal. It should be borne in mind too, that the chances of flushing rates as high as that recorded in 1985 occurring in the future, are limited; as a result, loadings of around 13 tonnes TP ann^{-1} corresponding to average flushing rates might be expected, compared with 21 tonnes in 1985. By the same token, SRP loadings of about 7 tonnes would be the expected norm compared with 12 tonnes in 1985.

As already explained in connection with the derivation of values plotted in Figure 32, predicted concentrations of P in the loch exhibit a more complex relationship with flushing rate; the line described by these values is included in Figure 34 along with the curves corresponding to the predictions of both the Dillon and Rigler and the OECD models discussed above. In essence the 2 sets of results are not markedly different, each showing the change in the form of the curve as the amount of P removed - and thus the ratio of diffuse inputs to total inputs - increases. The diagram also highlights the ameliorating effect of high flushing rates (such as that observed in 1985) on P concentrations in the loch, while point-source inputs are large. Again, as a consequence of the probability distribution of annual flushing rates, there is a far greater chance of the loch exhibiting higher mean levels of P than measured in 1985, than lower concentrations than this (even if no control on loading is exercised). It is also interesting to note the suggestion that the input of P could be reduced to a point after which only minor annual differences in P concentration would occur regardless of rainfall or flushing rate. The Dillon and Rigler model predicts that this would require the removal of 80% of P in both mill, and combined Kinross North and South sewage effluents (line F).

A similar pattern of results to those shown in Figure 34 is derived in Figure 35 for chlorophyll concentrations predicted by another OECD (1982) model to which the 1985 value fits very closely. As in the case of the OECD P concentration model, this equation predicts annual mean pigment concentrations $[\overline{chl}]$ from flushing-corrected TP loadings, ie

$$[\overline{chl}_a] = 0.43 \left[[\overline{TP}]_{in} \Big/ (1 + \sqrt{T(w)}) \right]^{0.88}$$

Substituting the 1985 values of 153 μg l^{-1} for $[\overline{TP}]_{in}$ and 0.3898 y for T(w), $[\overline{chl}]$ is 23.47 μg l^{-1} compared to 21.02 μg l^{-1} measured.

The advantage of considering predicted chlorophyll levels over P concentrations is that the pigment indicates biomass of algae - and these organisms rather than P per se cause the problems associated with eutrophication.

As with the predicted P concentrations (Figure 34) the situations described by lines D and F in Figure 35 are of major note. Not only does the 'cost-effectiveness' of option D make it an attractive proposition; it could reduce pigment levels to commonly around 20 μg l^{-1} - a concentration still characteristic of eutrophic waters in general (OECD 1982) but not typical of rich lakes as shallow as Loch Leven. Observations on the relationship between chlorophyll concentrations and water clarity at Loch Leven (Figure 36) are of further relevance here. In reducing mean annual concentrations to ca 20 μg l^{-1}, elimination of 80% of industrially-derived P takes pigment levels into the range where the rate of increase in water clarity per unit decrease in chlorophyll itself increases. These arguments lend considerable support for the action taken by FRPB in setting its consent - to achieve the situation predicted by option D.

The second possible option relates to line F in Figures 33-35 ie, removal of 80% of the P in a combined Kinross North and South STWs effluent, as well as 80% of the industrial P. According to the plots in Figures 34 and 35, this procedure would lead to P and chlorophyll levels effecting an even greater increase in water clarity (Figure 36). This second option should also be considered carefully, particularly if further housing development in the Loch Leven catchment is considered and loads to the sewage works increase. In view of the impending developments relating to the control of industrial effluent (option D), its effects can, and should, be assessed long before action for control options B, and thus F, could be contemplated.

8 CONCLUSIONS

The aim of a P control strategy is to increase water clarity and macrophyte and invertebrate diversity, developments desired by industrial users of L. Leven water, by tourists, conservationists and anglers. To this end, the prevention of entry of P in mill effluent to the loch should be given serious, and immediate, consideration. Following FRPB action in setting its consent, it seems likely that an 80% reduction in P loading from this source will soon be achieved. The time the loch would take to respond to the change, and reach within 5% of the predicted P (or chlorophyll) level - the 95% response time - approximates to 3T(w); thus ca 1.6 y for the long-term average value of 0.53 y (1/p or 1/1.88). OECD (1982) include a correction using R, the P retention coefficient discussed above, so that the 95% response time becomes 3T(w)x(1-R); where R is 0.6, this product becomes 0.64 y. A significant lowering of P and algal concentrations should thus be detected, after a period of between 8 and 19 months from the time the reduction comes into force (1 May 1987). It is absolutely essential that the situation is monitored. Only in this way can (i) the effects of this single control measure be assessed - by 1990, additional effects of the closure of Kinross South STWs would have to be taken into account, (ii) comparisons of observed and predicted changes in P loading, P and chlorophyll concentrations and macrophyte distribution be made, and (iii) our scientific understanding of the eutrophication process and ways to manage it be improved.

While the present study suggests that control of mill P would significantly reduce the level of the nutrient in the loch, concentrations would still be characteristic of eutrophic waters (cf. OECD 1982), though not of rich, shallow lakes. Further improvement would require greater control of P loading - and by 1990, P-stripping at Kinross North STWs would be logistically feasible. By then, however, the monitoring programme should determine whether this is necessary.

In reflecting the general current state of the art on nutrient load modelling, the present study gives all too scant consideration to the undoubtedly important influence of seasonal changes in loadings, concentrations and flushing rate - and, therefore, algal abundance and species composition. For instance, had the rainfall in 1985 been distributed over the year, in the manner in which it was in the more or less equally wet 1984 (Figure 37 compares the 2 years and shows the enormous contrast in summer rainfall), the blue-green algal species recorded may well have attained much higher population densities.

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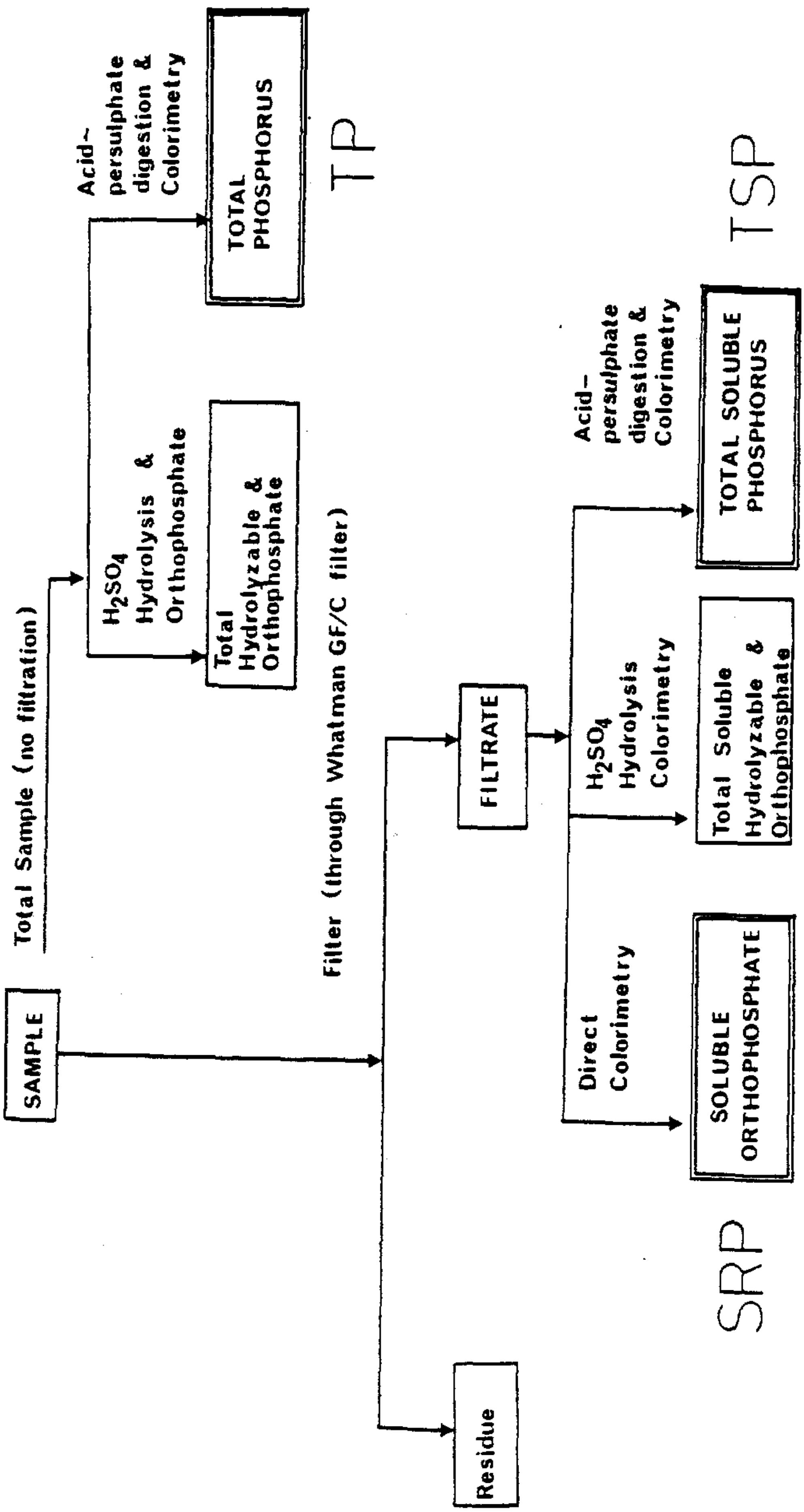
APPENDIX 1



ANALYTICAL SCHEME FOR DIFFERENTIATION OF PHOSPHORUS FORMS

APPENDIX 2

GLOSSARY

Names of sampling sites:

Two-letter codes are used, with one exception - 'L' for the loch near its outflow. The first letter, in upper case, refers to the stream or stream system, while the second letter, in lower case, indicates the position relative to others (where there is more than one station), with 'a' being nearest the loch, and 'b', 'c' and so on being furthest upstream. Examples are Na, Gc. For many streams the capital letter is the initial of the name:

C - Camel Burn
G - Gairney Water
K - Kinnesswood Burn
N - North Queich
P - Pow Burn
S - South Queich
U - Ury Burn

Exceptions are B, D, E and F for small drainage channels on the North-East shore, and H and W for 2 burns entering the loch a few hundred metres north of the South and North Queichs respectively.

Phosphorus fractions:

P - phosphorus
TP - total phosphorus
TSP - total soluble fraction
PP - particulate fraction
SRP - soluble reactive fraction
SURP - soluble un-reactive fraction

Phosphorus concentrations:

[TP], [SRP] etc. - total, soluble reactive phosphorus concentrations

$[TP]_l$ - concentration in the loch

$[TP]_{in}$ - concentration in the inflows

A bar over 'TP' etc. eg $[\overline{TP}]_l$ denotes mean concentration

Concentrations are expressed in $\mu g\ l^{-1}$ or $mg\ l^{-1}$ eg μg SRP l^{-1}, mg TP l

Phosphorus loadings:

TPL, SRPL etc. - loadings of TP, SRP etc.

Where appropriate, subscripts are used to indicate the source of the loading:

ir - from run-off - values derived from instantaneous flow data
cr - from run-off - values derived from continuous flow data
ind - from industry
sew - from sewage treatment works

rain - from rain falling directly on the loch surface

In addition, subscripts 'tot' and 'out' refer, respectively, to the total loading and the loading exported from the loch <u>via</u> its outflow.

Examples: TPL_{cr}, $SRPL_{sew}$, PPL_{out}

As loadings are rates of entry or export of P, units contain a time component. Thus $\mu g\ s^{-1}$, $mg\ s^{-1}$, $g\ s^{-1}$, $kg\ wk^{-1}$, $kg\ ann^{-1}$, $tonnes\ ann^{-1}$, and, where it is necessary to express loadings per unit area of loch surface, $mg\ m^{-2}\ ann^{-1}$ or $g\ m^{-2}\ ann^{-1}$.

<u>Physical features:</u>

A - loch area (m^2, km^2)

V - loch volume (m^3)

$\bar{z}$ - loch mean depth (m)

Q - flow rate ($l\ s^{-1}$, $m^3 s^{-1}$ or cumecs)

R - retention coefficient (no units, being the fraction of water or of P entering the loch that is retained by the loch - usually over 1 year)

ρ - flushing rate (per annum, $.ann^{-1}$, the volume of water entering the loch divided by V above)

T(w) - water residence time (years, y, the reciprocal of ρ above, ie the theoretical time that water equivalent to V above remains in the loch)

Table 1. (From Smith 1974)

Morphometry of Loch Leven

Mean depth	3.9m
Maximum depth	25.5m
Surface area	13.3 km^2
Volume	52.4 x 10^6 m^3
Length	5.9 km
Breadth	2.3 km
Length of shore line	18.5 km
Shore-line development	1.43
Water Balance:	
Annual average inflow	120.7 x 10^6 m^3
Mean renewal time	5.2 months

Table 2. Annual mean instantaneous flow rates ($\bar{Q}$) at sites in the Loch Leven watershed (see Figure 1 for locations). I. major sites recorded at 8-day intervals throughout a 12-month period; II. small drainage channels gauged only twice; III. ungauged burns (a) and areas of land with no well-defined drainage course (b): annual discharges from these sites are based on the discharge/catchment area relationship found for the gauged sites combined. The percentages that the flow and catchment area of each group of sites comprise of the whole watershed are also indicated (% Q/Q_{tot} and % A/A_{tot}).

Group	Site	$\bar{Q}$ ($\ell.s^{-1}$)	% Q/Q_{tot}	% A/A_{tot}
I	Na	1452.0		
	Sa	960.0		53.82
	Ga	767.0		
	Pa	199.0		
	Ca	55.1		
	Ka	47.6		
	Ua	22.8		
	Sub-Total	3503.5	88.8	88.5
II	Ea	44.2		
	Fa	34.6		
	Ba	33.1		
	Da	19.8		
	Sub-Total	131.7	3.7	3.6
IIIa	Ha			
	Wa			
b	Cavelstone			
	Vane Farm			
	Remainder of sector including Wa and Ca			
	Sub-Total	311.6	(7.9)	7.9
	Total	3946.8	100	100

Table 3. Annual (1985) mean flows ($\ell.s^{-1}$) estimated by 3 methods for each of 3 stations on feeder streams to Loch Leven. Methods: (i) from instantaneous records at the time of water sampling every 8 days; (ii) from the continuous (½ hourly) records for those sampling days, and (iii) from the continuous records for 365 days.

Recording Station	1985 mean flow ($\ell.s^{-1}$) Method (i)	Method (ii)	Method (iii)
Nd	900	875	878
Pc	119	115	119
Sc	960	913	946

Table 4. Correlation coefficients indicating the relationships between instantaneous measurements of flow of various sites at feeder waters to Loch Leven: * results for stations referred to in Figure 6.

	FLOWCA	FLOWGA	FLOWGB	FLOWGC	FLOWGT	FLOWKA	FLOWKB	FLOWKP	FLOWNA
FLOWGA	0.674								
FLOWGB	0.681	0.999							
FLOWGC	0.710	0.924	0.924						
FLOWGT	0.423	0.880	0.857	0.770					
FLOWKA	0.730	0.717	0.726	0.771	0.653				
FLOWKB	0.682	0.707	0.690	0.691	0.682	0.596			
FLOWKP	0.294	0.113	0.093	-0.014	0.024	0.713	-0.081		
FLOWNA	0.750	0.951	0.950	0.885	0.759	0.739	0.802	0.296	
FLOWND *	0.707	0.957	0.953	0.904	0.864	0.728	0.807	0.230	0.991
FLOWNH	0.771	0.960	0.959	0.927	0.848	0.868	0.694	0.535	0.988
FLOWPB *	0.809	0.917	0.916	0.885	0.710	0.810	0.796	0.306	0.959
FLOWPC	0.789	0.933	0.931	0.889	0.750	0.791	0.751	0.293	0.964
FLOWSC	0.703	0.938	0.933	0.887	0.776	0.719	0.822	0.331	0.978
FLOWUA *	0.784	0.829	0.830	0.860	0.678	0.843	0.888	0.149	0.900

	FLOWND	FLOWNH	FLOWPB	FLOWPC	FLOWSC
FLOWNH	0.983				
FLOWPB *	0.946	0.969			
FLOWPC	0.957	0.973	0.987		
FLOWSC	0.982	0.977	0.951	0.948	
FLOWUA *	0.895	0.928	0.932	0.898	0.877

Table 5. Analytical precision testing: Forth River Purification Board (F) and Institute of Terrestrial Ecology (I) results for SRP and TP determinations; SD is standard deviation.

Sample	Blank		Low standard		High standard		Sample		Spiked sample	
SRP	F	I	F	I	F	I	F	I	F	I
Mean ($\mu g\ l^{-1}$)	0	'-0.12'	502	39.1	4526	397.6	1016	88.2	3019	190.2
Within batch SD ($\mu g\ l^{-1}$)	0	0.187	4.5	0.5	57.7	1.9	7.1	0.7	14.8	1.2
Between batch SD ($\mu g\ l^{-1}$)	0	0.245	8.0	0.4	99.7	1.6	-	-	-	-
Total SD ($\mu g\ l^{-1}$)	0	0.307	9.1	0.6	115.2	2.3	22.9	0.8	66.3	1.4
Relative SD (%)	-	-	1.8	1.7	2.5	0.6	2.3	0.9	2.2	0.7
Mean spiking recovery (%)	-	-	-	-	-	-	-	-	101.2	98.5
TP										
Mean ($\mu g\ l^{-1}$)	0	'-0.18'	520	40.0	4651	401.2	1112	182.8	3156	250.6
Within batch SD ($\mu g\ l^{-1}$)	0	0.245	5.9	1.2	60.2	2.4	12.2	1.52	37.2	2.2
Between batch SD ($\mu g\ l^{-1}$)	0	0.198	7.6	0.1	121.8	2.2	-	-	-	-
Total SD ($\mu g\ l^{-1}$)	0	0.315	9.6	1.2	135.9	3.3	30.4	1.8	87.3	2.6
Relative SD (%)	-	-	1.8	2.9	2.9	0.8	2.7	1.0	2.8	1.0
Mean spiking recovery (%)	-	-	-	-	-	-	-	-	103.3	96.6

Table 6. Results of analytical bias testing for the determination of SRP and TP by the Forth River Purification Board (FRPB), the Institute of Terrestrial Ecology (ITE) and the Freshwater Biological Investigation Unit (FBIU). Figures are expressed as concentrations or percentages as appropriate.

Sample	Natural Water A	Natural Water B	Standard Solution
SRP			
Concentration ($\mu g\ l^{-1}$)	74.3	328.4	200.0
Maximum possible bias:			
ITE	0.95 $\mu g\ l^{-1}$	-0.79%	-1.06%
FBIU	-2.37 $\mu g\ l^{-1}$	1.07%	2.50%
FRPB	2.00 $\mu g\ l^{-1}$	-0.62%	2.31%
TP			
Concentration ($\mu g\ l^{-1}$)	88.3	408.5	250.0
Maximum possible bias:			
ITE	1.78 $\mu g\ l^{-1}$	-2.23%	-1.37%
FBIU	36.82 $\mu g\ l^{-1}$	34.14%	63.79%
FRPB	-1.54 $\mu g\ l^{-1}$	1.74%	-1.73%

Table 7. Equations from linear regressions of $\log_{10}$ instantaneous loadings of total P (μg TP s^{-1},y) on $\log_{10}$ flow (l s^{-1},x) of some feeder streams to Loch Leven; r^2 values indicate the proportion of variation in y accounted for by variation in x; s^2 values (listed only for sites for which continuous P loadings are eventually derived - see Table 10) are the squares of the standard deviation of y about the regression lines.

Water course and site*	Regression equation	r^2	s^2
Upper North Queich (Nd)	y = 1.4068x + 0.2389	0.866	0.0664
Hatton Burn (Nh)	y = 0.8803x + 2.2554	0.813	0.0753
Upper South Queich (Sc)	y = 1.6382x - 0.3619	0.897	0.0530
Pow Burn (Pb)	y = 1.3965x + 1.0264	0.729	0.0665
Pow Burn (Pc)	y = 1.2057x + 1.5882	0.808	0.0469
Gairney Water (Ga)	y = 1.2700x + 0.8868	0.889	-
site Gb	y = 1.1182x + 1.3453	0.845	0.0396
site Gc	y = 1.3094x + 0.7449	0.828	-
site Gt	y = 1.4063x + 0.8905	0.865	-
Camel Burn (Ca)	y = 1.0041x + 1.5804	0.550	-
Ury Burn (Ua)	y = 1.6741x + 1.0373	0.872	0.0644

* See Figure 1 for location.

Table 8. Loadings of phosphorus (kg $TPL_{ir}.ann^{-1}$) and run-off loss rates (kg TP $ha^{-1}.ann^{-1}$) from sites draining sub-catchments of the Loch Leven watershed not affected by marked point-source pollution.

Drainage system*	Subcatchment area (km^2)	TPL_{ir} (kg. ann^{-1})	TP loss rate (kg.$ha^{-1}.ann^{-1}$)
Upper S. Queich (Sc)	33.82	1669	0.493
Upper N. Queich (Nd)	26.20	1395	0.532
Hatton Burn (Nh)	7.25	571	0.788
Ury Burn (Ua)	1.73	131	0.757
Clash Burn (Ha)	2.36	179[a]	0.757[a]
Fochy Burn (Nf)	9.55	1535[b]	1.607[b]
Gairney Water (Gb)	30.71	1303	0.424
Gairney Tributary (Gt)	2.58	156	0.605
Vane Farm sector	4.39	216[c]	0.492[c]
Cavelstone sector	2.58	156[d]	0.605[d]
Pow Burn (Pb)	10.50	1157	1.102
Camel Burn (Ca)	1.24	51	0.411
'Wood' Burn (Wa)	0.50	13[e]	0.260[e]
remainder of sector 'Ca & Wa'	1.44	59[f]	0.411[f]
Drainage channel B	1.70	284	1.671
" " D	0.50	60	1.200
" " E	2.00	423	2.115
" " F	0.90	35	0.389
TOTAL B, D, E & F	5.10		

* see Figure 1 for the positions of these stations.

a assumes that the P loss rate from catchment area 'Ha & Ua' (4.09 km^2) is the same as that calculated from loadings measured at Ua.

b calculated from the difference between the loading at the mouth of the North Queich (Na) and the summed loadings at Nd, Nh and Milnathort STW.

c uses the loss rate calculated for a catchment of generally similar land use - Sc - and relates this to sector area.

d as c but uses value for Gt.

e assumes load is ca. 25% of that calculated for Ca; in comparison, measured TP concentrations are similar in Ca and Wa but the flow in Ca is visually estimated to be ca. 25% of that measured at Wa.

f area-based estimate as with c and d above but applying the loss-rate calculated from loadings measured at Ca.

Table 9. The mean % contributions of particulate and soluble fractions of P to the total P loadings at gauged sites on water courses entering Loch Leven; these together drain 84% of the total watershed.

Stream System*	Sub-catchment area (km^2)	% PP	% TSP	% SRP
		of TP loading.		
<u>South Queich</u>				
Upper S. Queich (Sc)	33.82	53	47	32
<u>North Queich</u>				
Upper N. Queich (Nd)	26.20	52	48	29
Hatton Burn (Nh)	7.25	45	55	44
Ury Burn (Ua)	1.73	21	79	71
<u>Gairney Water</u>				
Main Stream (Gb)	30.71	50	50	29
Tributary (Gt)	2.58	40	60	40
<u>Pow Burn</u>				
Main stream (Pb)	10.50	68	32	25
Camel Burn (Ca)	1.24	51	49	33
Channel B (Ba)	1.70	40	60	55
" D (Da)	0.50	61	39	37
" E (Ea)	2.00	65	35	31
" F (Fa)	0.90	79	21	10

* See Figure 1 for locations

Table 10. Continuous loadings of P (TPL_{cr} and $SRPL_{cr}$) based on 365 daily values derived as described in the text: (a) original estimates, (b) original estimates corrected as proposed by Ferguson (1986)** and (c) the ratio of estimates based on (8-day) instantaneous measurements to the values in (b).

Stream system*	TPL_{cr} ($kg\ TPL\ ann^{-1}$)			$SRPL_{cr}$ ($kg\ SRP\ ann^{-1}$)		
	(a)	(b)	(c)	(a)	(b)	(c)
<u>South Queich</u>						
Upper S. Queich (Sc)	1692	1947	0.86	558	656	0.82
<u>North Queich</u>						
Upper N. Queich (Nd)	1037	1234	1.13	367	470	0.85
Hatton Burn (Nh)	315	384	1.49	131	143	1.74
Ury Burn (Ua)	89	106	1.24	70	87	1.07
<u>Gairney Water</u>						
Main stream (Gb)	933	1037	1.26	286	307	1.23
<u>Pow Burn</u>						
Upper section (Pc)	429	485	1.48	191	203	1.03
Lower section (Pb)	575	681	1.70	253	266	1.09

* See Figure 1 for location

** The corrections are made by multiplying the original estimates (in (a)) by $e^{2.65s^2}$ where s^2 is the square of the standard deviation of y ($\log_{10} TPL_{ir}$) about the line of its regression on x ($\log_{10} Q$) as shown in Table 7.

