Evaluation of historical data on the nutrient status of Esthwaite Water, Cumbria

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Contents

1. Introduction

Esthwaite Water is a small lake (area 1 km^2) near Windermere which is generally agreed to be the most productive or eutrophic of the Cumbrian lakes. It has been studied extensively during the past 45 years by the Freshwater Biological Association. It is also a site of special scientific interest (SSSI) for the Nature Conservancy Council, in which large qualitative or quantitative changes of the biota could be matters for concern. Such changes are most likely to result from pronounced shifts in nutrient status.

This report reviews and interprets the past scientific records bearing on the changing nutrient status and plant production of the lake. Besides the element of historical description, attention is also given to probable causal relationships. Changes are mostly discussed in relation to time-scales from decades to seasons. Most information relates to concentrations of nutrients and plant numbers or biomass; reliable estimates of material fluxes, of undisputed importance, are few.

2. The lake : general features, and range of scientific studies

A bathymetric map shows three basins along the long axis of the lake (Fig. 1). The largest and deepest (15 m) basin contains the main sampling station (A). A summary of bathymetric information is provided by Ramsbottom (1973), including areas and volumes of successive depth-strata (Fig. 1).

Of the six inflow streams one, Black Beck, supplies about half of the total inflow. The drainage area of the lake (approx. 17 km^2) consists of valley pastureland on richer alluvial soil, flanked by upland heath on poorer soil which is partly afforested with conifers (Fig. 2). Adjoining Black Beck and the lake is the Esthwaite North Fen, a classic site for studies of the hydrosere (Pigott & Wilson 1978) and a NNR of the NCC.

Since 1971 there have been three abrupt developments which affect the lake. In November 1973 a sewage treatment plant was set up to receive sewage from Hawkshead (population c. 657) and discharge the treated effluent into Black Beck. This replaced, in part, a system of localized septic tanks. Effluent from the smaller villages of Near and Far Sawrey was also piped into the lake. In March 1981 the owner of the lake property instigated excavation of the outflow stream, the Cunsey Beck, which cut down the retaining sill and reduced the average lake level by about 0.5 m (Fig. 3). This caused the exposure of some littoral sediments, and would alter conditions for the Fen hydrosere. Finally, in spring 1981 an enterprise of fish-farming was begun by Mr N. Woodhouse which entailed the rearing and feeding of trout kept within floating cages anchored towards the south end of the lake. An extra supply of nutrients was thereby set up.

Between 1974 and 1980 the FBA made measurements of discharge from the lake, by dilution-gauging of the Cunsey outflow. A relationship between discharge and lake level was established (Fig. 4), from which the average retention time of the lake (as total volume/discharge) was calculated as approx. 90 days. This is equivalent to a specific dilution rate of 0.011 (ln units) day^{-1}, with a large range of approx. < 0.001 - 0.08 day^{-1} between high and low levels (Heller 1977). These values will still apply, although the discharge-level relationship cannot be applied to records of level after the latter was artificially lowered in March 1981. A new gauging has not yet been made.

The status of Esthwaite Water in relation to other Cumbrian lakes, as regards phosphorus content, phytoplankton abundance, and deep-water oxygen depletion in summer, is quantitatively illustrated by data from the early 1970's published by Jones (1972a) and reproduced here as Figs 5-7. All three indices of production agree with the position of Esthwaite at the upper end of the classic Pearsallian lake series (reviewed by Macan 1980). The same is

true for earlier measurements of light penetration in summer 1953 (Talling 1971), and for other correlates of algal production (Gorham et al. 1974), including winter concentrations of nitrate and soluble reactive phosphate (Sutcliffe et al. 1982, Carrick & Sutcliffe 1982: see Tables 1, 2).

Past scientific studies of the lake include the studies in 1939-40 by Mortimer (1941-2, 1971), and in 1971-72 by Heaney et al. (in prep.), on the annual cycle of stratification and associated chemical events of 'lake metabolism'. Work on plants includes the larger aquatic vegetation (Pearsall 1917-18); population cycles of some phytoplankters (Pearsall 1932, Lund 1949, 1950, 1954, 1961, 1966 and unpublished, Heaney et al. 1983, and in prep.); their modification by fungal or protozoan parasitism (Canter & Lund 1948, 1951, 1953, 1968); related changes in nutrients (Lund 1950, 1962, 1972, Heron 1961, Sutcliffe et al., 1982, Heaney et al., in prep.) and light penetration (Talling 1971, 1979, Heaney et al., in prep.); photosynthesis of phytoplankton (Talling 1957, Heller 1977, Harris, Heaney & Talling 1979) and its relation to the CO_2-system (Talling 1976); the distribution of phytoplankton horizontally (Heaney 1976, George & Heaney 1978, Heaney & Talling 1980) and vertically (Lund 1954, 1959, Heaney & Talling 1980, Talling 1971, 1981, Frempong 1981a, Heaney et al., in prep.); and its chemical composition (Frempong 1981a, Heaney et al., in prep.). Seasonal and longer-term changes of the zooplankton have been studied by Smyly (1961, 1972, 1973a,b, 1978, and unpublished) and Strachan (1980). Other predominantly biological work includes the distribution of ciliates in relation to profundal anoxia (Webb 1961, Goulder 1974, Finlay 1981, 1982, Bark 1981), the abundance and ecology of the rich benthic fauna of chironomids (Mundie 1965, and unpublished) and tubificids (Reynoldson unpublished), and the vertical distribution of bacteria and their activities (Collins 1960, Jones 1971, 1972a,b, 1977, 1978, 1981). Predominantly chemical studies have been directed to features of the Fe and Mn systems (Davison 1977, 1981, Davison & Heaney

1978, Davison et al. 1981, Davison, Woof & Rigg 1982, Tipping 1981a,b, Tipping, Woof & Cooke 1981, Tipping, Woof & Ohnstad 1982); to humic substances (Tipping & Woof 1983); to sedimenting seston (Pennington 1974), superficial sediments (Gorham 1958, Gorham et al. 1974), and deeper chemical stratigraphy (Mackereth 1965, 1966); to the inorganic composition of suspended particulates (Sholkovitz & Copeland 1982); and to the seasonal and longer-term variation of nutrients and major ions (Heron 1961, Sutcliffe et al. 1982). Aspects of diel variability involving phytoplankton, wind stress and thermal exchanges have been analysed by Talling (1971), George & Heaney (1978), Heaney & Talling (1980), and Frempong (1981a,b, 1982, 1983).

3. Winter maxima of nutrients : long-term changes

The seasonal growth of plants (mainly phytoplankton) in this productive lake induces large seasonal changes of nutrient concentrations, affecting measured forms of N (NO_3.N, NH_4.N), P (soluble reactive phosphate = PO_4.P), Si (as $Si(OH)_4$, often represented as the SiO_2 equivalent quantity), and C (free CO_2, HCO_3^- + CO_3^{2-} - CO_2, total CO_2). Seasonal losses of nitrate are also promoted by bacterial denitrification under warm summer conditions. Therefore, in order to make long-term comparisons of nutrient concentrations, it is useful to consider the winter season with minimal plant growth when the input quantities are little-utilized and summer deficits have been replaced. The data available refer to mean concentrations over the 0 - 5 m depth layer, obtained by an integrating tube-sampler.

Past comparisons of this type have been published by Lund (1972) and Sutcliffe et al. (1982), here reproduced as Figs. 8 and 9 and Tables 1 and 2. Both show some increases in the concentrations of NO_3.N and PO_4.P with time. Lund considered yearly maxima smoothed by calculating 5-year running means, for the years 1948 - 1967. Sutcliffe et al. record yearly mean winter-

spring values for NO_3.N and autumn-winter values for PO_4.P, plus the highest winter concentrations of both; these data cover the years 1946 - 1980. The seasonal maxima of PO_4.P precede those of NO_3.N, partly because vertical autumnal mixing introduces deep accumulations for PO_4.P but not for NO_3.N, and partly because the effect of uptake during spring algal increase is proportionately much greater for PO_4.P than for NO_3.N.

The longer series of analyses (Fig. 9, Tables 1 and 2) shows that the mean concentrations of both NO_3.N and PO_4.P increase markedly for most years after 1969. Before then the values of NO_3.N are less than 600 $\mu g\ l^{-1}$; those of PO_4.P are less than 7 $\mu g\ l^{-1}$, and with one exception less than 5 $\mu g\ l^{-1}$. In this earlier, pre-1969, period there are fluctuations of limited absolute (but large relative) magnitude in both nutrients, some represented also in the smoothed running - mean values used by Lund (Fig. 8). According to Lund (1972) there is then a definite time-related increase in the maximum PO_4.P concentrations, especially during 1948 - 1958, but no statistically significant overall increase in those of NO_3.N. The last feature is despite a known tendency of the analytical method used before 1964 to underestimate NO_3.N, roughly by about 30-40%.

After 1969 the mean winter-spring concentrations (and maximum values) of NO_3.N have often exceeded 1000 $\mu g\ l^{-1}$, as compared with pre-1969 concentrations below 600 $\mu g\ l^{-1}$. The post-1969 period is also marked by a majority of mean autumn-winter concentrations of PO_4.P above 8 $\mu g\ l^{-1}$, (Fig. 9, Table 2) with maximum values above 13 $\mu g\ l^{-1}$ since 1978 (Table 2). In 1980 a much less sensitive method for PO_4.P was adopted (omitting a solvent extraction), which was known to produce occasional overestimations. Consequently less reliance can be placed on maximum values recorded for 1980-1982. The mean autumn-winter concentrations for 1981 and 1982 (not in Table 1) were, 10.9 and 12.8 $\mu g\ l^{-1}$ respectively.

The variation between years in the circum-winter concentrations are conceivably also influenced by some algal uptake of N and P. Although generally low the accompanying algal populations are variable in density, as shown in Fig. 10 using chlorophyll *a* as index of quantity. In view of the known capacity of algal cells to store P in amounts roughly equivalent to their content of chlorophyll *a*, the higher winter concentrations of > 5 µg chl-*a* l^{-1} could appreciably reduce the corresponding concentrations of PO_4.P, generally 5-15 µg l^{-1}. Further, the winter contents of chlorophyll *a* in the 1970's have generally been lower than in the 1960's, often because of a reduced amount of the diatom *Melosira italica* subsp. *subarctica*, an alga which is favoured by unstratified circum-winter conditions (Lund 1964, 1965). Thus the evidence for a phytoplankton-independent increase in PO_4.P with time is ambivalent. There is no corresponding uncertainty about the post-1969 increase of NO_3.N, as other evidence (including seasonality in Windermere: Lund, Mackereth & Mortimer 1963) strongly indicates that the Esthwaite winter populations are too small to significantly reduce the relatively large circum-winter concentrations of NO_3.N.

The problems introduced by algal uptake of P can be partly overcome by considering the total phosphorus content (P_{tot}) of the lake water rather than PO_4.P. Total phosphorus is generally regarded as the best single index of trophic state, and many generalized quantitative relationships with phytoplankton production have been proposed (e.g. Dillon & Rigler 1974). Values, largely unpublished, are available from weekly or 2-weekly sampling since 1970 (e.g. Fig. 19, top). As they reflect particulate as well as 'dissolved' material these values are more irregularly distributed in time, being influenced by sporadic 'lumps' (e.g. zooplankters) and turbulent resuspension. Since 1970 almost all values lie between 10 and 55 µg P l^{-1}; the lowest values (< 20) occur mainly in May - June, and the highest ones (>

40) mainly in September. These seasonal trends are almost certainly due to both vertical redistributions and increased phosphorus input during summer from external (tourist-associated) sources. The minimum follows the sedimentation of the spring diatom crop, and the maximum the input of P-rich summer inflow which, with sediment-water exchange, increases the estimated whole-lake content of P by a factor of about 2 (1971-2 data; Heaney et al., in prep). The highest August-September concentrations, over 50 $\mu g\ l^{-1}$, are somewhat more numerous since 1975 than before. There is no clear long-term trend in the winter concentrations, which have been typically between 20 and 30 $\mu g\ l^{-1}$ since 1970. Pre-1970 measurements are few.

Winter maximum concentrations of dissolved silicon (here expressed as SiO_2) are important for determining the magnitude of spring diatom growth (e.g. Lund 1950). As noted by Sutcliffe at al. (1982), they show no marked long-term changes since yearly records began in 1946, and are typically in the range 2-3 mg $SiO_2\ l^{-1}$.

4. Seasonal cycles of nutrients and phytoplankton quantity

In this lake concentrations of the main sources of C, N, P, and Si undergo seasonal changes which are large in comparison with long-term trends and which include important determinants of cyclic phytoplankton abundance. Although there is no published account covering all these sources, one (Heaney et al., in prep.) is in preparation for the years 1971-2 and provides examples shown in Fig. 11. Lund (1950) illustrates seasonal variation in NO_3.N and Si (as SiO_2) for 1945-49; Heron (1961) variation in NO_3.N, Si (as SiO_2), and PO_4.P for 1958-60; and Harris et al. (1979) variation in NO_3.N, NH_4.N, Si (as SiO_2) and PO_4.P for 1976.

Dissolved silicon (present as silicic acid) is present at highest concentrations (2-3 mg $SiO_2\ l^{-1}$) in winter, typically December - February.

It is invariably almost entirely removed by diatom growth in late winter - spring, being usually reduced below 0.1 mg SiO_2 l^{-1} for the years (post-1956) in which a sufficiently sensitive analytical method was used. Its depletion usually leads to a mass death of spring planktonic diatoms, of which Asterionella formosa is the most typical species. Thereafter a slow replenishment from inflows occurs throughout the summer, which is favoured by flood-water input but is often set back by further, usually minor, episodes of diatom growth (some illustrated in Fig. 12). Another factor is the autumnal mixing up of deep water that is richer in this nutrient, although such influence is less for Si than for PO_4.P and especially NH_4.N.

Phosphate-phosphorus (PO_4.P) is also present in relatively high concentration in winter, with a long-term variability already described. Typically most of it has been removed in phytoplankton growth by early March, usually before a similarly strong reduction has occurred in any other measured nutrient. It is then common for low concentrations, at or near the lower limit of analytical measurement (0.6 µg P l^{-1}) to persist until late September, when entrainment with deeper and P-richer water occurs and phytoplankton crops decline. The summer epilimnion is thus PO_4.P-depleted, and although small increases are recorded sporadically, there is no marked or regular replenishment immediately after the spring algal maximum has subsided. As already noted, the total P is minimal at this time. The yearly maximum of Cladocera (mainly Daphnia hyalina) also occurs then (Fig. 11), and the associated P-excretion is likely to promote P turnover. Although the concentration of total P increases to late summer, the large algal populations then developed are strongly depleted in P relative to reliable measures of biomass such as particulate carbon and nitrogen (Fig. 13). The depletion was quantitatively large enough, when studied in 1971-2, to suggest that P was a growth rate-limiting element from the cell-composition criteria of Healey

(1975). This interpretation accords with the timing of the cessation of population increase. The low cellular P content may also condition low rates of photosynthesis per unit biomass (as chlorophyll a), as were measured for denser summer phytoplankton in 1967 (Talling unpublished), 1973-4 (Heller 1977) and 1971 (Talling 1976).

Nitrate-nitrogen (NO_3.N) is typically near maximum concentrations from January to March. It is relatively little depleted during the spring growth of phytoplankton, and concentrations equal to half the winter maxima are not encountered until June or July. During that season, however, concentrations fall rapidly to yield the seasonal minimum in August - September. This minimum varies considerably from year to year, ranging from values (< 20 $\mu g\ l^{-1}$) near the limit of analytical detection to more than 200 $\mu g\ l^{-1}$. Low minima have become increasingly frequent since 1968; there is some correlation with warm summers, stable summer stratification, and summer algal abundance (as chlorophyll a content). The last factor could indicate algal uptake, and the first two a bacterial denitrification promoted by higher temperature. Both these processes are known to be quantitatively important sinks for nitrate in eutrophic lake water. The magnitude of a third source of loss, uptake by aquatic macrophytes, is not known.

In most years concentrations rise again from late-September onwards although this recovery is considerably later in some years (e.g. November in 1971 and 1972, Fig. 11). A quantitative analysis of such variability has not been made, but qualitatively the important factors are fall in algal biomass, autumnal inputs in flood-water, and the promotion of net nitrification by lower temperatures and (see below) higher content of precursor-ammonium in oxygenated water.

Ammonium-nitrogen (NH_4.N) has a distinctive seasonal cycle of concentration in surface layers, with a single pronounced pulse after the autumnal mixing in September - October. This pulse originates from upward dispersion of the much higher concentrations accumulated within the summer hypolimnion. It typically reaches 100-300 µg NH_4.N l^{-1}, as compared with concentrations below 50 µg l^{-1} during most of the year. During the late summer phase of greatest algal biomass the concentrations are typically lowest and below 20 µg l^{-1}. They represent a N-compartment which is readily utilized by algae and hence likely to be transitory, in rapid flux. The autumnal pulse is typically developed when algal quantities have decreased, and its magnitude cannot be related to the size of subsequent algal crops.

The seasonal variation of total algal biomass is indicated by that of chlorophyll a, illustrated in Fig. 10. It shows a succession of major maxima, or growth/decline cycles, variable in number from year to year. A predominantly 2-cycle or biomodal pattern has been common, especially since 1969. It is constituted by a spring diatom maximum and a larger late-summer maximum of other algae, especially the dinoflagellate Ceratium hirundinella.

Additional sizeable maxima can be introduced in three main ways. Two species of diatoms, Asterionella formosa and Melosira italica subsp. subarctica, may contribute to the spring growth with two non-coincident maxima. After the spring and before the main summer maximum, other algae such as the chrysomonad Uroglena americana (in 1967) and Aphanizomenon flos-aquae (in 1983) may achieve large but usually short-lived maxima. Lastly, a growth-cycle may be inserted in autumn (October - November) after the vertical mixing or overturn, with diatoms, and often the blue-green alga Aphanizomenon flos-aquae, as principal components. With such additional maxima the plot of chlorophyll a concentration against time shows oscillations of higher frequency, with good examples for the years 1964 -

1967. After years of the bimodal pattern in the 1970's, the extra post-overturn maximum of diatoms and A. flos-aquae was re-developed in 1982. The last alga persisted through the winter into 1983 and produced an unusually large population in late spring - early summer, replacing the normal 'clear water' conditions.

The effects of algal production in the lake will be influenced by the number, duration, and especially the magnitude of the seasonal maxima. The usual spring diatom maximum dominated by Asterionella formosa is arrested sharply by silicon depletion. As the preceeding winter concentrations of this nutrient vary little, the achievable diatom growth is also of limited variability, with sedimentation and species composition as the main sources of variation. The larger summer-maximum is both more influential for water quality (e.g. turbidity) and more variable in density, even between successive years such as 1973 and 1974. The origin(s) of this variation is not well known, but the highest concentrations of chlorophyll a, above 100 $\mu g\ l^{-1}$, are achieved during fine summers of more stable stratification by communities dominated by Ceratium hirundinella. This motile dinoflagellate may well exploit deep accumulations of nitrogen and phosphorus, a supposition supported by evidence in Heaney et al. (in prep.), which are unlikely to develop sufficiently high in the water-column during the more windy, cool, and turbulent summers. If this hypothesis proves correct, at least the short-term variability in the highest densities of phytoplankton over the 0 - 5 m zone may be regulated by physical rather than chemical inputs. Another form of physical regulation applies to short-term redistributions of algal biomass by horizontal transport, which can generate high but local densities at the routine sampling station (e.g. Heaney & Talling 1980; Fig. 14). Long-term aspects are considered in section 6.

5. Depth-related aspects of seasonal cycles

Although the preceeding account of nutrients and algal production was primarily concerned with the upper (esp. 0 - 5 m) water of the lake, some relevant events in deeper layers could not be neglected. These are explicitly summarized below.

(a) The vertical separation of layers is primarily due to density differences associated with corresponding temperature differences. As in other stratified lakes one can recognise a division into an upper and warmer epilimnion, a deep and cold hypolimnion and an intermediate transition region, the metalimnion (including the strongest discontinuity or thermocline). Temperature stratification typically begins in late April and ends in early October, although the timing depends on weather conditions and may be altered by several weeks. Examples are given in Fig. 15.

(b) The vertical distribution of phytoplankton abundance (Figs. 16, 17, 18) is closely related to thermal stratification. Vertical near-uniformity of total biomass is usual in winter and spring before thermal stratification is developed, and the massive summer populations are broadly restricted to the upper warmer layer (epilimnion). However maxima can develop at depth (i) by sedimentation, especially of spring diatoms (ii) by growth there, especially during the clear-water phase in May-June (iii) by selective motility or buoyancy of algal cells, respectively exemplified by the flagellate Ceratium hirundinella (Fig. 16, and Harris et al. 1979, Heaney & Talling 1980) and the blue-green Oscillatoria agardhii (Lund 1959). Superficial scums of blue-green algae (including species of Anabaena, Aphanizomenon, and Microcystis) can also occur during calm weather.

(c) Deep accumulations of nutrients during summer stratification are notable for NH_4-N (N-source) and PO_4-P (P source). Examples, of distributions during 1971-2, are shown in Figs. 19, 20). Another N-source, NO_3-N, is lost from the anoxic deepest region (hypolimnion) but, in early summer, typically has a mid-water maximum (Fig. 20) influenced by algal uptake and bacterial nitrification plus denitrification.

(d) The vertical distribution of oxygen and carbon dioxide are inversely related during summer stratification. Oxygen is lost from the deepest water after mid-June, returning on autumnal mixing in late September - October. The vertical extension of anoxia varies from year to year (Fig. 26). Conversely, carbon dioxide and its combination in bicarbonate accumulate at depth. Some depth-accumulations of methane (CH_4) and hydrogen sulphide (H_2S, and its dissociation products HS^- and S^{2-}: Davison 1977, Davison & Heaney 1978) probably occur intermittently, but have not been regularly studied.

(e) Deep accumulations of iron and manganese, in the reduced divalent forms, are a regular feature of the summer stratification Example distributions are described by Mortimer (1941-2), Mackereth (1966), and Davison et al. (1981, 1982). For most of the summer precipitates of oxidised iron (hydrated ferric oxide) are conspicuous at and below the oxic-anoxic interface (Davison et al. 1981). As they readily adsorb phosphate, they constitute one barrier to the vertical transport of this nutrient.

(f) Light penetration into the lake varies seasonally, being reduced at times of dense phytoplankton and after autumnal mixing. The effective limit for appreciable plant growth is about 1% of the surface-incident light, a level often found about 6 m depth - but usually greater in June ('clear-water phase') and less in August - September (phytoplankton maximum). See also section 7(a).

6. Long-term variation of algal production

Here three aspects of algal production are of interest; they concern the total biomass, the biomass production of certain major species, and long-term qualitative changes of species occurrence.

(a) Total biomass

The best long-term record of total algal biomass is provided (Fig. 10) by average concentrations of chlorophyll a (uncorrected for degradation products) in the 0 - 5 m layer, measured generally at weekly intervals since 1964. The logarithmic scale used depicts maxima and minima with equal relative precision, and the seasonal frequency of maxima has already been discussed (section 3). The absolute magnitude of the main summer maxima shows slight systematic change with time during the last 19 years, with concentrations in excess of 100 mg m^{-3} (= $\mu g\ l^{-1}$) frequent in the 1970's but not in the 1960's. They indicate a highly productive water by world-standards. Some of these high maxima are likely to be partly caused by the localised accumulation of biomass at the sampling station after redistribution within the lake. Such a situation was demonstrated by Heaney & Talling (1980) for 15 September 1977, when movements of the dinoflagellate Ceratium hirundinella were involved (Fig. 14).

The earliest estimate of algal composition was made in 1928 by Pearsall (1932). Although his sampling was restricted to net collections near the surface of the water, and did not assess biomass, he observed that diatoms and blue-green algae predominated in summer. Quantitative estimates of algal biomass for the period 1946-1969 were made by Lund (1972) on the basis of dry weight, and also refer to the 0 - 5 m layer. He showed (Fig. 22 bottom) that the main summer maxima increased considerably and abruptly after 1963, largely due to a massive increase in Ceratium (Fig. 23). Thus there is evidence for

a large long-term increase in the summer biomass, but not during the last 19 years.

(b) Biomass of major species

The above-mentioned increase of the dinoflagellate Ceratium after 1963 exemplifies the importance of the behaviour of individual species. There is some evidence that its increase was partly at the expense of some previous summer dominants, notably the blue-greens Microcystis aeruginosa and Aphanizomenon flos-aquae. During the last decade these buoyant species have rarely constituted a large part of the summer maximum; another buoyant blue-green, Anabaena flos-aquae, was often important but still subordinate to the Ceratium. Diatoms, which dominate the spring maximum, are usually still more subordinate in summer but occasionally reach fairly high concentrations, as in Pearsall's 1928 study and in 1974 (Fig. 24). Such growths are dependent upon a sufficient replenishment by inflows of dissolved silicon, after its spring depletion, and probably a lack of CO_2-depletion by other competing algae (Talling 1976, and unpublished). The well-dispersed populations of such diatoms are not visually obtrusive.

(c) Species occurrence

Besides the quantitative aspects outlined above, the novel appearance of some species in quantity may signal long-term changes in the lake. Two such appearances relate to the centric diatom Stephanodiscus hantzschii and the green alga Chlorella sp., which were first seen in large quantity during the spring maxima of 1978 and 1974 respectively. The former species (a taxonomically difficult entity) is a recognized indicator of nutrient-rich waters, and has recurred in later spring maxima, sometimes in quantity. The Chlorella maximum of 1974 has not been repeated on the same scale. Its abundance in March 1974 followed the sewage installation in November 1973, apparently as a non-repeated transient.

7. Long-term variation of biological reactions upon the environment

Three such reactions are considered below.

(a) Transparency

The depth at which a white disc of 30 cm diameter (Secchi disc) disappears from view when lowered into the water gives a useful measure of transparency. Values thus obtained from 1971-82 are given in Fig. 25, which shows no obvious long term trend. However there are large but rather regular seasonal fluctuations, with decreased transparency in spring and especially in late summer due to increasing phytoplankton density, and also immediately after autumn overturn from resuspension of solids and precipitation of ferric hydroxide. The period of greatest transparency, and so of light penetration, is normally in May-June after sedimentation of the spring diatom growth. The depth of the euphotic zone in which appreciable photosynthesis occurs, and often defined as the depth at which 1% of the surface-penetrating irradiance (photosynthetically active radiation) is found, is roughly between 2 and 3 times the Secchi disc transparency.

