Institute of
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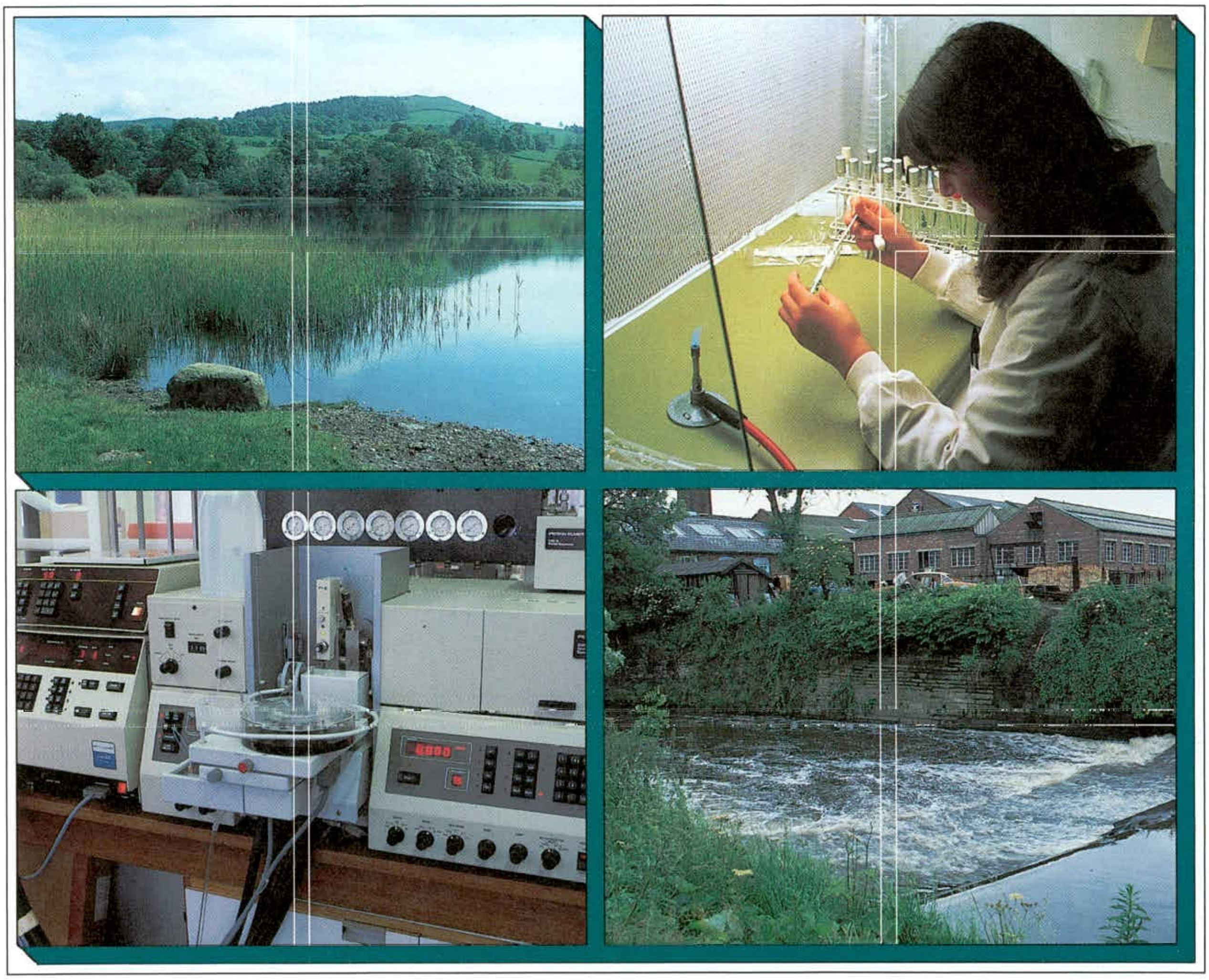
Principal Investigators:

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

Report to the Water Research Centre (March 1992)