FIGURES 1-37

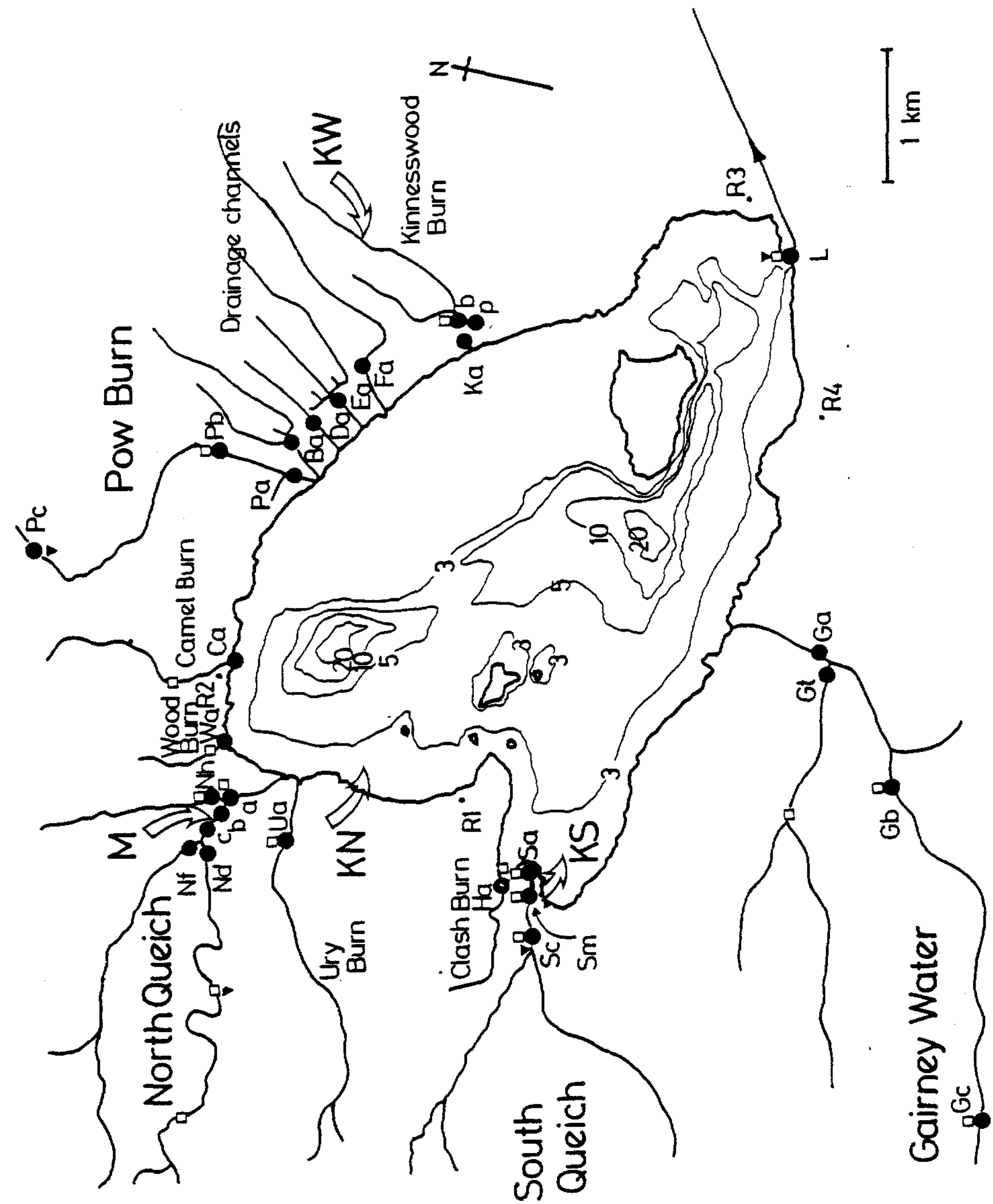


Figure 1 Loch Leven showing depth contours (in metres), position of sites sampled during the P loading study (● , labelled with codes used in other Figures and in the text) staff gauges (□), level recorders (▼). Outfalls of treated sewage effluent (open arrows) - M, Milnathort; KW, Kinnesswood; KN and KS, Kinross North and South. Industrial effluent enters the South Queich at Sm. Rain collectors were positioned at sites R1-R4.

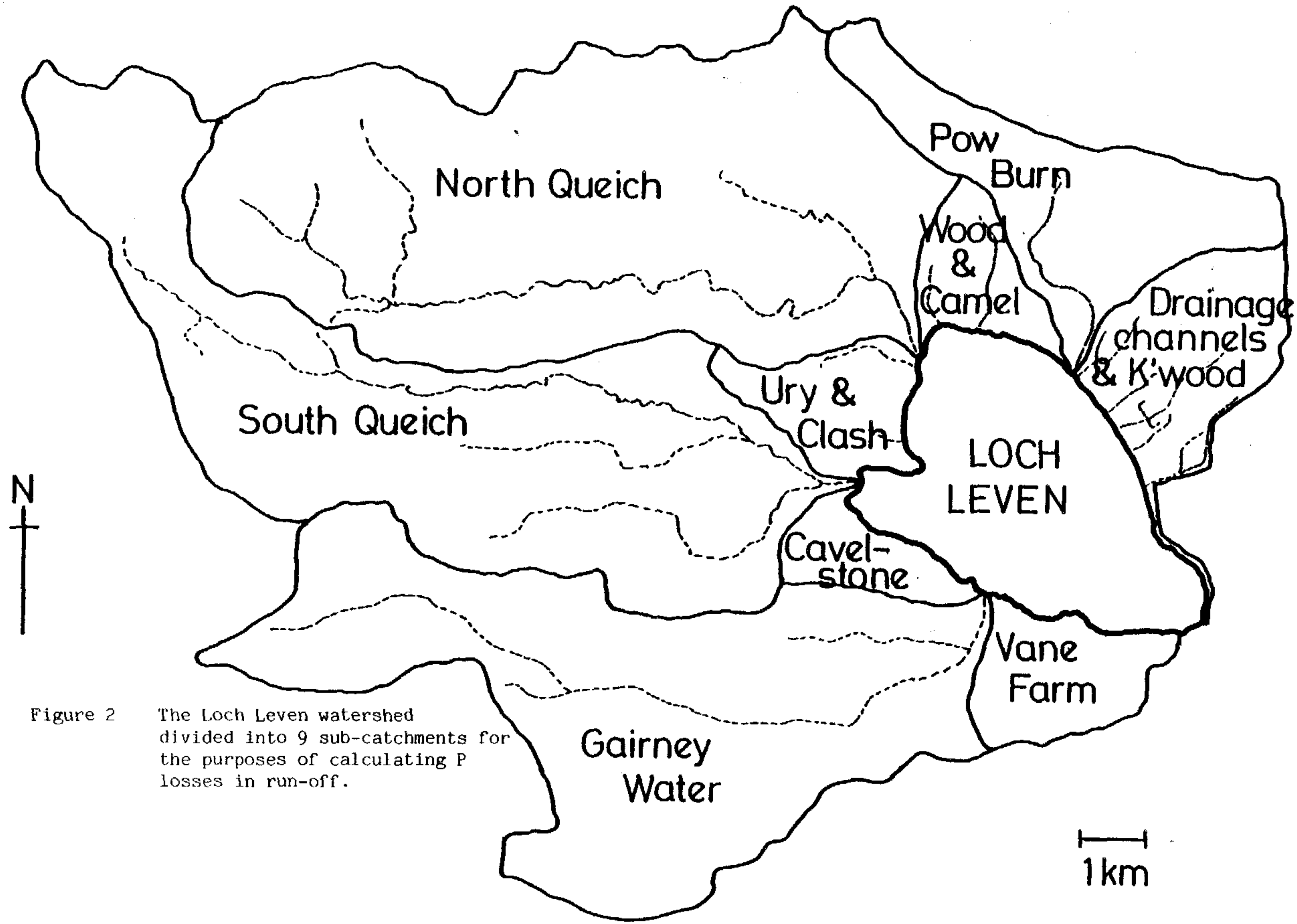


Figure 2 The Loch Leven watershed divided into 9 sub-catchments for the purposes of calculating P losses in run-off.

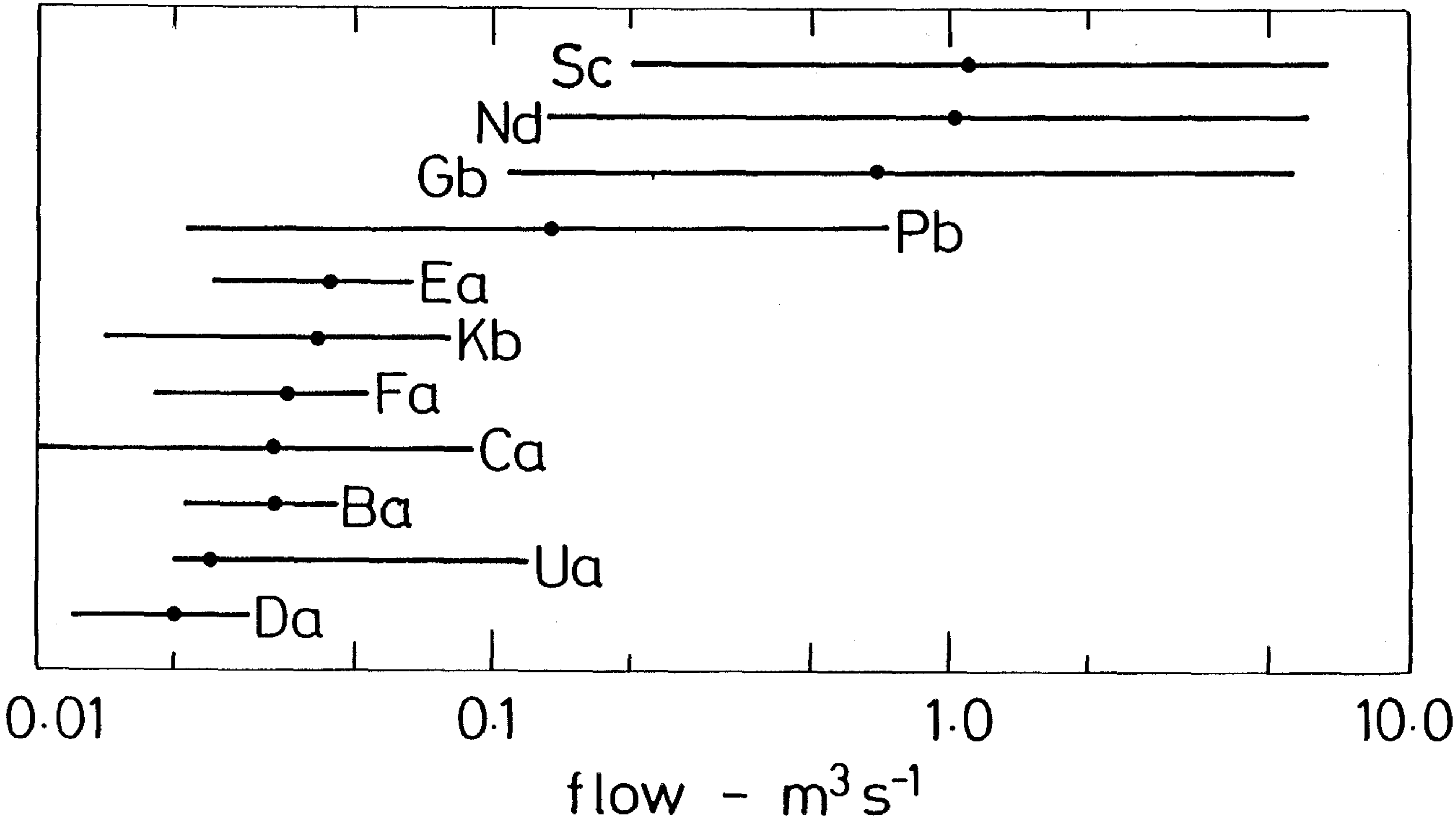


Figure 3 Mean flow (●) and ranges in flow on streams entering Loch Leven.

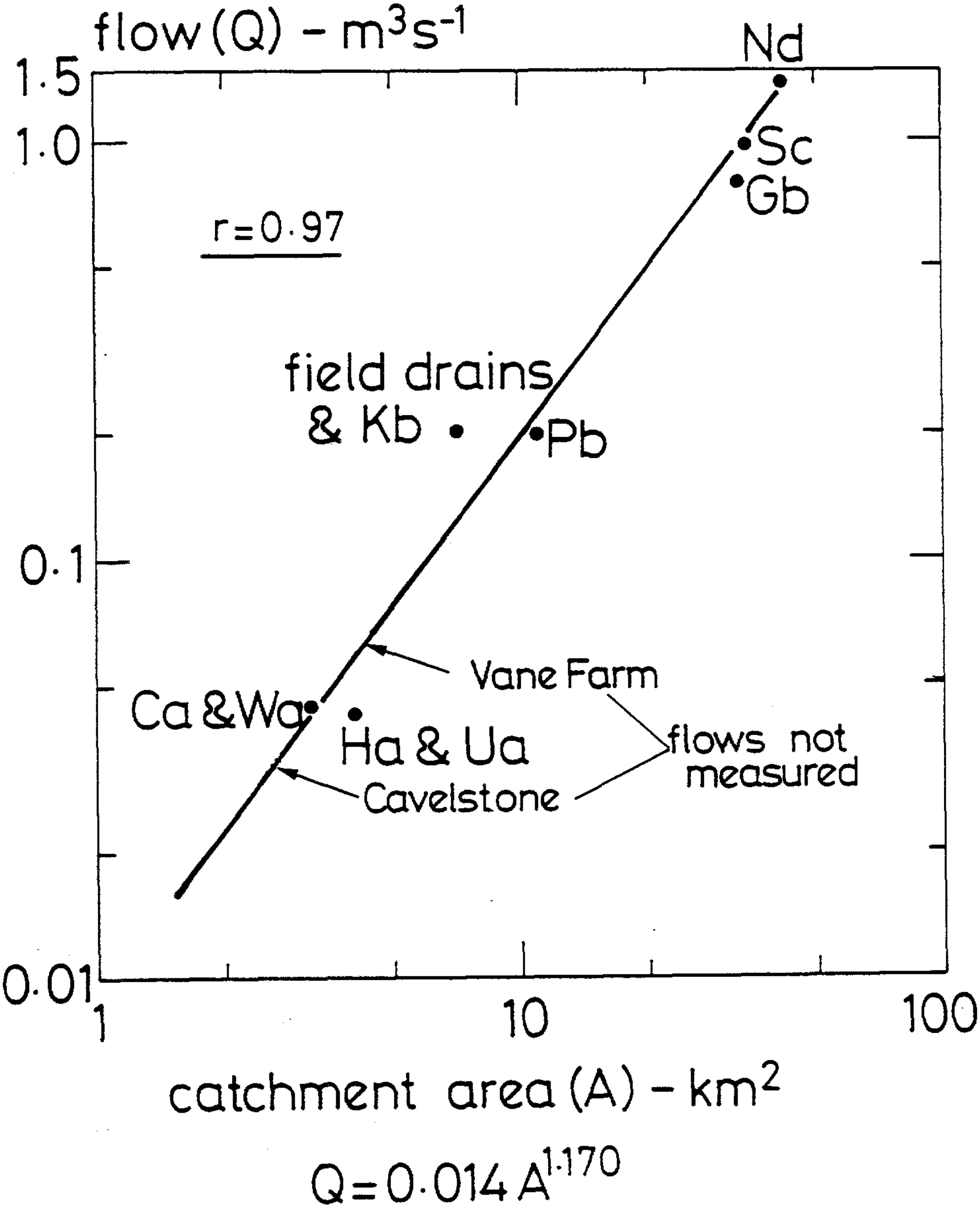


Figure 4 The relationship between mean discharge (Q) and drainage area (A) of various sub-catchments in the Loch Leven watershed; discharges are the average of instantaneous flows measured at 8-day intervals throughout 1985. No well-defined water courses exist in the Vane Farm and Cavelstone areas; water losses from these catchments (arrows) are predicted from the equation.

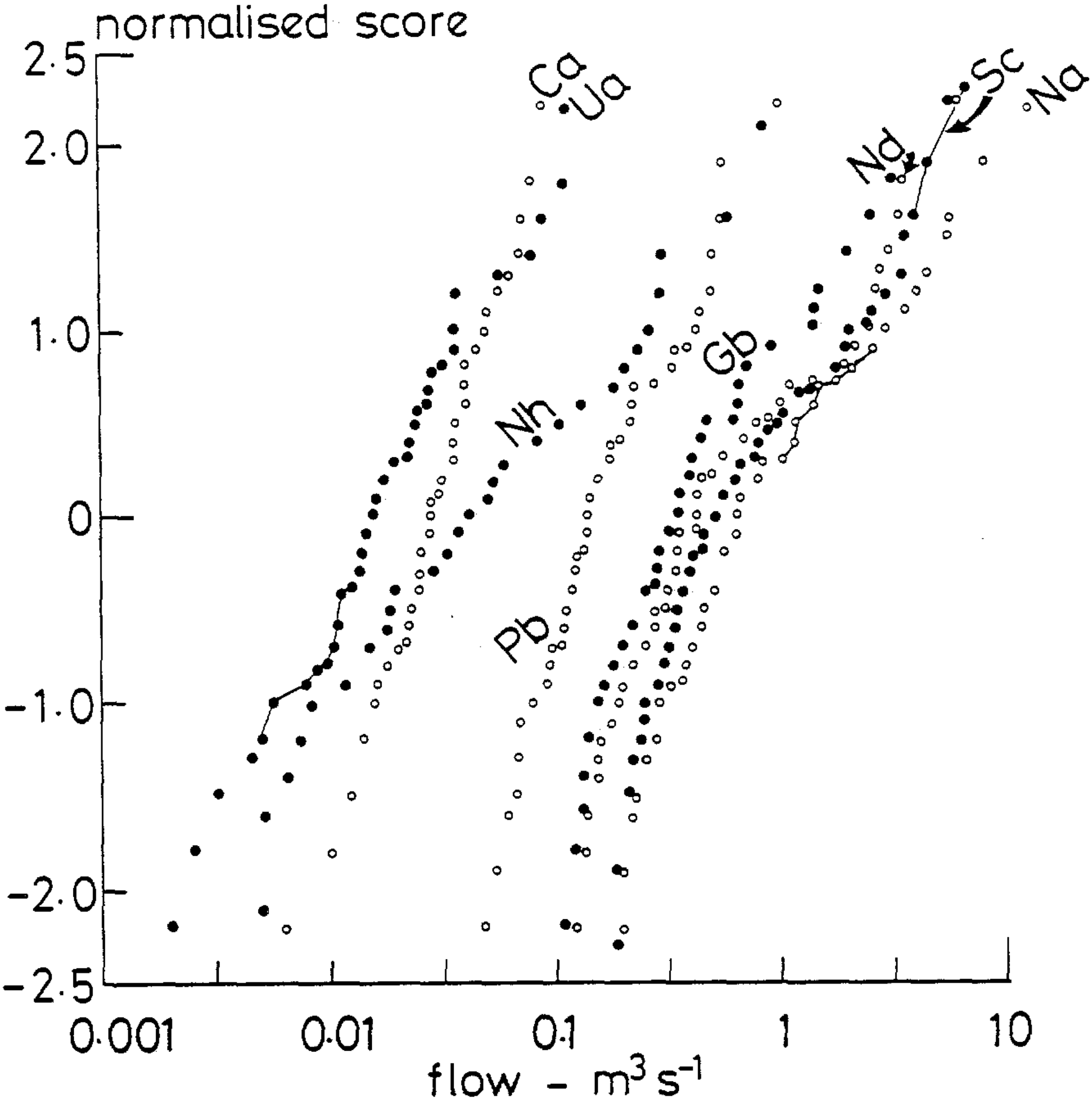


Figure 5(a) Instantaneous flow values plotted against their normalised scores to illustrate the frequency distribution of flows of various waters entering Loch Leven: plots describing more or less straight lines indicate a log-normal distribution while marked departures from linearity indicate discontinuous series of flow values.

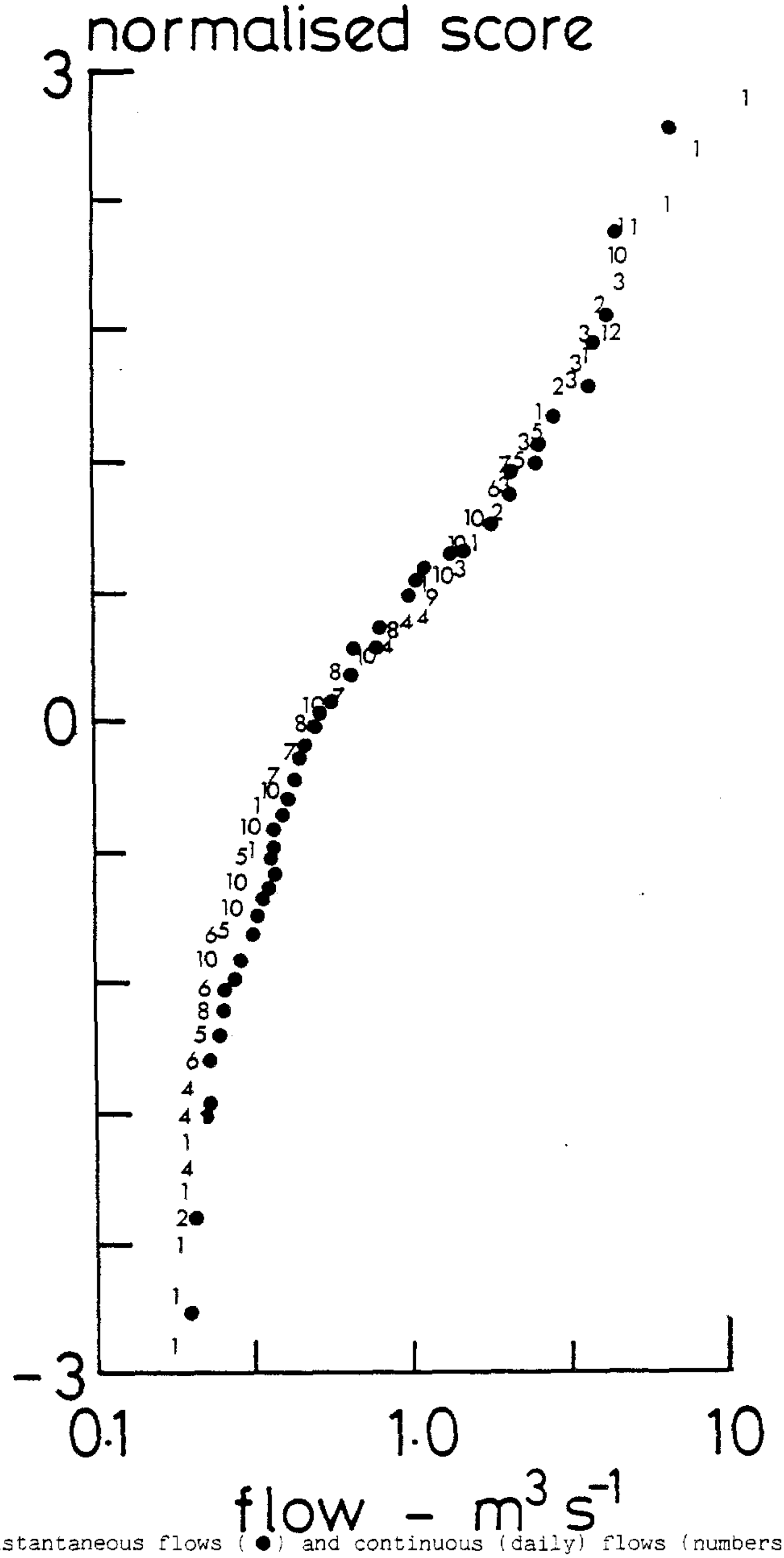


Figure 5(b) Instantaneous flows (●) and continuous (daily) flows (numbers) at site Sc on the South Queich plotted as in Figure 5(a); the similarity between the plots suggests that, in this case, water discharges and nutrient loadings based on instantaneous (8-daily) flow values, are not markedly different from those derived from continuous flow data - i.e. daily values each being the mean of 24 or 48 recordings.

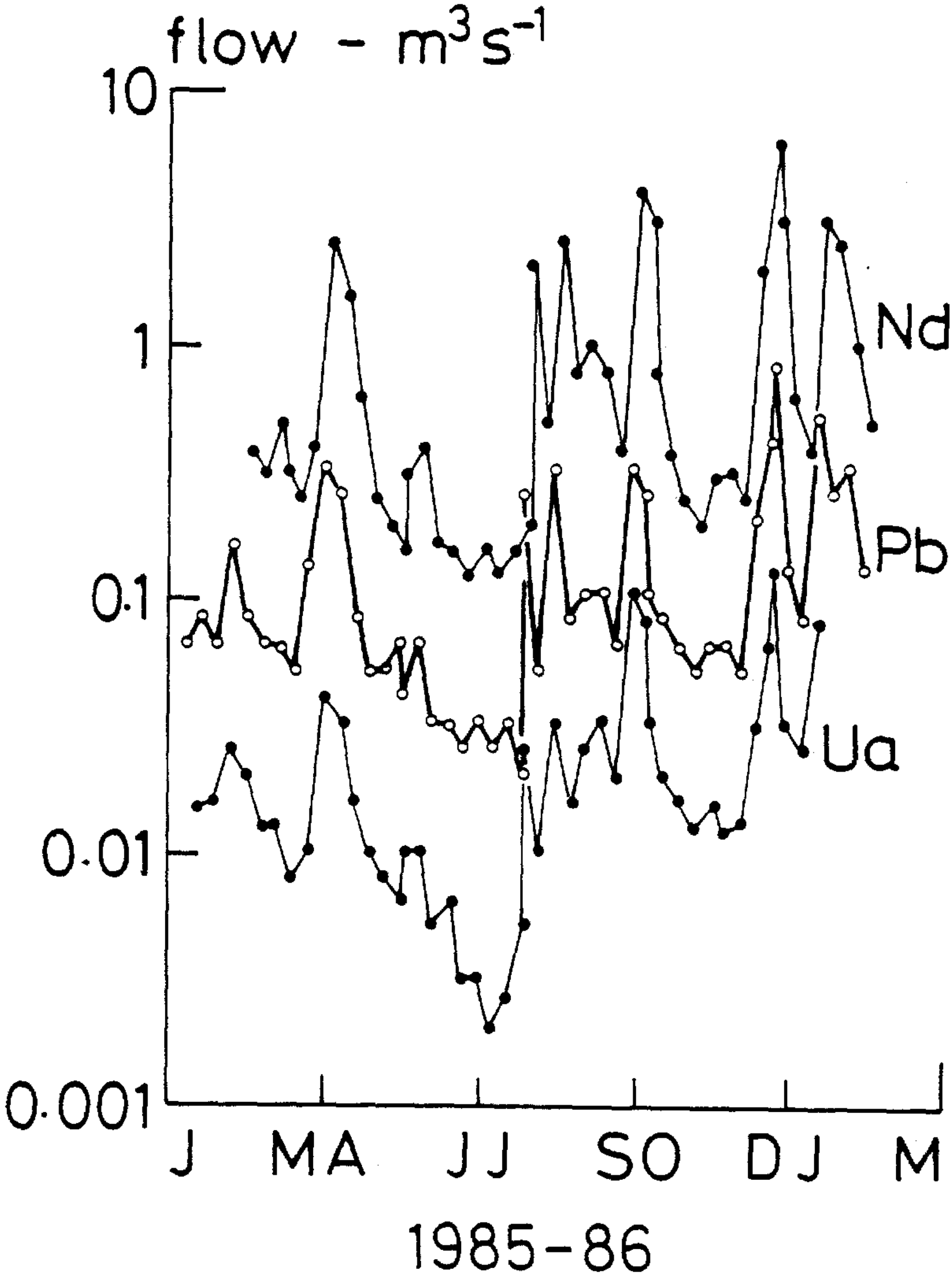


Figure 6 Seasonal variation in instantaneous flows on 3 Loch Leven feeder waters contrasting in size.