Different colours of light are differentially attenuated with depth in lakes. For Esthwaite Water Talling (1971) showed that there is a relationship between the vertical extinction coefficient (a measure of rate of attenuation) for the most penetrating orange light and algal abundance measured by the concentration of chlorophyll a. The increment in coefficient per unit chlorophyll a during dense populations of Ceratium hirundinella was only about half that obtained for Asterionella formosa in nearby Windermere. The low value may possibly be due to a 'sieve effect' with large Ceratium cells, so that some of the pigment is relatively ineffective for light interception. This in turn permits a higher upper limit of chlorophyll a in unit area of the euphotic zone. Such a characteristic of Ceratium may enable it to more effectively utilize the available nutrients (e.g. P) and

obtain a greater biomass before becoming light-limited through self-shading.

(b) Deep-water anoxia

As already described (section 4) this condition develops during each summer stratification, with the average time limits (mid June - early October) subject to some modification by weather conditions. Much greater variation, however, appears in its maximum vertical extent which has been followed at station A for many years since 1939 (Fig. 26). The upward extension of anoxia above the 10 m horizon, shown by black areas in Fig. 26, has clearly increased greatly since 1940 and to some extent since 1966. This overall trend with time is subject to large year-to-year variations enforced by weather conditions, as in cool windy summers the depth of the upper mixed (and oxygenated) layer is increased. The converse is true of calm, hot summers such as that of 1976, when anoxia reached up to 3.5 m on 27 August. At this time some 57% of the entire lake volume was probably anoxic.

The biological implications of such vertically extended anoxia include the restriction of habitat for aerobic organisms unless (like some chironomid larvae and tubificid worms) exceptionally tolerant to anoxia. Chemically, the greatly extended area of sediment overlain by anoxic water is likely to enhance the internal summer loading of nutrients from sediment to lake water, as Mortimer (1941-2) showed in this lake that exchange was accelerated under anoxic conditions at the sediment surface.

Lastly, the maximum vertical extent of anoxia is closely related to the maximum summer areal deficit of oxygen, which has long been recognised as a measure - if an imperfect one - of organic production per unit area of lake. The time-related increase in anoxia is therefore indicative of increased production, which is likely to be a step-wise rather than a regular progression.

(c) CO_2-depletion with high pH

An often unregarded consequence of high plant production is the photosynthetic depletion of carbon dioxide, which can lead to high values of pH especially in poorly buffered waters. Esthwaite Water provides an example in which pH values of 10 or more are often reached in surface water during the summer algal maxima. In extreme instances values around pH 10.5 were recorded, as in August 1973 (George & Heaney 1978), and values above 9.5 occur almost every year at station A since systematic records began in 1966. Such high values were not recorded in the few early measurements, such as those by Mortimer in 1939-40, but some appreciable elevations (to pH 8.4 - 9.4) were noted by Collins (unpublished) during 1957-62. An example time-sequence for 1971 and 1972 is given in Fig. 27, top.

Possible zoological consequences of such alkaline shifts are not known, but there is experimental demonstration for this lake that they must impose a seasonal barrier to the growth of some phytoplankters (Talling 1976) and probably many macrophytes (Maberly, private communication).

8. Aquatic macrophytes

There has been no sustained work on the aquatic macrophytes comparable to that on the phytoplankton. The information available is (i) a published survey of the species and their spatial distribution in 1914-6 (Pearsall 1917), (ii) published long-term surveys in the area adjoining the North Fen (Tansley 1939, Pigott & Wilson 1978), (iii) brief unpublished surveys of horizontal and vertical distribution in some areas, using SCUBA diving, by Jupp (in 1974) and Wade, (iv) intensive field sampling and laboratory experimentation on species (mainly the moss Fontinalis antipyretica) collected from near the main (Black Beck) inflow from 1982-83 by Maberly (unpublished), (v) the species list of Stokoe (1983), and (vi) incidental observations by staff of the Freshwater Biological Association.

At the present time it appears that well-developed beds of submerged macrophytes are few, as compared with the neighbouring lake Windermere and the early 1914-6 survey of Esthwaite Water by Pearsall. The most obvious development is in the shallow bay beside the main inflow, with Fontinalis, Elodea canadensis and Potamogeton obtusifolius all abundant there. SCUBA-collecting by Jupp off the steeper north-west shore indicated a lower depth-limit of 3.8 m for the beds (mainly of a Potamogeton sp.) there, and this is matched closely by other observations of Wade and Maberly.

There is therefore a possibility, though unsupported by adequate quantitative sampling, that submerged macrophytes have declined during the period in which the summer phytoplankton has become denser. If true, possible causes include the effect of phytoplankton on reducing light penetration to submerged weed-beds, and its effect on depleting the concentrations of carbon dioxide available for macrophyte photosynthesis. The unpublished work of Maberly suggest that the latter may be important, and that the location of some weed-beds near the main inflow may relate in part to replenishment of CO_2-sources there from the inflow or from the organic shallow-water sediments. Reduction of light penetration in the lake by phytoplankton is quantitatively predictable, using estimations of chlorophyll a concentration (Talling 1971, Heaney et al, in prep.). Whether these factors have adversely affected macrophyte growth and survival is conjectural and will depend on the species, its phenology and the depth occupied.

9. Chemical budgets and fluxes

Reasonably accurate nutrient budgets for the lake are not available, nor are they being undertaken at present. Nevertheless, estimates of phosphorus and nitrogen inputs have been made from (a) direct chemical analysis of the inflows together with information of water flux over one twelve-month period in the late 'sixties, (b) the likely input of phosphorus from the major point

source, Hawkshead sewage works, based on the stable population and tourist visitors to Hawkshead village and making assumptions for the contribution of phosphorus per capita to the sewage works, (c) the nitrogen input from the Hawkshead sewage works obtained from chemical analyses of effluent and flow discharge data. Phosphorus is not normally determined in the sewage effluent although some measurements are being made by the North West Water Authority at Bank Holidays and summer weekends during 1983.

(a) Phosphorus and nitrogen loadings from chemical analyses of lake inflows and water flux

The late F.J.H. Mackereth (unpublished) used analyses of PO_4-P and NO_3-N in the major inflows and outflow of the lake at weekly intervals from June 1968 to June 1969, and associated water flux data, to estimate phosphorus and nitrogen inputs. The water flux was calculated from records of lake level and the relationship between outflow rate and lake level determined from the data of the former Lancashire Water Authority (Fig. 4). It was assumed that the main inflow, Black Beck, contributed half the total flow to the lake. Table 3 shows that the estimated annual inputs of PO_4-P and NO_3-N were 589 and 15,045 kg respectively or 0.59 and 15.0 g per m^2 lake surface (area = 1.0 km^2). Black Beck, into which the then septic tanks of Hawkshead overflowed, contributed most of the PO_4-P (85%) and of the NO_3-N (73%). This calculation underestimates the P-input by neglecting other P-fractions, particulate-P and soluble organic P. These fractions have not been measured in the inflows.

(b) Estimate of the phosphorus loading from Hawkshead sewage works for 1982

From (a) above, the main phosphorus input to the lake is via the major inflow, Black Beck. Before November 1973, Hawkshead village and the outlying district at the head of Esthwaite Water were connected to septic tanks. After

this date the village was connected to a mains sewerage system and sewage works which discharges into Black Beck. As Hawkshead is a tourist centre, traffic congestion has made it necessary to prohibit parking in the village from 1981, and all cars and coaches are now restricted to a large 'pay and display' car park on the edge of the village. The numbers of vehicles using this park are recorded by the South Lakeland District Council. By making the following assumptions, these records, together with the population census of Hawkshead and estimates of overnight stays at the nearby campsite, permit a desk estimate of the sewage phosphorus contribution to the lake for 1982. It is based on the following estimates and assumptions:

(i) The number of people visiting Hawkshead by car and coach far exceeds those arriving by other means, such as on foot, bicycle or bus, and the car park returns give a reasonable estimate of the day-time visitors; records are not kept from November to Easter when tourist traffic is low.

(ii) There are, on average, 2.4 persons per car and 40 persons per coach (Lake District Special Planning Board).

(iii) All visitors use the toilet facilities.

(iv) The campsite of 72 pitches has a 50% occupancy with 3 persons per pitch.

(v) the stable, non-tourist population of Hawkshead is 657 (1981 government census).

(vi) The per capita contribution of phosphorus for residents is 1.8 g day^{-1} in domestic sewage of which the ratio of detergent phosphorus to excreted phosphorus is 1:1 (Alexander & Stevens 1976). As most visitors to the car park (90%) are short-stay (< 2 hours), a value of 0.3 g P day^{-1} is taken as the average value for visitors.

Table 4 shows the calculations of phosphorus input to and outflow from the sewage works from various sources. The annual contribution to the sewage works is calculated as 548 kg P $year^{-1}$. Assuming that 88% of the phosphorus is soluble (Alexander & Stevens 1976), and not removed in treatment but

discharged to Black Beck, then the annual phosphorus loading to Esthwaite Water from the sewage works effluent was 482 kg, or 0.48 g per m^2 lake area.

The greatest loading of phosphorus is during the summer period when tourist traffic is heaviest. This is also the time of stratification and heavy algal growth when the lake is most sensitive to high nutrient inputs. Table 4 also shows the calculated input of phosphorus from Easter - 31 October as 321 kg P or 67% of the annual loading.

In these calculations no allowance is made for hotel guests, use of holiday homes and flats not included in the census, people not arriving by car or coach, and car park users in winter. Also not included is phosphorus from agricultural sources. All these factors will tend to underestimation of the phosphorus loading to the lake.

(c) Nitrogen loading from the Hawkshead sewage works, 1981-1982

Although phosphorus is not analysed in the sewage effluent, both ammonium-nitrogen and total oxidised nitrogen (TON, NO_3+NO_2-N) are. Results of these analyses and measurements of effluent flow for 1981 and 1982 are given in Figs. 28 and 29, from which may be obtained the annual and seasonal discharge of both forms of nitrogen. During winter and autumn almost all the nitrogen is in the oxidised form but from April to September the situation is reversed and the major proportion of nitrogen leaves the sewage works as NH_4-N. The weekly volume of effluent discharged is generally less during the spring - summer period.

The number of analysed samples was small and each was a spot sample taken around mid-day, rather than a composite sample collected over 24 hours. The ability of such a sampling procedure to accurately reflect the changes in nitrogen concentration, both daily and seasonally, is questionable. Nevertheless, as the results show striking seasonal differences of the

nitrogen forms and sewage flow rates, nitrogen loadings to Esthwaite Water from the sewage effluent have been calculated (Table 5). In these calculations mean values of TON and NH_4-N concentration were obtained separately for April - September and for the remainder of the year, and used with the weekly flow data to obtain the nitrogen loadings. In doing so maximum nitrogen discharges at week-ends and Bank Holiday periods are likely to be under-represented.

The total estimated yearly loadings of inorganic nitrogen for 1981 and 1982 from Hawkshead sewage works were 2762 and 1563 kg N, or 2.76 and 1.56 g m^{-2} lake area, respectively. Assuming no major changes in land usage and intensity of fertilizer application since 1968-69, the recent contribution of total inorganic nitrogen from the sewage works is only about 0.10 - 0.18 of the total estimated input of NO_3-N to the lake. This implied preponderance of non-sewage inputs for N is in clear contrast to the situation with phosphorus, where the main input to the lake is from sewage effluent rather than agricultural or 'natural' sources.

(d) Phosphorus and nitrogen fluxes in relation to summer algal increase

In Esthwaite Water both dissolved inorganic nitrogen and phosphorus are usually at very low concentrations at the time of high algal biomass in summer. It is therefore desirable to determine the extent to which fluxes of these elements, both external and internal, may sustain algal production.

An example from late-August 1972 is given in Table 6, where weekly external inputs and lakewater concentrations of PO_4-P, NO_3+NO_2-N and NH_4-N are related to the particulate (mainly phytoplankton) phosphorus and nitrogen content of the upper mixed layer. Weekly loadings of P and N are used as the population doubling of the main phytoplankters at this time is approximately of this duration. Although weekly nutrient inputs will vary in response to factors such as the influx of visitors, time of fertilizer

application and rainfall, mean values over the period of stratification are the best approximations available. Table 6 shows that for phosphorus the weekly input plus lakewater content of PO_4-P averaged only 28% of the standing stock of particulate phosphorus, whereas for nitrogen the weekly input plus lakewater content of inorganic nitrogen averaged 79% of the particulate nitrogen content. Therefore, despite the external loadings of P and N, there would have been insufficient of each element to sustain the postulated rate of population increase unless there was a lowering of cellular P and N contents. This conclusion would be strengthened if appreciable quantities of P and N are removed by macrophytes and by microbial uptake at the shallow-water sediments, and weakened if a converse 'internal loading' from sediment to epilimnetic water is marked. A calculation of bacterial denitrification is given later.

Nutrient forms of P and N may be internally supplied within the lake by regeneration in both the epilimnion and the hypolimnion. The magnitude of epilimnetic regeneration is not known but during late summer it is likely to be small for both elements as the large algal cells present are not grazed and there is little microscopic evidence of other appreciable sources of mortality. However, one may speculate (Heaney 1976) that appreciable quantities of phosphorus may be released from the littoral sediments as such release accelerates in alkaline conditions (Mackereth 1953) which are produced by photosynthesis in the summer epilimnion (Talling 1976).

Both P and N are appreciably regenerated in the hypolimnion, and PO_4-P and NH_4-N reach high concentrations in the deeper layers. The availability of these deep reserves of phosphorus for plant production is not clear. The normally dominant phytoplankter, Ceratium, does not enter the anoxic and phosphorus-rich layers. A partial chemical barrier, of uncertain quantitative significance, is constituted by the adsorbtion of phosphate onto ferric hydroxide particles prevalent at the oxic/anoxic boundary. This does not

prevent P-enrichment of epilimnetic water by entrainment during the thermocline depression in late September.

The uptake of ammonium-nitrogen from the hypolimnion during 1972 was assessed by Heaney et al. (in prep.). During mid-summer (July-August) they calculated that 25-50% of the nitrogen demand of new cells of Ceratium during this period could have been sustained by direct uptake of NH_4-N by cells entering the hypolimnion before anoxic conditions developed. They also calculated that the diffusion of NH_4-N across the thermocline during August would supply only c. 2% of the nitrogen necessary to sustain the earlier rate of population doubling every 5 days. By analogy, the supply of phosphorus by turbulent diffusion would also be small.

These estimates of external and internal nutrient fluxes suggest that, at the time of high phytoplankton biomass during late summer, the supply of phosphorus and (in lesser degree) nitrogen is insufficient to support appreciable exponential growth. The major nutrient carbon can also, at times, limit algal growth; the dominant Ceratium and some blue-green algae have the capacity to utilize virtually all the total inorganic carbon (Talling 1976, Heaney et al. in prep.). Carbon is approximately 50% of organic dry weight and this content appears to vary little. In contrast, cellular N and especially P may decrease as the flux into cells diminishes in relation to growth rate. Such changes, exemplified by decreases in particulate N:C and especially P:C ratios of the plankton, have been found in the lake in late summer (Fig. 13). They suggest that the external and internal lake fluxes of P, and possibly of N, can be insufficient to sustain further exponential growth in late summer.

Silicon is probably the element which usually limits the spring growth of diatoms (Lund 1950). It may also be important in controlling the size of their summer populations in conditions otherwise conducive for their growth (Heaney, Talling & Sinton in preparation).

(e) Bacterial denitrification and phytoplankton uptake of NO_3-N

The major seasonal decline of NO_3-N in the lake takes place as the result of uptake and utilization by phytoplankton and macrophytes and by microbial denitrification. An estimate of the relative significance of the last process for Esthwaite Water may be obtained from the determination of nitrate entering the lake and measurements of microbial denitrification (as N_2 gas evolved) from a nearby but similarly productive lake, Blelham Tarn (Jones & Simon 1981).

In the calculation (Table 7) it is assumed that an upper value of denitrification in the profundal sediment during stratification is typical for sediment throughout the lake during the period of complete mixing. The nitrate input is that determined for Esthwaite Water for 1968-69 (15,045 kg). Denitrification is calculated to account for only a small proportion (c. 10%) of the annual nitrate entering the lake.

Table 8 shows an attempt to assess relative losses of NO_3-N due to microbial denitrification, uptake by phytoplankton, and through the outflow of the lake for the main period of NO_3-N decline and algal increase during 1971 (mid-May to end of August). Microbial denitrification and NO_3-N input were determined as above and the NO_3-N available assessed as the lake content of NO_3-N during mid-May added to the NO_3-N input. Net uptake by phytoplankton, as increase of particulate N in the epilimnion, and loss of NO_3-N through the outflow, were obtained from Heaney et al. (in prep). Microbial denitrification accounted for 12% of the NO_3-N available compared to 29% for phytoplankton uptake and 30% for washout. The remainder will represent errors associated with the estimates and NO_3-N utilization by macrophytes. Even considering the assumptions involved in such estimates, the latter are likely to indicate the relative magnitude of losses of nitrate from the lake.

(f) Phosphorus loading in relation to other lakes and nutrient-crop relationships

Several quantitative relationships have been proposed between summer phytoplankton production and phosphorus loading for lakes where this element is likely to limit growth. Two of the simpler models are applied below to Esthwaite Water.

Using Vollenweider's earliest graphical relationship of phosphorus loading, phytoplankton abundance (as chlorophyll a), and mean lake depth (Vollenweider 1968), Fig. 30 places Esthwaite Water in relation to other northern temperate lakes. The lake is clearly eutrophic on Vollenweider's assessment, but less so than, for example, Lough Neagh, N. Ireland. Over the years Vollenweider has refined his mathematical model of phosphorus loading to also take account of sedimentation and flushing processes (e.g. Vollenweider & Kerekes 1980). However, as the average inflow concentration of total phosphorus is not known for Esthwaite Water it is not possible to apply his models further. Nevertheless, Vollenweider & Kerekes (1980) provide a preliminary classification of lakes in the OECD eutrophication programme based on a number of variables, into which the position of Esthwaite Water is indicated in Table 9. Thus, on the basis of lake nitrogen and phosphorus levels the lake may be considered mesotrophic, but is eutrophic with regard to chlorophyll a concentration and water clarity.

Dillon & Rigler (1974) also considered empirically the phosphorus-chlorophyll a relationship in lakes. They produced the predictive equation (1) below for lakes such as Esthwaite Water where the ratio by weight of total Kjeldahl nitrogen to total phosphorus, measured at the spring overturn, is greater than 12:

$$\log_{10} [\text{chl } a] = 1.449 \log_{10} [P \text{ max}] - 1.136 \qquad (1)$$

where [chl a] is the mean summer concentration of chlorophyll a (middle of May to the beginning of September), [P max] the maximum total phosphorus

concentration in spring. For Esthwaite Water from 1970 - 1979 the mean concentration ($\bar{x}$) and standard deviation (s) for total phosphorus for March and April were $\bar{x}$ = 21.2, s = 4.5 and $\bar{x}$ = 21.3, s = 3.9 µg P l^{-1} respectively. Taking a mean spring value of total phosphorus of 21.3 µg P l^{-1} and substituting into equation 1, a mean summer concentration of chlorophyll a of 6.2 mg m^{-3} is obtained. This is very much less than that measured (Fig. 10).

Esthwaite Water may fit Dillon & Rigler's predictive equation poorly for several reasons: (1) the normal summer phytoplankton periodicity encompasses both a phase of low biomass ('clear water phase') and an annual maximum, rendering 'summer averages' meaningless; (2) both direct measurements and calculations of weekly P input to the lake show there is a considerable additional loading during the summer from tourism, and internal loading may also accelerate; (3) the mobile populations in summer, mainly the dinoflagellate Ceratium hirundinella, are effective in growing at the expense of their cellular reserves to yield a large but P-depleted biomass.

10. General discussion

This exposition would ideally account for the observed historical changes in terms of estimated inputs to the lake and mechanisms (including species responses) known to be operating within it. Ideally also, predictive relationships would be offered by which future changes could be gauged. The factual reality falls short of these goals, but the following comments are offered.

There is evidence, of varying rigour, that the nutrient (N, P) status of the lake has increased over the last three decades. The biggest absolute changes, not necessarily the largest relative or biologically influential ones, have occurred in the last decade and involve both NO_3-N and forms of P. Possibly the earliest detectable evidence of nutrient increase concerned

PO_4-P between 1948 and 1958, although the case is weakened by a deficiency of total P analyses and a possible influence of algal uptake on the winter PO_4-P index. However, it seems that the first recognisable increase in NO_3-N followed that of P. Thus decisive step-wise episodes of general nutrient increase are not evident, e.g. as well-defined responses to such discrete events as the introduction of piped sewage in November 1973.

Similarly, there is evidence for an increase in the main summer production of phytoplankton over the same three decades. In this a discrete step in 1963-4 can be recognised, when very large populations of the dinoflagellate Ceratium hirundinella first developed. Two capabilities of this organism, for vertical migration and for forming P-depleted biomass, were probably important for its subsequent success. Nevertheless, continued substantial summer maxima of blue-green algae, and occasional ones of some diatoms, were also significant in the two more productive recent decades.

Lake enrichment, or 'eutrophication', is often judged subjectively based on the appearance of lake surface waters. Esthwaite Water has, over the recent past, borne large populations of the less conspicuous and often deeply dispersed Ceratium. A much smaller biomass of floating blue-green algae, as can occasionally occur, may arouse much greater public anxiety. Thus algal quality, and its possible change, is of great importance for amenity. The relative infrequency of observed scums of blue-green algae during summer is also influenced by the daily wind pattern. During anticyclonic weather the wind normally freshens at about 08.00 hours reaching a maximum velocity near 14.00 hours G.M.T. (George & Heaney 1978), causing appreciable mixing in the surface layers and dispersal of algal accumulations.

Both external and internal forms of nutrient loading have been considered. Although the estimates of external loading are crude, they are probably adequate to establish that the lake would be expected to be productive to very productive on this basis alone. Although the internal

loading represents a redistribution rather than a primary input, it could be crucial in temporally delimiting nutrient delivery to a specific biological event. The late-summer maximum of algae is by far the most important such event as regards biomass, water quality, and effects on the general lake biota. Focussed on August and September, it occurs when processes of internal loading are probably most marked. Three of them may involve elements of positive feed-back, in that past production promotes more supply of nutrients for future production. These possible feed-backs are: upward resupply by entrainment of nutrients accumulated in the hypolimnion; promotion of sediment-water exchanges by vertically extended anoxia; and a similar promotion of P-release from shallow-water sediments bathed in highly alkaline water resulting from intense photosynthesis.

The seasonal succession involves such a variety of situations and organisms that the concept of a single, generally limiting nutrient is not tenable. The biggest challenge is again posed by the late-summer algal maximum. This crop was, when studied, strongly P-depleted and this constitutes strong circumstantial evidence for P-limitation of growth rate. By comparison, the observed N-depletion in the crop was relatively slight, but the accompanying depletion of inorganic N content in the surface water and predominance of cellular N there are still suggestive of potential N limitation. Inorganic C is also depleted to an extent which probably blocks the growth of many species found earlier, but not immediately that of such summer dominants as Ceratium and Microcystis. Nevertheless, this nutrient can be ultimately replenished from both the atmosphere and from deep reserves at times of weather-induced mixing. Lastly, the densest crops are probably not far short of concentrations at which self-shading might be expected to produce light-limitation of population growth (Heaney et al., in prep., Talling 1971). Given increased inputs of P, limitation might well shift to one or more of these other factors.

Some appraisal of the various influential factors has been attempted here but a further, more quantitative study of their relative importance, together with more rigorous estimates of external and internal nutrient inputs, would assist in making management decisions for the lake.

11. Summary

(a) Esthwaite Water is probably the most productive of the Cumbrian lakes, and in recent decades has shown evidence of nutrient enrichment with increased abundance of phytoplankton. These changes, with implications for water quality and other biota, are examined from long-term (1939-83) records of the Freshwater Biological Association.

(b) An outline is given of the general geographical features of the lake, its water replacement and level-discharge relationship, the range of past scientific studies, and human manipulations in the last decade. The latter include the input of piped sewage from a new treatment plant (Nov. 1973), the lowering of lake level by c. 0.5 m (March 1981), and the introduction of a fish-farming venture (1981).

(c) Surveys of the mean circum-winter concentrations of N (as NO_3-N) and P (as PO_4-P), when biological uptake is minimal, provide evidence of long-term increases in these nutrients. Some small increase in PO_4-P possibly occurred during 1948-1958, and after 1969 larger increases in both nutrients were found, to values above 8 μg PO_4-P l^{-1} and 600 μg NO_3-N l^{-1}. Corresponding winter concentrations of total phosphorus, available only since 1970, are typically 20-30 μg l^{-1} - a low level for such a productive lake. However it is seasonally increased, by external and internal loading, to above 40 μg l^{-1} in September. Winter concentrations of another nutrient, Si,

which often limits the spring diatom growth, have changed little over the years.

(d) There are large seasonal variations in the concentrations of nutrients, in part induced by the seasonal growth of phytoplankton. Dissolved silicon is almost entirely removed each spring by diatom growth. Its replacement, mainly from inflow, during the summer is often slow and a factor limiting further diatom maxima in that season. Soluble reactive phosphate (PO_4-P) is also reduced in spring from relatively high winter concentrations, and levels near or below the analytical limit of detection (c. 0.6 µg l^{-1}) prevail until late September. An increase is then induced by the decline of phytoplankton, the upward entrainment of deep water rich in PO_4-P and replacement from inflow. Concentrations of total phosphorus are minimal about June, probably because of sedimentation after spring diatom growth. Although maximal in surface water during the summer abundance of phytoplankton, these algal crops are so depleted in P that a growth-limiting role of the element is likely. Nitrate-nitrogen (NO_3-N) is in relatively high concentrations during winter-spring, but a summer decrease produces minima that are variable from year to year, sometimes being very low (< 20 µg l^{-1}). Ammonium-nitrogen shows a large autumnal pulse due to upward mixing of deep water. Concentrations in other seasons are inconsiderable (< 50 µg l^{-1}), but probably imply a rapid turnover at times of phytoplankton abundance.