Natural Environment Research Council

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

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CONTENTS

1. AIMS, APPROACHES AND RATIONALE

The main aim of this study is to assess the relative importance of factors controlling the sensitivity and responses of lentic waters to nutrient loads, ie what levels of nutrients and what types of organisms and abundances will result, per unit of nutrient supply. The factors of major interest include weather regimes, especially rainfall, features such as the shape and size of a lake basin, and chemical properties determining water colour and total ionicity, for example. For present purposes, the nutrient of main concern is phosphorus (P), because it is the chemical factor most commonly limiting the growth of planktonic algae in Northern Temperate waters. Nitrogen in the form of nitrate (NO_3), and silica (SiO_2) are also observed to drop to growth-limiting levels at certain times of the year (Bailey-Watts 1990), but these shortages affect particular types of algae - not all plants as in the case of P. Indeed, N-fixing blue-green tend to predominate in summer when nitrate is reduced to low levels, by microbial de-nitrification (Bailey-Watts 1986 and Bailey-Watts, Kirika, May and Jones 1990 for Loch leven, and Bailey-Watts, Lyle, Kirika and Wise 1987 for Coldingham Loch), and silica availability controls the timing and magnitude of diatom maxima (e.g. Bailey-Watts, Smith and Kirika 1989). The links between nutrient supply and lake response are thus assessed here, mainly on the basis of relationships between P loading, and the in-lake concentrations of P and phytoplankton biomass; the latter is considered primarily with reference to the concentrations of chlorophyll *a* recognising that algal blooms are of widespread concern. However, there are other manifestations of eutrophication which are of at least local importance; these include overgrowths of rooted vegetation and/or attached algae, and hypolimnetic de-oxygenation (Bailey-Watts, May, Kirika and Lyle in press).

Relationships between P loading and in-lake P and chlorophyll *a* concentrations (encompassing data from Loch Leven, Scotland, and Lough Neagh, Northern Ireland) have been well-documented by Vollenweider and co-workers (Vollenweider 1968, OECD 1982). The P loading - P concentration regression equations take the following forms:

$$[P]_\lambda = a[P]_j^b \qquad (1)$$

$$\text{or} \quad [P]_\lambda = a[P]_j/(1+\sqrt{T(w)})]^b \qquad (2)$$

where

$[P]_\lambda$ is the annual mean concentration of P in a lake (in $\mu g\ l^{-1}$ or $mg\ m^{-3}$)

$[P]_j$ is the external loading expressed as the mean concentration in the inflowing water (also $\mu g\ l^{-1}$ etc.)

the term $1/(1+\sqrt{T(w)})$ is a flushing correction, with T(w) being the water residence time (in years, ie the reciprocal of flushing rate *p*)

a and *b* are constants, which differed between the sets of OECD project data as shown in **Table 1** along with the coefficients of determination (r^2) and the number (n) of sites corresponding to each set.

TABLE 1. Regression equations described by OECD project data*, relating annual mean in-lake concentrations ($[P]_\lambda$) of phosphorus to (i) annual mean inflow concentrations ($[P]_j$) and (ii) flushing-corrected inflow concentrations (X, where $X = [P]_j/(1 + \sqrt{T(w)})$; the coefficient of determination (r^2), and the number of sites contributing to each study (n) are shown.

Project	(i)	(ii)
Combined OECD	$[P]_\lambda = 0.76\ [P]_j^{0.83}$ $r^2 = 0.76$; $n = 87$	$[P]_\lambda = 1.55\ X^{0.82}$ $r^2 = 0.86$; $n = 87$
Alpine	$[P]_\lambda = 0.86\ [P]_j^{0.80}$ $r^2 = 0.85$; $n = 18$	$[P]_\lambda = 1.58\ X^{0.83}$ $r^2 = 0.86$; $n = 18$
U.S.A.	$[P]_\lambda = 0.66\ [P]_j^{0.85}$ $r^2 = 0.71$; $n = 31$	$[P]_\lambda = 1.95\ X^{0.79}$ $r^2 = 0.90$; $n = 31$
Shallow Lakes and Reservoirs	$[P]_\lambda = 0.65\ [P]_j^{0.88}$ $r^2 = 0.83$; $n = 24$	$[P]_\lambda = 1.22\ X^{0.88}$ $r^2 = 0.90$; $n = 24$

*OECD 1982

Equation (1) differs from Equation (2) in the factor that corrects for flushing. The lower the value of T(w), the higher this term. **Table 2** illustrates this with data obtained from desk analysis of maps of rainfall, evaporation, catchment areas and bathymetry of 4 Scottish lochs contrasting considerably as regards T(w) - Bailey-Watts, May, Kirika and Lyle (in prep.).