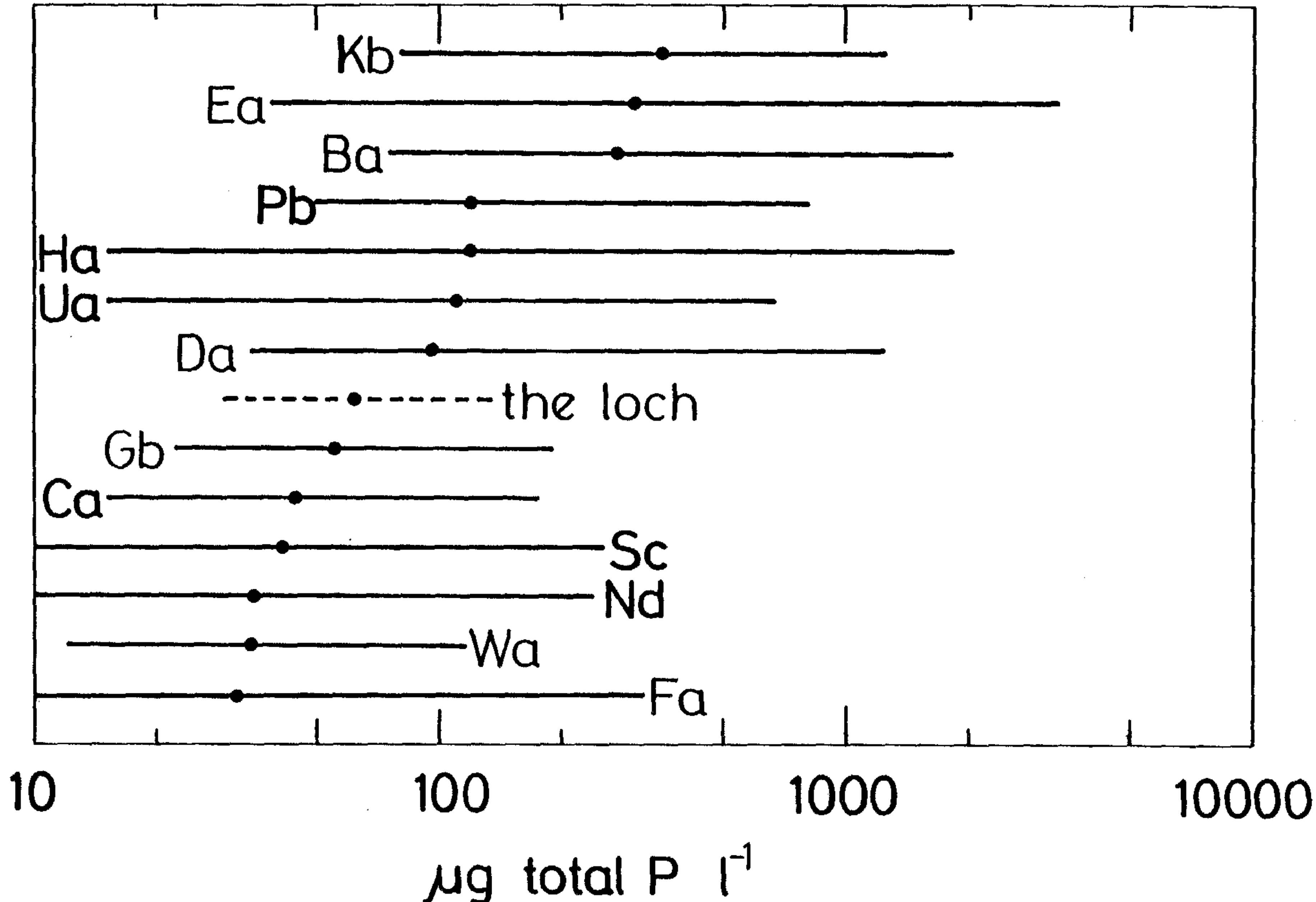


Figure 7 Mean (●) and range of total phosphorus concentration in streams entering Loch Leven and in the loch itself.

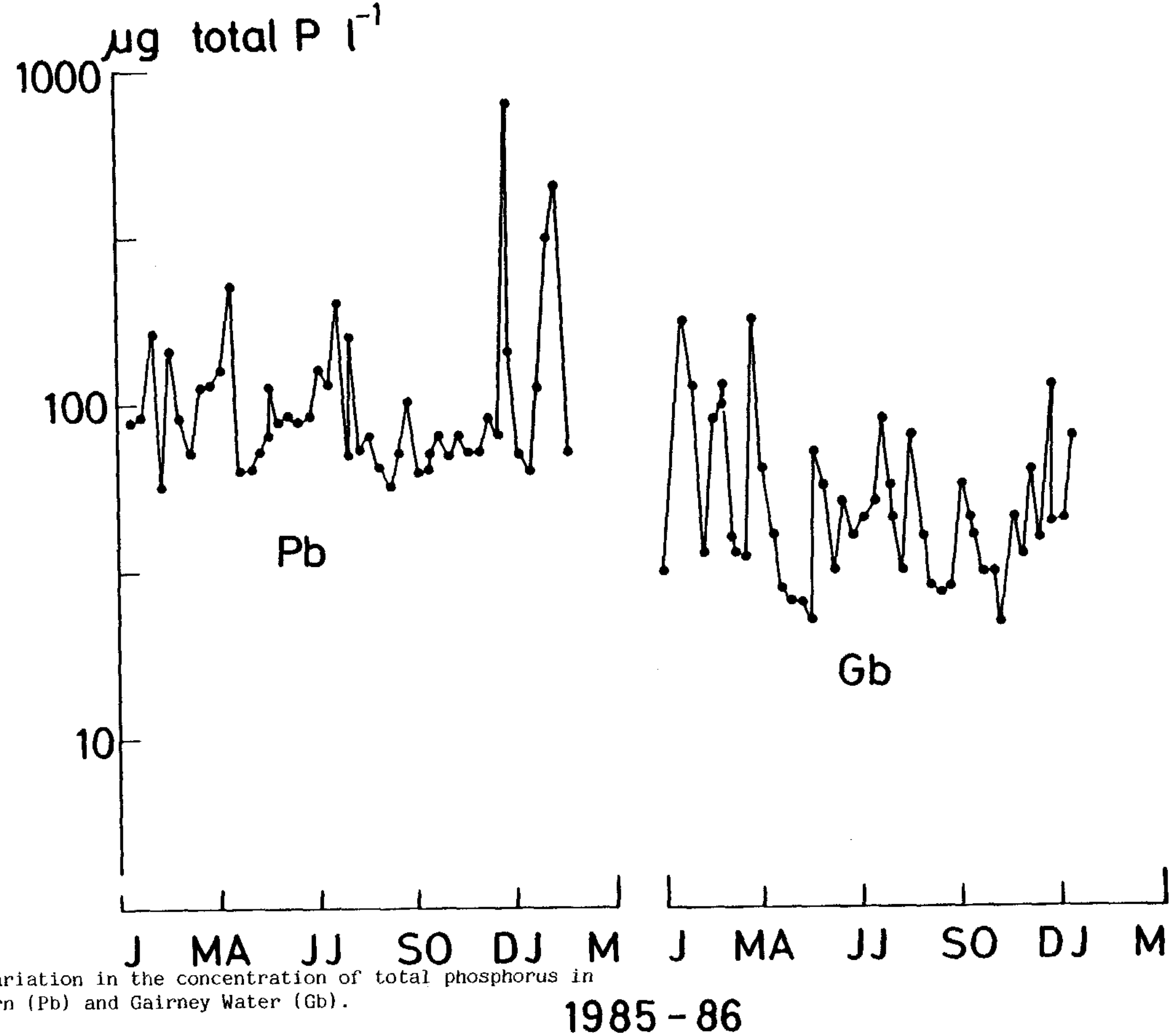


Figure 8(a) Seasonal variation in the concentration of total phosphorus *in* the Pow Burn (Pb) and Gairney Water (Gb).

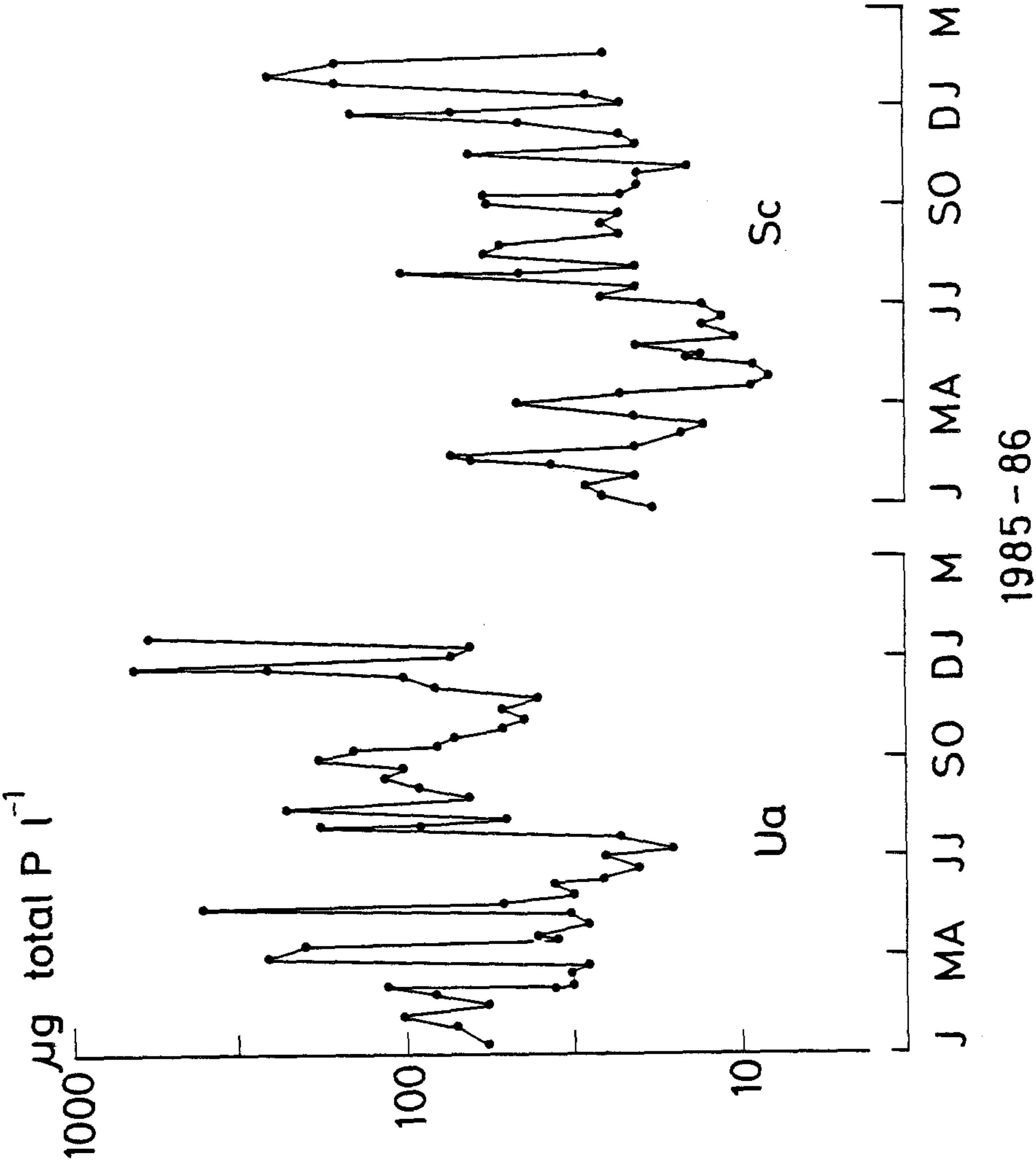


Figure 8(b). As Figure 8(a) for the Ury Burn (Ua) and the upper South Queich (Sc).

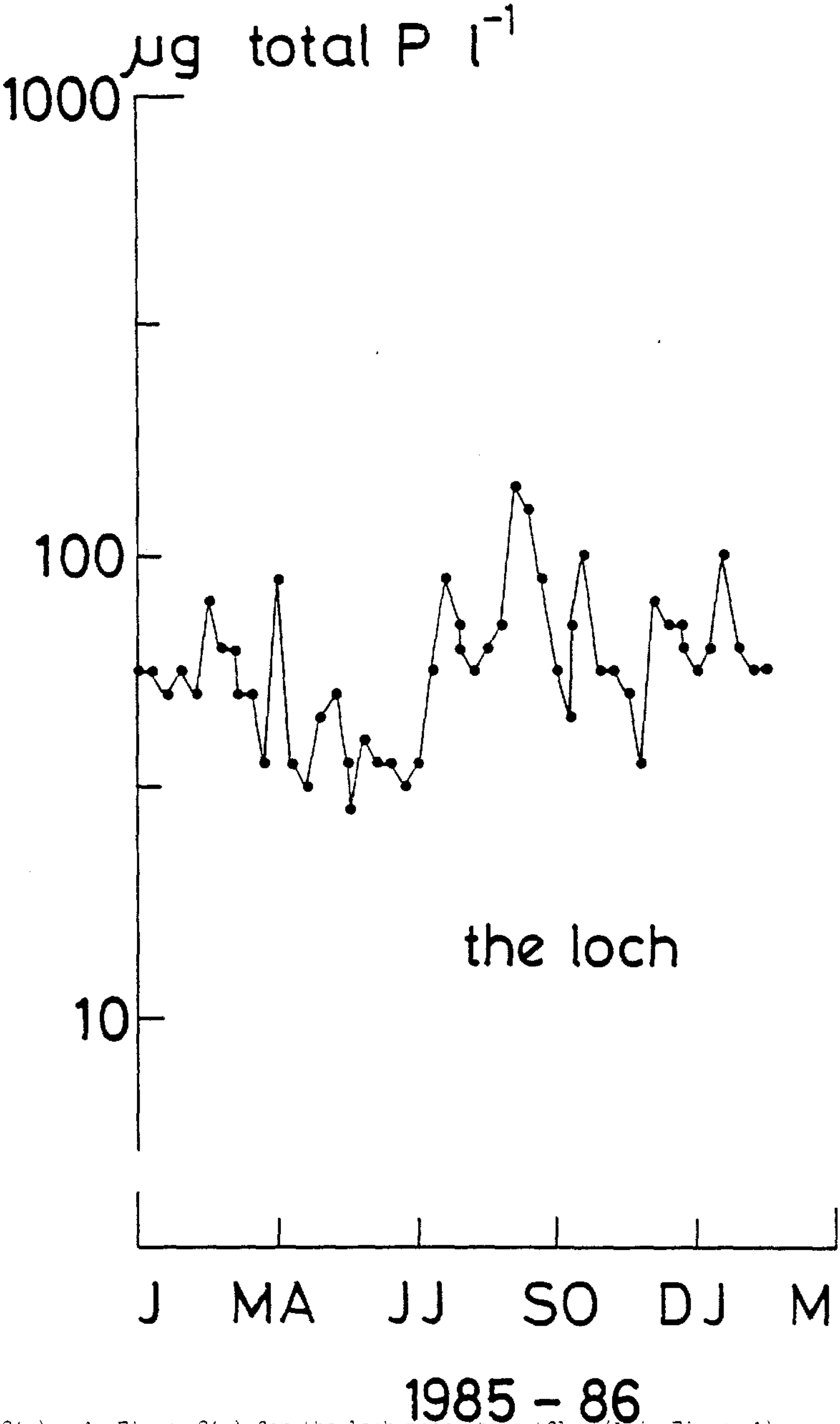


Figure 8(c) As Figure 8(a) for the loch near its outflow (L in Figure 1).

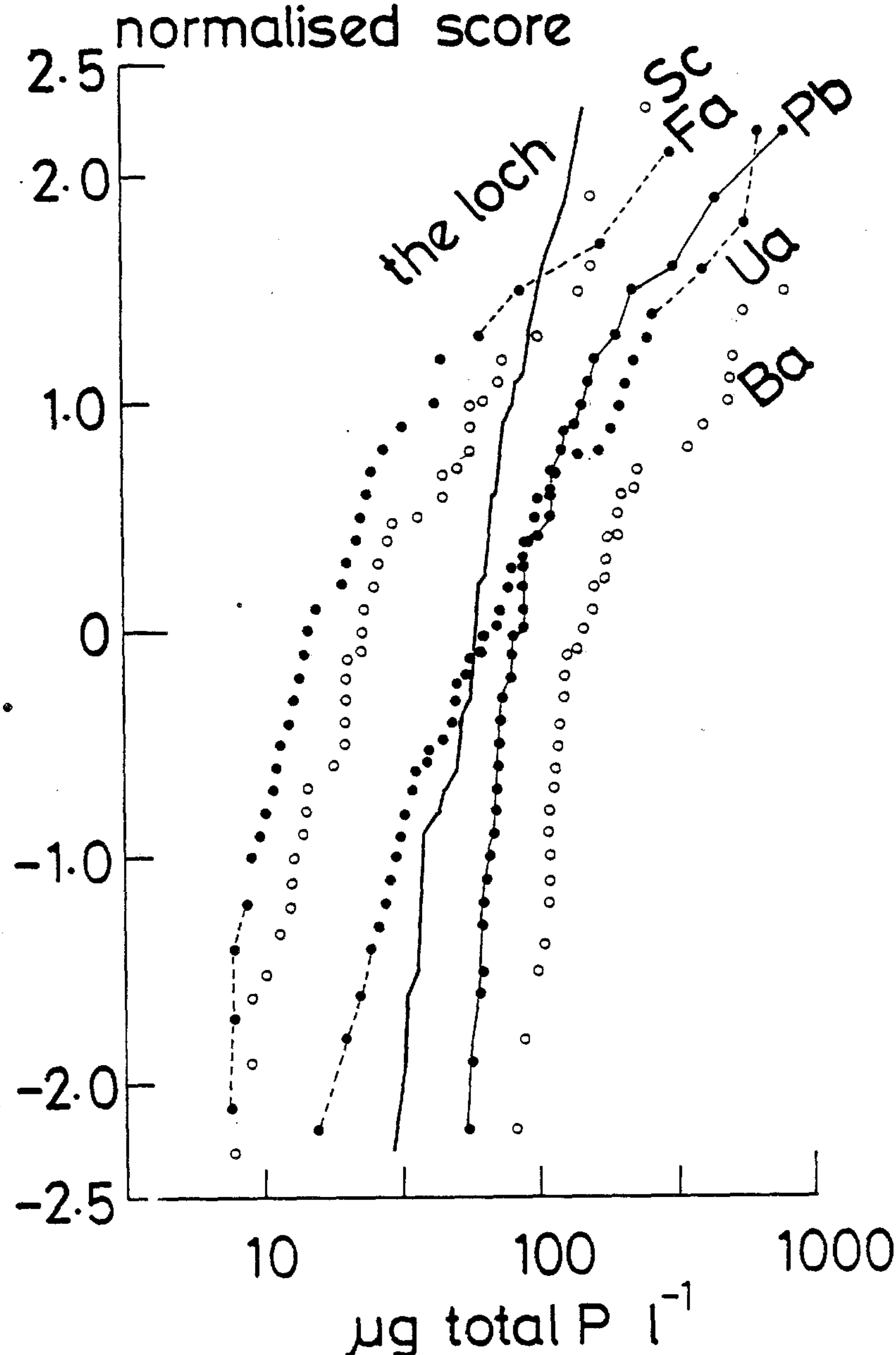


Figure 9 Values for total phosphorus concentration in 1985, plotted against their normalised scores to illustrate the frequency distribution in the loch and some of its feeder waters: upper South Queich (Sc), Ury Burn (Ua), Pow Burn (Pb) and drainage channels 'B' (Ba) and 'F' (Fa). See Figure 5(a) for explanation.

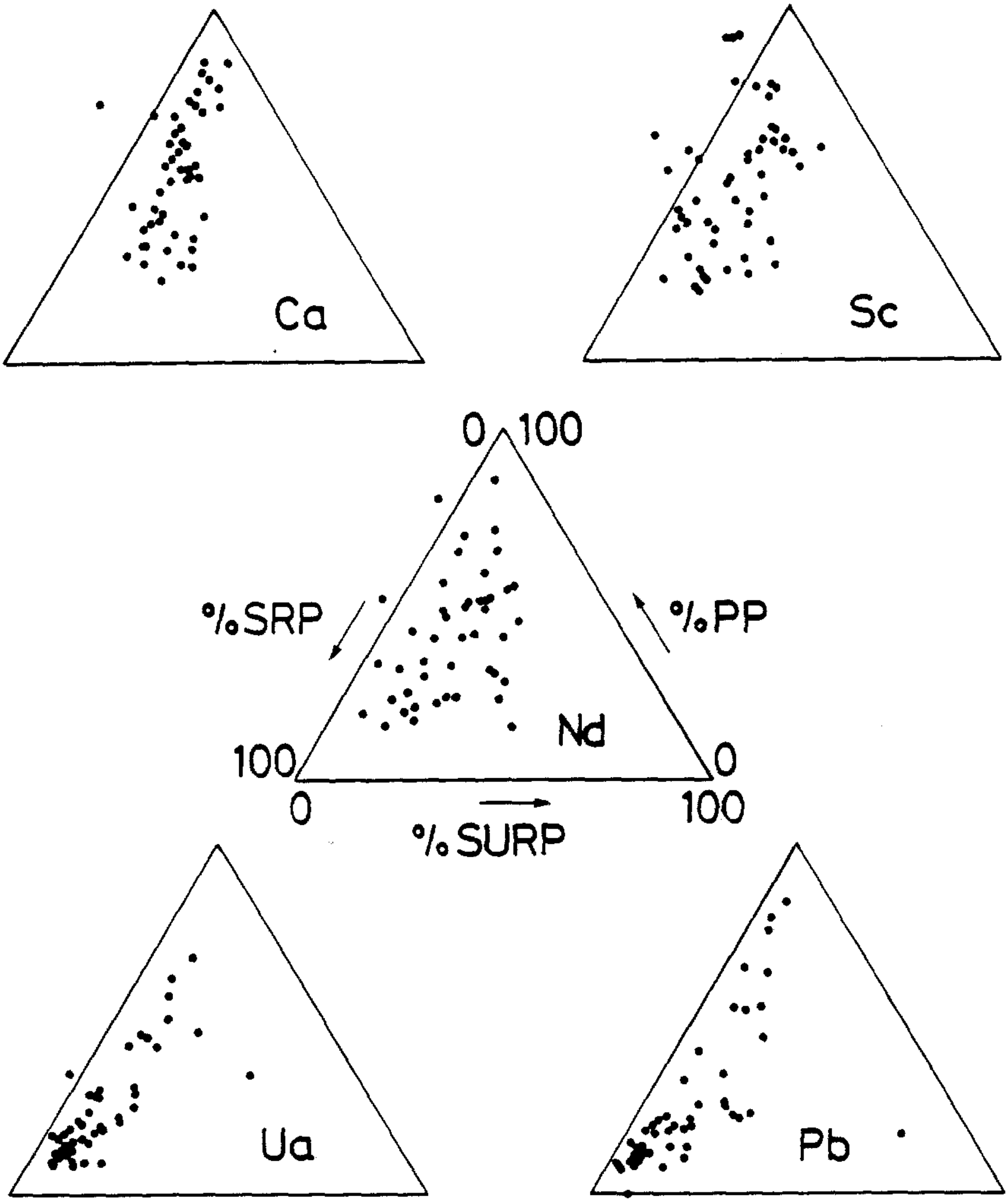


Figure 10 Triangular plots illustrating annual variation in the proportions of soluble reactive phosphorus (SRP), soluble unreactive phosphorus (soluble 'organic' phosphorus - SURP) and particulate phosphorus (PP) in the total phosphorus of a variety of feeder streams (Camel Burn, Ca; upper North Queich, Nd; other waters as Figure 9); points near the apex of the triangle correspond to high percentages of PP, points near the base of the triangle to high proportions of soluble P - SRP, left hand corner and SURP, right hand corner.

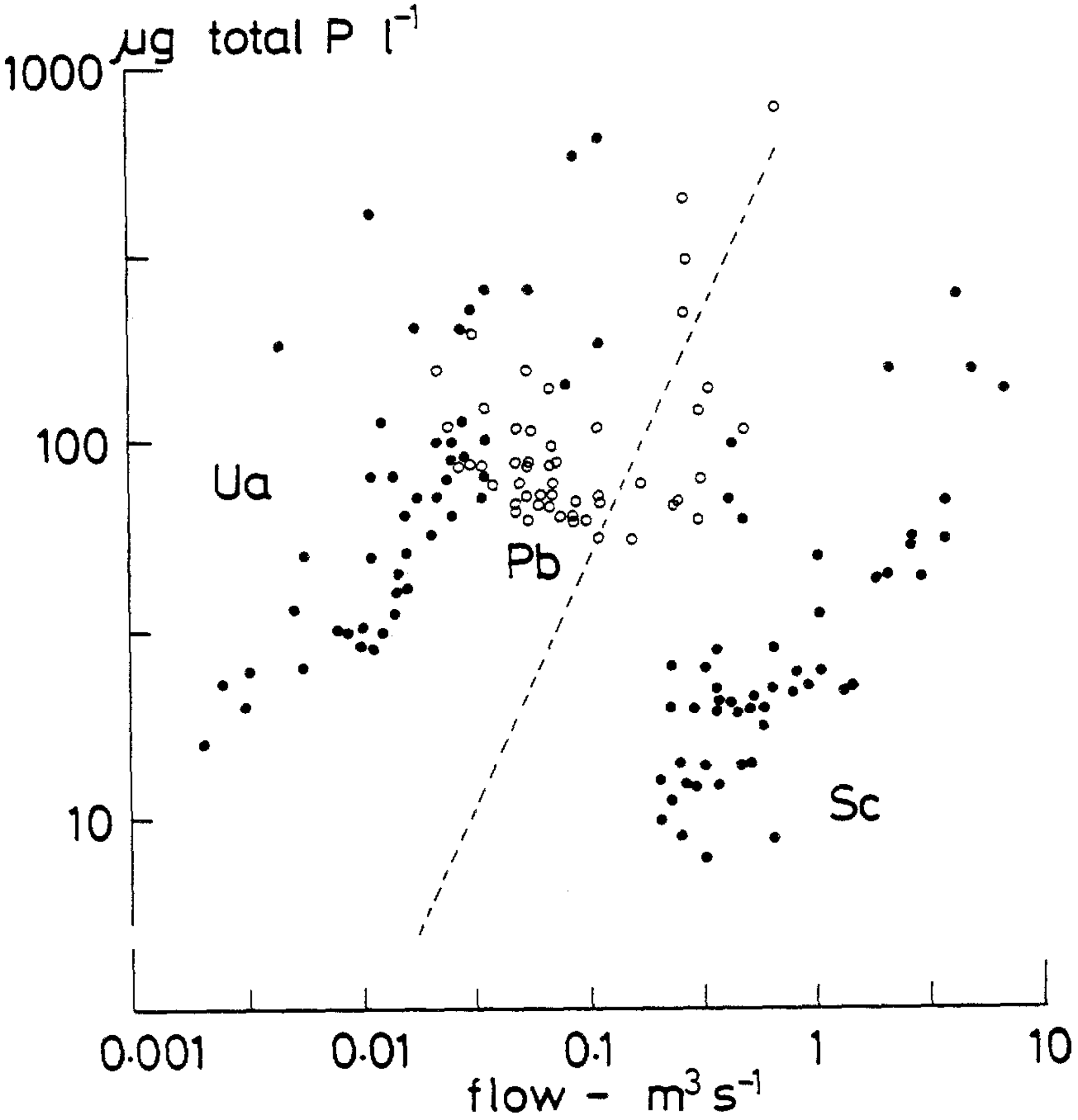


Figure 11 Relationships between the concentration of phosphorus and flow in 3 feeder streams of contrasting size (site codes as Figure 10); dotted line highlights the contrast between 2 series of solid points.