The seasonal variation of phytoplankton biomass regularly includes a smaller spring and a larger late-summer maximum. The latter, now usually dominated by the dinoflagellate Ceratium hirundinella, has the greatest effects on water quality and other biota. It is governed by year-to-year differences in physical as well as chemical factors. Additional maxima in spring, early summer, and late autumn can occur but have been less frequent in recent years.

(e) Depth-related aspects of seasonal cycles in the lake are briefly reviewed. Vertical stratification is primarily conditioned by divergences in temperature, typically from late April to early October, with some modifications from weather conditions. Phytoplankton development is mainly in the upper layers during summer, although deep maxima can develop from several causes. Deep accumulations of several nutrients (especially PO_4-P, NH_4-N) invariably form during summer stratification, and their dispersal at autumnal mixing then gives some enrichment to the upper water. Nitrate-N is consumed in the upper layers by algal uptake and microbial denitrification and by the latter process in the anoxic deep water. Carbon dioxide accumulates in deep water, as do Fe^{2+} and Mn^{2+}. Light penetration with depth varies seasonally, being reduced at times of dense phytoplankton, and after autumnal mixing. There is usually a 'clear-water phase' in June, when algal growth may occur in deeper layers.

(f) The long-term variation of algal production involves estimates (for the 0 - 5 m layer) of total phytoplankton biomass, of biomass of major species, and of certain changes in species-composition. There has been minor increase in biomass concentrations of the main summer maxima since records of chlorophyll concentration began in 1964. There was also an abrupt increase in 1963-4 associated with much denser populations of the dinoflagellate Ceratium hirundinella. Year-to-year fluctuations in the size of these maxima exist which are influenced (inter alia) by weather conditions and local accumulations of biomass at the sampling station. The highest values, above 100 µg chlorophyll a l^{-1}, are indicative of a very productive water. The biomass of some blue-green algae and diatoms can make significant contributions to the summer algal growth. For the spring growth, besides the traditional dominants two other species appeared in quantity during the 1970's, possibly in relation to nutrient enrichment.

(g) The long-term variations of some biological reactions upon the lake water have also been traced. Transparency (by Secchi disc) shows regular late-summer reductions by dense phytoplankton, preceded by remarkably clear water in May-June. Deep-water anoxia is also of regular summer occurrence, but its vertical extent varies from year-to-year with weather conditions and from decade to decade with increasing organic production of the lake. Thus in August 1976, a hot summer, anoxia extended up to 3.5 m depth and most of the lake volume was probably anoxic. Such extension may increase the internal loading of nutrients (NH_4-N, PO_4-P) from sediments. Depletion of CO_2, with surface pH above 9.5, now develops in almost every summer by photosynthesis of the dense phytoplankton. Smaller but marked elevations of pH (8.4 - 9.4) were first noted in 1957-62, but not recorded in 1939-40. The pH increase may possibly also promote internal loading of PO_4-P from the sediments.

(h) Submerged macrophytes have been little studied, but the available sources are summarized. The lower depth limit is about 3.8 m, and weed-beds are especially well developed in shallows near the main inflow. There are some subjective indications that macrophytes may have declined in quantity during the recent decades with phytoplankton increase. Possible causes include the summer reduction in light penetration and depletion of carbon dioxide available for photosynthesis.

(i) The annual inputs of nutrients to the lake have been very roughly estimated from analyses of inflows (N and P), from human population within the drainage base (P), and from analyses of sewage effluent (N). The two estimates (total, and Hawkshead sewage, inputs) for PO_4-P are of similar magnitude, 0.59 and 0.48 g per m^2 lake surface, whereas the estimate of

sewage N (NO_3-N + NH_4-N) is much less (1.56 g m^{-2}) than the total estimate for inorganic NO_3-N input (15.0 g m^{-2}). Thus sewage is probably the dominant source of P but not of N. Estimates of the fluxes and availability of P and N are compared with amounts required for appreciable exponential growth of the dense summer phytoplankton. Neither element is in adequate supply, with P much the least; this is consistent with the small N and large P depletion in the maximal phytoplankton biomass. By analogy with a neighbouring lake, bacterial denitrification is estimated to account for only about 12% of the nitrate available, and net algal uptake some 29%. Under stratified summer conditions the vertical movement of N & P from deep water is probably small, unless promoted by movements of the motile algal dominant, Ceratium hirundinella. Conditions in Esthwaite Water are compared with relationships proposed between P loading, depth and trophic state (Vollenweider) or spring P concentration and summer chlorophyll concentration (Dillon & Rigler). The actual late summer crops of phytoplankton are greatly in excess of those predicted, for which several reasons are offered - including enhancement from external and internal sources of summer loading.

(j) Attention is drawn to the apparent non-coincidence of more abrupt phytoplankton increase (1963-4) and more continued and generally later rise of nutrient status; to the importance of characteristics of one organism, Ceratium hirundinella; to the environmental significance of phytoplankton quality as well as its quantity; to system sensitivity to positive feed-backs which may augment internal loading of nutrients (especially P); to the imminence of several potential limitations to the largest algal crops, though that of P is probably now predominant; and to the necessity for further quantitative study before management decisions can be safely made.

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References

Bark, A.W. 1981. The temporal and spatial distribution of planktonic and benthic protozoan communities in a small productive lake. Hydrobiologia 85, 239-255.

Canter, H.M. & Lund, J.W.G. 1948. Studies on plankton parasites 1. Fluctuations in the numbers of Asterionella formosa Hass. in relation to fungal epidemics. New Phytol. 47, 238-261.

Canter, H.M. & Lund, J.W.G. 1951. Studies on plankton parasites. III. Examples of the interaction between parasitism and other factors determining the growth of diatoms. Ann. Bot. Lond, n.s. 15, 359-371.

Canter, H.M. & Lund, J.W.G. 1953. Studies on plankton parasites. II. The parasitism of diatoms with special reference to lakes in the English Lake District. Trans. Brit. mycol. Soc. 36, 13-37.

Canter, H.M. & Lund, J.W.G. 1968. The importance of Protozoa in controlling the abundance of planktonic algae in lakes. Proc. Linn. Soc. Lond. 179, 203-219.

Carrick, T.R. & Sutcliffe, D.W. 1982. Concentrations of major ions in lakes and tarns of the English Lake District. Occ. Publs Freshwat. biol. Ass. 16, pp. 168.

Collins, V.G. 1960. The distribution and ecology of gram-negative organisms other than Enterobacteriaceae in lakes. J. appl. Bact. 23, 510-516.

Davison, W. 1977. The polarographic measurement of O_2, Fe^{2+}, Mn^{2+} and S^{2-} in hypolimnetic water. Limnol. Oceanogr. 22, 746-753.

Davison, W. 1981. Supply of iron and manganese to an anoxic lake basin. Nature, 290, 241-43.

Davison, W. & Heaney, S.I. 1978. Ferrous iron-sulfide interactions in anoxic hypolimnetic waters. Limnol. Oceanogr. 23, 1194-1200.

Davison, W., Heaney, S.I., Talling, J.F. & Rigg, E. 1981. Seasonal transformations and movements of iron in a productive English lake with deep-water anoxia. Schweiz. Z. Hydrol. 42, 196-224.

Davison, W., Woof, C. & Rigg, E. 1982. The dynamics of iron and manganese in a seasonally anoxic lake; direct measurements of fluxes using sedimentation traps. Limnol. Oceanogr. 27, 987-1003.

Dillon, P.J. & Rigler, F.H. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19, 767-773.

Finlay, B.J. 1981. Oxygen availability and seasonal migrations of ciliated protozoa in a freshwater lake. J. gen. Microbiol. 123, 173-178.

Finlay, B.J. 1982. Effects of seasonal anoxia on the community of benthic ciliated protozoa in a productive lake. Arch. Protistenk. 125, 215-222.

Frempong, E. 1981a. Diel periodicity in the chemical composition of lake phytoplankton. Arch. Hydrobiol. 92, 457-495.

Frempong, E. 1981b. Diel variaton in the abundance, vertical distribution, and species composition of phytoplankton in a eutrophic English Lake. J. Ecol. 69, 919-939.

Frempong, E. 1982. The space-time resolution of phased cell division in natural populations of the freshwater dinoflagellate Ceratium hirundinella. Int. Revue ges. Hydrobiol. 67, 323-339.

Frempong, E. 1983. Diel aspects of the thermal structure and energy budget of a small English lake. Freshwat. Biol., 13, 89-102.

George, D.G. & Heaney, S.I. 1978. Factors affecting the spatial distribution of phytoplankton in a small productive lake. J. Ecol. 66, 133-155.

Gorham, E. 1958. Observations on the formation and breakdown of the oxidized microzone at the mud surface in lakes. Limnol. Oceanogr. 3, 291-298.

Gorham, E., Lund, J.W.G., Sanger, J.E. & Dean, W.E. Jr. 1974. Some relationships between algal standing crop, water chemistry, and sediment chemistry in the English Lakes. Limnol. Oceanogr. 19, 601-617.

Goulder, R. 1974. The seasonal and spatial distribution of some benthic ciliated protozoa in Esthwaite Water. Freshwat. Biol. 4 , 127-147.

Harris, G.P., Heaney, S.I. & Talling, J.F. 1979. Physiological and environmental constraints in the ecology of the planktonic dinoflagellate Ceratium hirundinella. Freshwat. Biol. 9, 413-428.

Healey, F.P. 1975. Physiological indicators of nutrient deficiency in algae. Fish. Mar. Serv. Res. Div. Tech. Rep. 585, 30 pp.

Heaney, S.I. 1976. Temporal and spatial distribution of the dinoflagellate Ceratium hirundinella O.F. Müller within a small productive lake. Freshwat. Biol. 6, 531-542.

Heaney, S.I., Chapman, D.V. & Morison, H.R. 1983. The role of the cyst stage in the seasonal growth of the dinoflagellate Ceratium hirundinella within a small productive lake. Br. phycol. J. 18, 47-59.

Heaney, S.I., Smyly, W.J.P. & Talling J.F. Interrelations in a system of physical, chemical and biological changes within a productive lake during summer stratification. (in prep).

Heaney, S.I. & Talling, J.F. 1980. Dynamic aspects of dinoflagellate distribution patterns in a small productive lake. J. Ecol. 68, 75-94.

Heller, M.D. 1977. Interrelationships between photosynthetic production and population dynamics of freshwater algae in a productive English lake. Ph.D. thesis, University of Lancaster.

Heron, J. 1961. The seasonal variation of phosphate, silicate and nitrate in waters of the English Lake District. Limnol. Oceanogr. 6, 338-346.

Jones, J.G. 1971. Studies on freshwater bacteria : factors which influence the population and its activity. J. Ecol. 59, 593-613.

Jones, J.G. 1972a. Studies of freshwater bacteria : association with algae and alkaline phosphatase activity. J. Ecol. 60, 59-75.

Jones, J.G. 1972b. Studies on freshwater micro-organisms; phosphatase activity in lakes of differing degrees of eutrophication. J. Ecol. 60, 777-791.

Jones, J.G. 1977. The effect of environmental factors on estimated viable and total populations of planktonic bacteria in lakes and experimental enclosures. Freshwat. Biol. 7, 67-91.

Jones, J.G. 1978. The distribution of some freshwater planktonic bacteria in two stratified eutrophic lakes. Freshwat. Biol. 8, 127-140.

Jones, J.G. 1981. The population ecology of iron bacteria (genus Ochrobium) in a stratified eutrophic lake. J. gen. Microbiol. 125, 85-93.

Jones, J.G. & Simon, B.M. 1981. Differences in microbial decomposition processes in profundal and littoral lake sediments, with particular reference to the nitrogen cycle. J. gen. Microbiol., 123, 297-312.

Lund, J.W.G. 1949. Studies on Asterionella I. The origin and nature of the cells producing seasonal maxima. J. Ecol. 37, 389-419.

Lund, J.W.G. 1950. Studies on Asterionella formosa Hass. II. Nutrient depletion and the spring maximum. Part 1. Observations on Windermere, Esthwaite Water and Blelham Tarn. J. Ecol., 38, 1-14.

Lund, J.W.G. 1954. The seasonal cycle of the plankton diatom, Melosira italica (Ehr.) Kütz. subsp. subarctica O. Mull. J. Ecol. 42, 151-179.

Lund, J.W.G. 1959. Buoyancy in relation to the ecology of the freshwater plankton. Br. phycol. Bull., 1, 1-17.

Lund, J.W.G. 1961. The periodicity of μ-algae in three English Lakes. Verh. int. Ver. Limnol. 14, 147-154.

Lund, J.W.G. 1962. A rarely recorded but very common British alga, Rhodomonas minuta Skuja. Brit. phycol. Bull. 2, 133-139.

Lund, J.W.G. 1966. The importance of turbulence in the periodicity of certain freshwater species of the genus Melosira. Bot. Zh. SSSR. 51, 176-187. (Russian).

Lund, J.W.G. 1972. Changes in the biomass of blue-green and other algae in an English lake from 1945-1969. Taxonomy and Biology of Blue-green Algae (Ed. by T.W. Desikachary), pp. 305-327. University of Madras.

Lund, J.W.G. 1981. Investigations of phytoplankton with special reference to water usage. Occ. Publs Freshwat. biol. Ass., 13, 64pp.

Lund, J.W.G., Mackereth, F.J.H. & Mortimer, C.H., 1963. Changes in depth and time of certain chemical and physical conditions and of the standing crop of Asterionella formosa Hass. in the North Basin of Windermere in 1947. Phil. Trans. R. Soc. Ser. B 246, 255-290.

Macan, T.T. 1970. Biological studies of the English lakes. London. Longman. pp. 260.

Mackereth, F.J.H. 1965. Chemical investigation of lake sediments and their interpretation. Proc. R. Soc. (B), 161, 295-309.

Mackereth, F.J.H. 1966. Some chemical observations and post-glacial lake sediments. Phil. Trans. R. Soc. (B), 250, 165-213.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes; I and II. J. Ecol. 29, 280-329.

Mortimer, C.H. 1942. The exchange of dissolved substances between mud and water in lakes. III and IV. J. Ecol. 30, 147-201.

Mortimer, C.H. 1971. Chemical exchanges between sediments and water in the Great Lakes - speculations on probable regulatory mechanisms. Limnol. Oceanogr. 16, 387-404.

Mundie, J.H. 1965. (Data in Director's Report). Ann. Rep. Freshwat. Biol. Ass. 33, 39.

Pearsall, W.H. 1917-18. The aquatic and marsh vegetation of Esthwaite Water. J. Ecol. 5, 180-202, 6, 53-74.

Pearsall, W.H. 1932. Phytoplankton in the English Lakes. II. The composition of the phytoplankton in relation to dissolved substances. J. Ecol. 20, 241-261.

Pennington, W. 1974. Seston and sediment formation in five Lake District lakes. J. Ecol. 62, 215-251.

Pigott, C.D. & Wilson, J.F. 1978. The vegetation of North Fen at Esthwaite in 1967-9. Proc. R. Soc. (B) 200, 331-351.

Ramsbottom, A.E. 1976. Depth charts of the Cumbrian Lakes. Sci. Publ. Freshwat. Biol. Ass. 33, 39pp.

Scholkovitz, E.R. & Copland, D. 1982. The chemistry of suspended matter in Esthwaite Water, a biologically productive lake with seasonally anoxic hypolimnion. Geochim. cosmochim. Acta , 46, 393-410.

Smyly, W.J.P. 1961. The life-cycle of the freshwater copepod Cyclops leuckarti Claus in Esthwaite Water. J. Anim. Ecol. 30, 153-171.

Smyly, W.J.P. 1972. The crustacean zooplankton, past and present, in Esthwaite Water. Verh. int. Verein. theor. angew. Limnol. 18, 320-326.

Smyly, W.J.P. 1973a. Decrease in abundance of the freshwater ostracod Cypria opthalmica (Jurine), (Crustacea, Ostracoda) in a small lake. Naturalist, Hull 925, 65-69.

Smyly, W.J.P. 1973b. Bionomics of Cyclops strenuus abyssorum Sars (Copepoda: Cyclopoida). Oecologia 11, 163-186.

Smyly, W.J.P. 1978. Limnetic crustaceans in Esthwaite Water and adjacent lakes. Rep. Freshwat. biol. Ass. 46, 70-73.

Stokoe, R. 1983. Aquatic macrophytes in the tarns and lakes of Cumbria. Occ. Publ. Freshwat. Biol. Ass. 18, 60 pp.

Strachan, I.M. 1980. Feeding and population ecology of Acanthocyclops bicuspidatus (sensu stricta) (Claus) in Esthwaite Water, Cumbria. Ph.D. thesis, University of Lancaster.

Sutcliffe, D.W., Carrick, T.R., Heron, J., Rigg, E., Talling, J.F., Woof, C. & Lund, J.W.G. 1982. Long-term and seasonal changes in the chemical composition of precipitation and surface waters of lakes and tarns in the English Lake District. Freshwat. Biol. 12, 451-506.

Talling, J.F. (1957). Photosynthetic characteristics of some freshwater plankton diatoms in relation to underwater radiation. New Phytol. 56, 29-50.

Talling, J.F. 1971. The underwater light climate as a controlling factor in the production ecology of freshwater phytoplankton. Mitt. int. Verein. theor. angew. Limnol. 19, 214-243.

Talling, J.F. 1976. The depletion of carbon dioxide from lake water by phytoplankton. J. Ecol. 64, 79-121.

Talling, J.F. 1979. Factor interactions and implications for the prediction of lake metabolism. Ergebn. Limnol. 13, 96-109.

Talling, J.F. 1981. The development of attenuance depth-profiling to follow the changing distribution of phytoplankton and other particulate material in a productive English lake. Arch. Hydrobiol. 93, 1-20.

Tansley, A.G. 1939. The British Islands and their vegetation. Cambridge Univ. Press. 930 pp.

Tipping, E. 1981a. Adsorption by goethite (α-FeOOH) of humic substances from three different lakes. Chem. Geol., 33, 81-89.

Tipping, E. 1981b. The adsorption of aquatic humic substances by iron oxides. Geochim. cosmochim. Acta, 45, 191-199.

Tipping, E. & Wood, C. 1983. Seasonal variations in the concentrations of humic substances in a soft-water lake. Limnol Oceanogr. 28, 168-172.

Tipping, E. Woof, C. & Cooke, D. 1981. Iron oxide from a seasonally anoxic lake. Geochim. cosmochim. Acta , 45, 1411-1419.

Tipping, E. Woof, C. & Ohnstad, M. 1982. Forms of iron in the oxygenated waters of Esthwaite Water, U.K. Hydrobiologia, 91/92, 383-393.

Vollenweider, R.A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors in eutrophication. O.E.C.D. report DAS/CSI 68, 27.

Vollenweider, R.A. & Kerckes, J. 1980. The loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. Prog. Wat. Tech. 12, 5-38.

Webb, M.G. 1961. The effects of thermal stratification on the distribution of benthic Protozoa in Esthwaite Water. J. Anim. Ecol. 30, 137-151.

Wood, R.B. & Gibson, C.E. 1973. Eutrophication and Lough Neagh. Water Research, 7, 173-187.

From: Sutcliffe et al. 1982

APPENDIX C (PO_4-P, $\mu g\ l^{-1}$)

Year	Blelham	HC	Esthwaite	HC	Windermere (N)	HC	Windermere (S)	HC
1945	—	—	—	—	3.3 (6) 1.1	5.0	—	—
1946	1.7 (12) 0.7	4.4	2.5 (10) 0.5	3.8	2.2 (12) 0.2	3.0	2.5 (12) 0.5	3.6
1947	3.7 (10) 0.8	5	2.5 (10) 0.4	3	2.3 (10) 0.3	3.0	2.3 (10) 0.5	4.0
1948	1.8 (11) 0.2	2	1.9 (12) 0.4	3	1.8 (9) 0.3	2.5	2.2 (9) 0.5	3.2
1949	1.2 (10) 0.3	1.5	1.3 (10) 0.2	2	1.1 (9) 0.1	1.4	1.1 (8) 0.2	2.0
1950	1.2 (6) 0.4	2	1.1 (10) 0.1	1.3	1.1 (7) 0.4	2.0	1.5 (7) 0.4	2.0
1951	1.0 (5) 0	1	1.1 (10) 0.1	1.3	1.1 (7) 0.3	1.5	1.0 (6) 0.2	1.5
1952	1.0 (9) 0	1	1.0 (10) 0	1	1.0 (11) 0	1.0	1.0 (11) 0	1.0
1953	1.6 (8) 0.4	2	1.4 (10) 0.3	2	1.4 (11) 0.3	2.0	1.4 (12) 0.4	3.0
1954	4.7 (10) 0.9	5	2.9 (12) 0.9	6	4.3 (8) 0.9	6.0	2.7 (9) 2.0	5.0
1955	3.4 (8) 1.2	6	4.7 (10) 0.3	5	1.8 (10) 0.6	3.0	2.6 (9) 0.6	4.0
1956	—	—	—	—	—	—	—	—
1957	1.9 (7) 1.4	4.1	1.4 (8) 0.5	2.2	0.3 (7) 0.2	7.0	0.4 (8) 0.2	0.9
1958	3.6 (4) 1.4	4.7	—	—	2.5 (4) 0.6	2.7	3.1 (4) 2.6	5.2
1959	2.8 (8) 0.8	4.4	1.7 (11) 1.1	2.5	2.8 (11) 0.2	3.7	2.1 (11) 0.5	3.1
1960	5.3 (12) 1.3	9.2	5.5 (11) 1.1	7.1	2.1 (13) 0.3	2.9	2.6 (13) 0.6	4.0
1961	4.0 (10) 0.9	6.7	1.6 (12) 0.5	3.1	2.5 (10) 0.4	3.6	3.3 (12) 0.4	4.3
1962	4.9 (6) 1.0	5.7	2.3 (6) 0.6	9.0	4.7 (7) 0.9	5.8	2.9 (9) 0.6	4.3
1963	2.3 (11) 0.9	4.6	1.9 (11) 0.5	3.1	2.3 (10) 0.7	4.2	1.0 (11) 0.4	1.8
1964	6.6 (10) 1.1	8.7	1.0 (10) 0.2	3.1	3.3 (12) 0.2	3.7	4.4 (12) 0.7	5.6
1965	7.3 (5) 4.9	13.7	3.7 (6) 0.7	8.0	3.2 (12) 0.5	4.1	7.0 (12) 0.3	7.9
1966	8.4 (11) 1.8	13.5	1.8 (10) 0.5	4.9	2.8 (13) 0.4	3.6	5.5 (13) 0.5	6.6
1967	9.1 (11) 1.5	12.0	1.6 (10) 0.3	4.1	2.6 (13) 0.4	3.9	7.0 (12) 0.5	8.7
1968	5.1 (9) 1.1	8.7	2.2 (11) 0.6	4.8	3.3 (12) 0.3	4.4	7.5 (12) 0.6	9.0
1969	10.9 (10) 0.9	13.0	2.1 (10) 0.5	3.2	4.0 (11) 0.5	5.2	10.3 (11) 0.8	1.3
1970	10.1 (6) 1.5	13.0	5.1 (10) 1.0	7.5	3.4 (12) 0.4	4.7	8.7 (10) 1.0	10.0
1971	7.6 (11) 1.2	9.1	4.0 (9) 0.9	6.4	4.3 (10) 0.3	4.9	12.8 (12) 1.2	15.0
1972	9.7 (12) 1.0	17.0	7.5 (11) 0.3	8.0	5.2 (11) 0.5	6.7	18.3 (12) 0.8	20.0
1973	11.0 (10) 1.2	17.0	3.8 (11) 0.8	7.5	4.5 (9) 0.3	5.0	13.2 (12) 1.2	15.4
1974	8.0 (12) 0.8	10.2	7.5 (8) 0.5	8.2	3.7 (9) 0.3	4.4	11.5 (12) 1.3	14.6
1975	7.4 (11) 1.1	9.4	9.3 (9) 1.2	10.8	3.9 (9) 0.3	4.3	13.5 (12) 0.9	15.9
1976	4.5 (12) 0.5	5.7	6.0 (11) 0.6	7.6	4.1 (10) 0.4	4.7	17.3 (11) 1.8	21.6
1977	2.0 (7) 1.3	4.7	5.7 (11) 0.8	8.3	3.7 (9) 0.3	4.4	16.6 (7) 1.5	18.0
1978	5.8 (10) 0.9	8.0	12.9 (8) 1.1	13.9	4.4 (8) 0.4	6.5	15.5 (7) 0.9	16.5
1979	7.7 (4) 5.4	12.0	11.6 (7) 1.4	13.3	5.9 (10) 0.4	6.6	16.2 (12) 0.5	17.3
1980	6.9 (6) 1.7	9.9	11.9 (5) 3.5	16.1	3.8 (6) 0.8	4.9	11.8 (6) 4.0	17.5

Table 1. Long-term variation in the autumn-winter concentrations of PO_4-P in four English lake basins, including Esthwaite Water (0-5 m layer). The four values given for each basin are: mean value of consecutive determinations in the period, number of determinations used, interval (±) for 95% confidence limits, and highest concentration (HC).