TABLE 2. Values for the term $(1 + \sqrt{T(w)})$ and its reciprocal, calculated for 4 Scottish lochs ranging in water retention times from 0.045 to 1.493 ($T(w)$ in years)

Loch	T(w)	$(1+\sqrt{T(w)})$	$1/(1+\sqrt{T(w)})$
Insh	0.045	1.21	0.83
Veyatie	0.154	1.39	0.72
Leven	0.532	1.73	0.58
Shiel	1.493	2.22	0.45

In contrast to the 33-fold range in T(w) over these sites, the flushing correction terms differ by less than 2-fold.

The lines described by the equations of type (1) above and listed in Table 1, are very similar; **Figure 1** compares 2 of these. Considerable differences are revealed however, between curves corresponding to different flushing regimes as described by the equations of type (2) referred to above; to illustrate this **Figure 2** uses ranges of values that are well-representative of Scottish systems (Bailey-Watts 1990; Bailey-Watts, Sargent, Kirika and Smith 1987; Bailey-Watts and Kirika 1991). Indeed, long-term records of the discharge of water from Loch Leven, show annual flushing rates ranging from *ca* 0.5 to 2.5 y^{-1} (Ledger and Sargent, 1984) equivalent to T(w) values of 0.4 to 2.0; moreover, within almost any year, monthly flushing rates varied some 10-fold, and in the long-term, flushing in any one month, e.g. July, February, has also varied over an order of magnitude (Bailey-Watts, Kirika, May and Jones 1990).

Regression equations relating annual mean chlorophyll *a* concentrations with the flushing-corrected, average annual inflow concentrations of P are of the same form as Equation (2)

(OECD 1982). As an example, the equation obtained from the Shallow Lakes and Reservoirs report is as follows:

$$[Chl] = 0.43([P]_j/(1+\sqrt{T(w)}))^{0.88}$$

Substituting values obtained for $[P]_j$ and T(w) at Loch Leven in 1985 (Bailey-Watts, Sargent, Kirika and Smith 1987), an annual mean pigment concentration of 23.5 $\mu g\ l^{-1}$ is obtained. This compared very favourably with the figure of 21.0 $\mu g\ l^{-1}$ calculated from the chlorophyll values measured every 8 days throughout 1985.

Much of the remainder of this report is concerned with the factors that cause a system to deviate from the lines of best fit described by these equations. As in many of the cases examined under the OECD programme points fall in either the top left hand or bottom right hand areas of graphs of the type illustrated in Figures 1 and 2. The analysis is somewhat subjective, because actual data are very limited. Indeed less than 10 waters in Northern Britain have been the subject of loading studies, with inflows and open water being sampled at the appropriate short time intervals: Lough Neagh in N. Ireland (Foy, Smith and Stevens 1982; Gibson, Smith and Stewart 1992) and the Scottish lochs, Leven (Bailey-Watts and Kirika 1987; Bailey-Watts, Sargent, Kirika and Smith 1987), Eye (Bailey-Watts and Kirika 1991), Lowes, Balgavies and Forfar (Harper and Stewart 1987) and the Loch of Strathbeg (Hancock 1982). However,these works, and a number of desk studies of land use of other catchments combined with occasional sampling for water quality characterisation (Gibson 1976, 1986, 1989; Bailey-Watts, May, Kirika and Lyle, in prep) amount to a considerable body of limnological experience.