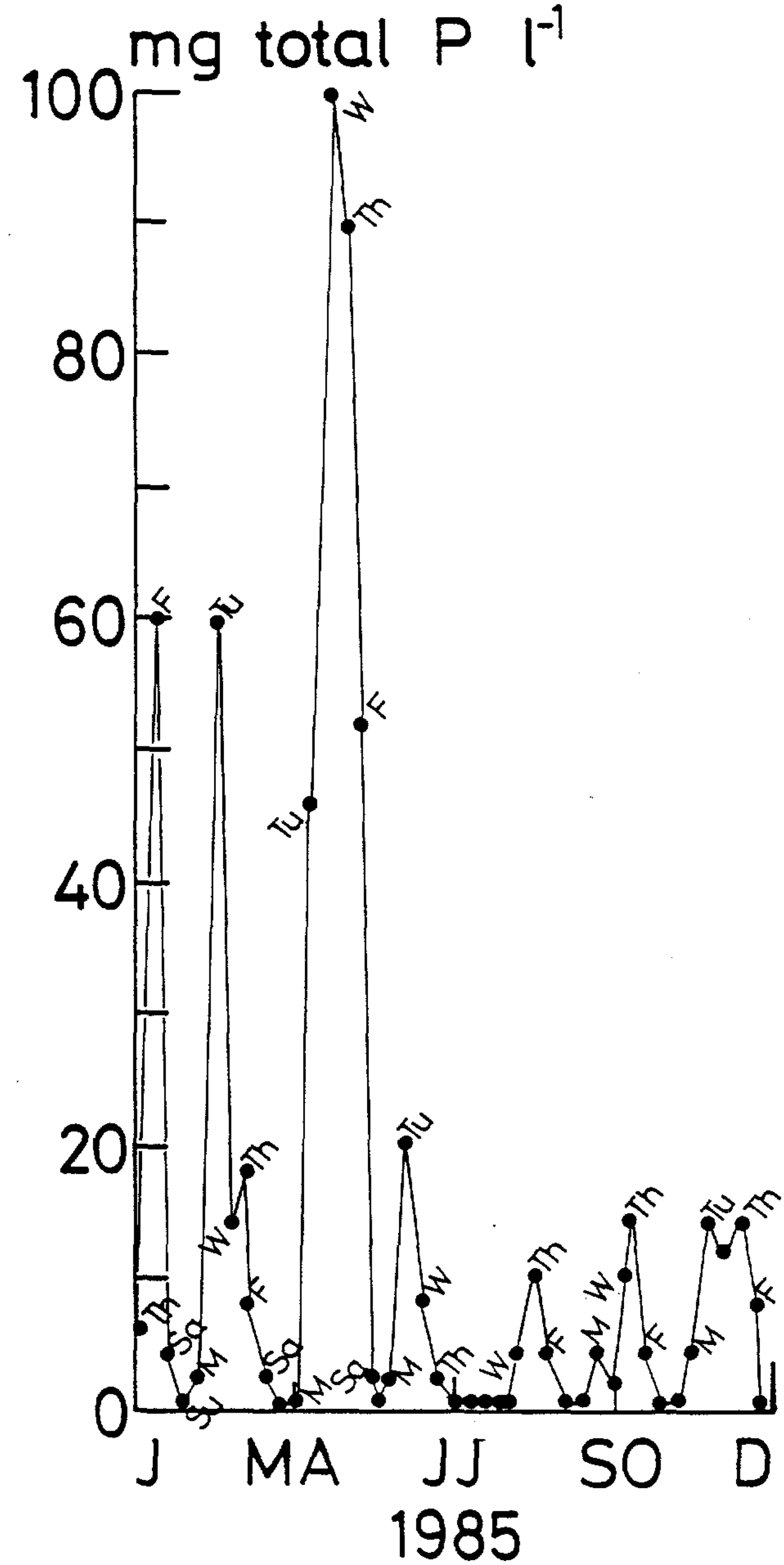


Figure 12 Fluctuations in the concentration of total phosphorus in industrial effluent before it enters the South Queich: values correspond to instantaneous samples taken at 8-day intervals throughout the year; note variation associated with day of the week.

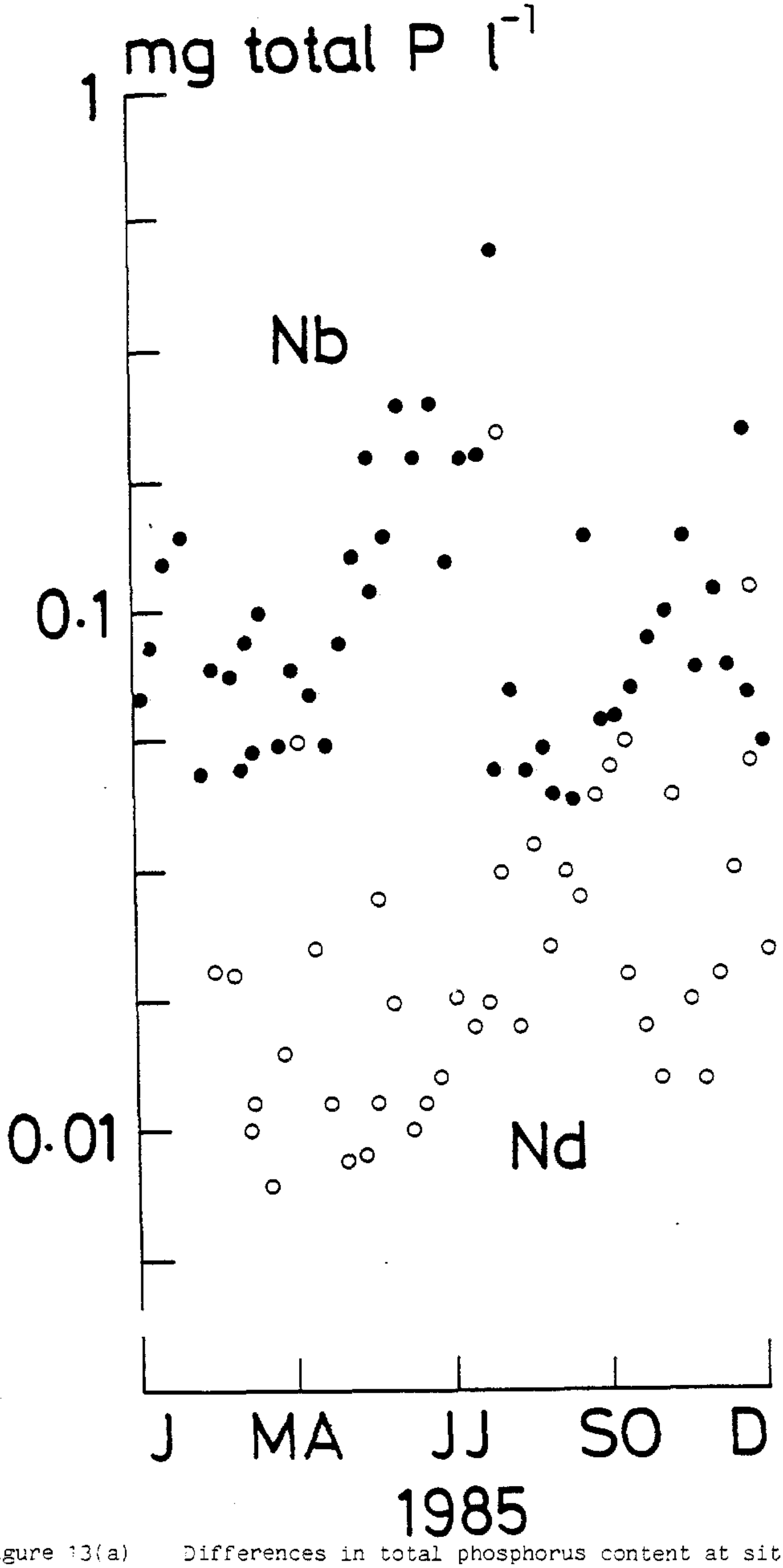


Figure 13(a) Differences in total phosphorus content at sites above (Nd) and below (Nb) the entry of Milnathort STW effluent to the North Queich: 8-day interval sampling.

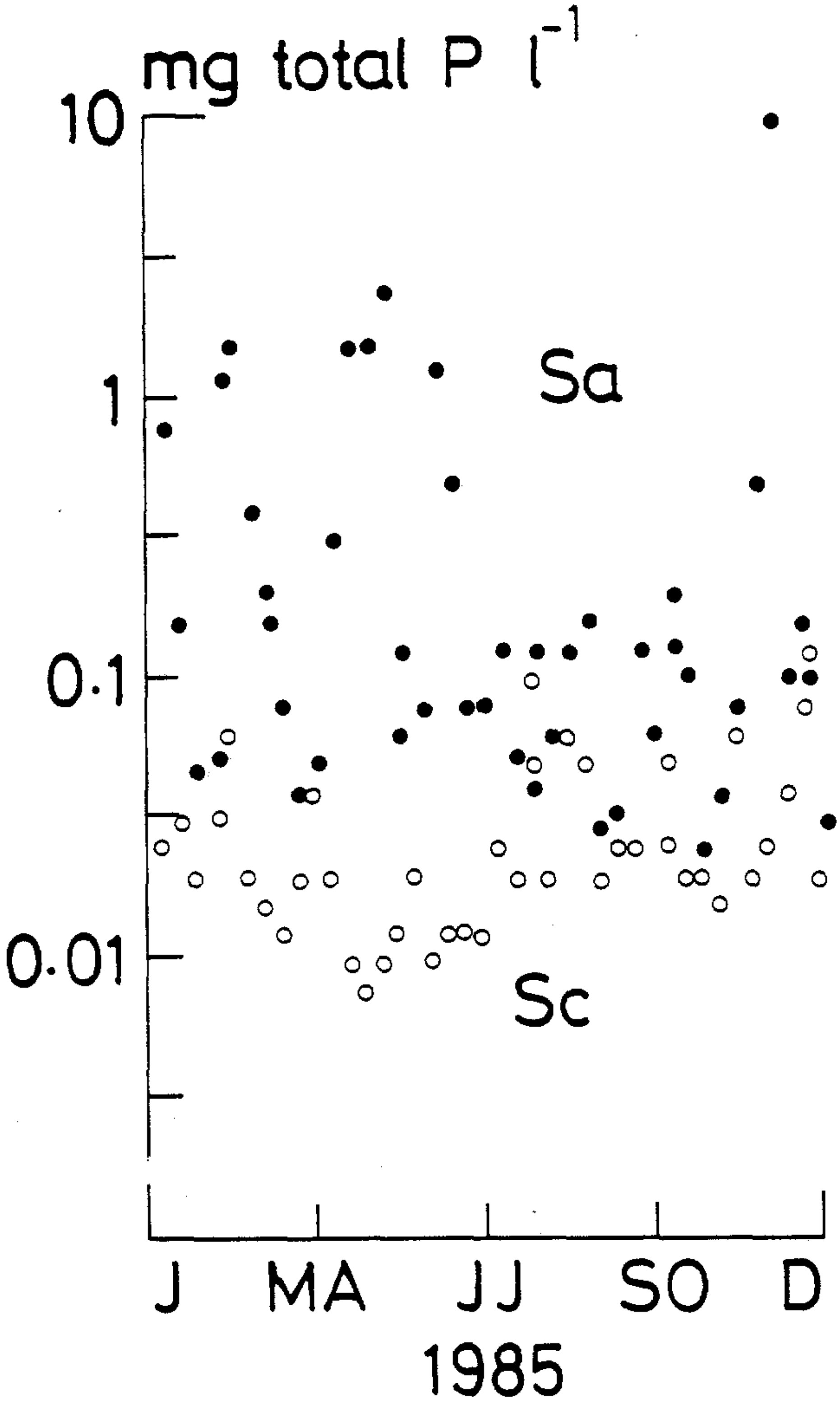


Figure 13(b) As Figure 13(a) at sites above (Sc) and below (Sa) the outfall of industrial waste on the South Queich.

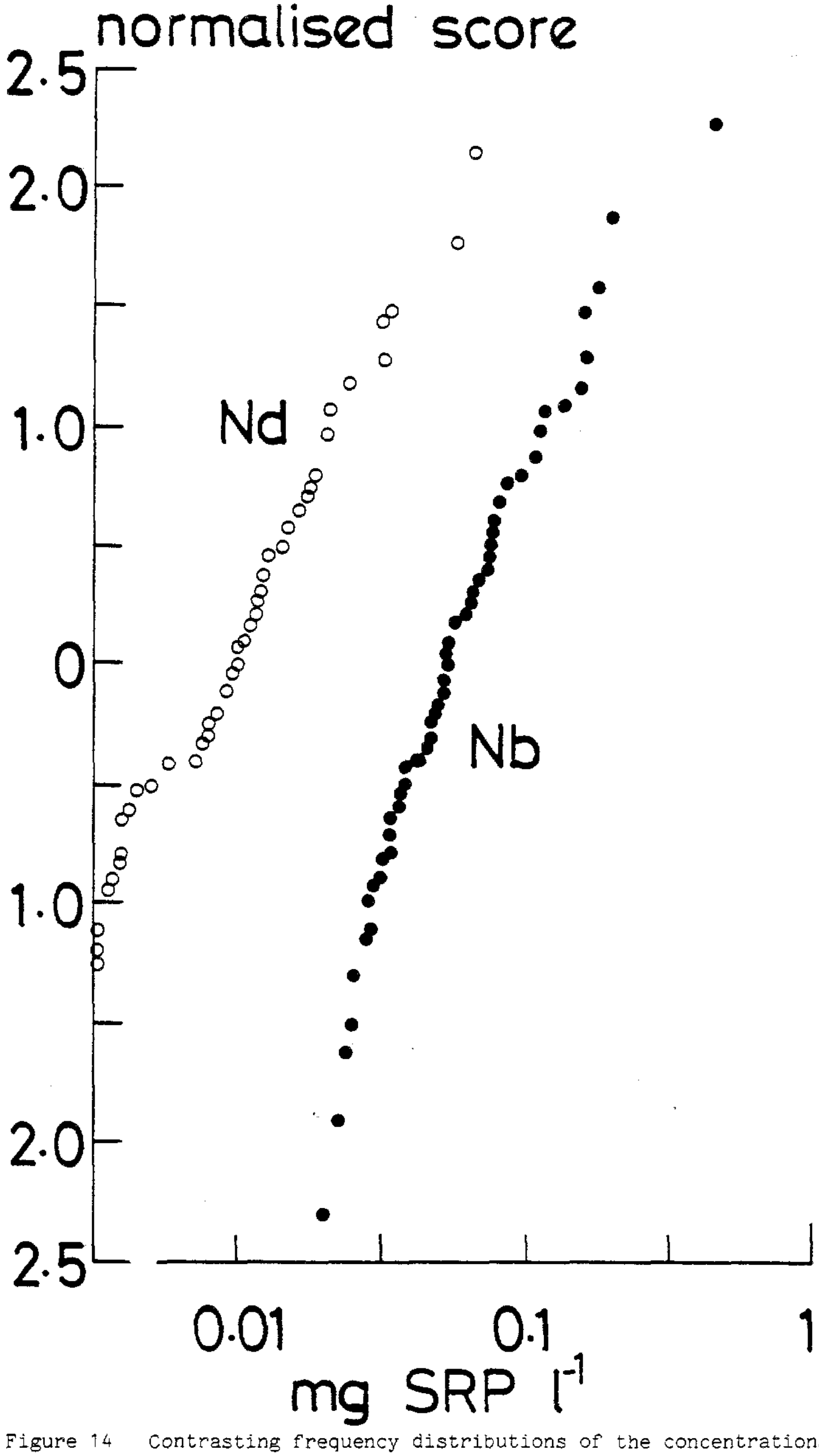


Figure 14 Contrasting frequency distributions of the concentrations of soluble reactive phosphorus (SRP) recorded at 8-day intervals on the North Queich above (Nd) and below (Nb) the input of Milnathort STW effluent and adjoining tributaries.

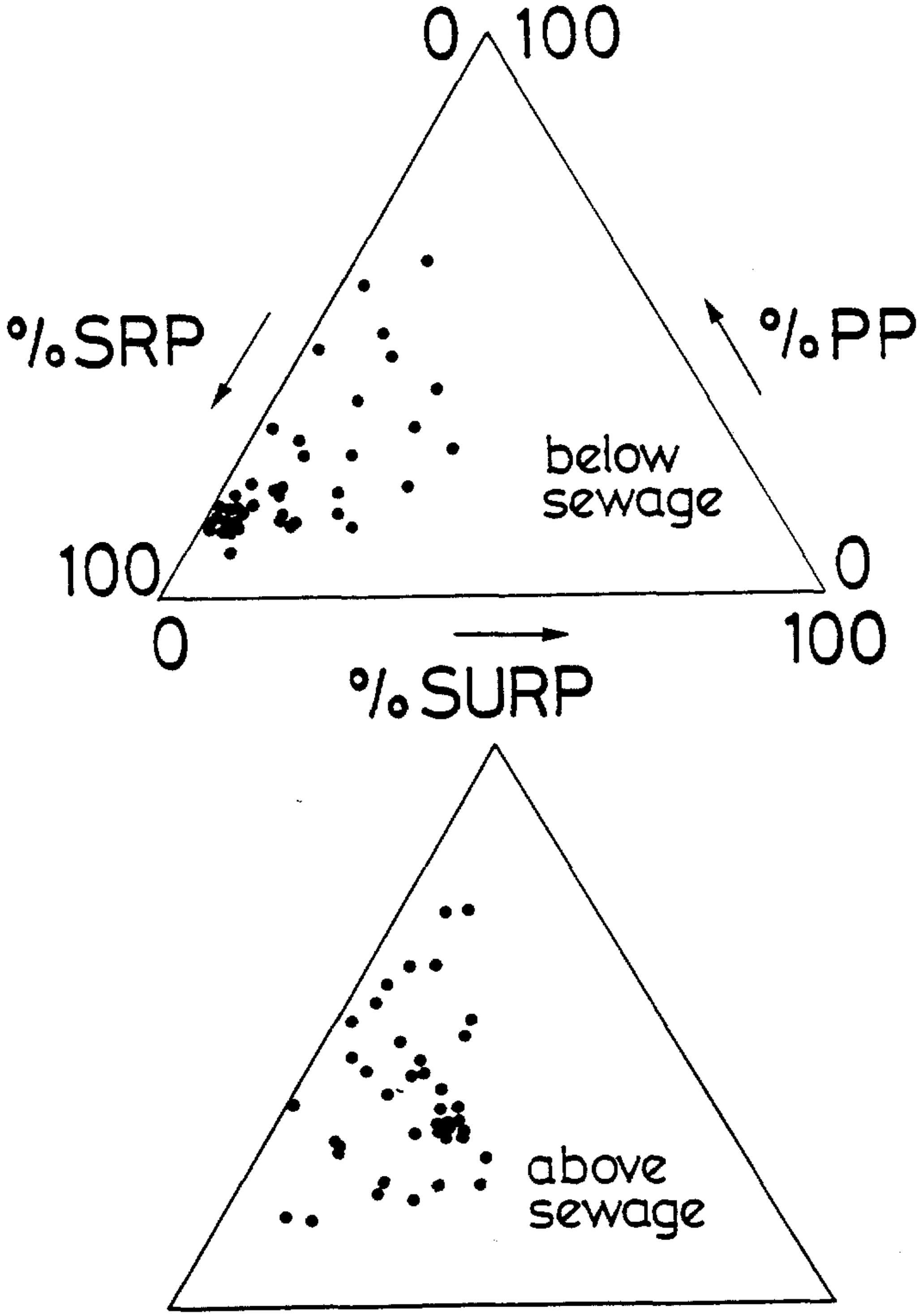


Figure 15(a) Triangular plots (see Figure 10 for explanation) illustrating shifts in the proportions of different fractions of P (expressed as concentrations) in the North Queich, between reaches above and below the input of treated effluent from Milnathort STW.

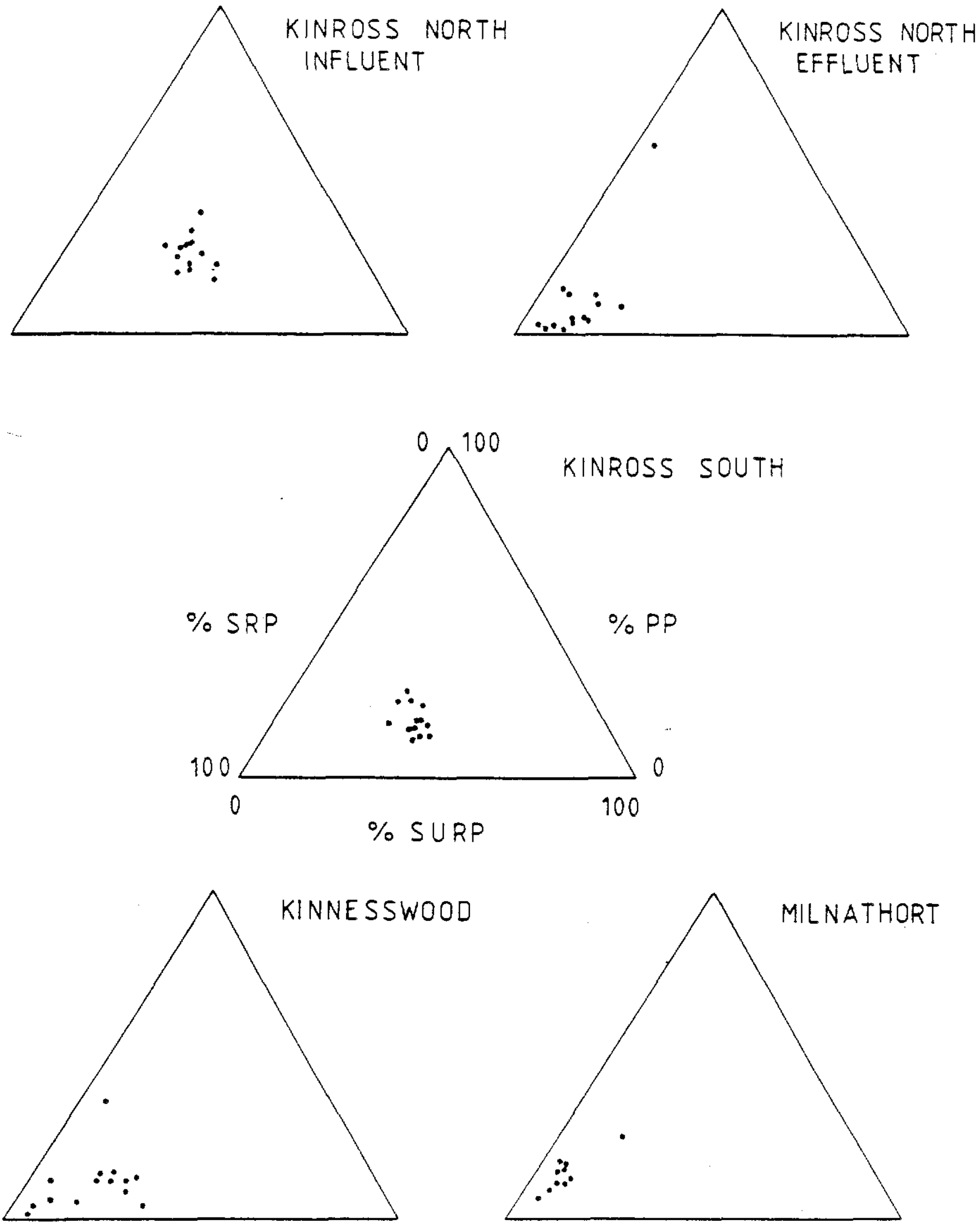


Figure 15(b) As Figure 15(a) for different fractions of P (expressed as mean daily loadings) in influent and effluent waters of different STWs.

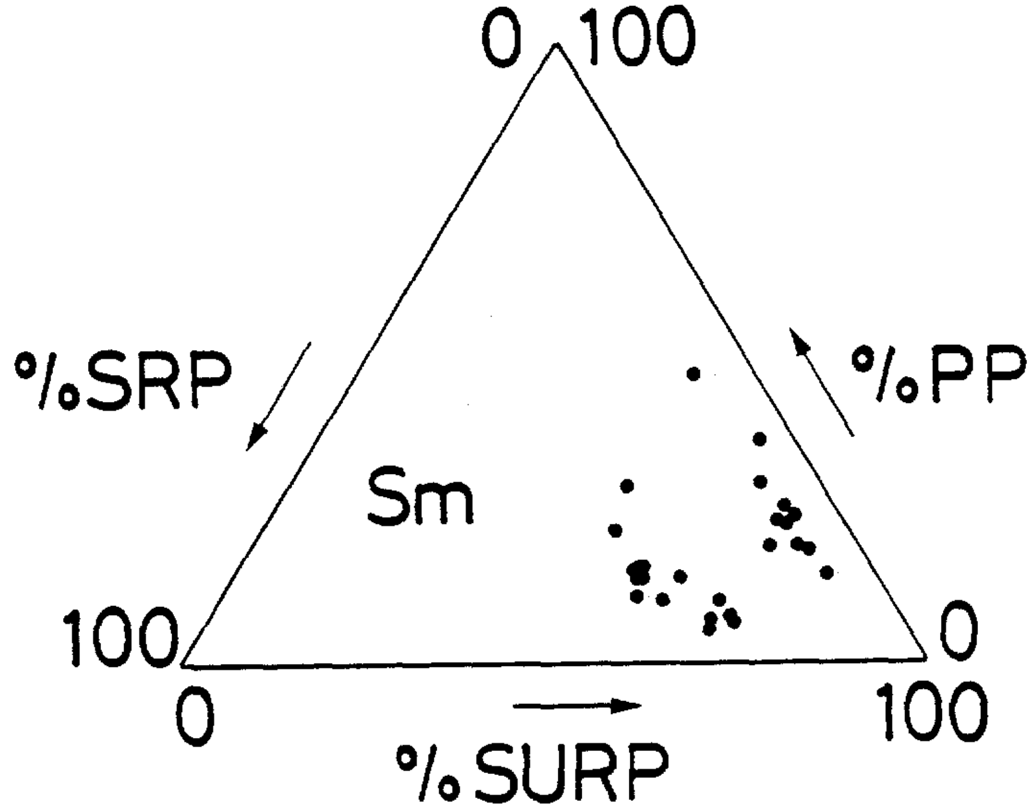


Figure 16 Triangular plot illustrating the high % SURP/TP in industrial effluent: compare the distribution of points in this Figure with that of eg Figure 15.