From: Sutcliffe et al. 1982

APPENDIX B (NO_3-N, $\mu g\ l^{-1}$)

Year	Blelham	HC	Esthwaite	HC	Windermere (N)	HC	Windermere (S)	HC
1939	—	—	—	—	310 (9) 40	360	320 (7) 40	370
1945	—	—	—	—	240 (9) 40	300	230 (9) 35	330
1946	460 (10) 65	610	350 (10) 50	510	280 (14) 15	330	300 (10) 20	340
1947	450 (10) 30	500	410 (10) 60	560	370 (11) 25	400	320 (9) 50	400
1948	520 (10) 50	640	490 (10) 35	600	340 (10) 35	400	380 (10) 45	480
1949	450 (10) 80	650	320 (10) 35	380	320 (12) 15	350	290 (10) 20	350
1950	400 (10) 40	500	430 (10) 55	600	328 (15) 10	360	310 (10) 15	400
1951	360 (10) 30	420	360 (10) 20	420	335 (12) 10	360	320 (10) 15	350
1952	390 (10) 20	440	360 (10) 30	460	320 (11) 20	370	310 (10) 15	380
1953	370 (10) 20	400	320 (10) 25	370	300 (10) 20	330	300 (10) 20	340
1954	470 (10) 40	530	430 (10) 35	500	320 (10) 20	380	320 (10) 20	360
1955	270 (10) 20	330	260 (10) 15	300	260 (13) 20	290	250 (10) 10	270
1956	650 (10) 20	700	360 (10) 30	420	290 (12) 20	330	280 (10) 20	330
1957	490 (10) 30	560	400 (10) 40	500	305 (12) 25	370	330 (10) 20	400
1958	540 (10) 40	610	460 (10) 35	580	270 (13) 15	320	290 (10) 20	360
1959	740 (10) 40	820	400 (10) 30	460	320 (13) 25	390	330 (10) 45	450
1960	370 (10) 30	430	370 (10) 30	450	250 (15) 15	310	290 (10) 20	360
1961	490 (10) 20	510	340 (10) 20	390	240 (11) 20	300	290 (10) 15	320
1962	480 (10) 40	540	420 (10) 20	470	300 (11) 20	340	330 (10) 15	350
1963	520 (10) 60	620	450 (10) 25	490	340 (10) 20	380	270 (10) 15	320
1964	560 (10) 20	600	420 (10) 20	470	380 (11) 10	420	370 (10) 30	420
1965	490 (10) 20	530	540 (10) 15	590	410 (10) 20	470	440 (9) 10	460
1966	570 (10) 15	600	540 (10) 20	600	430 (10) 10	450	440 (10) 50	450
1967	510 (10) 30	570	480 (10) 25	530	360 (10) 15	380	350 (10) 20	390
1968	690 (10) 30	730	510 (10) 20	550	360 (10) 15	380	370 (10) 15	390
1969	860 (10) 40	920	620 (10) 25	690	410 (10) 10	440	420 (10) 15	450
1970	1270 (10) 50	1390	980 (10) 30	1080	580 (10) 30	640	510 (10) 65	640
1971	1010 (10) 70	1140	900 (10) 50	1030	550 (10) 15	570	560 (10) 20	600
1972	1000 (10) 70	1120	880 (10) 40	990	590 (11) 20	640	600 (10) 20	660
1973	548 (10) 32	590	547 (10) 38	613	424 (14) 9	455	496 (10) 10	530
1974	719 (10) 48	812	713 (10) 45	809	411 (12) 25	476	448 (10) 27	504
1975	412 (10) 28	490	509 (10) 18	546	356 (9) 13	378	438 (10) 14	462
1976	716 (8) 25	767	621 (10) 48	720	398 (9) 29	454	441 (7) 14	473
1977	1206 (10) 78	1337	1098 (10) 41	1156	633 (8) 14	651	653 (9) 27	705
1978	764 (5) 70	863	782 (5) 14	796	520 (12) 18	599	552 (10) 19	611
1979	1420 (10)182	1813	1151 (8) 56	1258	727 (10) 55	795	666 (10) 50	719
1980	617 (11) 54	755	644 (11) 69	826	567 (10) 41	654	538 (9) 39	619

Table 2. Long-term variation in the winter-spring concentrations of NO_3-N in four English lake basins, including Esthwaite Water (0-5 m layer). The four values given for each basin are: mean value of consecutive determinations in the period, number of determinations used, interval (±) for 95% confidence limits, and highest concentration (HC). No adjustment made for the differences in analytical methods noted in text.

1968	A	B	C	D	E	F	G	H	I	J	K	L
June	20	34	17	17	199	1128	31	495	3·4	19	0·53	8·4
June	27	98	49	49	158	566	19	208	7·7	28	0·93	10·4
July	4	343	171	171	18·1	340	8	130	3·1	58	1·36	22·2
July	11	503	251	251	88·0	686	14	300	22·0	172	3·50	75·5
July	18	234	117	117	148	1970	16	375	17·3	230	1·88	43·9
July	25	99	49	49	180	1140	38	565	8·8	56	1·86	27·7
Aug.	1	53	26	26	367	2136	30	620	9·5	55	0·78	16·1
Aug.	8	34	17	17	425	1500	31	1170	7·2	26	0·53	19·9
Aug.	15	21	10·5	10·5	225	1110	120	1250	2·4	12	1·26	13·1
Aug.	22	109	55	55	133	955	21	287	7·5	53	1·15	15·8
Aug.	29	132	66	66	540	1695	48	587	35·6	112	3·16	38·8
Sept.	5	153	77	77	68	917	27	200	5·2	71	2·08	15·4
Sept.	12	224	112	112	155	(1000)	21	(200)	17·3	112	2·35	22·4
Sept.	19	303	151	151	106	632	13	212	16·0	95	1·96	32·0
Sept.	26	1195	597	597	16·8	554	5	120	10·0	330	2·98	71·5
Oct.	3	2570	1285	1285	10·2	271	3·8	160	13·1	350	4·90	206
Oct.	10	1067	533	533	50·0	775	5	175	26·6	410	2·66	93·6
Oct.	17	844	422	422	15·3	617	3·8	170	6·5	260	1·60	71·8
Oct.	24	450	225	225	28·6	710	2·8	185	6·5	160	0·63	41·6
Oct.	31	650	325	325	8·3	584	2·4	140	2·7	190	0·78	45·5
Nov.	7	1920	960	960	15·9	770	2·5	212	15·3	740	2·40	203
Nov.	14	367	183	183	25·4	844	3·5	175	4·7	154	0·64	32·0
Nov.	21	199	100	100	22·0	854	5·1	215	2·2	85	0·51	21·5
Nov.	28	1090	545	545	10·5	587	3·0	235	5·7	320	1·63	128
Dec.	5	554	277	277	46·6	1000	8·0	317	12·9	277	2·22	88·0
Dec.	12	313	157	157	21·8	1350	4·6	337	3·4	212	0·72	52·8
Dec.	19	262	131	131	24·4	(800)	3·0	(200)	3·2	105	0·39	26·2
Dec.	26	893	447	447	(15)	(600)	(3·0)	(200)	6·7	270	1·34	89·5
1969												
Jan.	2	405	203	203	14·3	1290	4·5	520	2·9	260	0·92	106
Jan.	9	228	114	114	64·6	680	10·5	345	7·4	78	1·20	39·6
Jan.	16	313	157	157	11·6	1005	3·2	475	1·8	158	0·50	74·8
Jan.	23	1514	757	757	9·2	1360	3·7	615	7·0	1030	2·80	465
Jan.	30	1090	545	545	8·3	1070	2·9	491	4·5	580	1·58	268
Feb.	6	686	343	343	19·7	1020	4·6	517	6·8	350	1·58	177
Feb.	13	478	239	239	15·0	960	4·2	497	3·6	230	1·00	119
Feb.	20	351	175	175	(15·0)	(1000)	3·4	445	2·6	175	0·60	78·0
Feb.	27	312	156	156	20·6	1130	3·4	445	3·2	176	0·53	69·5
Mar.	6	259	129	129	35·6	1150	5·7	477	4·6	148	0·74	61·5
Mar.	13	194	97	97	40·6	1050	7·0	750	4·0	102	0·68	72·6
Mar.	20	260	130	130	38·7	955	5·5	611	5·0	124	0·72	79·5
Mar.	27	266	133	133	67·6	1080	6·9	670	9·0	144	0·92	89·0
April	3	434	217	217	44·0	1390	3·4	574	9·5	300	0·74	125
April	10	426	213	213	33·0	850	3·7	350	7·0	180	0·79	74·5
April	17	743	371	371	24·0	1120	3·7	330	8·9	415	1·37	123
April	24	477	238	238	75·2	1070	4·0	350	17·8	256	0·95	83·5
May	1	441	220	220	33·1	930	6·7	370	7·3	204	1·47	81·5
May	8	310	155	155	36·1	790	5·7	352	5·6	183	0·88	54·5
May	15	608	304	304	15·8	795	4·5	280	4·8	242	1·36	85·0
May	22	435	217	217	124	1325	4·4	405	27·0	286	0·95	88·0
May	29	277	139	139	166	755	32	197	23·1	105	4·45	27·4
June	5	315	158	158	(150)	(1000)	(30)	(300)	23·7	158	4·75	47·5
June	12	283	142	142	184	1115	38·2	554	26·2	159	5·42	78·5

A Week ending.
B Cunsey flow 10^3 m^3 (week).
C E.B. flow 10^3 m^3 (week).
D Other inflows 10^3 m^3 (week).
E Conc. P E.B. µg/lP.
F Conc. NO_3 E.B. µg/lN.
G Mean conc. P µg/l. others.
H Mean conc. N µg/l. others.
I E.B. P input (kg.)
J E.B. N input (kg.)
K Others input P (kg.)
L Others input N (kg.)

L. volume = 6.4×10^6 m^3
assumed E.B. flow = ½ total flow.

Total flow Cunsey = 25.8×10^6 m^3
flow E.B. 12.9×10^6 m^3
flow others 12.9×10^6 m^3

E.B. input P Total PO_4P 505·6 kg.
E.B. NO_3N input Total 10,945 kg.
Others input P Total PO_4P 83·6 kg.
Others input NO_3N 4,100 kg.

Theoretical average lake comp. 22·8 µg/l PO_4P
580 µg/l NO_3N
if E.B. had same composition as other inflow streams (i.e. sewage removed) the composition of Esthwaite would be 6·5 µg/l PO_4P
318 µg/l NO_3N
years average E.B. composition 39·2 µg/l PO_4P
845 µg/l NO_3N

Total PO_4-P input to lake 589.2 kg
Total NO_3-N input to lake 15,045 kg
Lake area 1.00 km^2
PO_4-P loading 0.59 g m^{-2}
NO_3-N loading 15.0 g m^{-2}

Table 4 Estimated contributions from various sources at Hawkshead of phosphorus to sewage works, 1982.

		kg P
residents, 657		
annual phosphorus contribution 657 x 365 x 1.8	=	432
campsite, 72 pitches, 3 persons per pitch, 205 days,		
50% occupancy 72 x 3 x 205 x 1.8 ÷ 2	=	40
cars, 97058, 2.4 persons per car		
97058 x 2.4 x 0.3	=	70
coaches, 517, 40 persons per coach		
517 x 40 x 0.3	=	6
Annual loading to sewage works	=	548

Assuming that 88% of phosphorus is soluble and passes through the sewage works and discharged into Black Beck, then the 1982 phosphorus loading to Esthwaite Water from this source was 482 kg.

Easter - 31 October	kg P
residents, 657 x 205 x 1.8	249
campsite, 72 x 3 x 205 x 1.8 ÷ 2	40
cars, 97058 x 2.4 x 0.3	70
coaches, 517 x 40 x 0.3	6
phosphorus input to sewage works	= 365
phosphorus input to Esthwaite water (88%)	= 321

Table 5 Nitrogen loadings to Esthwaite Water from Hawkshead sewage works for 1981, 1982.

Year			kg N
1981	Total oxidised nitrogen (TON) + NH_4-N		
	January - March + October - December	TON	951
	" " " "	NH_4-N	200
	April - September	TON	42
	" "	NH_4-N	1569
			2762
1982	January - March + October - December	TON	764
	" " " "	NH_4-N	105
	April - September	TON	42
	" "	NH_4-N	652
			1563

Table 6 Example of the contents of PO_4-P, NO_3+NO_2-N and NH_4-N in the upper mixed layer of Esthwaite Water (after hypsographic correction for decreasing volume with depth), and the calculated weekly external input of these nutrients in relation to the total particulate phosphorus and nitrogen in the lake near the time of maximum summer biomass.

date internal content	depth	constituent	kg
30 August 1972	0 - 6 m	NO_3-N	432
"	"	NH_4-N	84
"	"	PO_4-P	20.5
weekly external input			
		NO_3-N*	183
		NH_4-N†	61
		PO_4-P*	11
external input + internal content			
		NO_3+NH_4-N	760
		PO_4-P	31.5
30 August 1982, internal content	0 - 6 m layer	particulate N	960
"	"	" P	113
external input + internal contents as % of standing stock in 0 - 6 m layer		N	79%
		P	28%

* 1968 - 1969 nutrient input analyses (Mackereth)

† 1982 effluent discharge by Hawkshead sewage works

Table 7 Estimates of bacterial denitrification in sediments of Esthwaite Water using the annual nitrate input from a study in 1968-69 and sediment-derived rates of microbial denitrification, as N_2 gas released, from Blelham Tarn (Jones & Simon 1981).

littoral sediments April - September	
N_2 gas released	7.0 mg N m^{-2} day^{-1}
area of littoral sediments (above 5 m depth)	450 x 10^3 m^2
daily denitrification by littoral sediments	3.15 kg d^{-1}
denitrification by littoral sediments April - September	576 kg
profundal sediments April - September	
N_2 gas released	3.3 mg N m^{-2} day^{-1}
areas of profundal sediments (below 5 m depth)	550 x 10^3 m^2
daily denitrification by profundal sediments	1.82 kg day^{-1}
denitrification by profundal sediments April - September	332 kg

Assuming that for the non-stratified period, October - March, the daily rate of denitrification for all sediments is the same as the summer rate in the profundal then,

N_2 gas released	3.3 mg N m^{-2} day^{-1}
lake area	1 km^2
daily denitrification of lake sediments	3.3 kg day^{-1}
denitrification by profundal sediments October - March	600 kg
Annual denitrification by lake sediments	1508 kg
Annual nitrate input to lake	15,045 kg
Annual sedimentary denitrification as % of nitrate input	10%

Table 8 Nitrate losses as percentages of nitrate availability in Esthwaite Water from mid May - end August 1971

NO_3-N input mid May - August (Mackereth 1968-69)	2061 kg*	
NO_3-N lake content mid May 1971	2342 kg**	
NO_3-N loading mid May - August 1971	4403 kg	
		% total
microbial denitrification mid May - August	532 kg	12
increase in planktonic particulate N mid May - August	1274 kg	29
outflow NO_3-N mid May - August	1324 kg*	30
		71

* NO_3-N determined by cadmium reduction method

** NO_3-N determined by phenol disulphonic acid method in 1971. The determinations assumed to be 70% of values obtained from cadmium reduction method and revised upwards accordingly.

TABLE 9

Preliminary classification table for the assessment of trophic state in lakes in the OECD eutrophication program. Trophic state is assigned based on the opinion of the investigator of each lake. Linear Averages and Standard Deviations.

Variable (Annual mean values)	Oligotrophic	Mesotrophic	Eutrophic	Hyper Eutrophic
Total phosphorus mg/m^3	mean = 8.6 ± 3.69 N = 20 range 3 - 16.1	25.1 ± 16.33 N = 18 5.6 - 80	113. ± 89.4 N = 73 8.8 - 386	---------- N = 2 750 - 1200
Total nitrogen mg/m^3	775 ± 476 N = 11 306 - 1630	798 ± 364 N = 7 361 - 1387	2367 ± 1536 N = 38 300 - 6100	
Chlorophyll a mg/m^3	1.8 ± 0.92 N = 21 0.2 - 3.6	5.2 ± 2.46 N = 16 2.2 - 11.0	19.1 ± 15.87 N = 72 2 - 78	N = 2 100 - 150
Chlorophyll a maximum value mg/m^3	4.62 ± 1.96 N = 15 1.3 - 8.6	16.0 ± 6.44 N = 11 5.9 - 24.8	63.7 ± 61.34 N = 46 9.5 - 275.	
Secchi depth m	11.1 ± 6.95 N = 13 5.4 - 28.0	4.8 ± 2.25 N = 21 1.9 - 8.1	2.84± 1.64 N = 79 0.8 - 7.0	N = 2 0.4 - 0.5

* Esthwaite Water

From: Vollenweider and Kerekes 1980

TABLE 9

From: Ramsbottom 1976

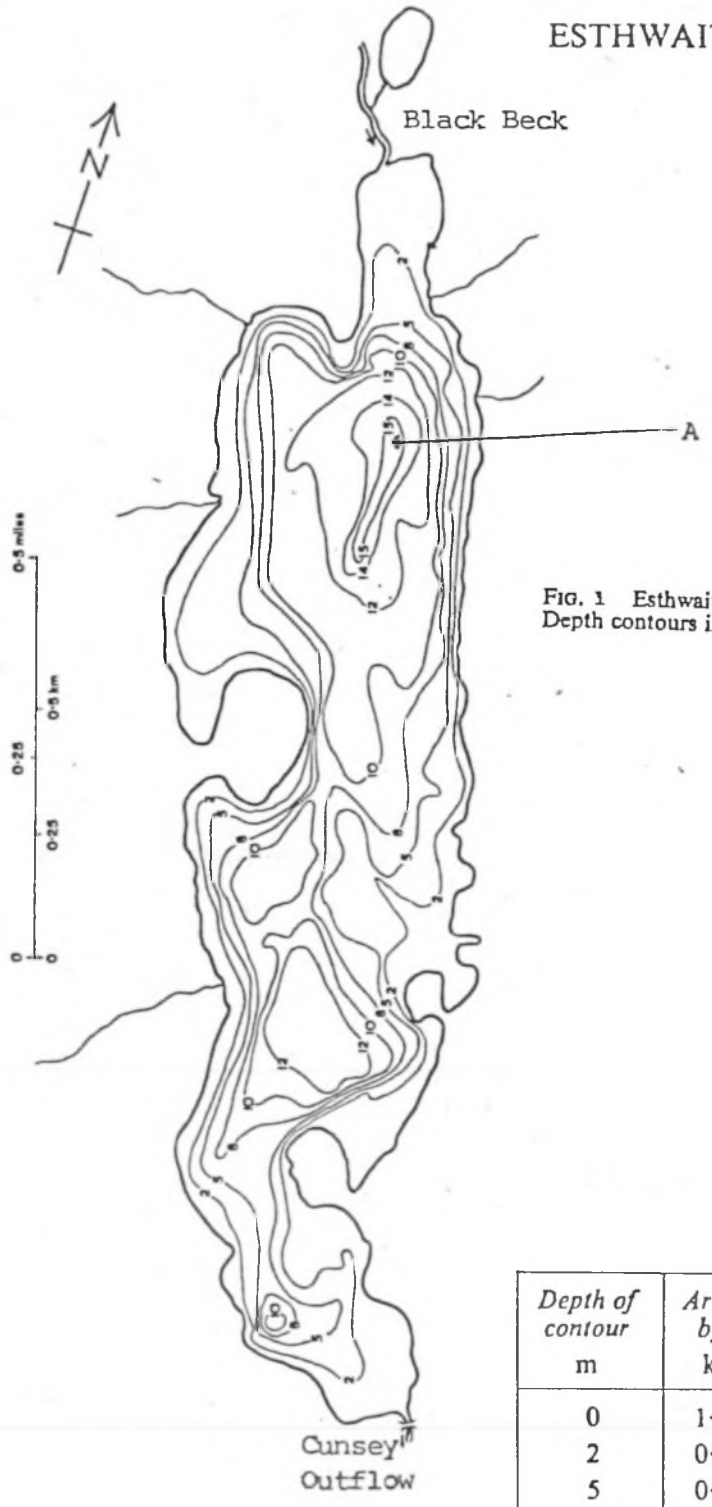


FIG. 1 Esthwaite Water.
Depth contours in metres.

ESTHWAITE WATER

Depth of contour m	*Area enclosed by contour* km²	%	*Layer* m	*Volume of layer* m³×10³	%
0	1·004	100	0-2	1·758	27
2	0·754	75	2-5	1·962	30
5	0·554	55	5-8	1·445	22
8	0·409	41	8-10	0·698	11
10	0·289	29	10-12	0·411	6
12	0·122	12	12-14	0·150	2
14	0·028	3	14-15	0·018	0·3
15	0·008	0·8	15-15·5	0·002	0·1
15·5 (bottom)					

Max. depth: 15·5 m Mean depth: 6·4 m Total Volume: 6·4 m³×10⁶

Fig. 2. Aerial photograph of Esthwaite Water (Ordnance Survey photograph).

FIG. 3(A)

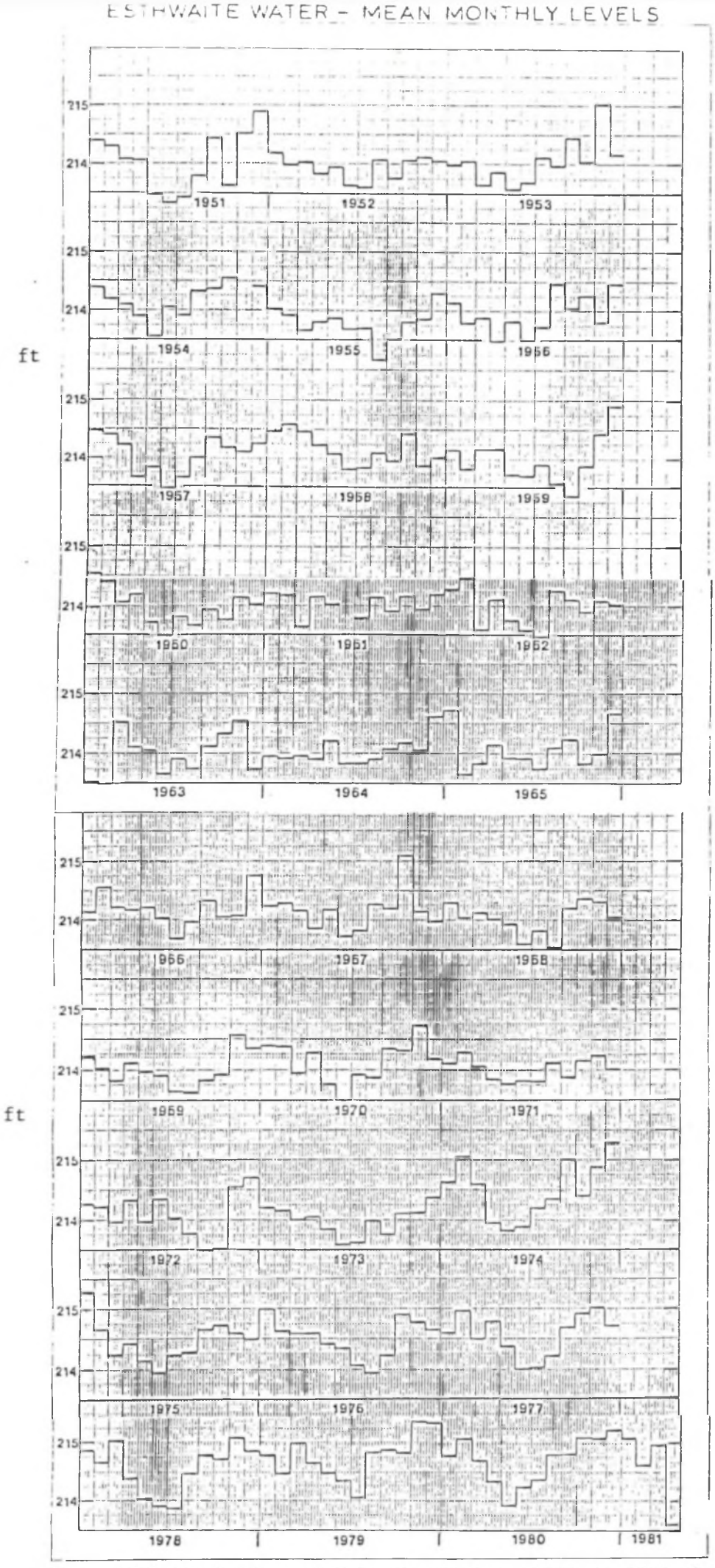
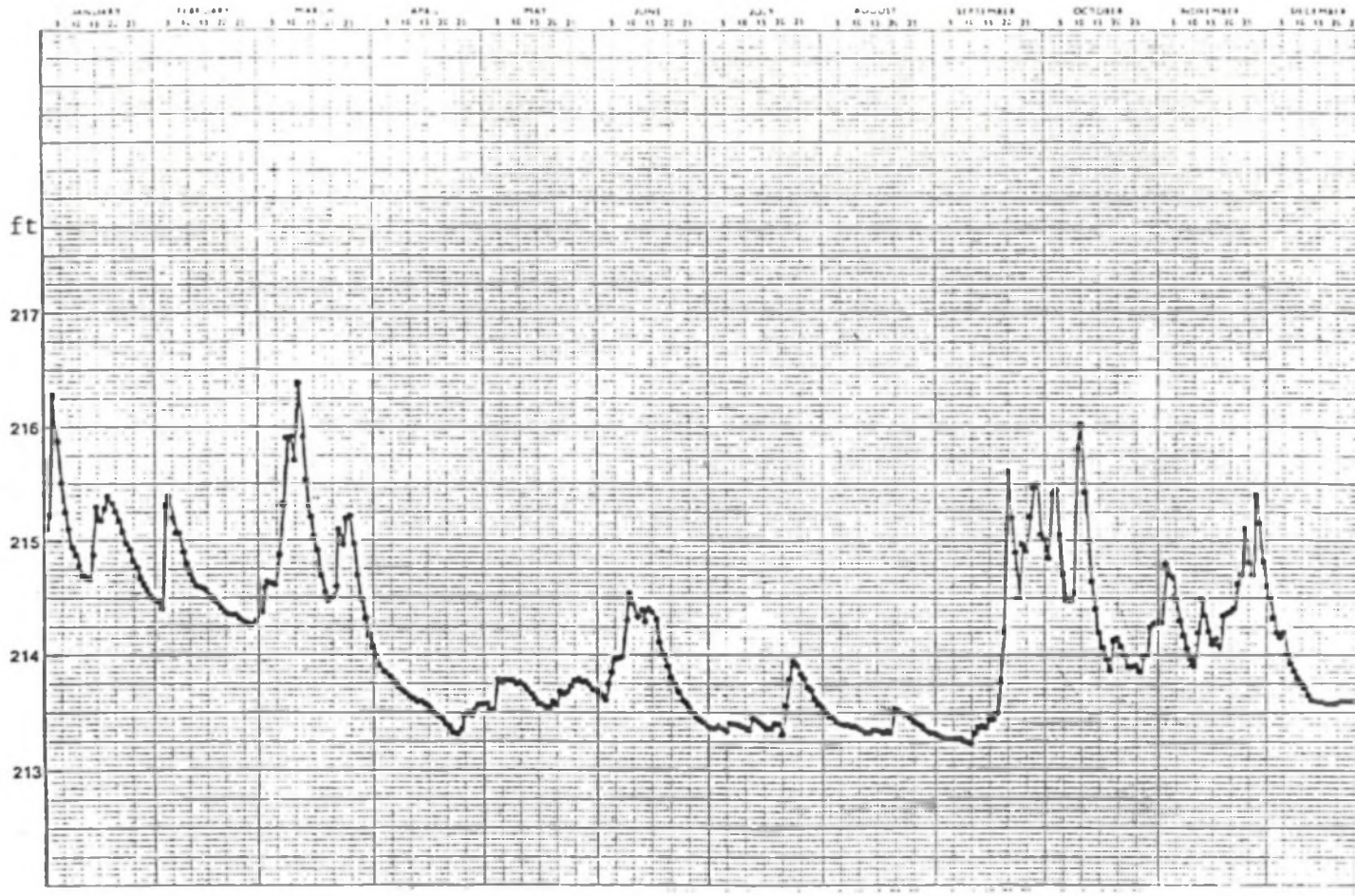


Fig. 3(a) Variation of lake level from 1951 to 1981 (note: one or more shifts in gauge zero are suspected).

ESTHWAITE WATER — DAILY LEVELS 1981



ESTHWAITE WATER - DAILY LEVELS 1982

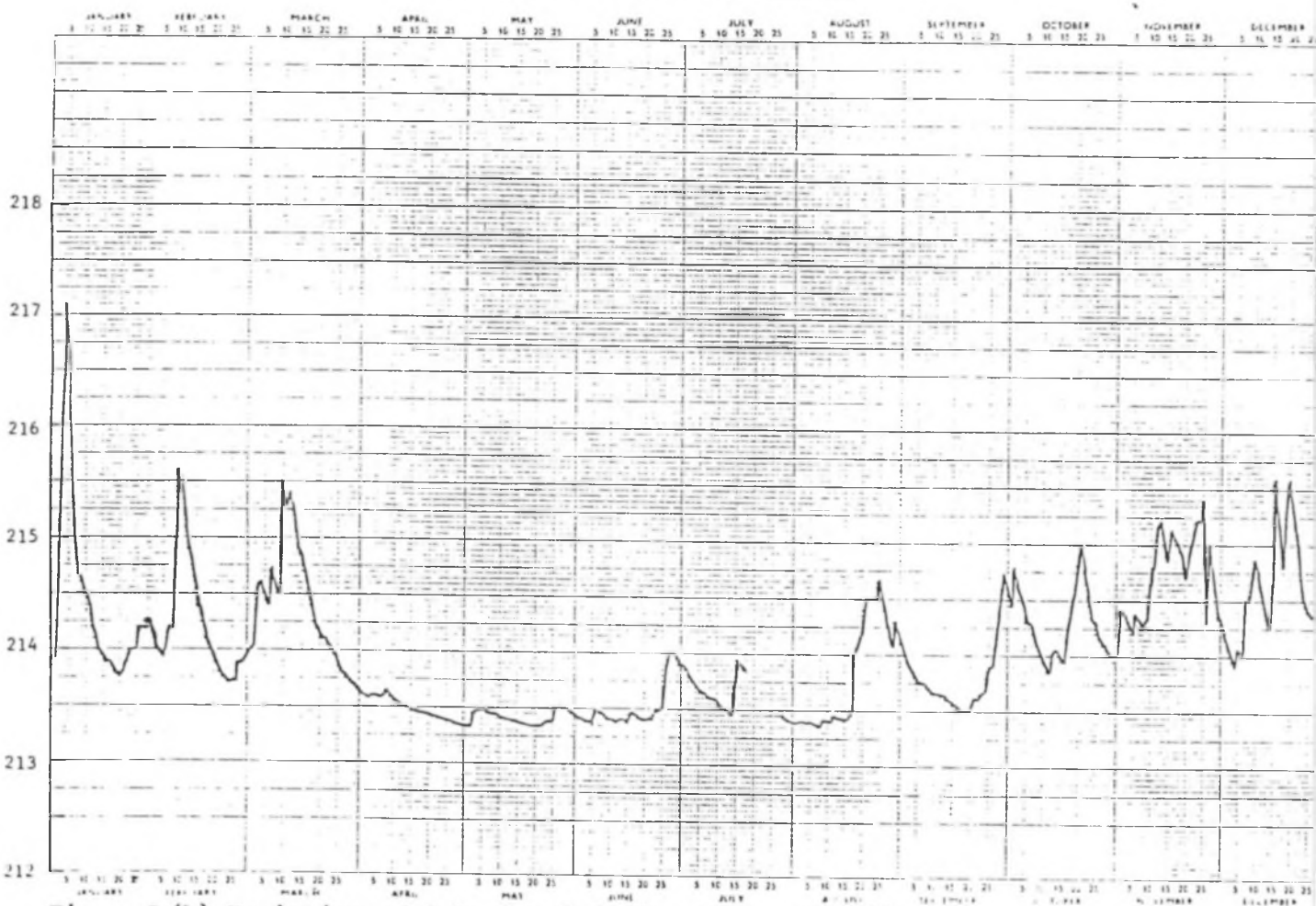


Fig. 3 (b) Variation of lake level during 1981 and 1982.

From: Heller 1977 and unpublished

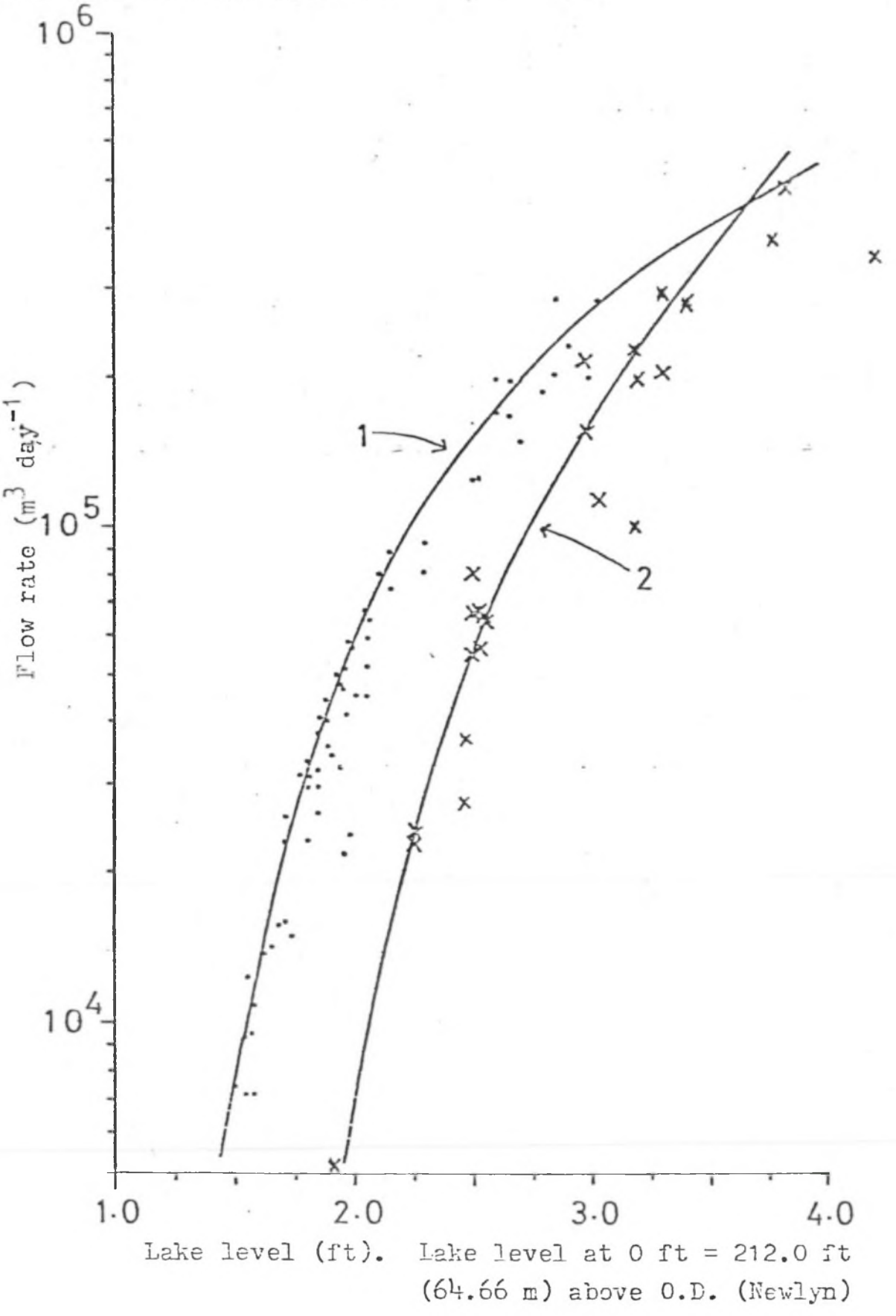


Fig. 4. Relationship between ouflow rate and lake level, estimated from 1966-9 data of the River Authority (., 1) and in 1974-80 by dilution gauging (x, 2).

From: Jones 1972

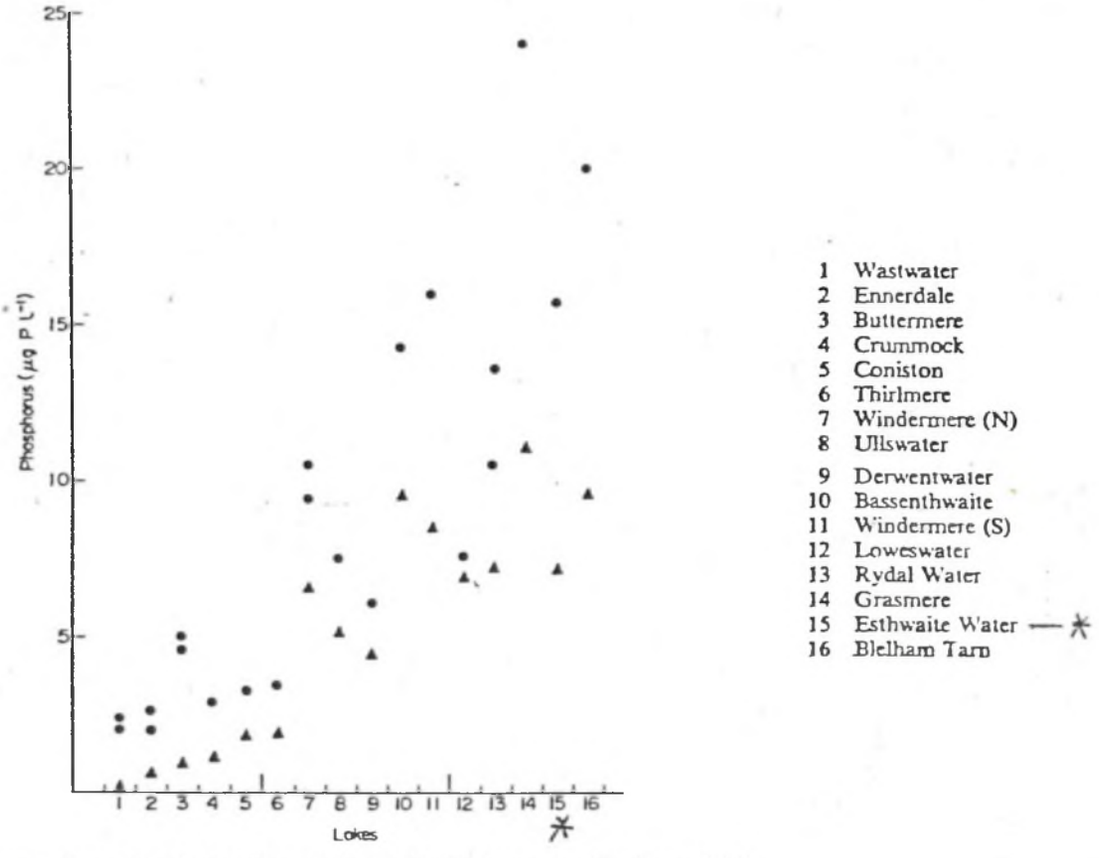
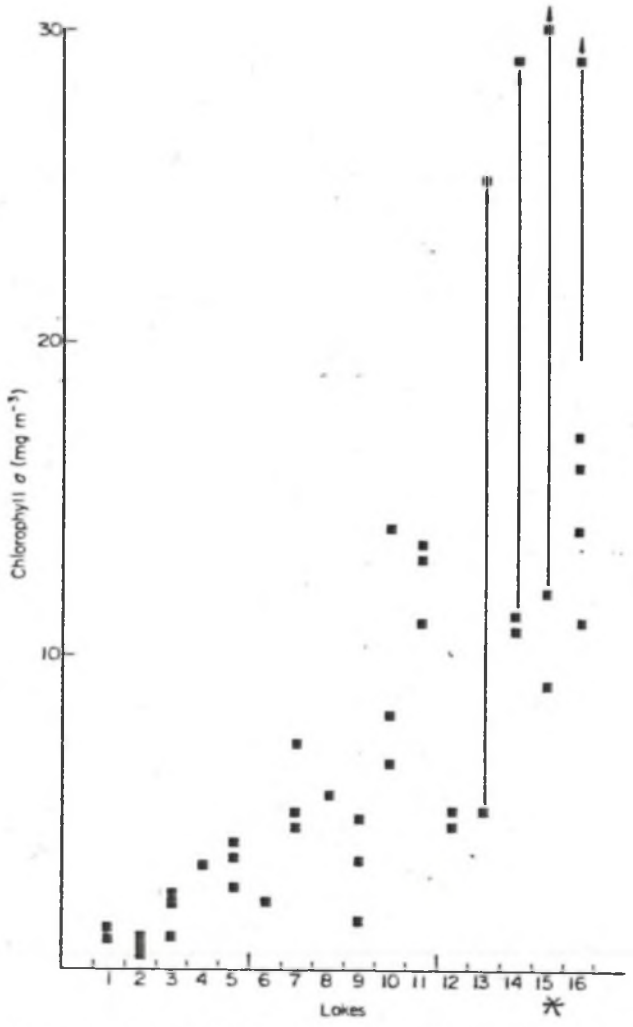


FIG. 5 Concentrations of total phosphorus (●) and dissolved organic phosphorus (▲) in the epilimnion of sixteen lakes (see Table 1) during 1971.

From: Jones 1972



1 Wastwater
2 Ennerdale
3 Buttermere
4 Crummock
5 Coniston
6 Thirlmere
7 Windermere (N)
8 Ullswater
9 Derwentwater
10 Bassenthwaite
11 Windermere (S)
12 Loweswater
13 Rydal Water
14 Grasmere
15 Esthwaite Water
16 Blelham Tarn

FIG. 6 Estimates of chlorophyll *a* concentration in epilimnetic samples of sixteen lakes (see Table 1). Arrows indicate range of values or where levels exceed the scale of the graph.

From: Jones 1972

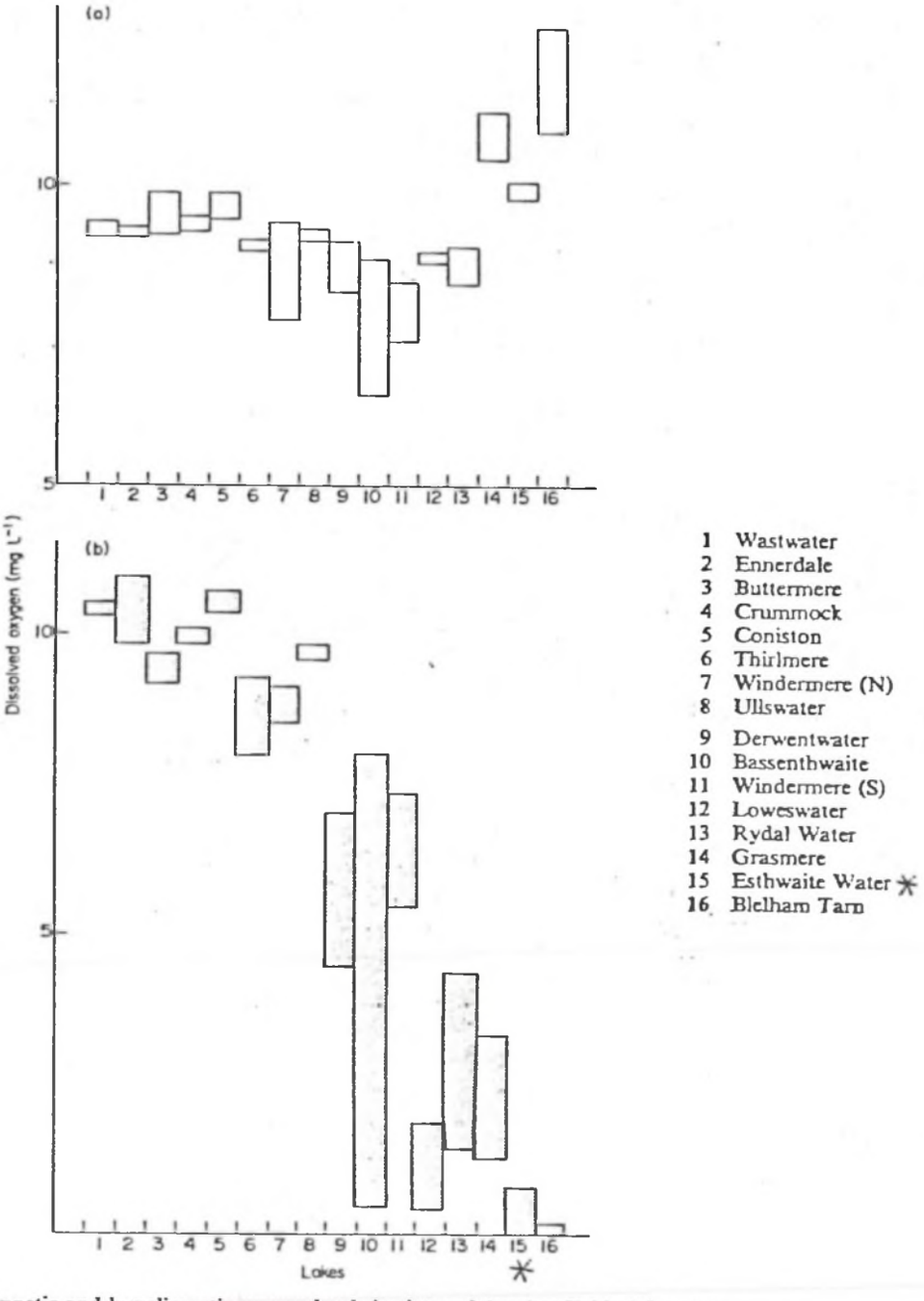


FIG. 7 Epilimnetic and hypolimnetic oxygen levels in sixteen lakes (see Table 1 for numbering of lakes) during 1971. (a) Open columns, epilimnion; (b) stippled columns, hypolimnion.

From: Lund 1972

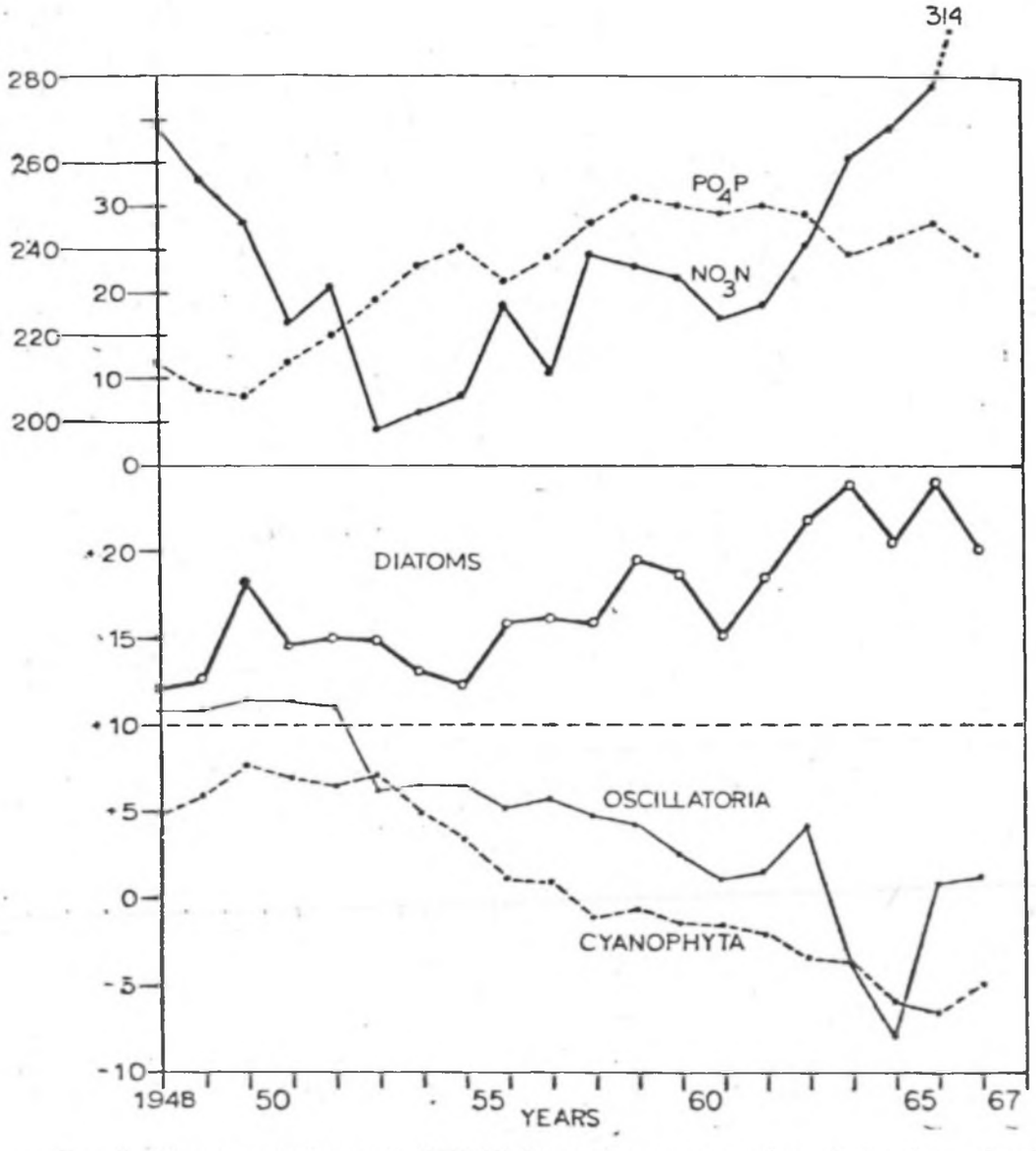


FIG. 8 Five-year running totals (1946–69) for maximum concentrations of phosphate phosphorus (PO_4P, vertical scale 0–30) and nitrate nitrogen (NO_3N, vertical scale 200–280) in $\mu g l^{-1}$, and for the increase or decrease of diatoms (vertical scale +10 to +20), Cyanophyta and of *Oscillatoria agardhii* var. *isothrix* (vertical scale +10 to −10) after the overturn.

From: Sutcliffe et al. 1982

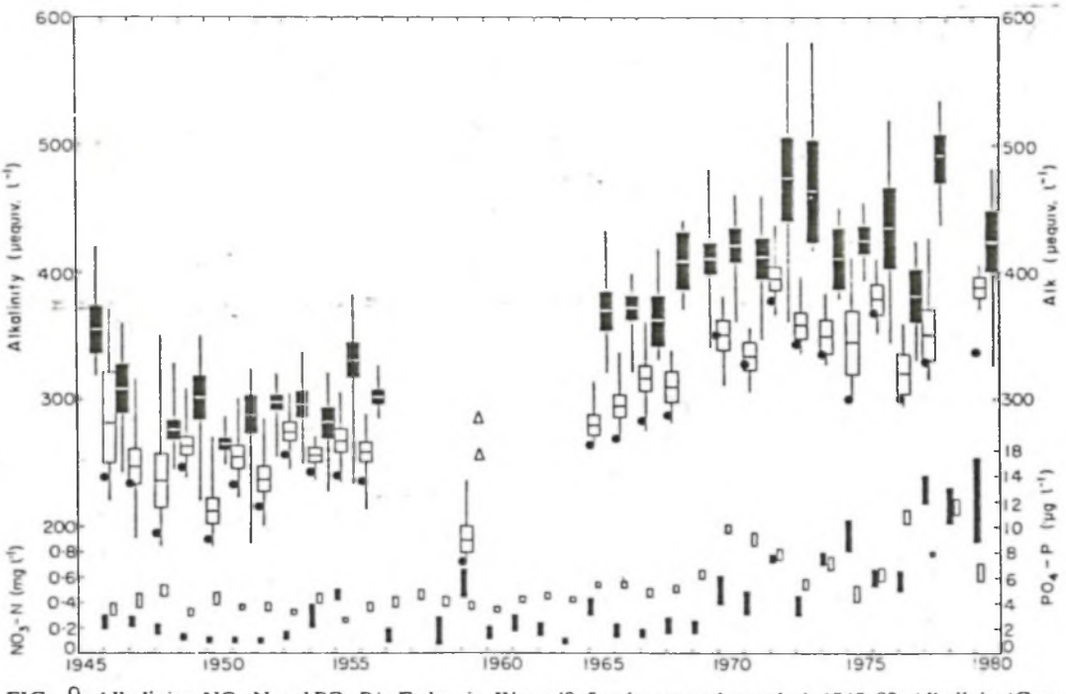


FIG. 9 Alkalinity, NO_3-N and PO_4-P in Esthwaite Water (0–5 m integrated samples), 1945–80. Alkalinity (Gran titration values, μequiv. l^{-1}): 95% CL for the mean (boxes) and range (vertical lines) for January–June (unshaded) and July–December (shaded); ●, means for each winter–spring period; Δ, individual measurements. Values for 1968 based on seven measurements in the period 29 August–3 October; July–December 1969 and all 1970 based on water samples (Jenkin corer) from just above the sediment surface at 3 and 5 m depth (combined values) (Goulder, 1974 and personal communication); 1978 based on measurements made by Ann Wightman; 1980 April–June and July–December. NO_3-N (mg l^{-1}): ▯, 95% CL of the means for winter–spring. PO_4-P (μg l^{-1}): ▮, 95% CL of the means for autumn–winter. Further details in the Appendix.