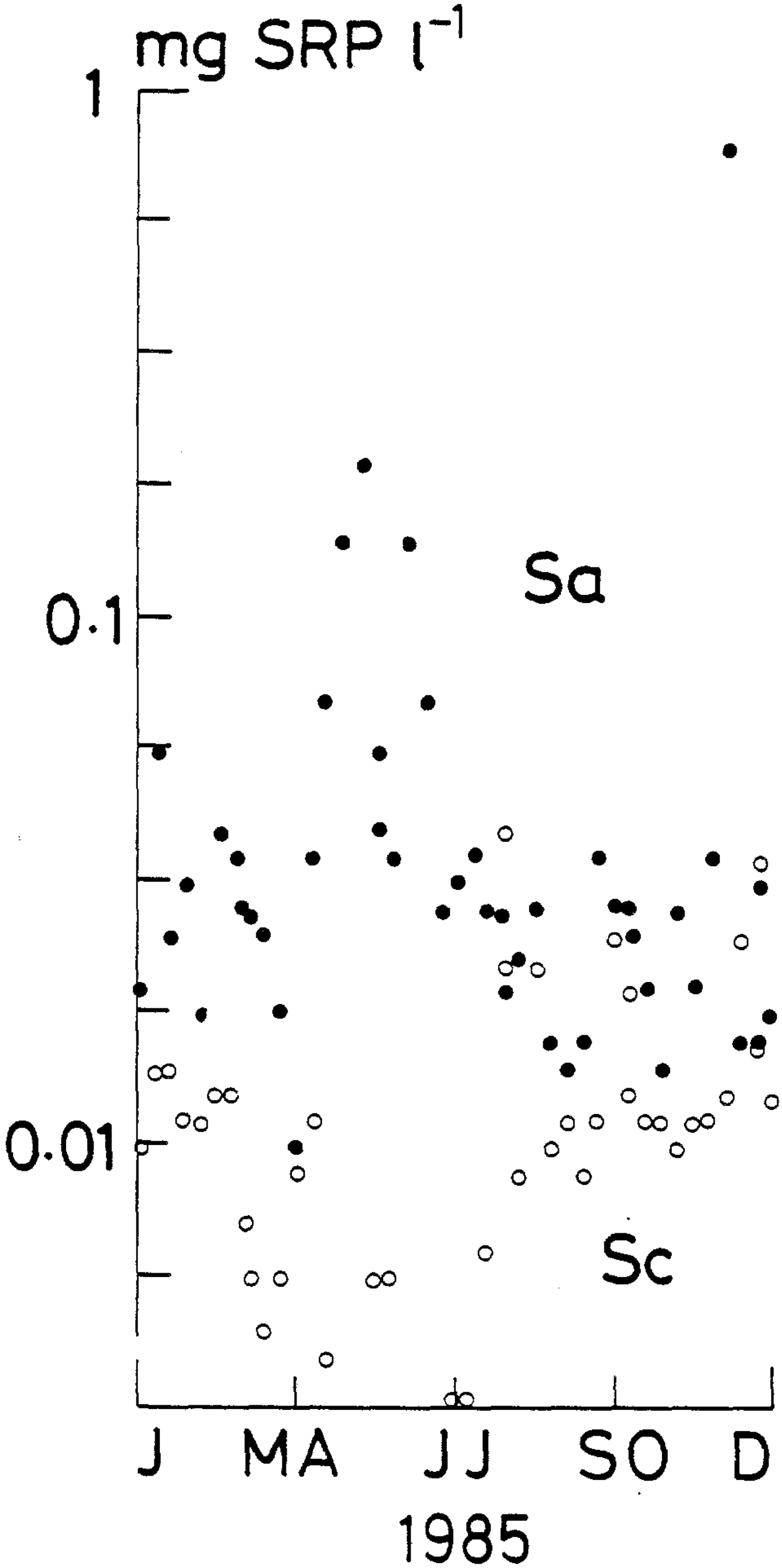


Figure 17 The influence of the outfall of industrial effluent on soluble reactive phosphorus (SRP) concentrations of the South Queich: sites immediately above (Sc) and below (Sa) input compared.

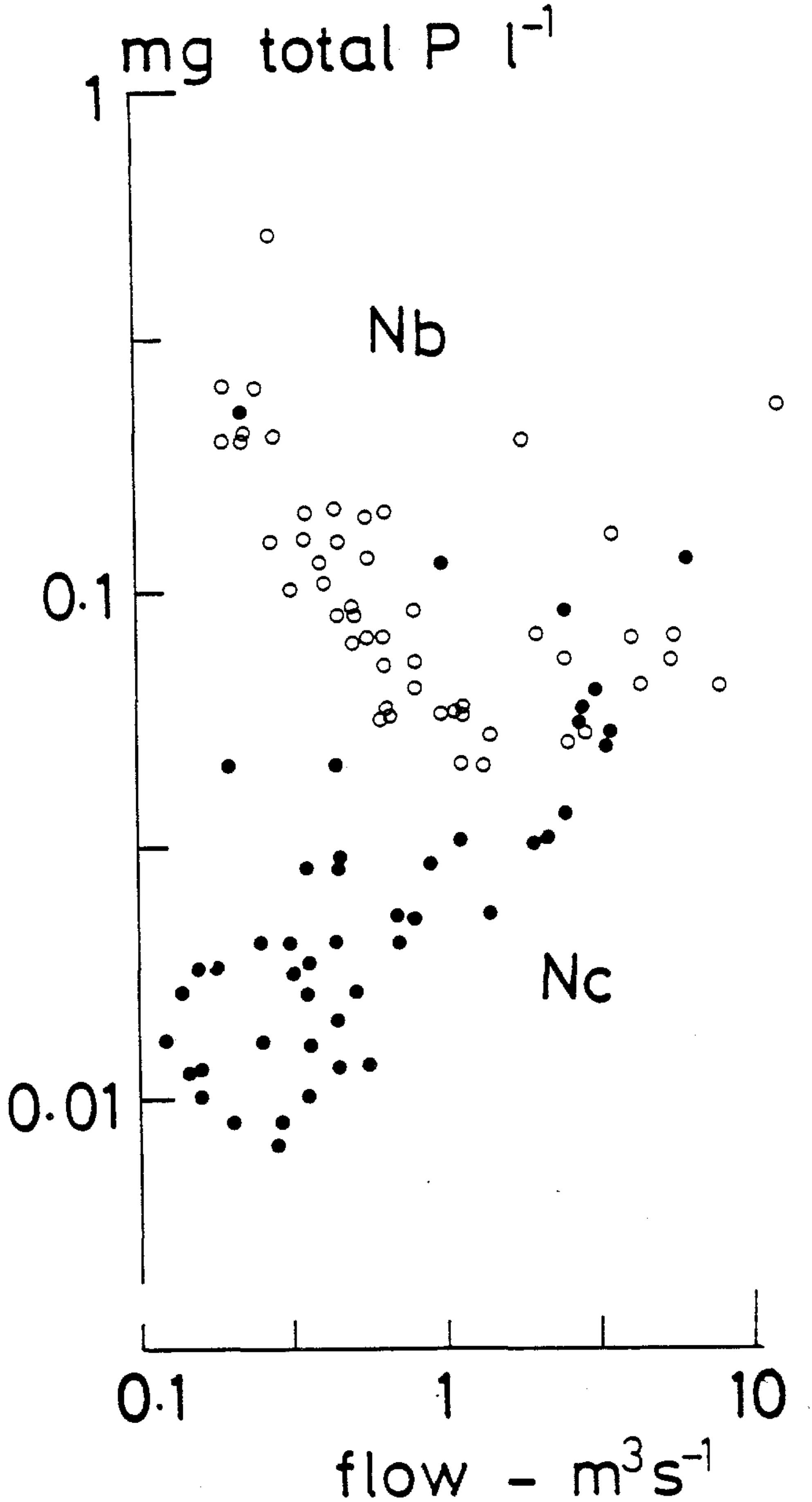


Figure 18 Contrasts in the relationship between phosphorus concentration and flow above (Nc) and below (Nb) the input of effluent from Milnathort STW on the North Queich.

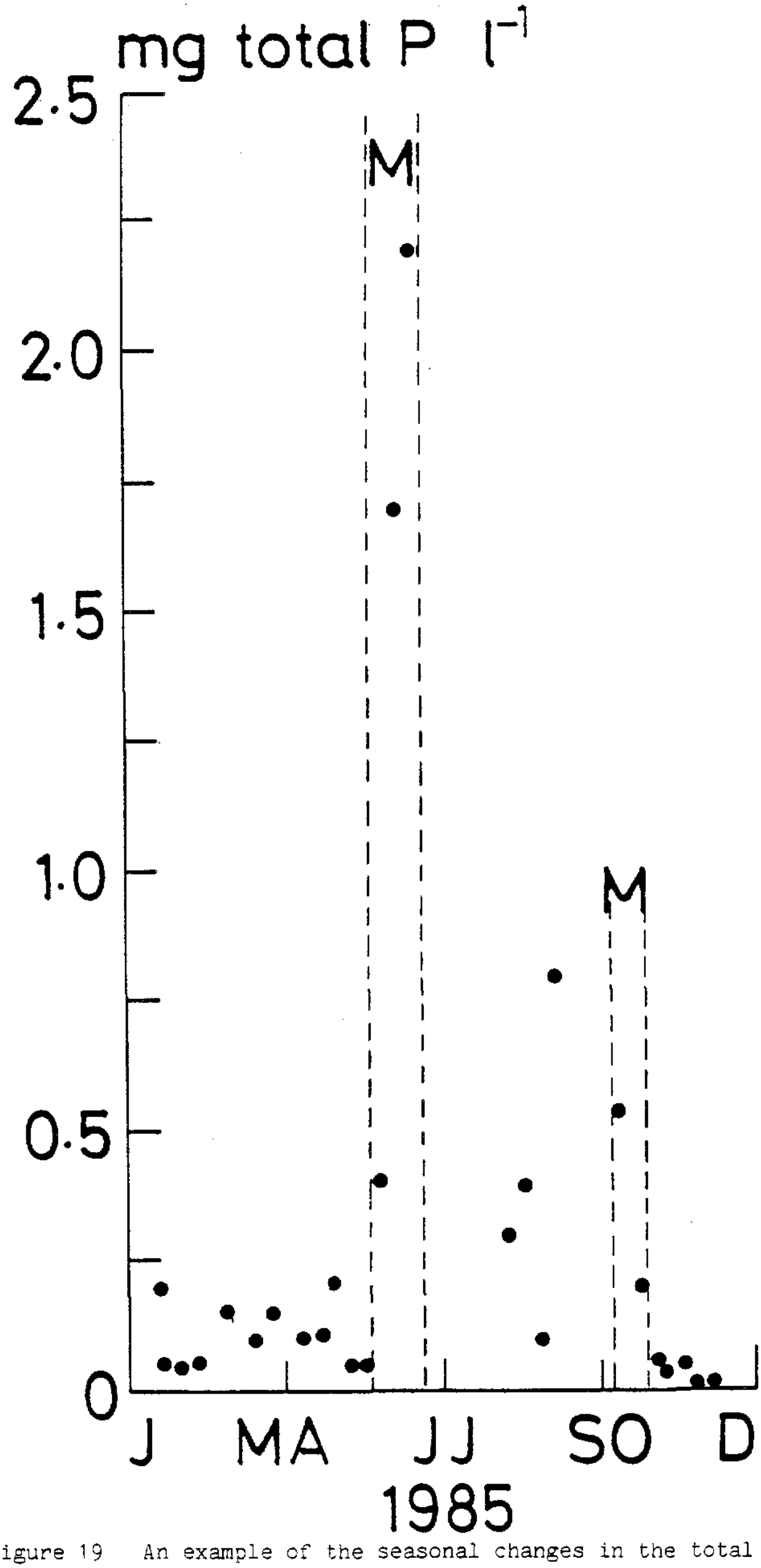


Figure 19 An example of the seasonal changes in the total phosphorus concentration of rain water collected as described in the text; periods during which midges (Chironomidae) contaminated the samples, are indicated by 'M'.

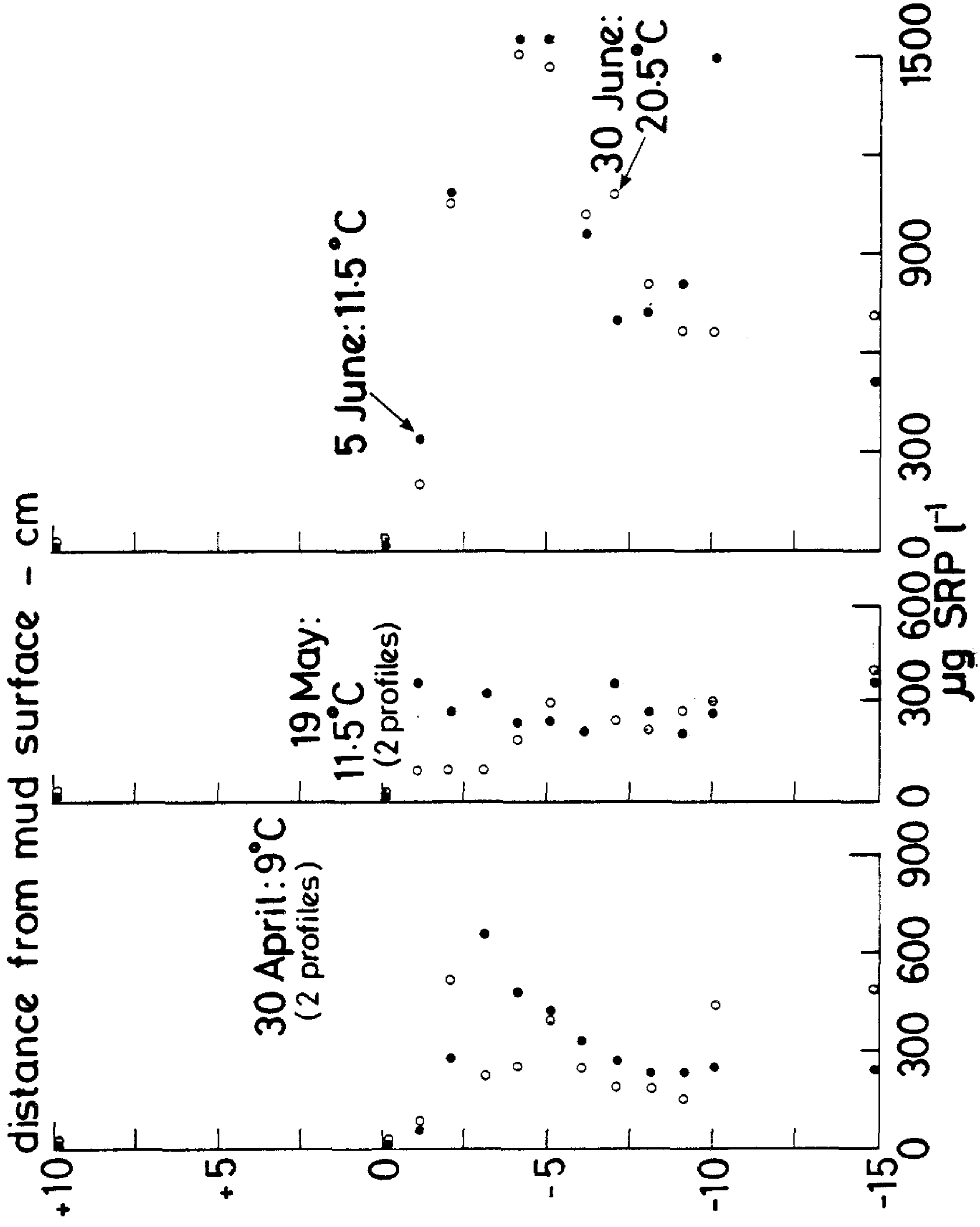


Figure 20 Vertical profiles of soluble reactive phosphorus concentration in mud interstitial water and overlying loch water on various dates April to June 1986; prevailing temperatures are also shown.

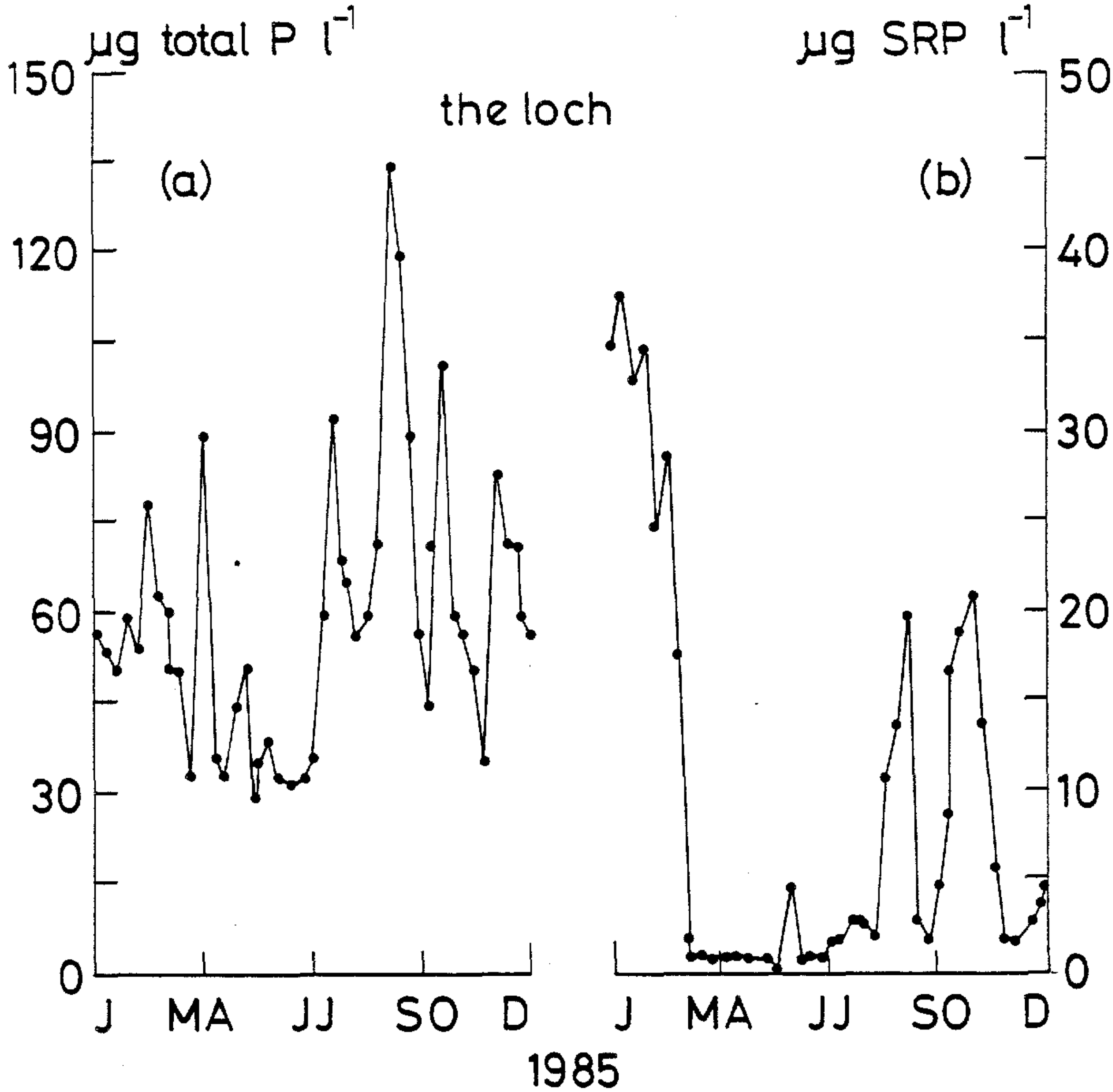


Figure 21 Seasonal changes in 1985, of the concentrations of total phosphorus (a) and soluble reactive phosphorus (b) in Loch Leven at site L near the outflow.

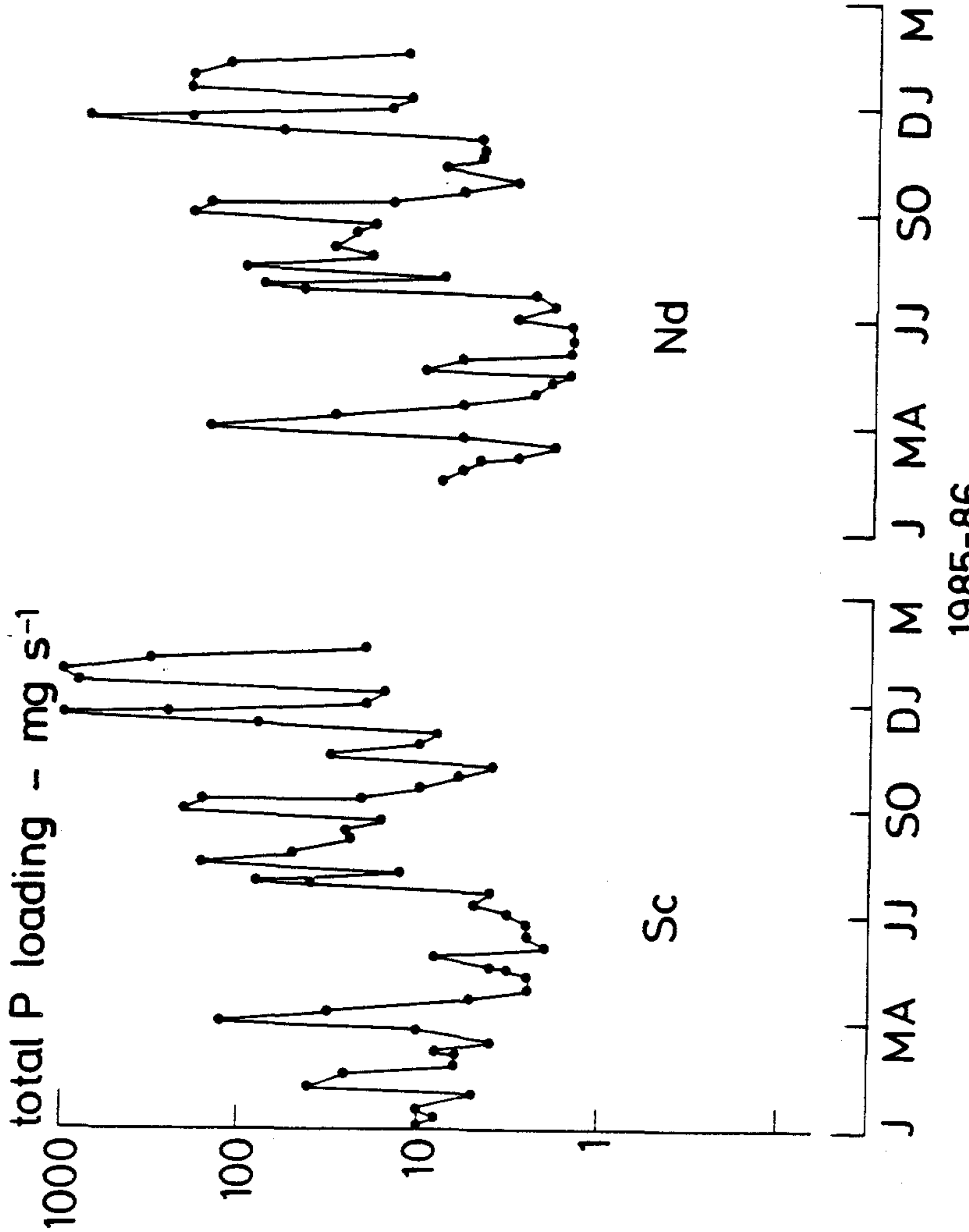


Figure 22(a) Seasonal variation in total phosphorus loading at sites above point-source inputs on 2 large rivers: based on instantaneous loadings from sampling at 8-day intervals.

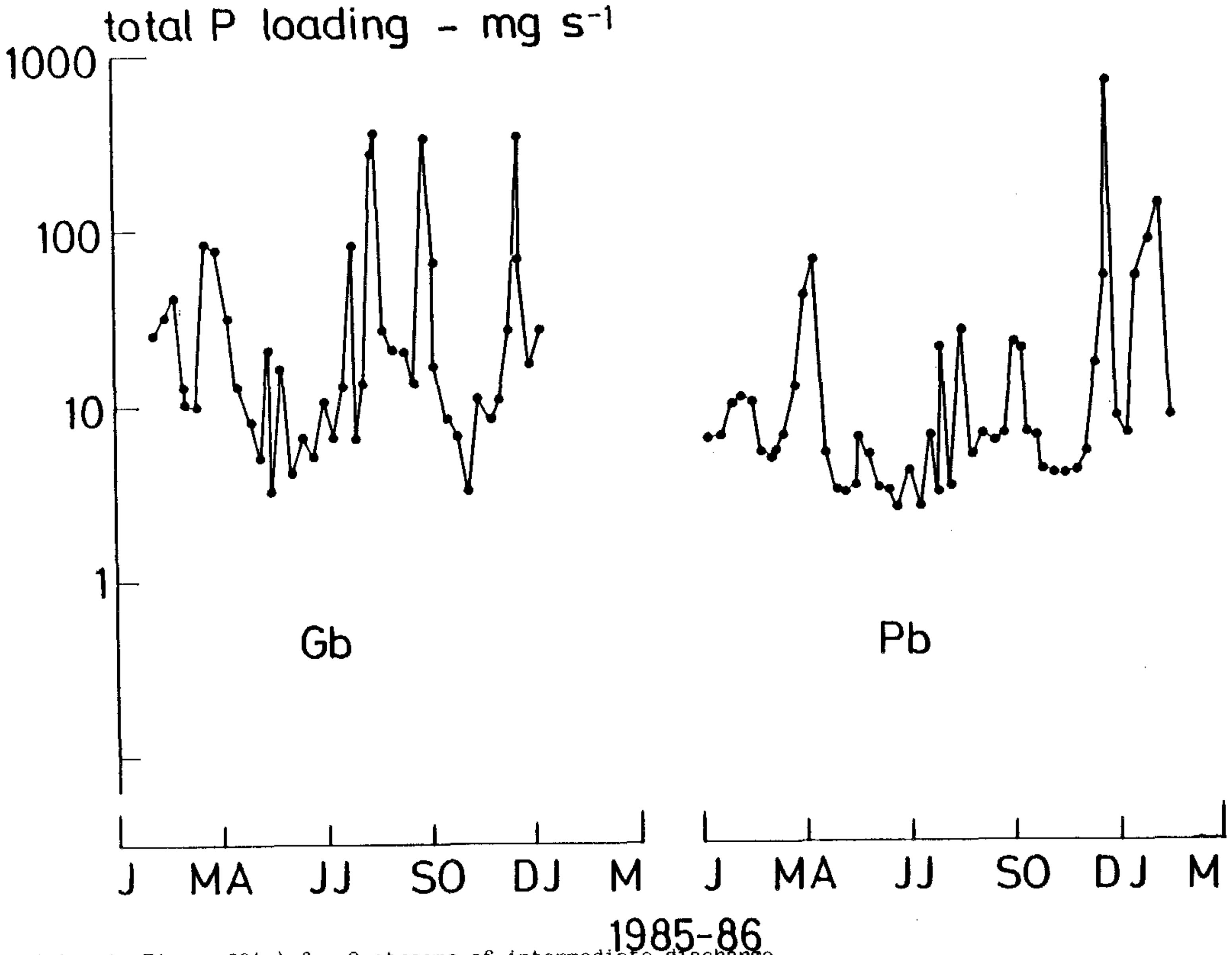


Figure 22(b) As Figure 22(a) for 2 streams of intermediate discharge.