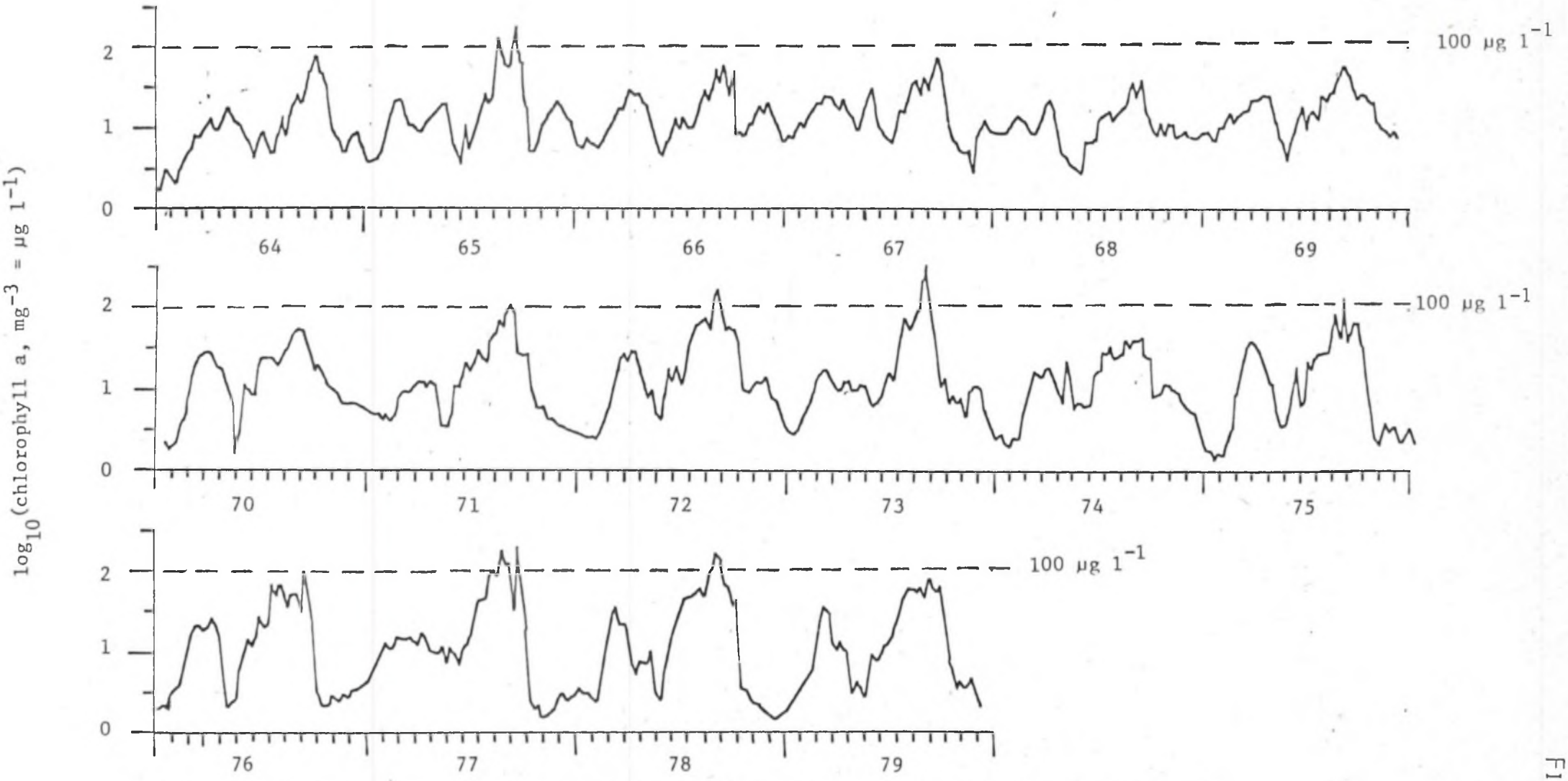


Fig. 10. Variation of the mean concentration of chlorophyll a in the 0-5 m layer, 1964-1979.

FIG. 10

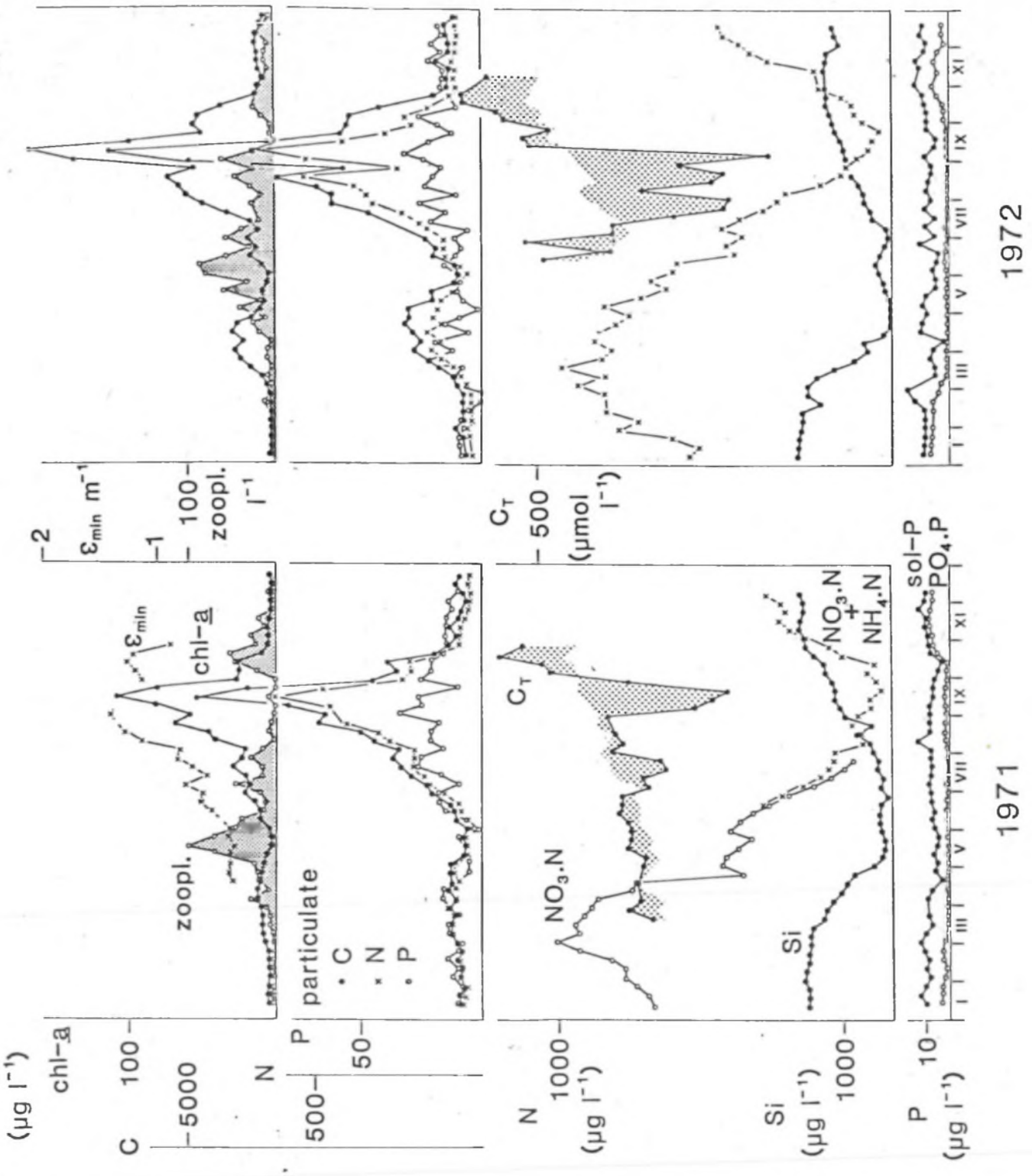


Fig. 11. Seasonal variation during 1971 and 1972 of concentrations of phytoplankton (as chlorophyll a) and larger zooplankton (zoopl., shaded); light attenuation as minimum coefficient for the visible spectrum (ε_{min}); concentrations of particulate C, N, and P; and concentrations of total inorganic carbon (C_T, with excess or deficit to the free CO_2 end-point shaded), NO_3.N or (x) NO_3.N + NH_4.N, Si, soluble P, and PO_4.P.

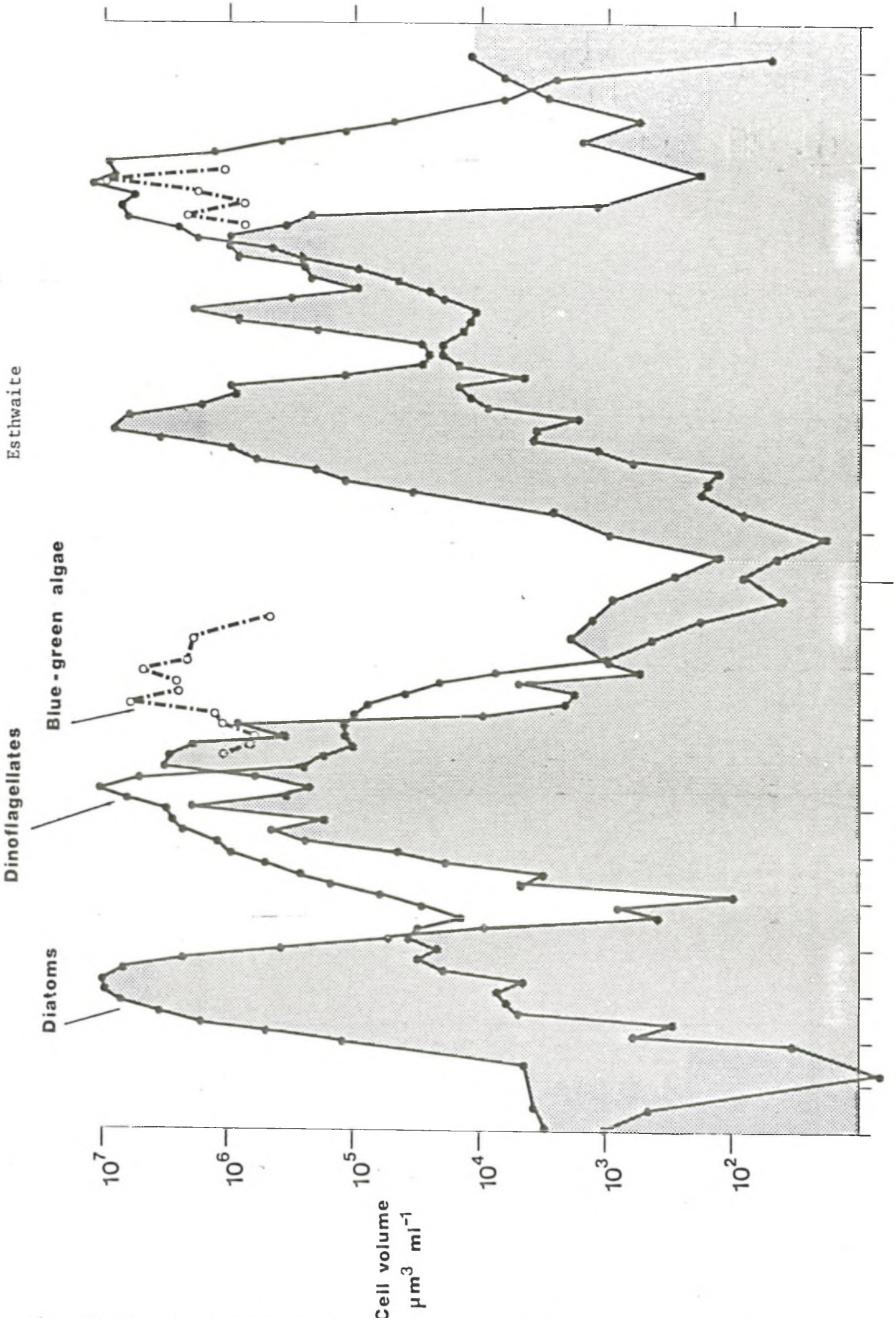


Fig. 12. The comparative abundance of diatoms (shaded), dinoflagellates, and blue-green algae during 1980–1981, expressed as cell volume in the 0–5 m layer.

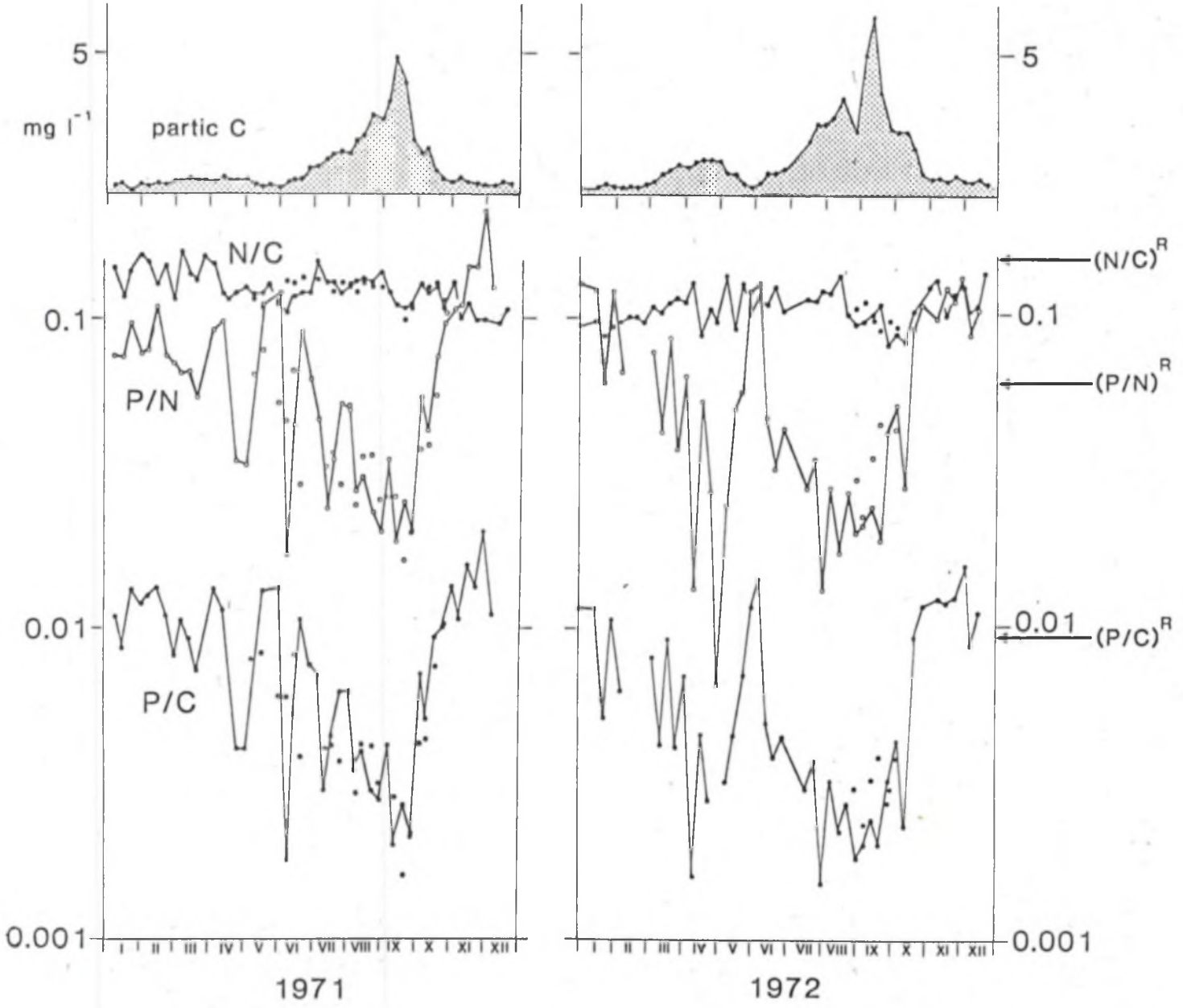


Fig. 13. (Above) the seasonal variation of particulate matter (as C content) in the 0-5 m layer during 1971 and 1972, with (below) its ratios (by atoms) of N/C, P/N, and P/C. Arrows indicate 'average' Redfield ratios for plankton.

From: Heaney and Talling 1981

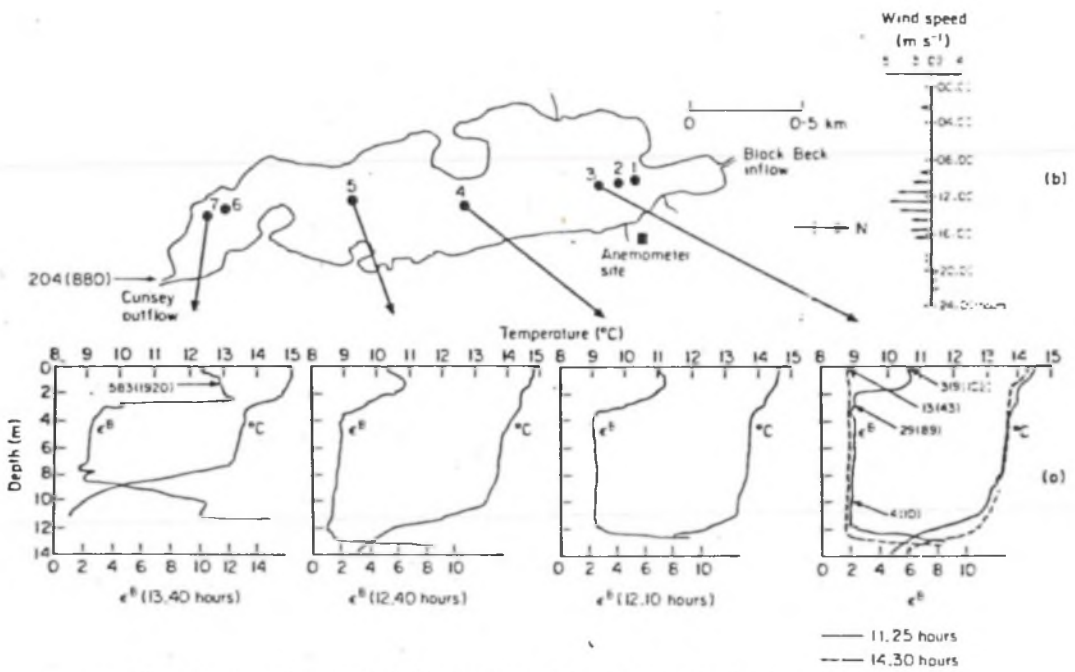


FIG. 14 The changing vertical distribution of *Ceratium* and related variables along the lake on 15 September 1977: (a) continuous vertical profiles of temperature and horizontal beam coefficient (ϵ^B, m^{-1}) at the times and stations indicated, with concentrations of chlorophyll *a* (mg m^{-3}), and of *Ceratium hirundinella* (cells ml^{-1}) in parentheses, given for the depths indicated by arrows; (b) average hourly wind vectors.

From: Davison et al. 1980

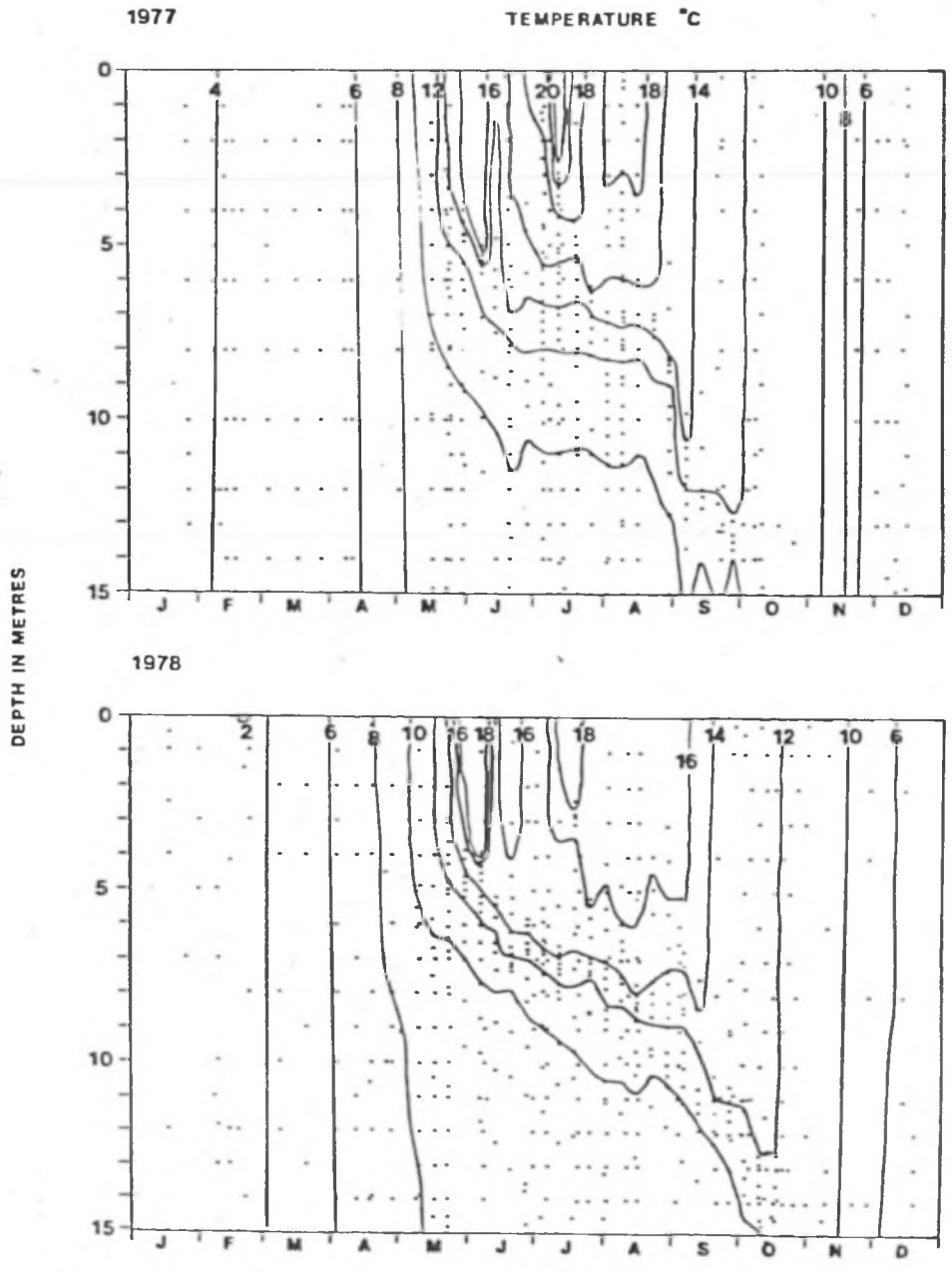


Fig. 15 The depth-time distribution of temperature (isotherms in °C) during 1977 and 1978.

From: Heaney and Talling 1981

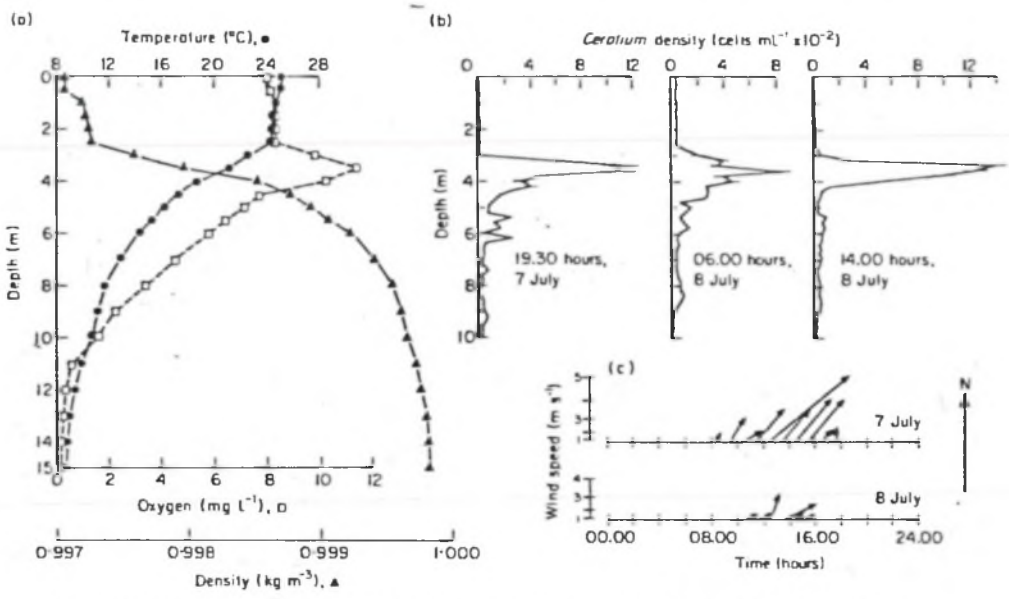


FIG.16 Distribution of *Ceratium* and controlling factors on 7–8 July 1976: (a) stratification at station 3 of temperature (●), oxygen (□) and water density (▲) at 10.50 hours, 8 July; (b) vertical profiles of *Ceratium* density; horizontal lines show 95% confidence limits; (c) average hourly wind vectors plotted on a non-linear square-law scale; the perpendicular height from the baseline indicates the wind speed.

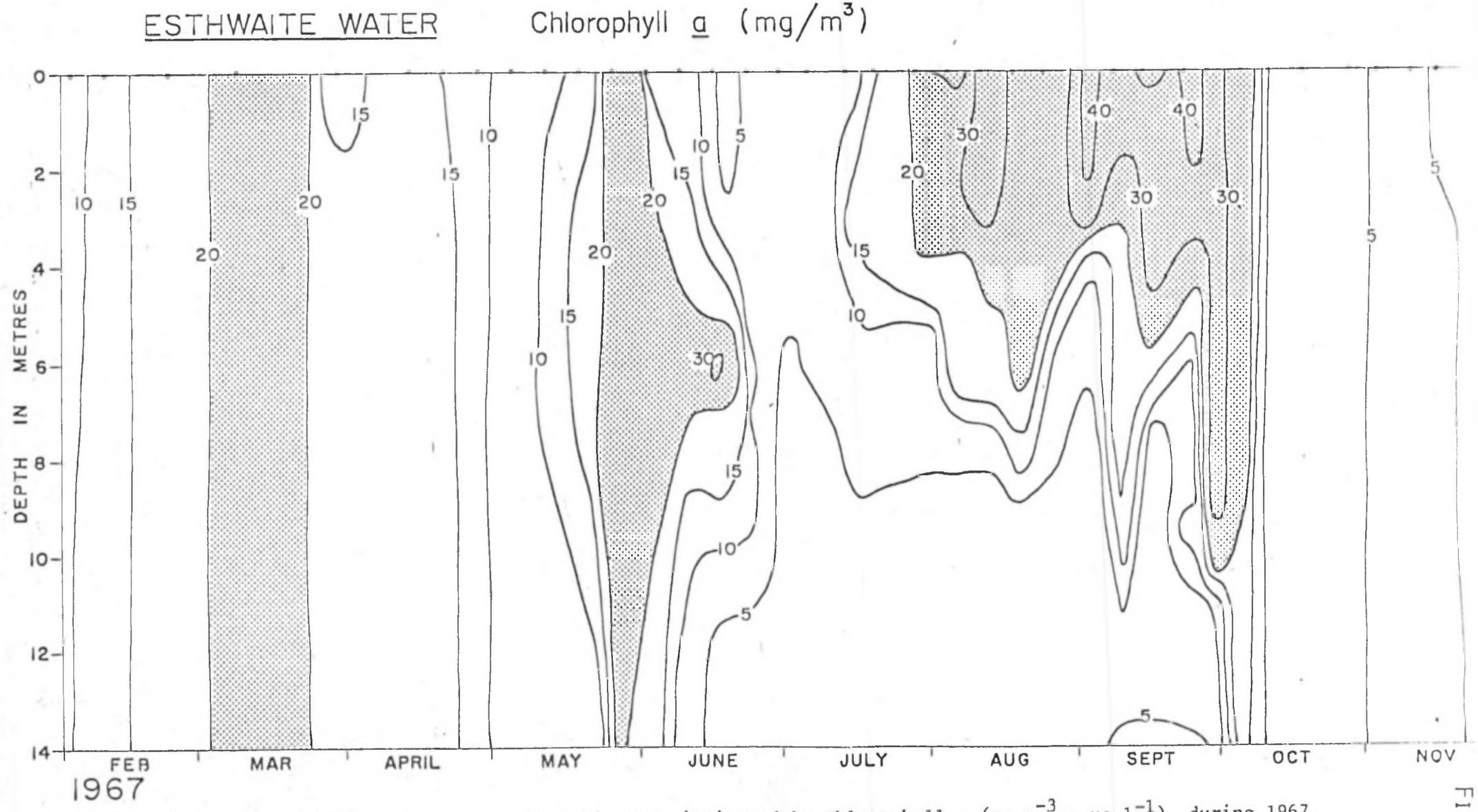


Fig. 17. Depth-time variation of phytoplankton biomass, indicated by chlorophyll a (mg m^{-3} = $\mu g\ l^{-1}$), during 1967.

FIG. . .

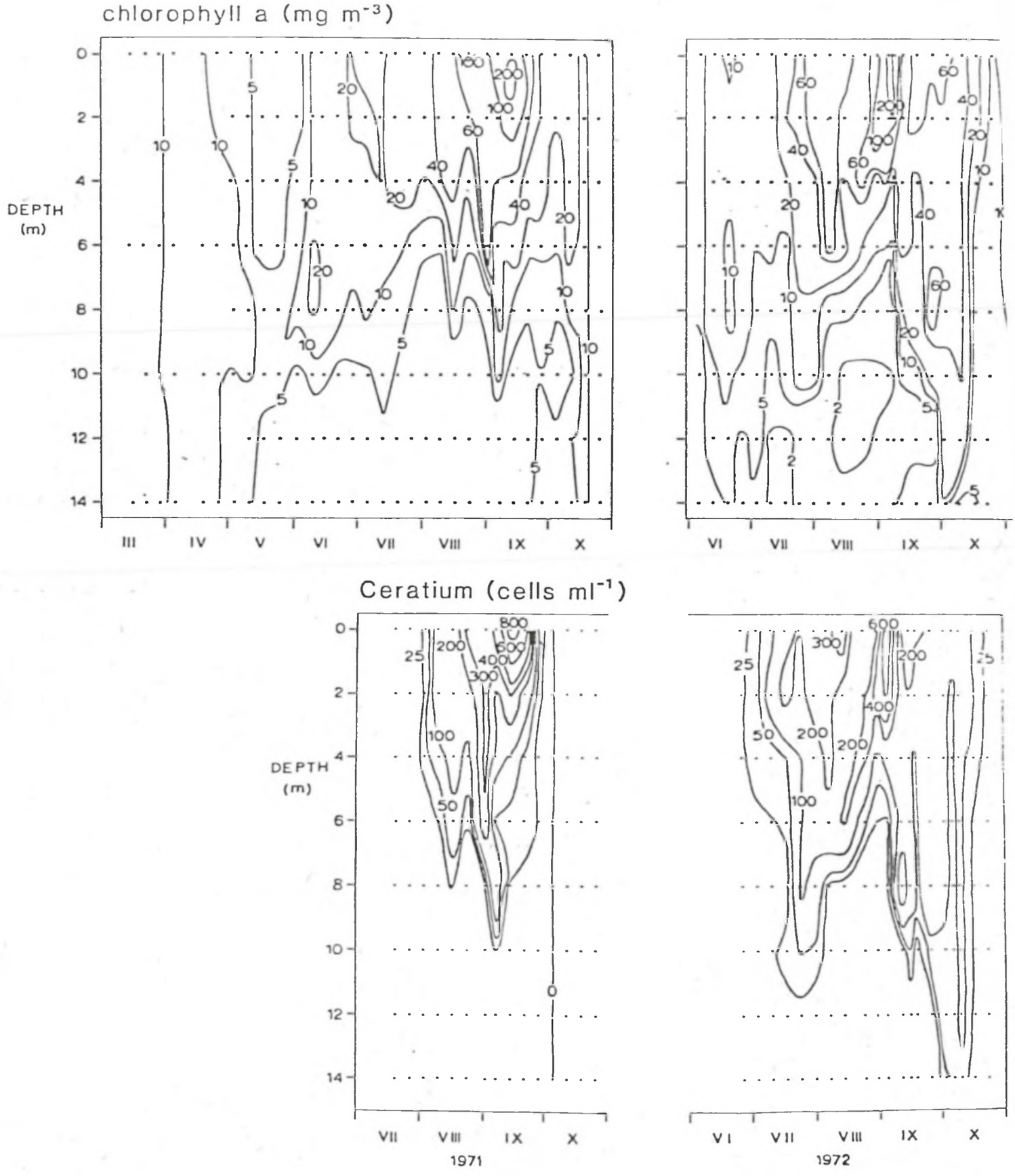


Fig. 18. Depth-time distributions of (above) chlorophyll a concentration (mg m^{-3} = µg l^{-1}), and of Ceratium numbers, during the summers of 1971 and 1972.

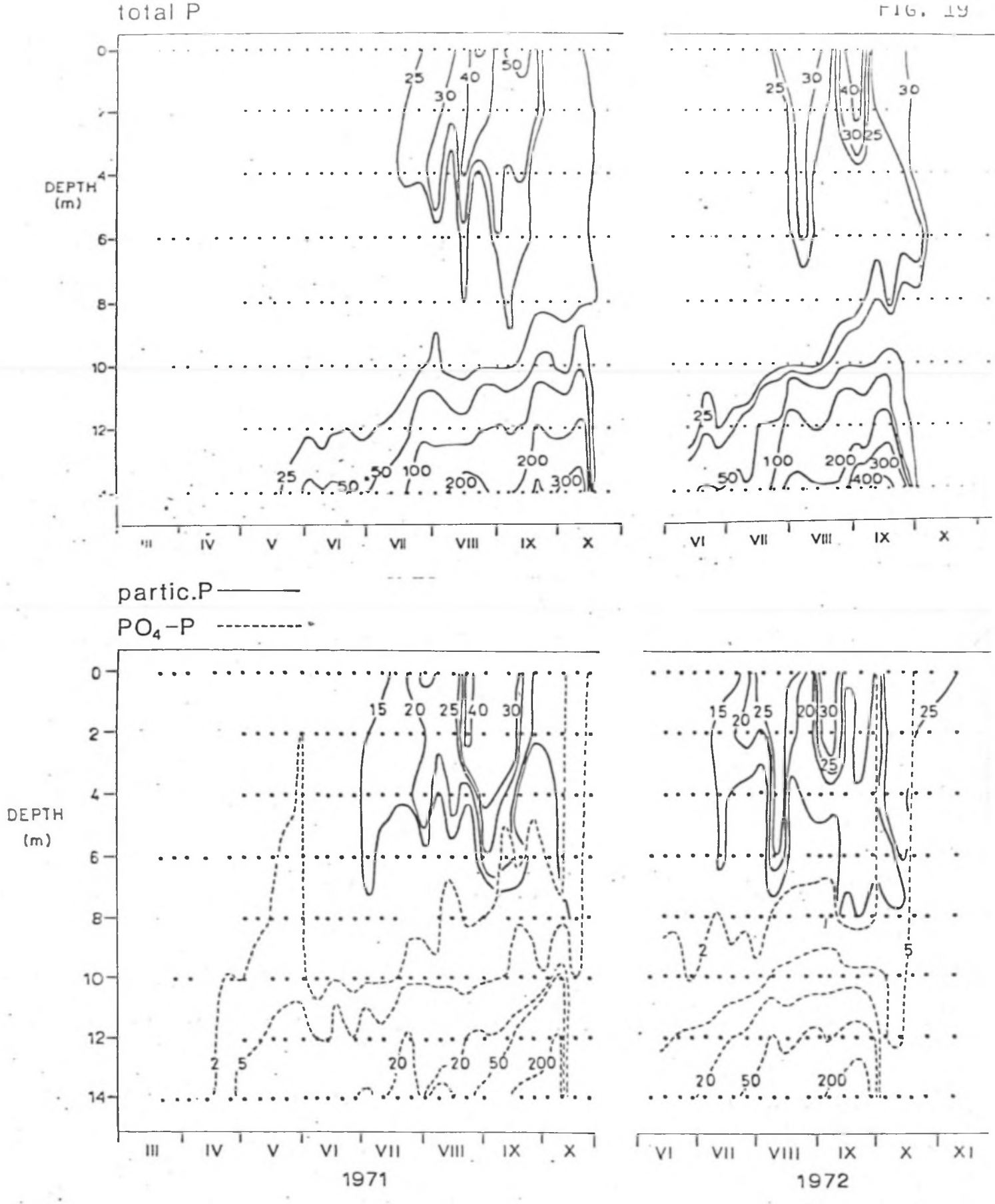


Fig. 19. Depth-time distributions of three forms of P (total, particulate and PO_4-P) during the summers of 1971 and 1972.

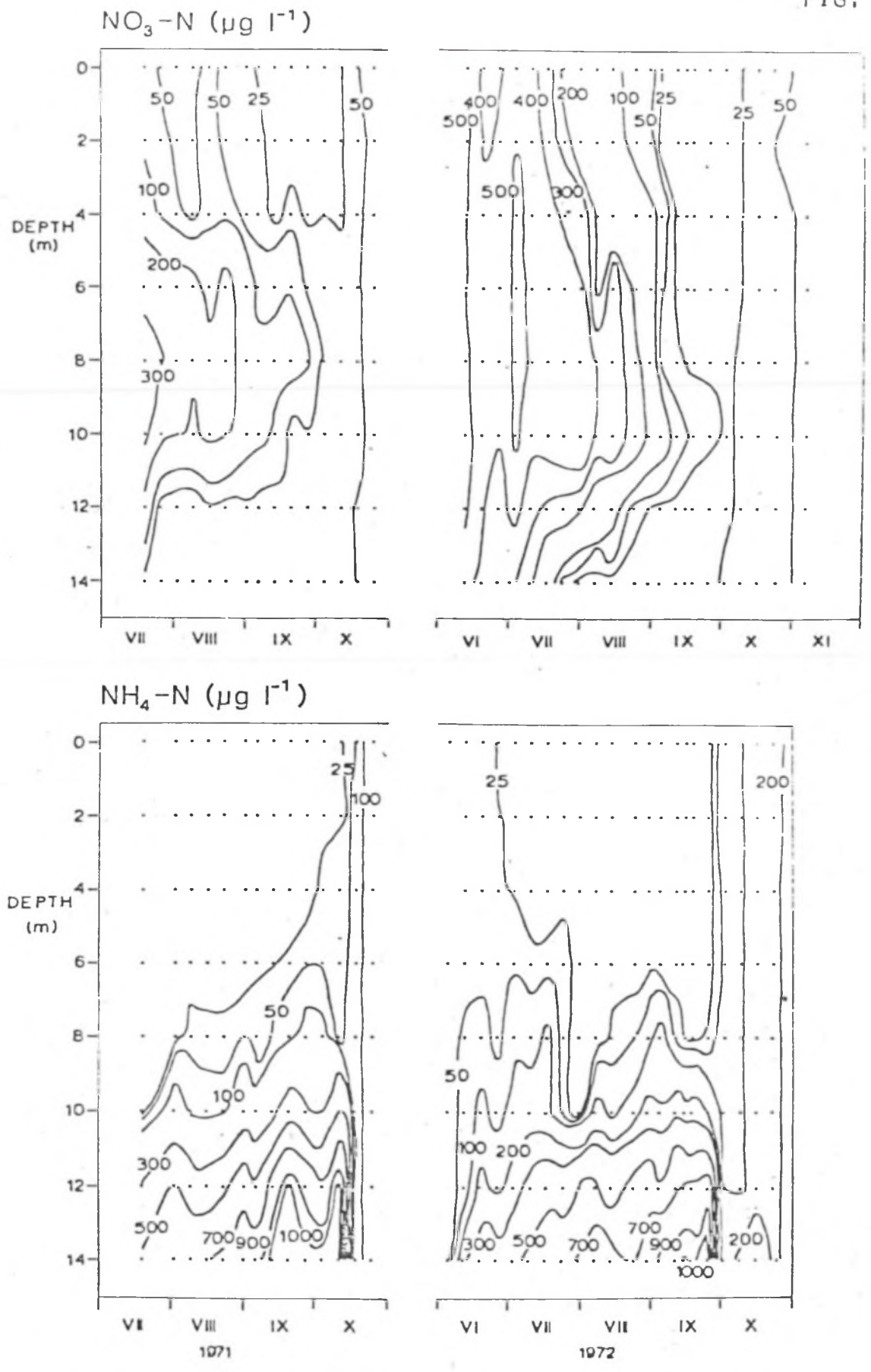


Fig. 20. Depth-time distributions of NO_3-N and NH_4-N during the summers of 1971 and 1972.

From: Davison et al. 1980

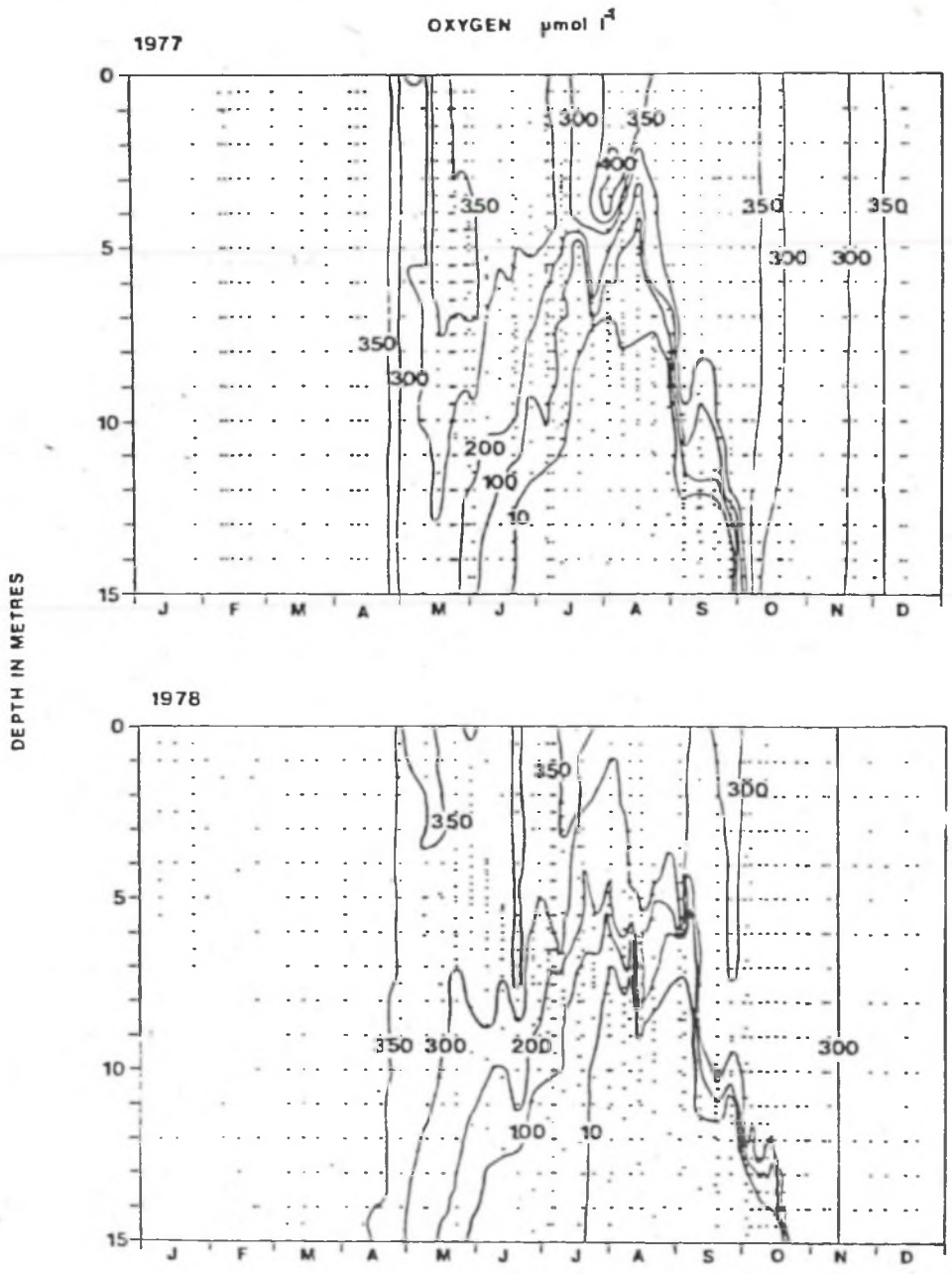
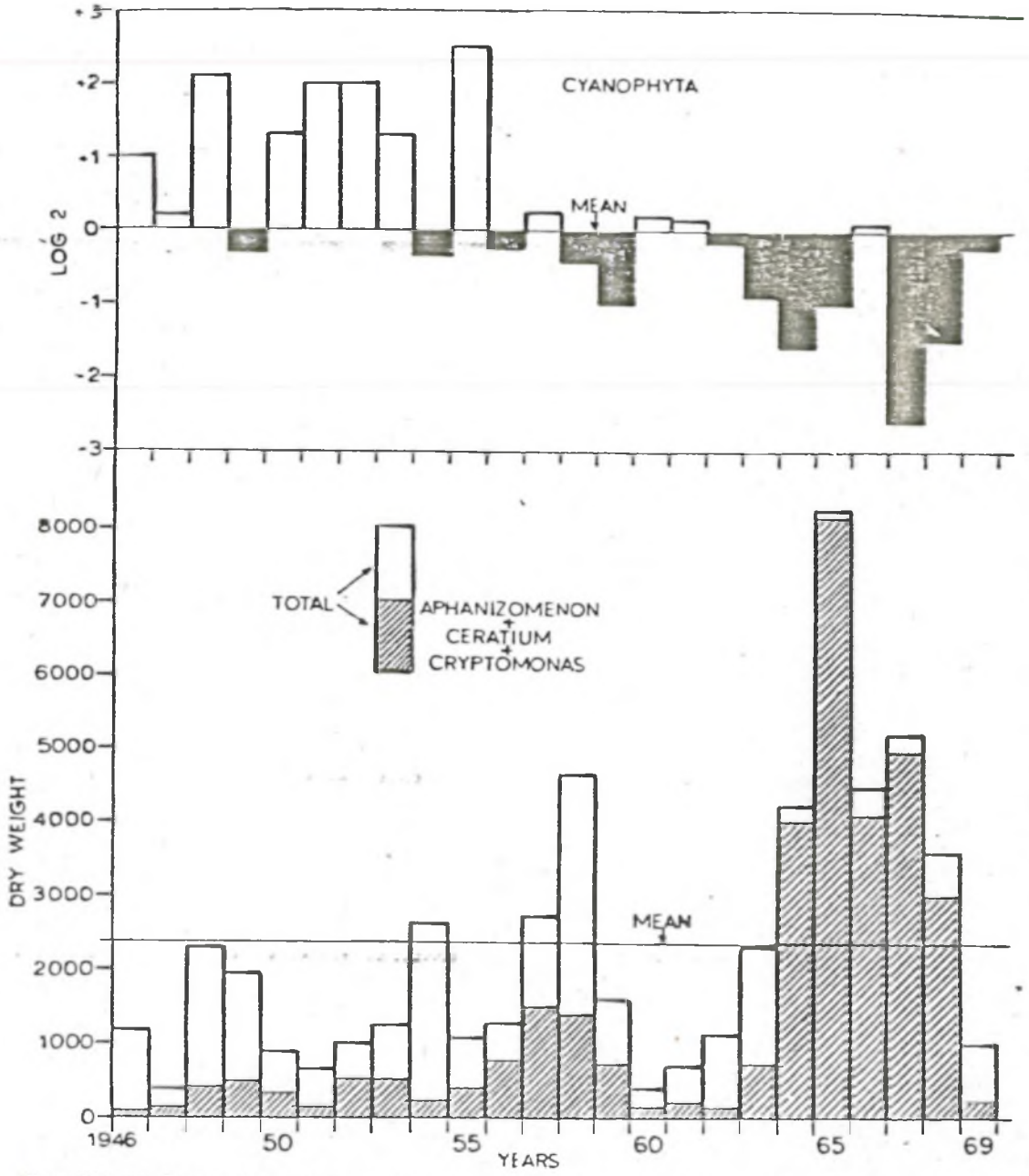


Fig. 21. The depth-time distribution of dissolved oxygen (isopleths in µ moles l^{-1}) during 1977 and 1978.

From: Lund 1972



TEXT-FIG. 22 Pre-overturn algal maxima compared with post-overturn growth of Cyanophyta. Upper columns increase (unshaded) or decrease (black) of Cyanophyta after the overturn up to 31 December on a log$_2$ scale and based on dry weights (*see* Fig. 2). Bottom columns, sum of maxima of the populations of the major components of the phytoplankton in the four weeks before the overturn; shaded portion of columns, sum of the maxima of the populations of *Aphanizomenon*, *Ceratium* and *Cryptomonas*.

From: Lund 1981, with additions

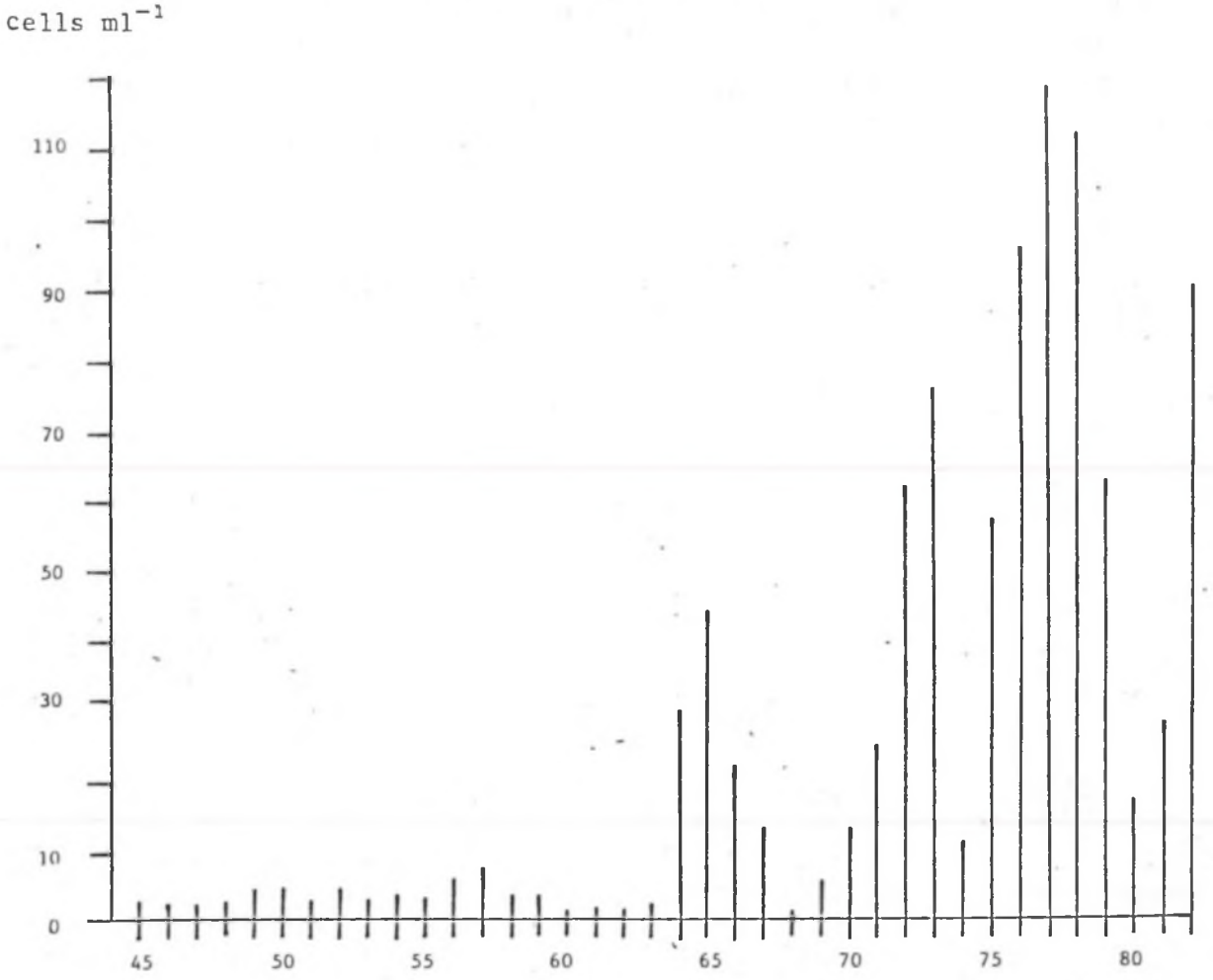


Fig. 23. Long-term changes (1945-1982) in the mean weekly concentration of Ceratium hirundinella in the 0 - 5 m layer (occurrence dates only).