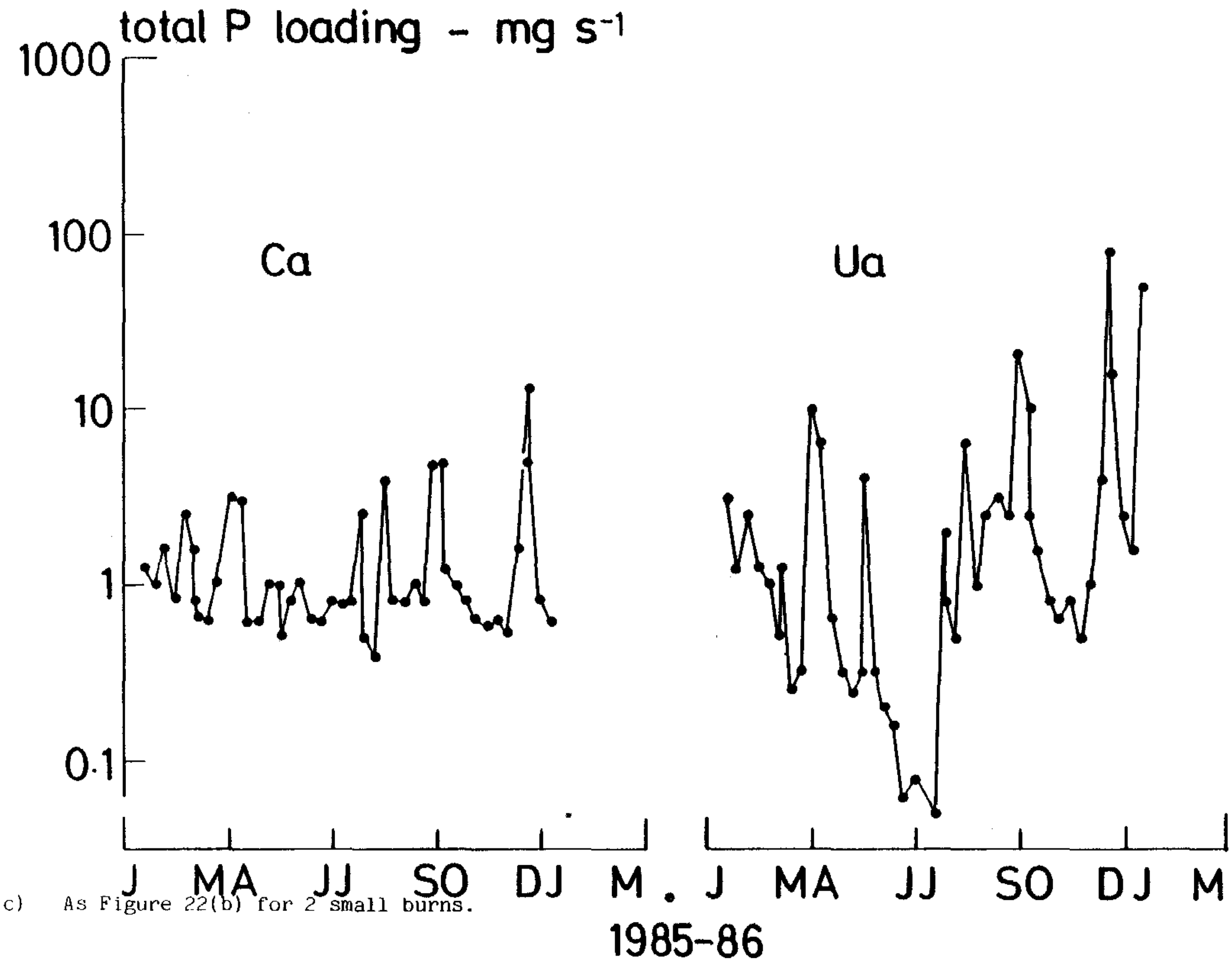


Figure 22(c) As Figure 22(b) for 2 small burns.

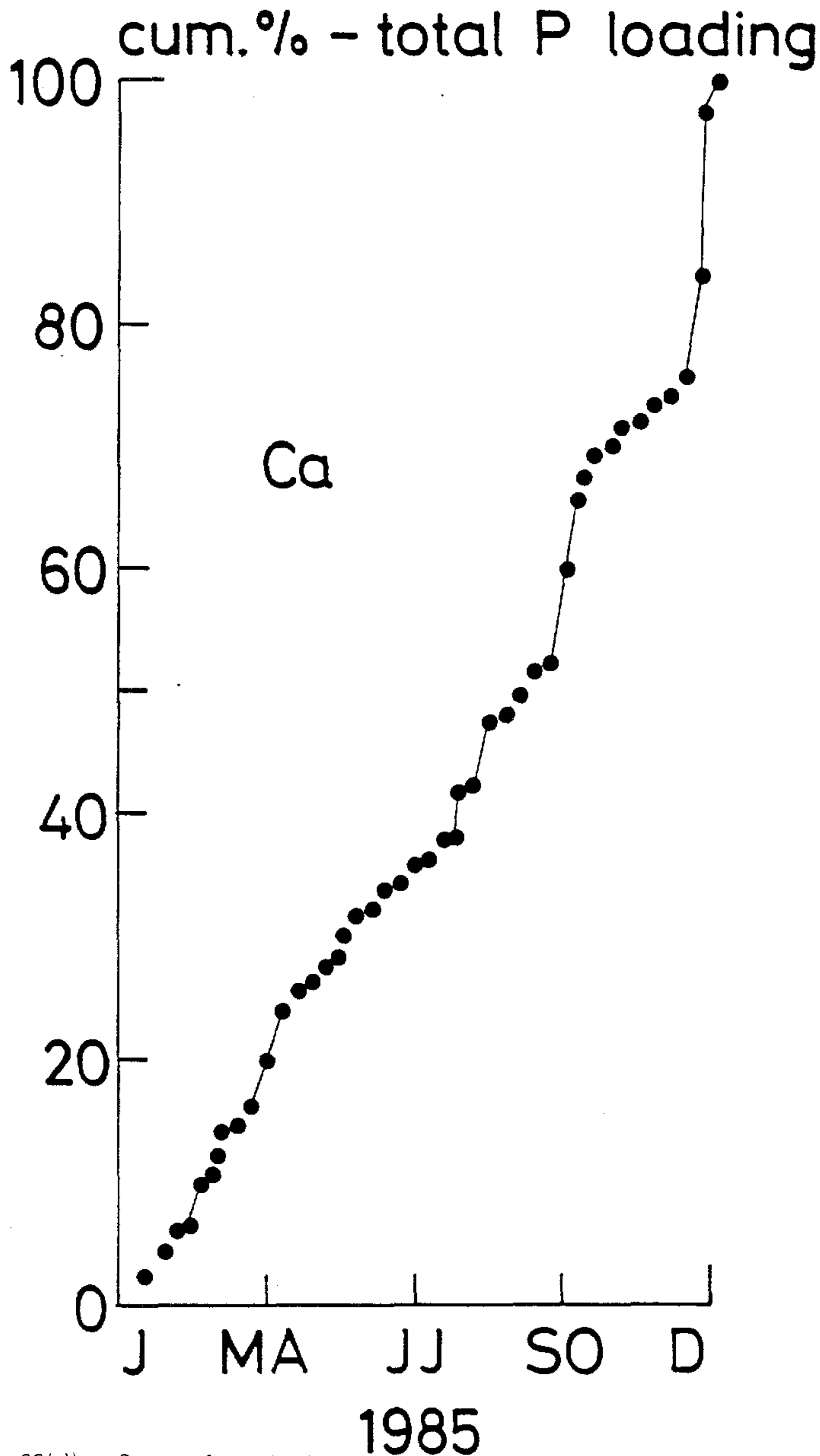


Figure 22(d) Seasonal variation in the annual accumulation (cum %) of total phosphorus loading from the Clash Burn to illustrate the importance of short episodes of high loads; loadings as defined in Figure 22(a).

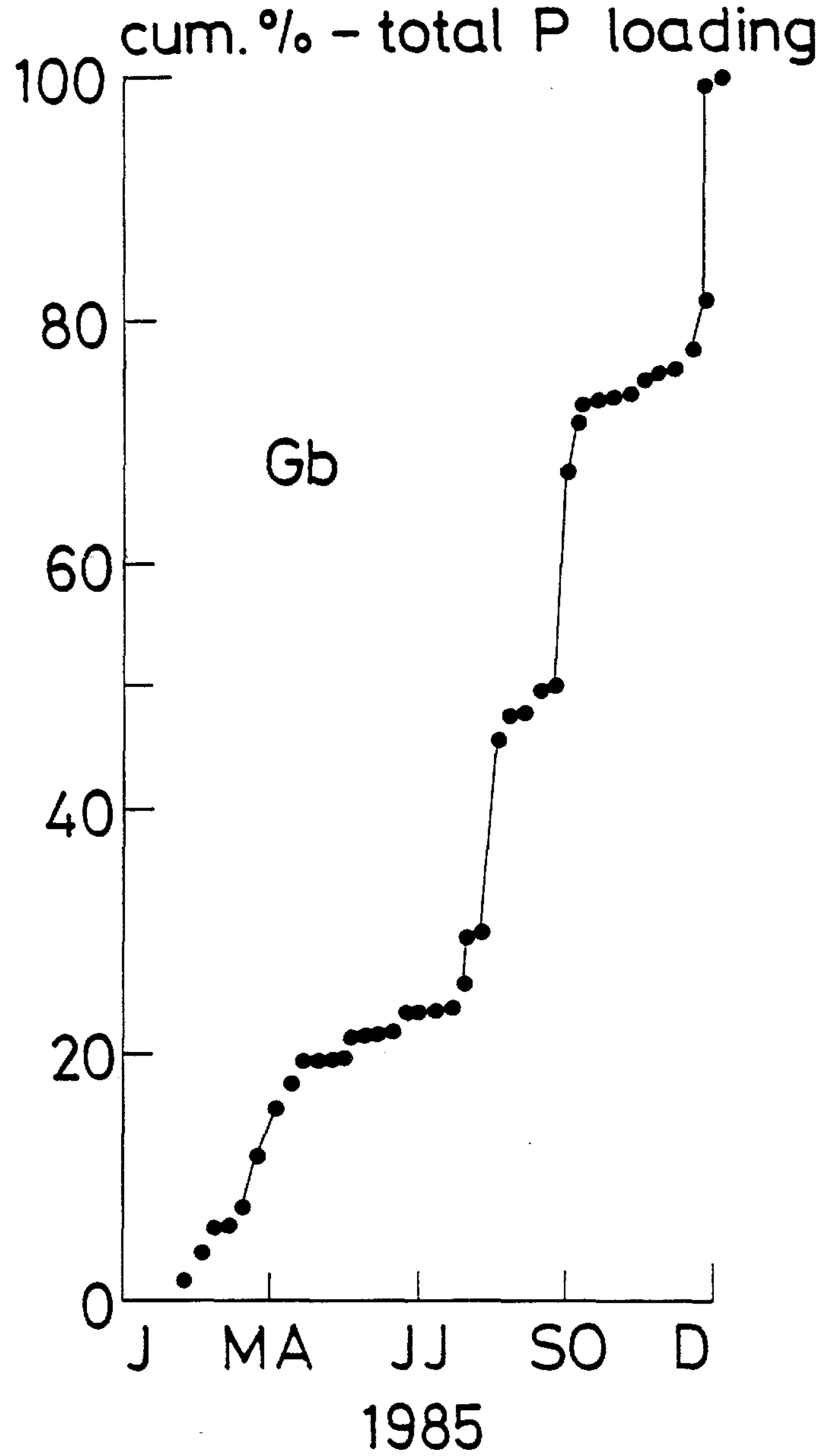


Figure 22(e) As Figure 22(d) for the Gairney Water

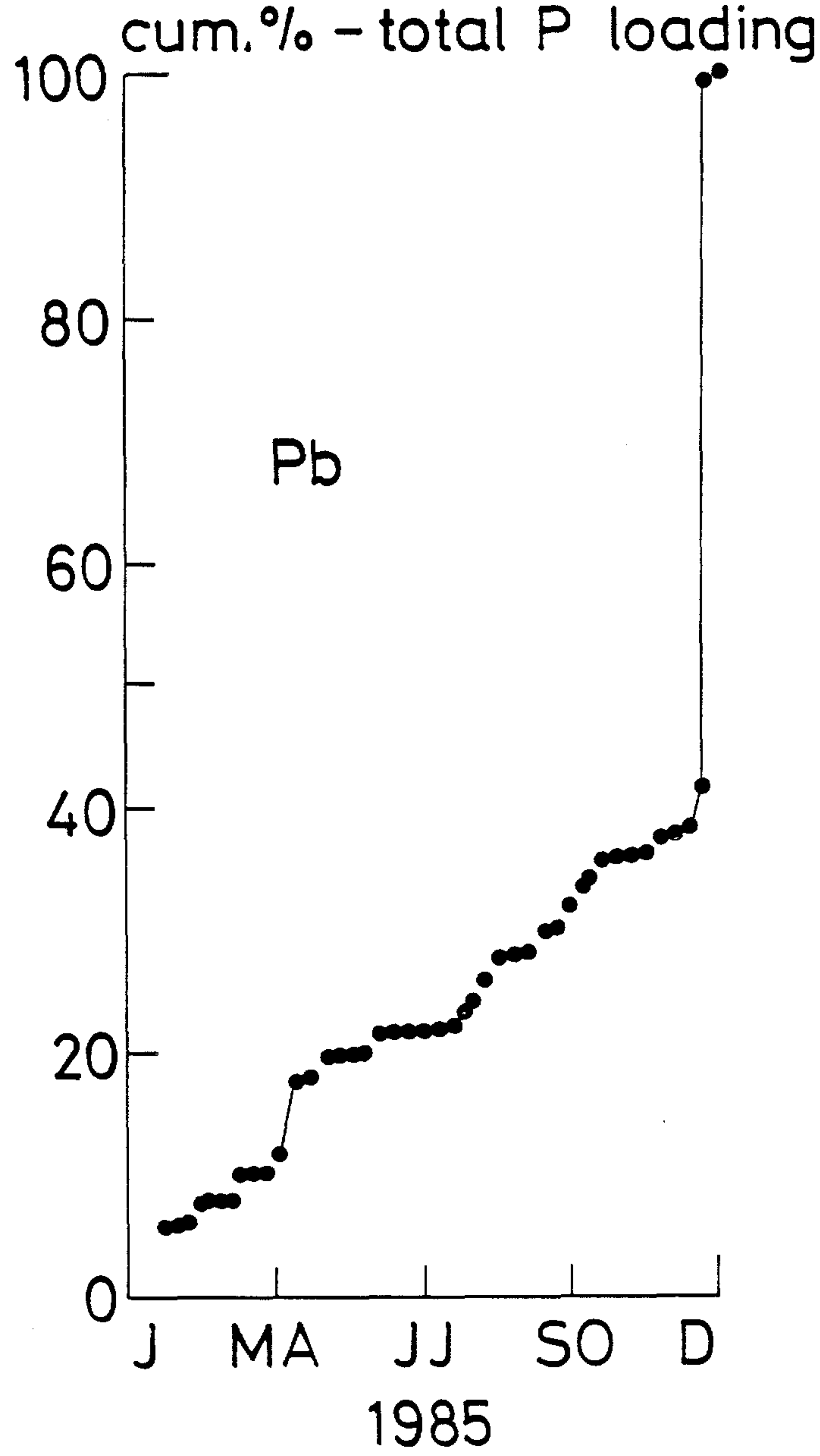


Figure 22(f) As Figure 22(d) for the Pow Burn.

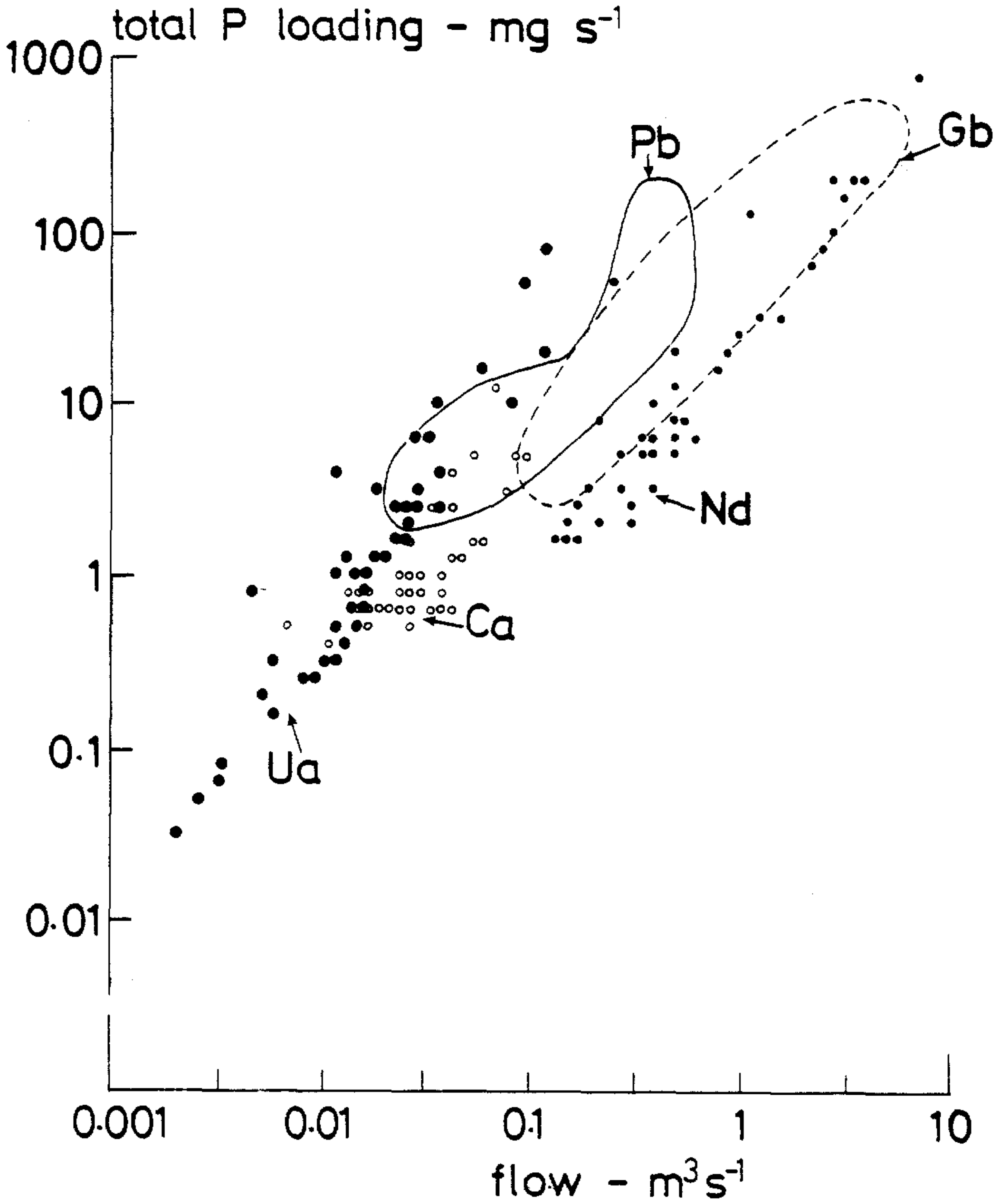
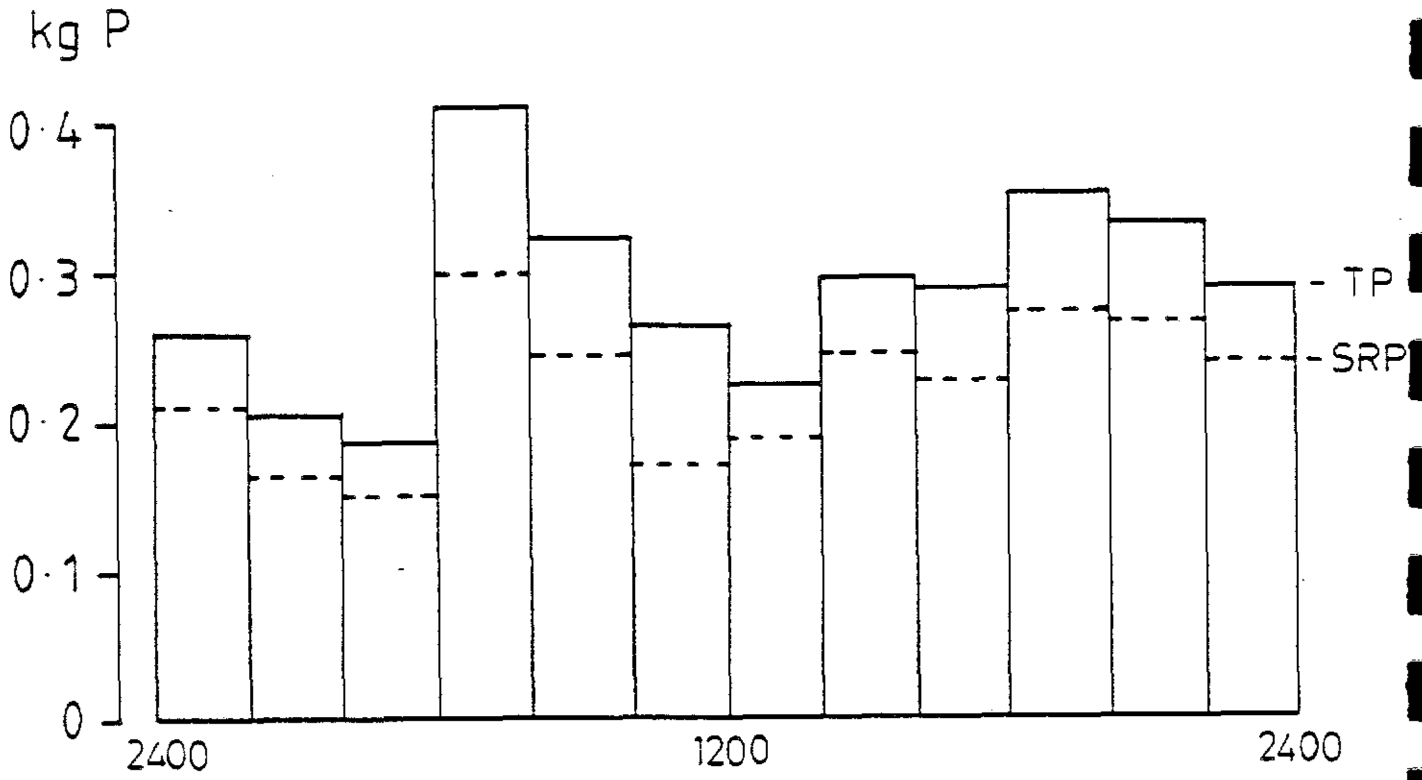


Figure 23 The relationship between total phosphorus loading and flow in 5 feeder waters of Loch Leven. Site codes as shown in previous Figures.

DIURNAL VARIATION - P LOADING

MILNATHORT S T W



KINNESSWOOD S T W

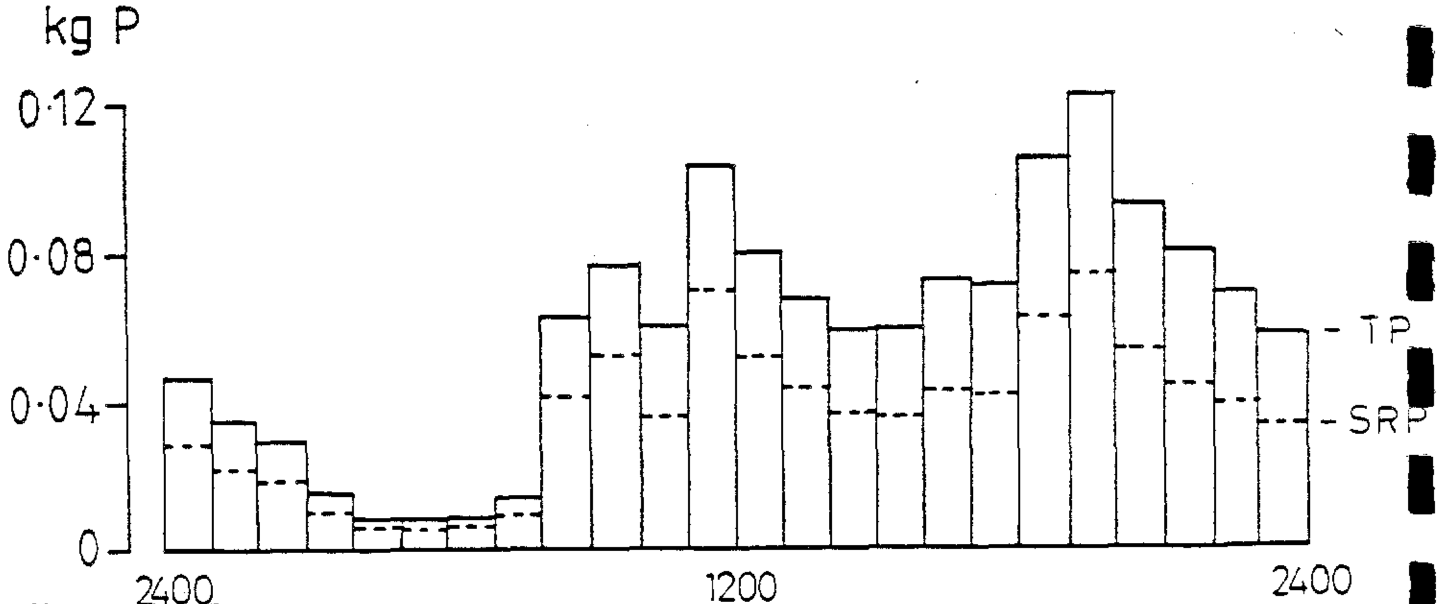


Figure 24 Bar-charts of diurnal variation in TP loading (as kg P) over 2-hour intervals at Milnathort STW (23-24 October 1986) and over 1-hour intervals at Kinnesswood STW (23-24 August 1986).

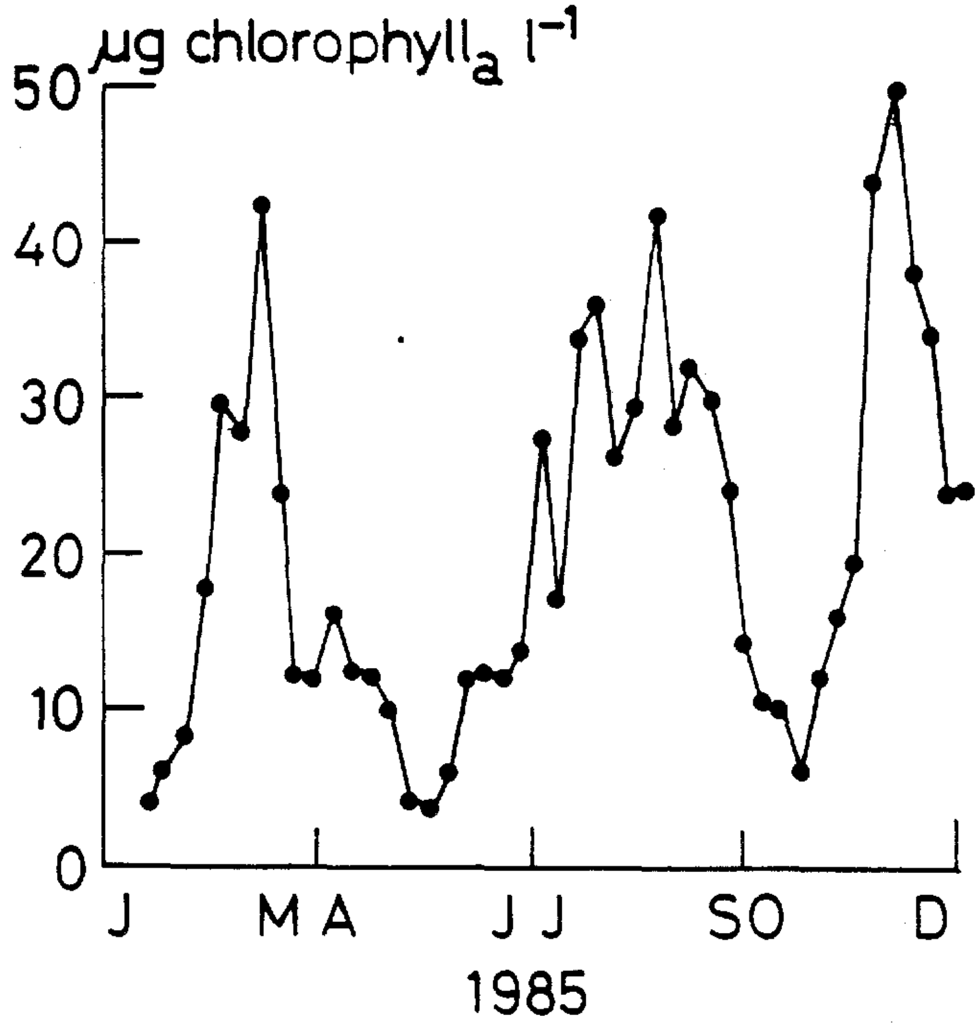
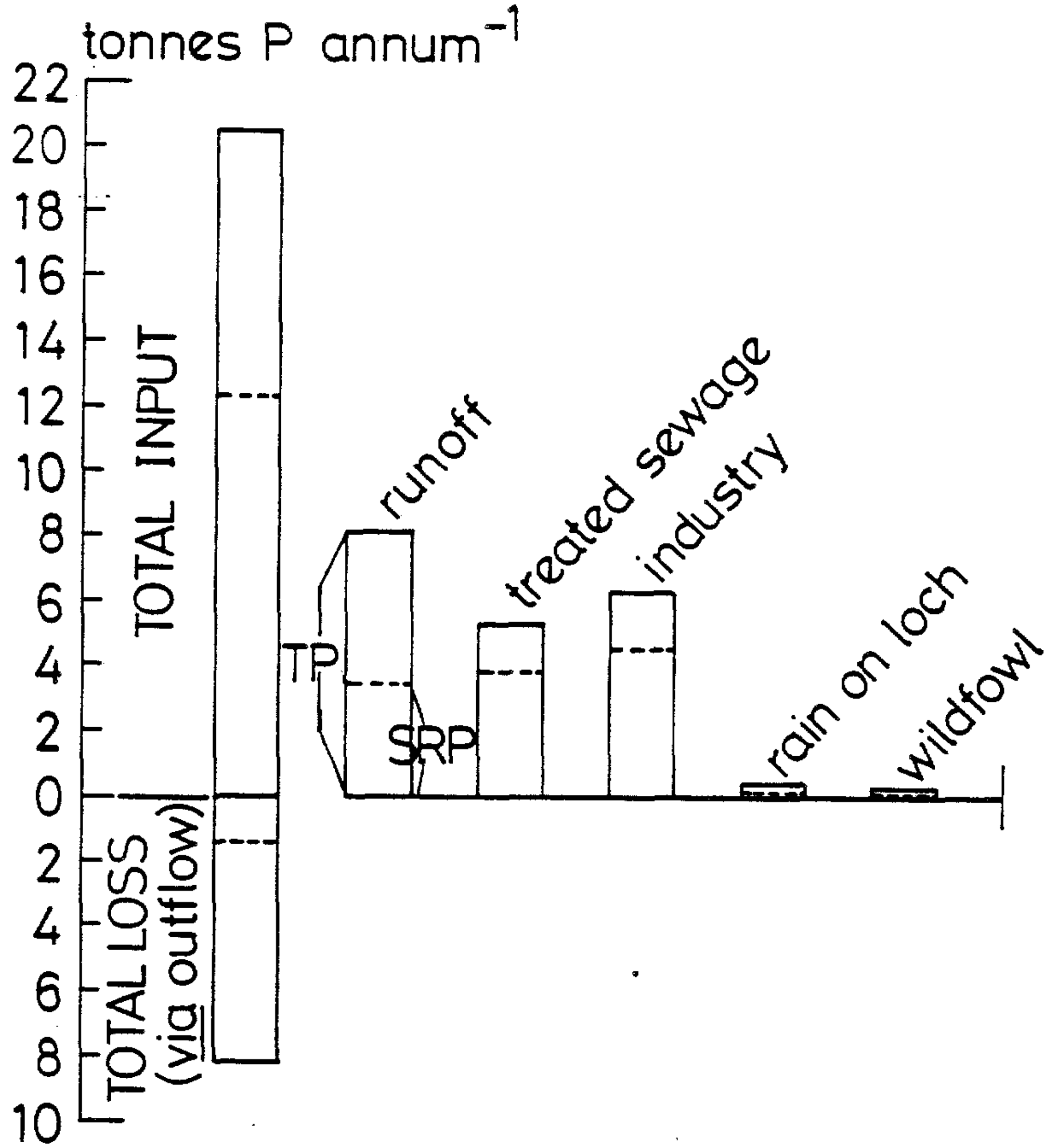
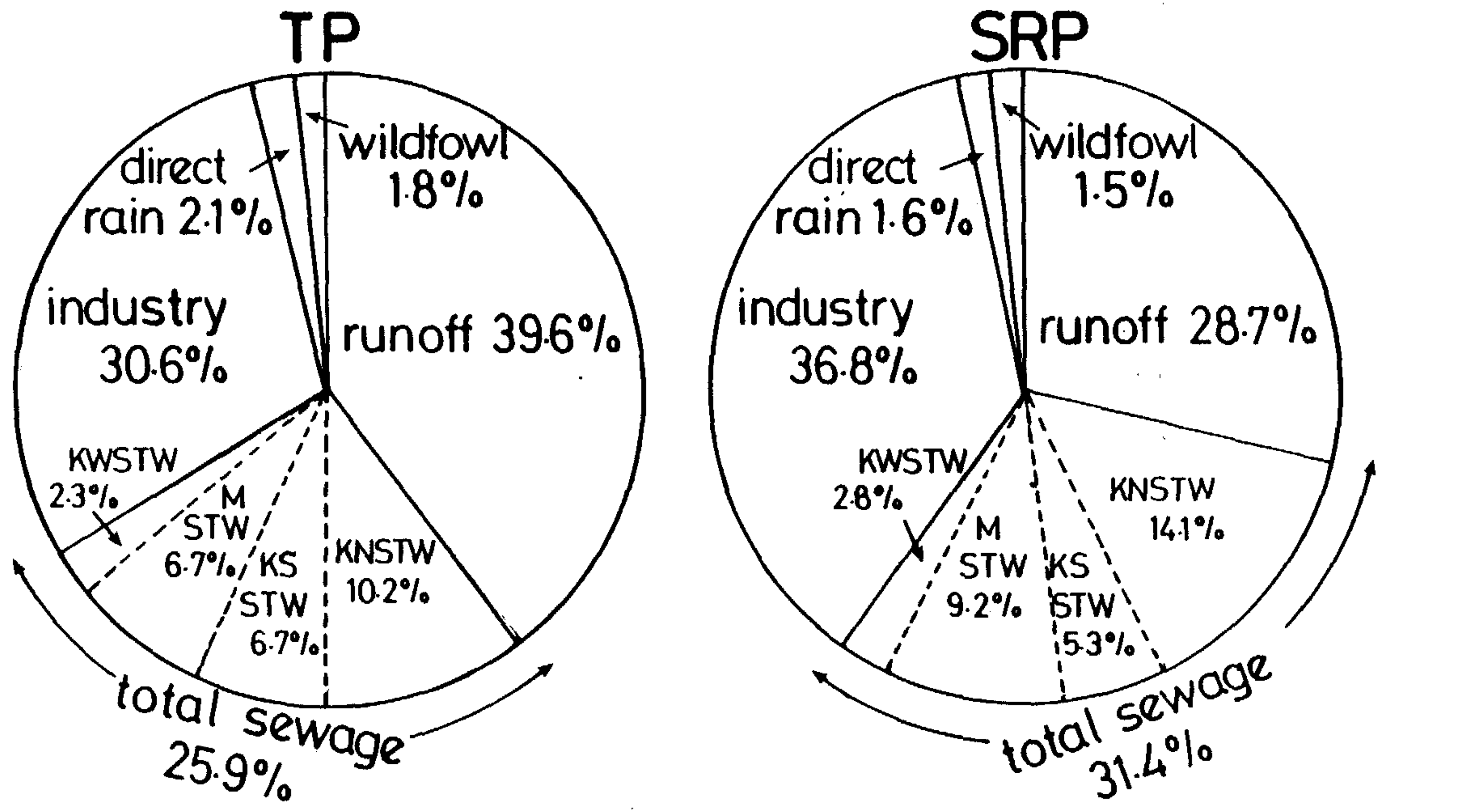


Figure 25 Fluctuations in the biomass of phytoplankton as indicated by chlorophyll a concentration in Loch Leven 1985.



External loadings of P to L. Leven 1985

Figure 26 Bar chart illustrating the annual (1985) loadings of total phosphorus (TP) and of soluble reactive phosphorus (SRP) to Loch Leven from various external sources; the annual export of the nutrient from the loch *via* its outflow is also compared to the total input.



%contributions to the total loadings of TP and SRP to L.Leven - 1985

Figure 27 Date included in Figure 26 re-plotted to illustrate the % contributions of each external source to the total annual loadings of TP and of SRP to Loch Leven (for 1985); for the input in treated sewage effluent, the separate loadings from each of the 4 sewage treatment works are also shown.

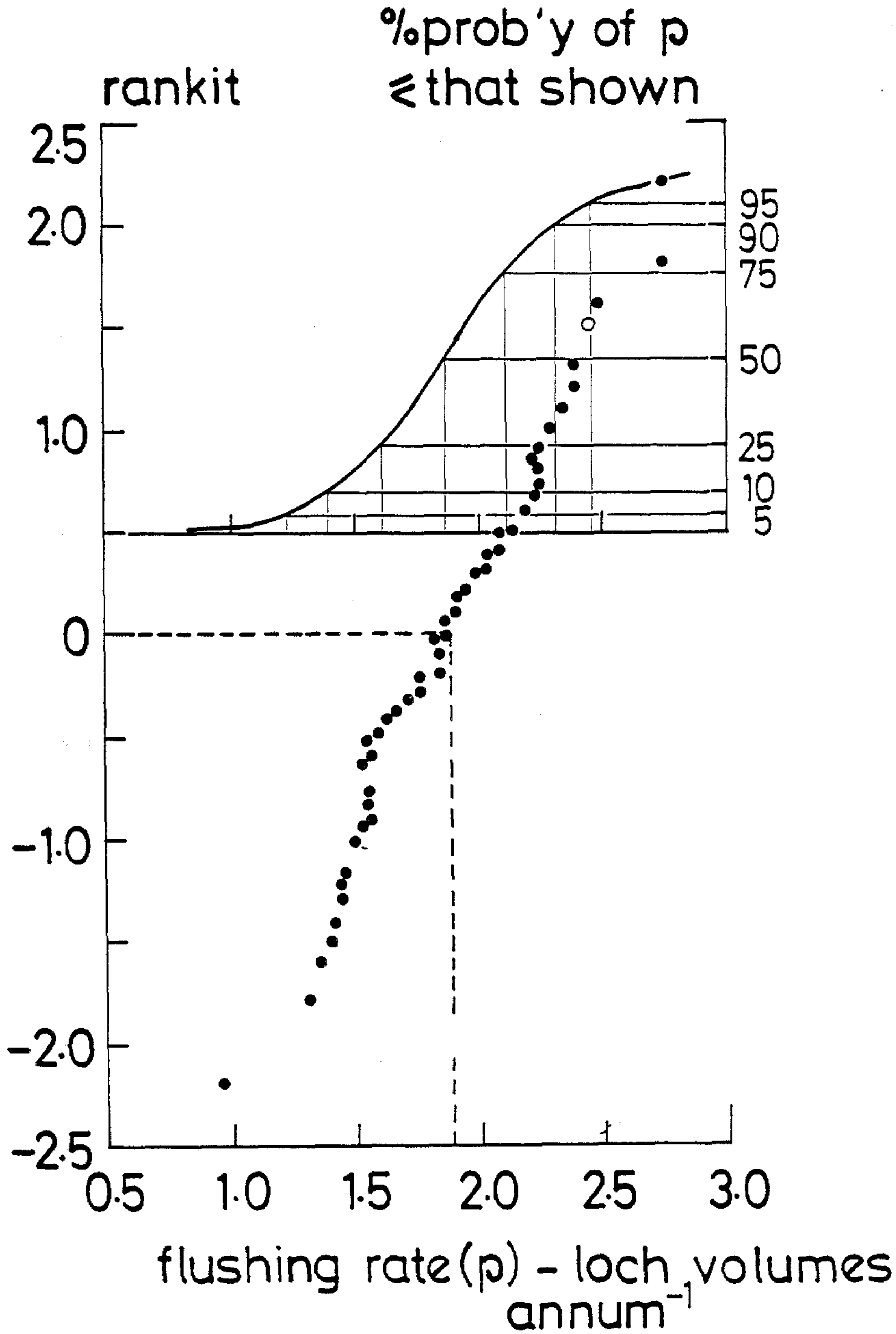


Figure 28 Main panel: 49 years' data on annual flushing rates at Loch Leven plotted against rankits: the approximation to a straight line indicates a normal frequency distribution. Upper panel: cumulative frequency distribution showing the % probability of occurrence of different flushing rates. See also Figures 30-35.

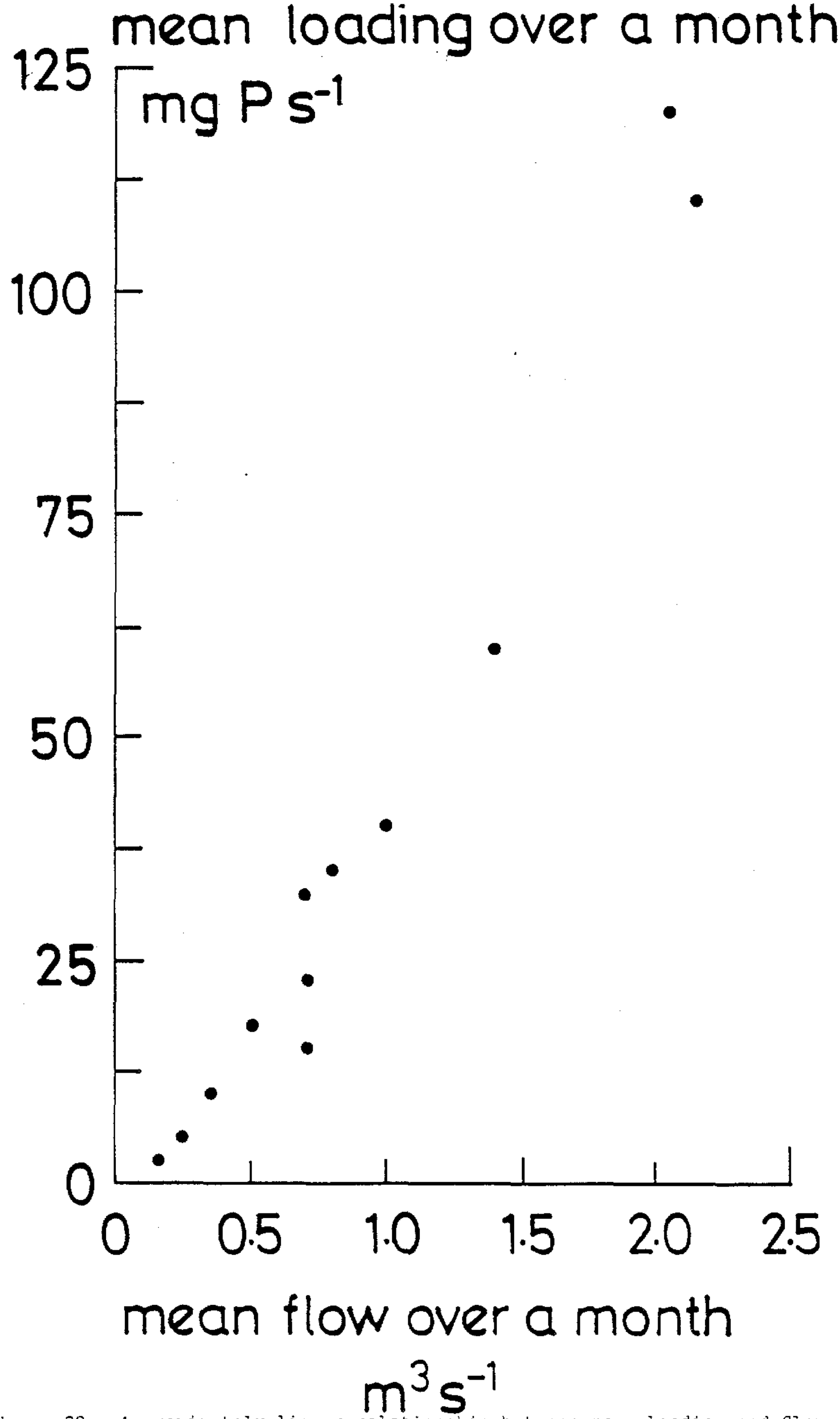


Figure 29 Approximately linear relationship between mean loading and flow rates for monthly periods; calculated from continuous (daily) data for site Nd.

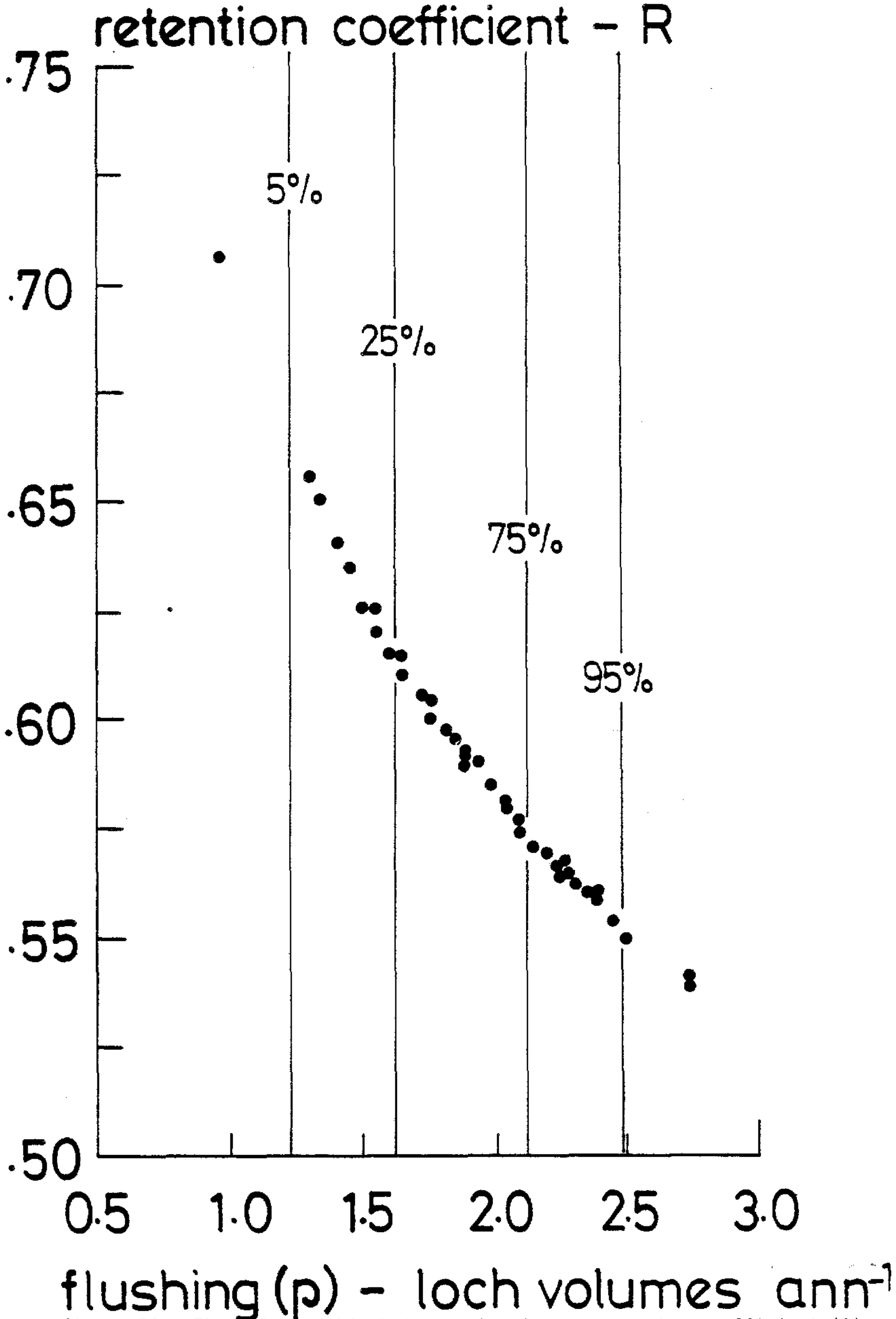


Figure 30 The relationship between phosphorus retention coefficient (R) and annual flushing rate (p) according to the Dillon and Rigler model, and using the spectrum of p values referred to in Figure 28; vertical lines indicate % chances of occurrence of p values equal to, or less than those shown.

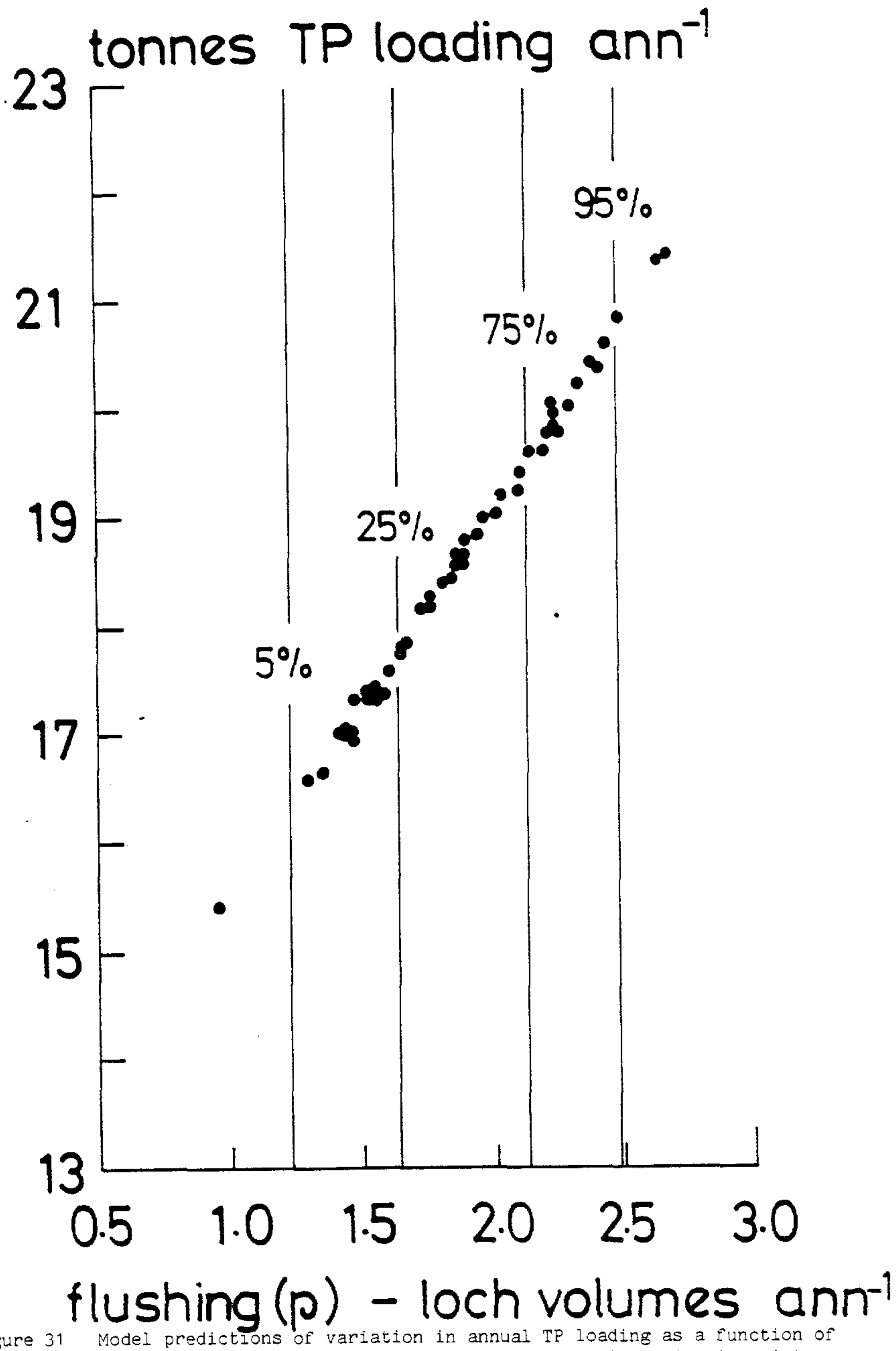


Figure 31 Model predictions of variation in annual TP loading as a function of flushing rate (p) - the situation assuming no alteration in point-source and wildfowl inputs of P; the % chances of occurrence of flushing rates equal to, or less than those shown are also indicated.