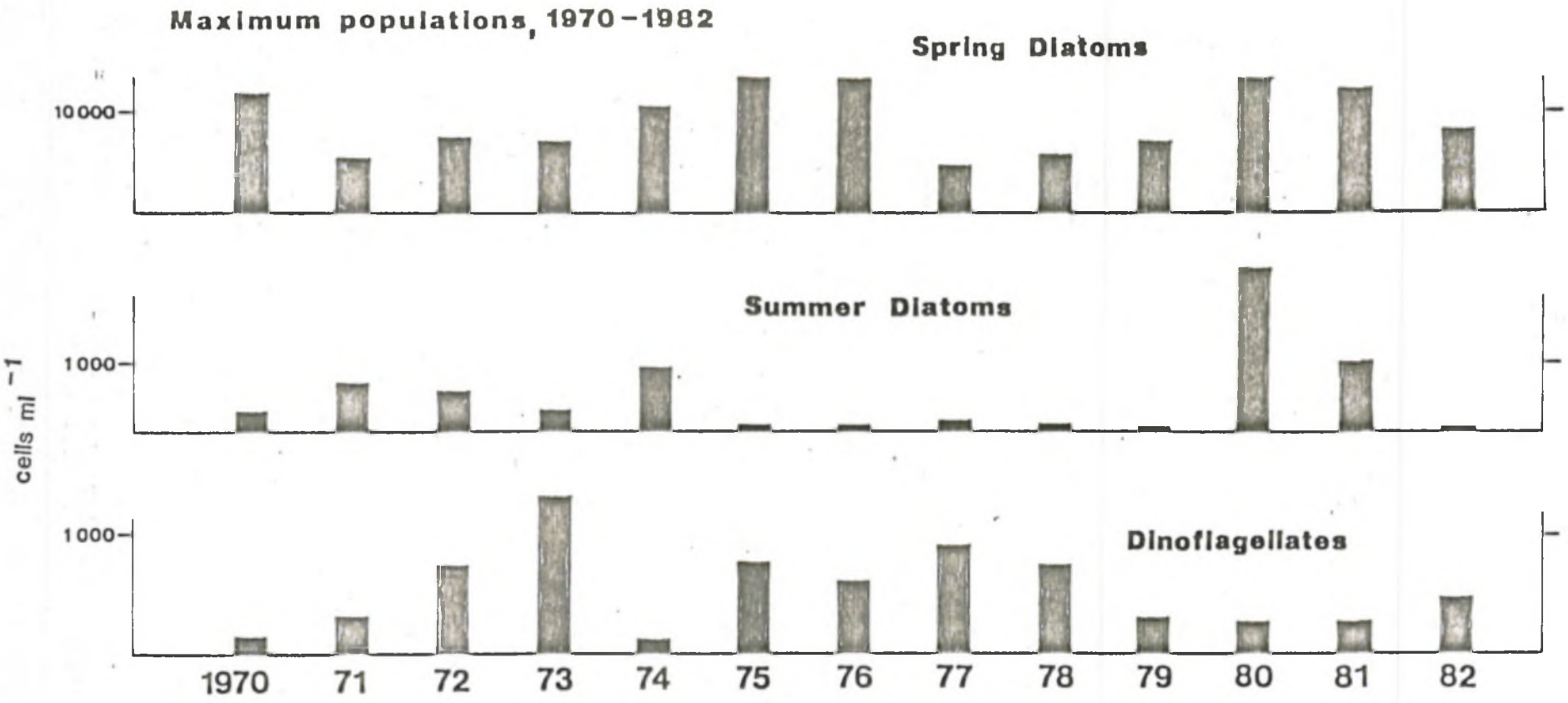


Fig. 24. The maximum cell concentrations reached by spring diatoms, summer diatoms, and (late summer) dinoflagellates (Ceratium hirundinella) during 1970-1982.

FIG. 24

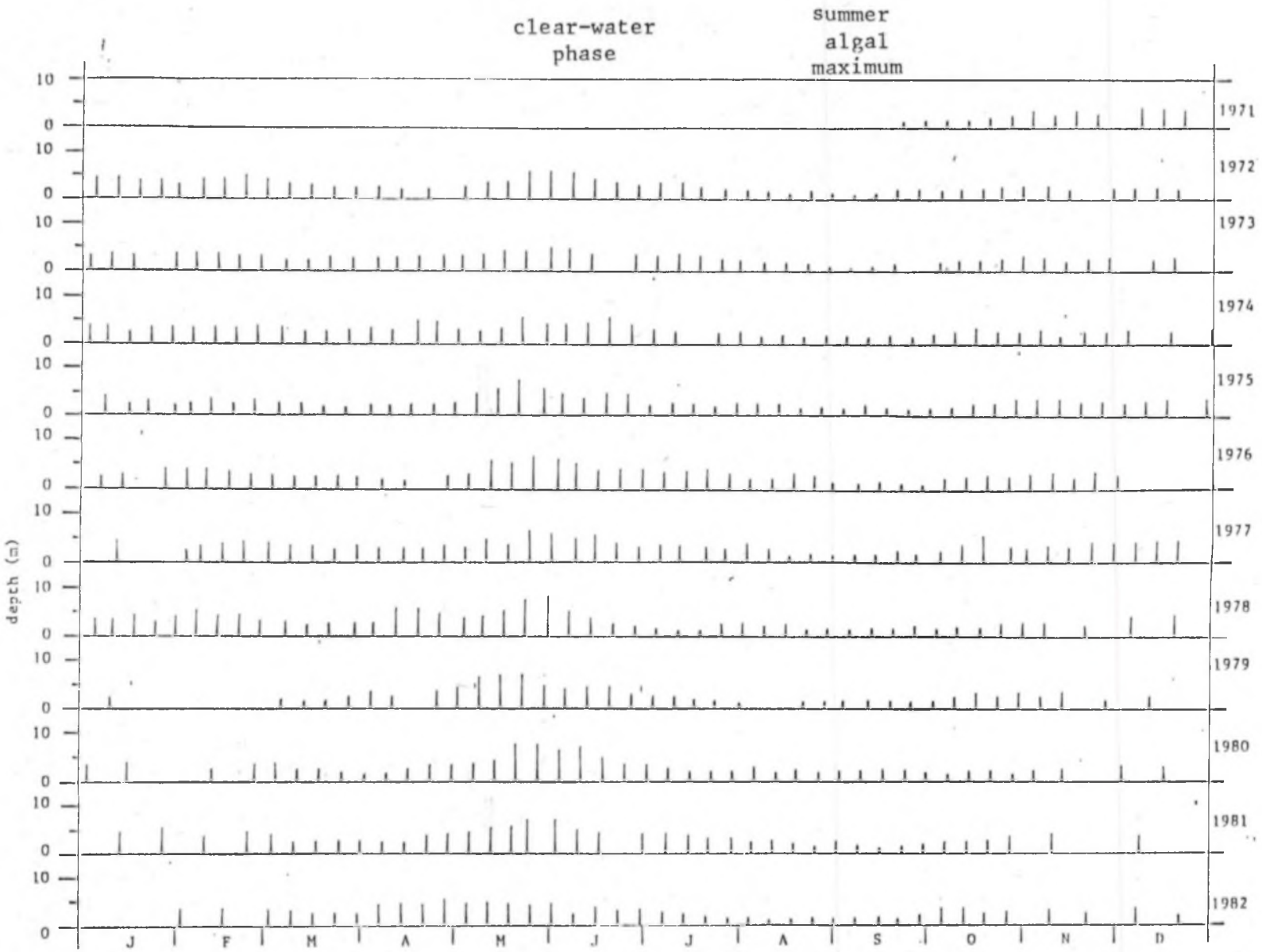


Fig. 25. Seasonal variation of transparency (Secchi disc) from 1971 to 1982.

FIG. 25

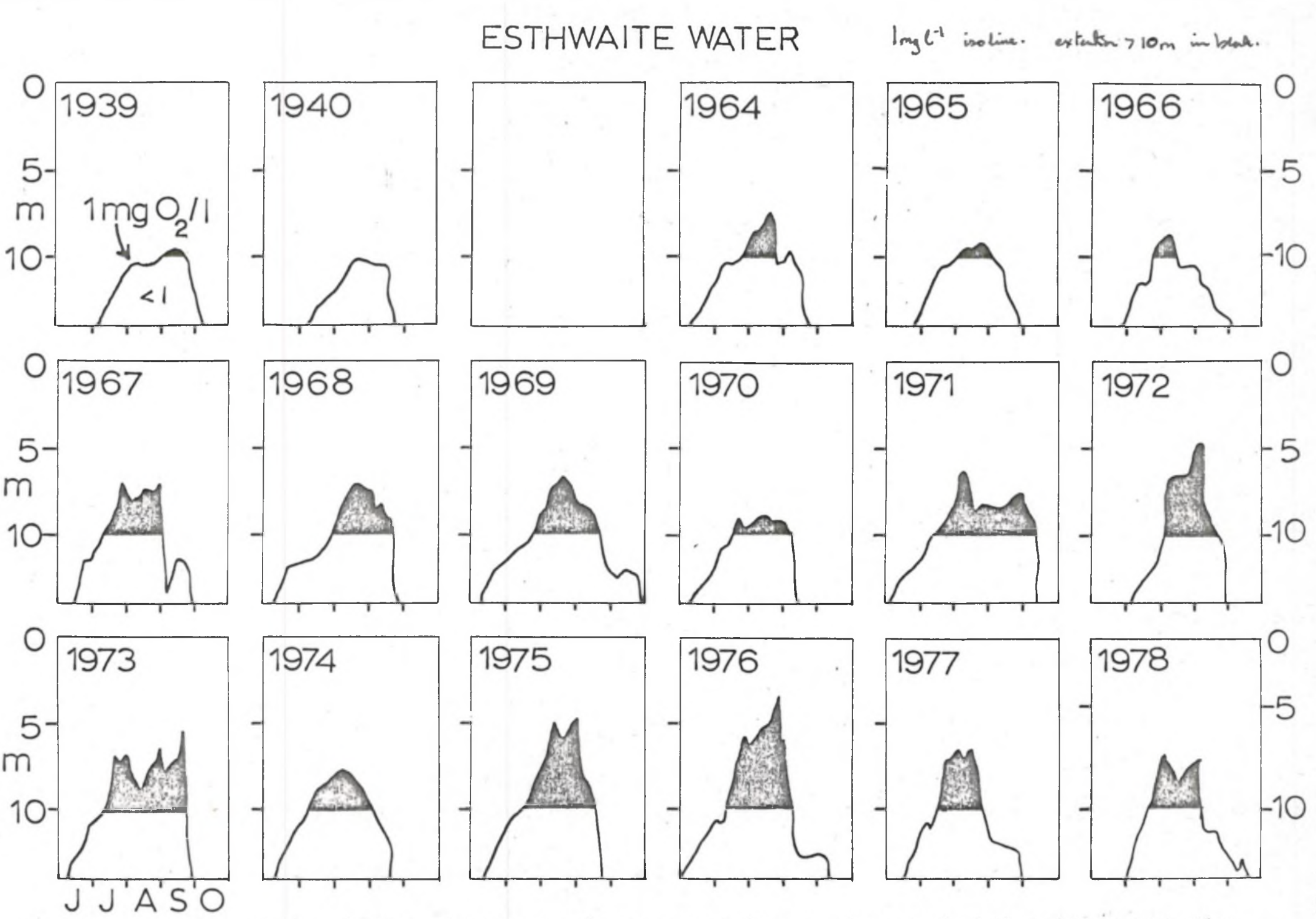


Fig. 26. Depth-time variation, in June - October of various years, of the approach to anoxia indicated by concentrations below 1 mg l^{-1}. Extensions above 10 m depth are shown in black.

FIG. 26

From: Talling 1976

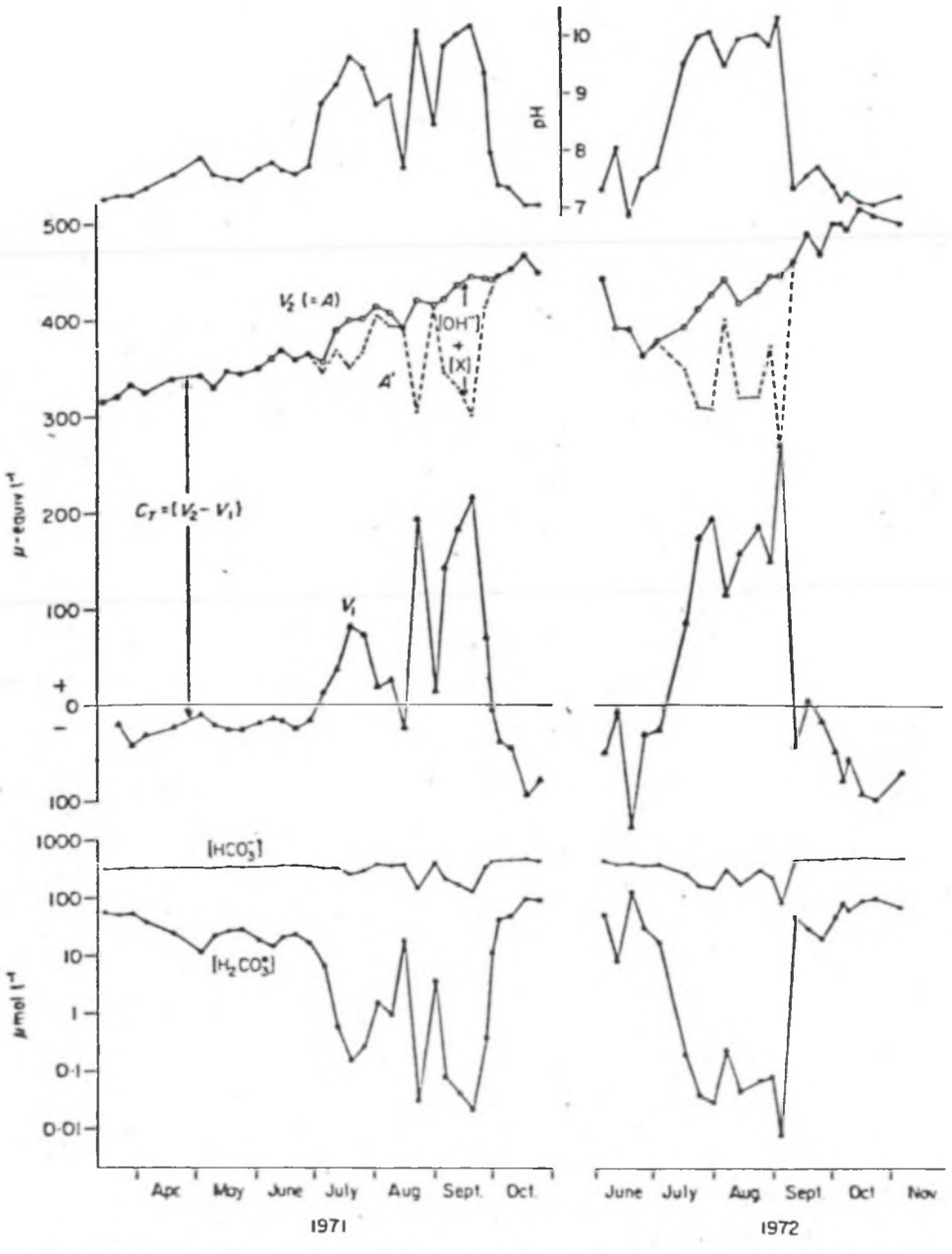


FIG 27 Esthwaite Water, 1971, 1972: changes with time in the surface water of pH (corrected to 20° C), measured total alkalinity V_2 (= A), calculated carbonate alkalinity (A'), measured V_1, total CO_2 ($C_T = V_2 - V_1$), and calculated concentrations of HCO_3^- and $H_2CO_3^*$.

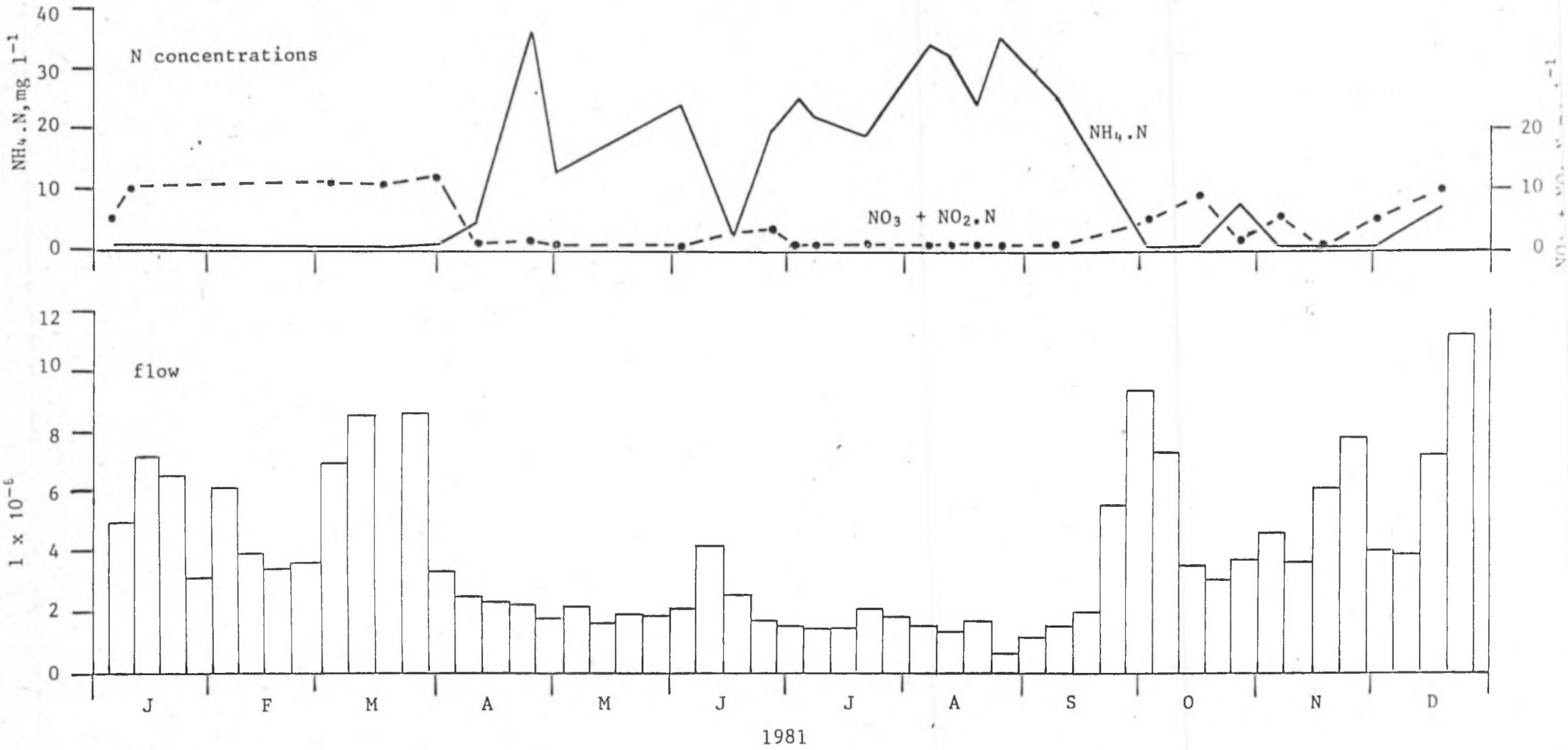


Fig. 28. Seasonal variation during 1981 of (above) concentrations of NH_4-N and of NO_3+NO_2-N in Hawkshead sewage effluent, and (below) corresponding weekly rates of flow of effluent

FIG. 28

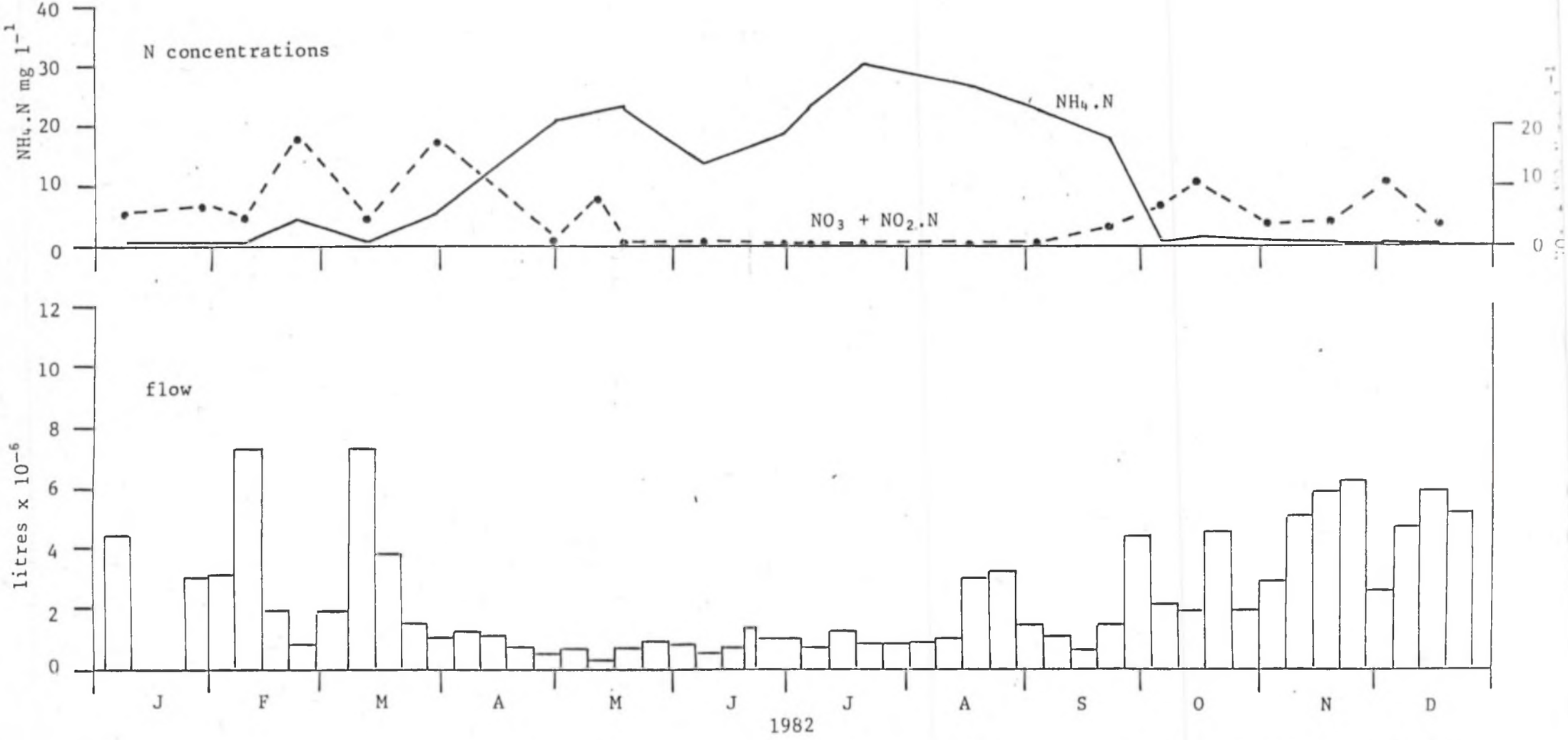


Fig. 29. Seasonal variation during 1982 of (above) concentrations of NH_4-N and of NO_3+NO_2-N in Hawkshead sewage effluent, and (below) corresponding weekly rates of flow of effluent.

FIG. 2

From: Wood & Gibson 1973

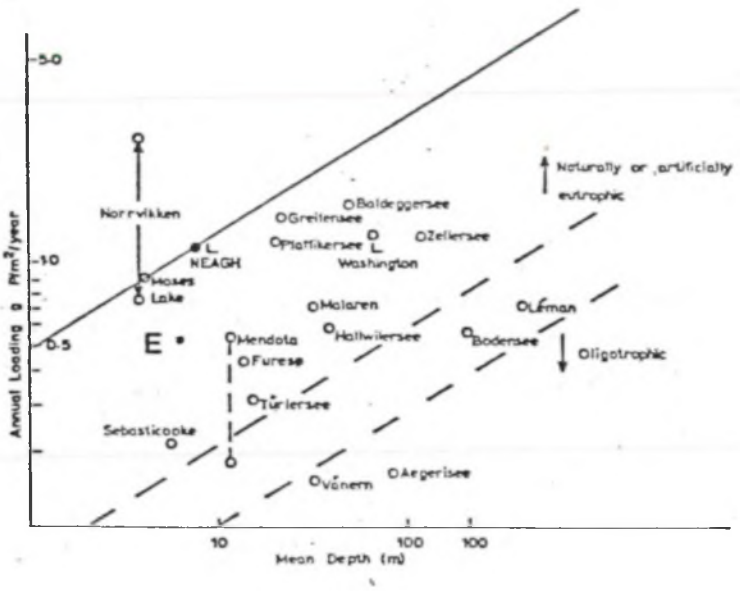


Fig. 30. Annual phosphorus loading and mean depths of some lakes compared to L. Neagh and Esthwaite Water (E).