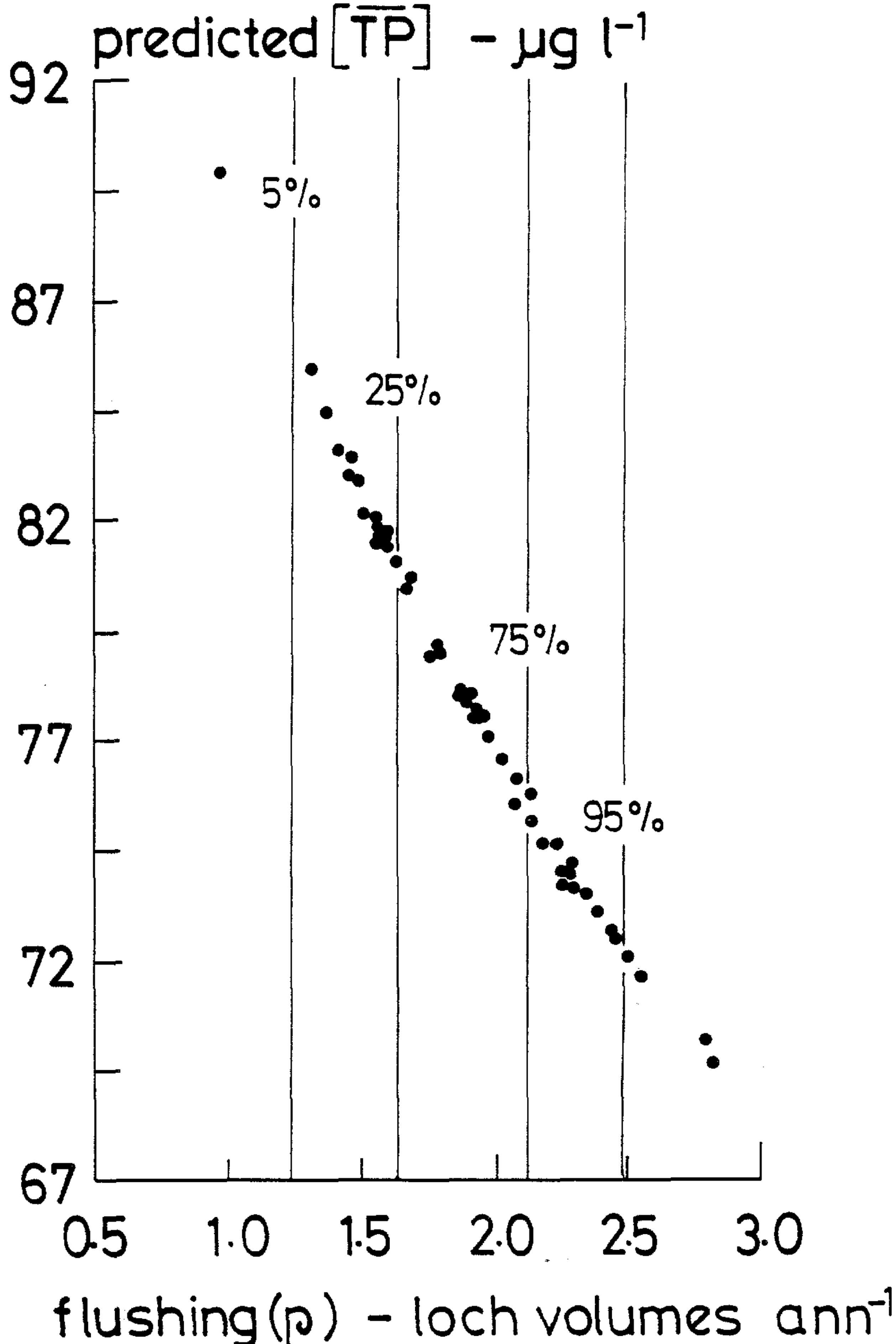


Figure 32 As Figure 31 for predicted TP concentrations in the loch.

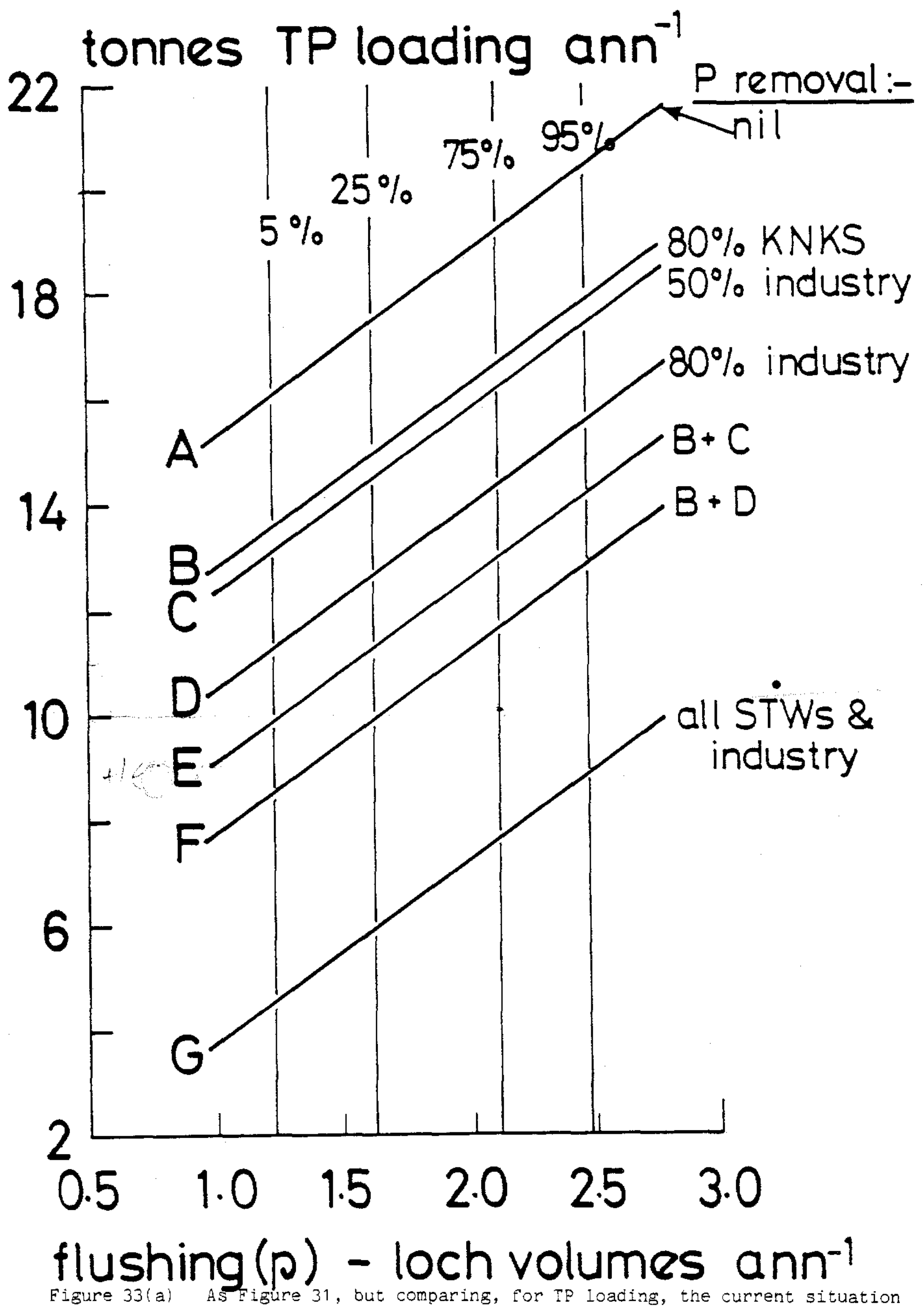


Figure 33(a) As Figure 31, but comparing, for TP loading, the current situation (with no alteration in point-source and wildfowl inputs of P) with predictions of the effects of 6 alternative measures of reducing P supplies to the loch; KNKS, *Kinross North and South STWs*.

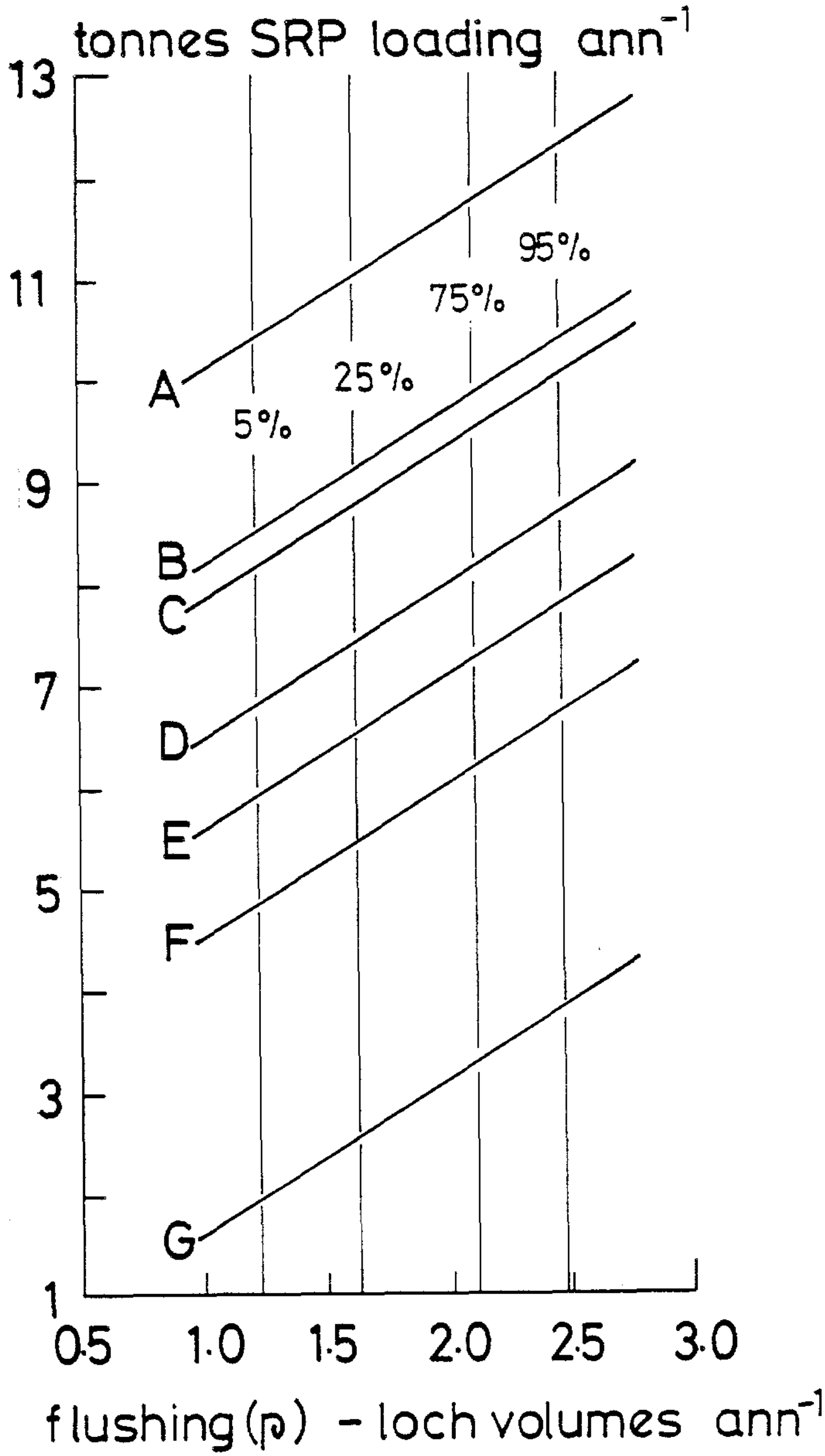


Figure 33(b) As Figure 33(a) for SRP loading.

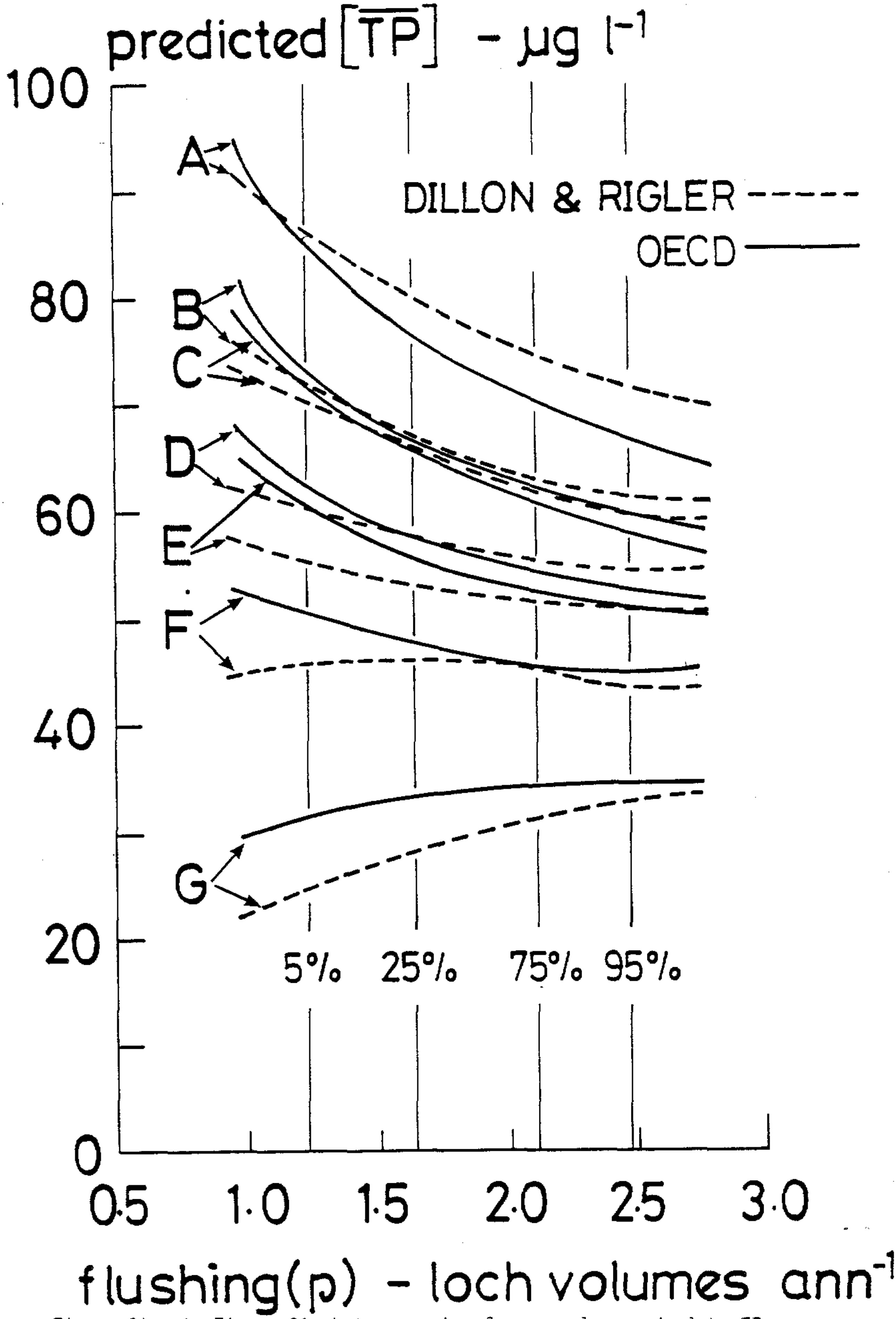


Figure 34 As Figure 31, but comparing for annual mean in-lake TP concentration, the predictions, by 2 models, of the effects of the 6 alternative measures of reducing P supplies to the loch as defined in Figure 33(a).

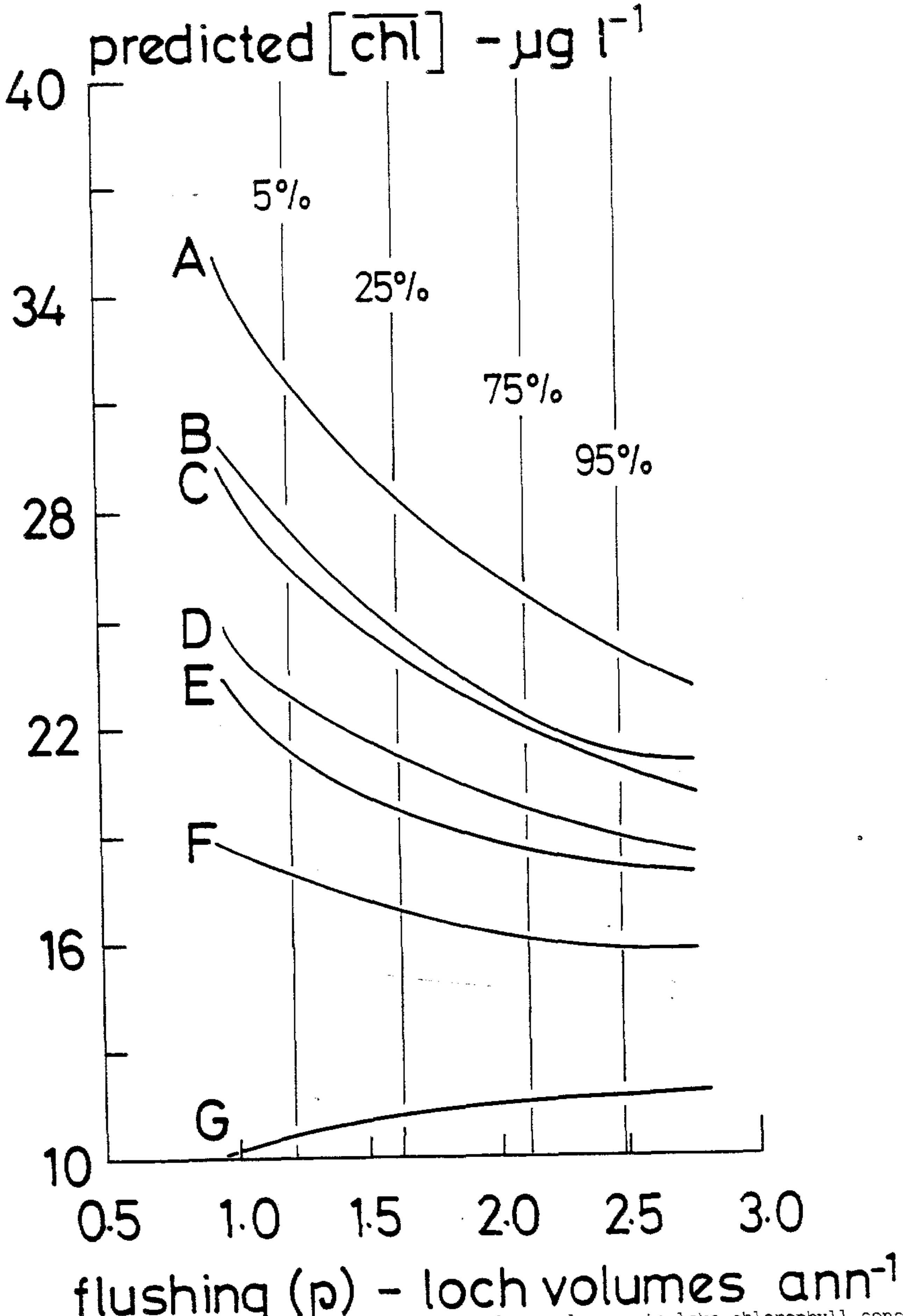


Figure 35 As Figure 34 for predicted annual mean in-lake chlorophyll concentration, but OECD model only.

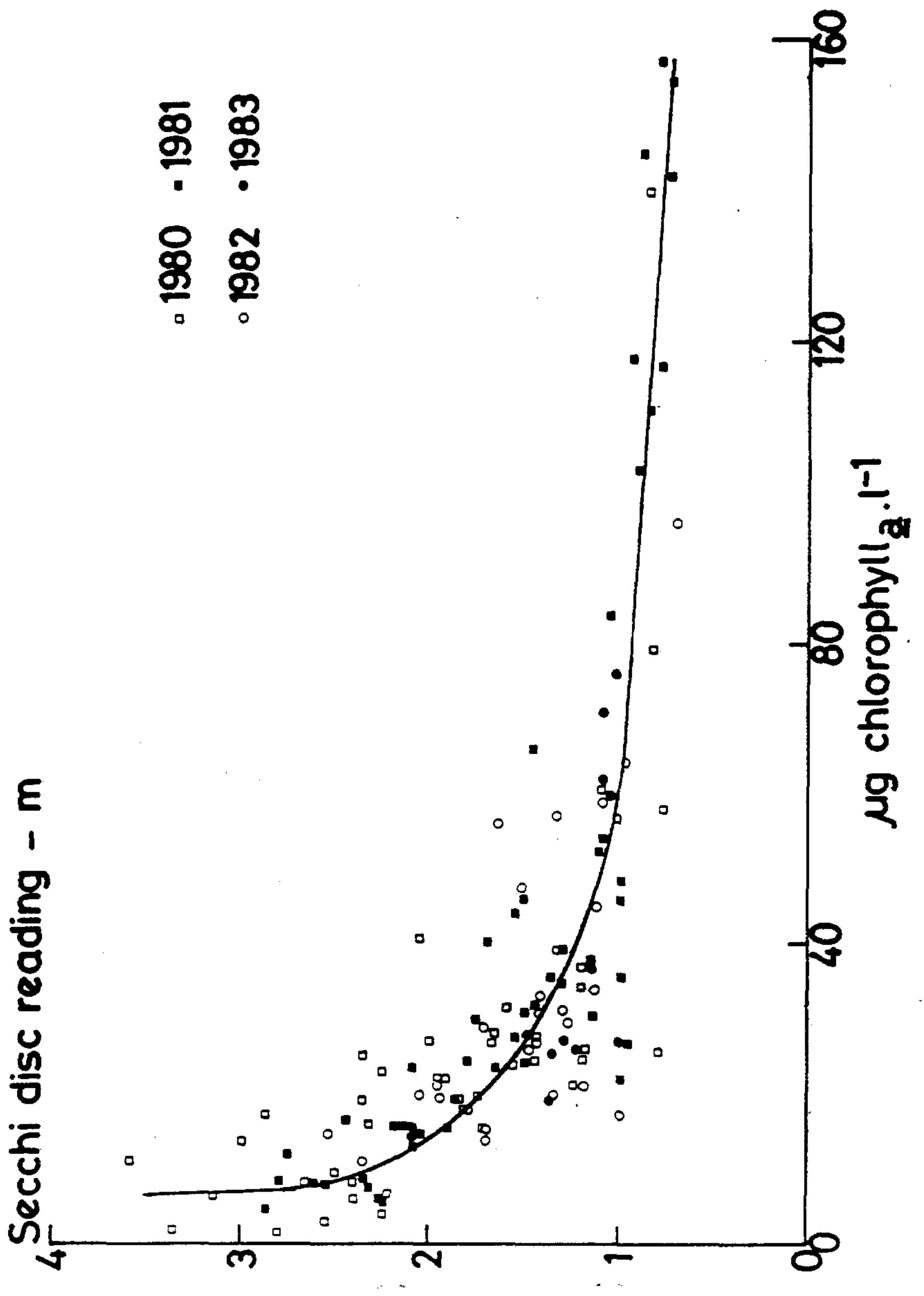


Figure 36 Relationship between water clarity (Secchi disc reading) and chlorophyll concentration at Loch Leven - 4 years' observations.

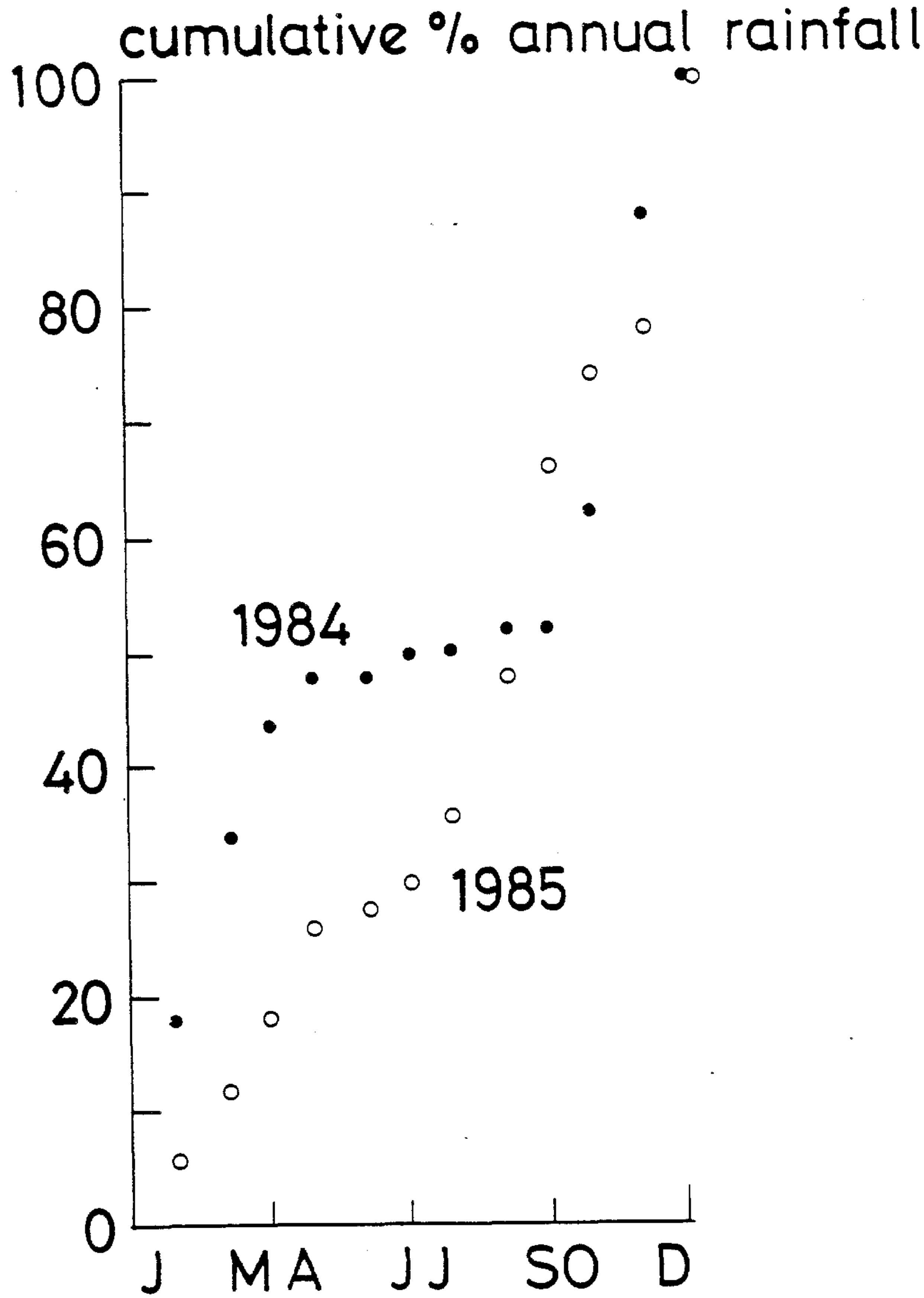


Figure 37 Contrasts in the seasonal accumulation of annual rainfall in the Loch Leven area, between 1984 and 1985.