The following section summarises the concern over, and the extent of accelerated eutrophication in Scotland and Northern Ireland, while section 3 discusses the factors determining P loadings, and in-lake concentrations of P and chlorophyll. The final section is an overview of those factors causing a system to deviate significantly from model predictions of P loading-concentration and P-chlorophyll relationships.

2. THE EXTENT OF ENHANCED NUTRIENT ENRICHMENT OF STANDING WATERS IN SCOTLAND AND NORTHERN IRELAND

As elsewhere, the accelerated enrichment of lentic waters in Northern Britain takes many forms and the classic eutrophication sites where agriculture and/or urban wastes dominate have already been mentioned. However, the problem in Scotland and N. Ireland is perhaps different with the expansion of (i) afforestation (Gibson 1976; Bailey-Watts, Kirika and Howell 1988), (ii) cage-fish farming (Phillips *et al.* 1990) and (iii) the gradual increase in the numbers of dwellings with septic tank facilities. Indeed, it is possible that the actual increase in, or threat of, eutrophication from these sources, and the effect on rural Scotland and Northern Ireland, is giving rise to as much concern, as the problems stemming from the more conventional sources of nutrients such as urban waste and agriculture. In spite of the relative lack of sound quantitative data the potential effects of these practices are regarded very seriously; in Scotland, for example, many of the waters under threat constitute domestic supplies needing hitherto, very rudimentary treatment (Greene 1987; Greene and Taylor 1989); this situation reflects the bias - conscious or otherwise - of the fish-farmer and the forestry developer towards the more pristine landscapes. A number of waters associated with these industries are known to support algal populations that might not otherwise have been expected (Gibson 1976, 1986, 1989 for N. Ireland; Bailey-Watts, May, Kirika and Lyle, in prep, for Scotland). There are many 'hot-spots' of stream inflow concentrations of > 10^2 μg P l^{-1} in Scotland, although over the country as a whole levels of 10 μg P l^{-1} or less prevail. Indeed, values exceeding 1 **mg** l^{-1} have been recorded by e.g. Swift (1987) in a stream draining a recently-fertilised forest area.

Experimental evidence suggests that additions of P alone to the waters, can effect increases in algal numbers (Bailey-Watts, Kirika and Howell 1988); it is thus likely that other nutrient requirements can be met from existing surces. In this connection, however, it should be borne in mind that nutrient-rich wastes rarely consist of P alone; forests are commonly fertilised with mixtures of phosphorus, nitrogen and potassium, and phosphate itself is often applied as a salt of calcium (Taylor 1986), and fish-cages release nitrogenous as well as phosphatic wastes (Phillips, Bailey-Watts *et al.* 1990).

Land put over to various agricultural uses, is commonly limed, and this may reduce potential P losses. Thus, at Loch Leven, for example, while P runoff from agricultural land is considerable, the amounts lost from this source are equivalent to < 5% of the amounts of fertiliser P applied (Bailey-Watts and Kirika 1987).

3. FACTORS DETERMINING PHOSPHORUS (P) LOADINGS, AND IN-LAKE P AND CHLOROPHYLL CONCENTRATIONS

It is worth considering separately the factors controlling (i) inflow P, (ii) lake P and (iii) chlorophyll concentrations, as these determine the positioning of a point on graphs such as those shown in Figure 1. A number of interacting factors is involved, so it is useful to distinguish between those controlling eutrophication *per se*, ie the actual rates and seasonal patterns of P input to a system (as defined by $[P]_j$), and those determining the immediate in-lake result of this, ie $[P]\lambda$. Then, the determinants of the major biological manifestations of the enrichment can be discussed with reference to chlorophyll.

3.1 Factors controlling the levels of, and seasonal fluctuations in feeder stream P

Two classes of factors control P levels in feeder waters. One includes the numbers of people, and the types and extent of agricultural practices and industries such as forestry that lead to an export of P. The other group of factors controls the actual transport of the nutrient to the streams, and it is useful to distinguish here between runoff-dependent and runoff-independent transport. The rainfall regime and the catchment terrain as regards slope, soil structure and vegetation cover, influence the runoff-related inputs.

Contrastingly, these factors have little effect on the loadings of P from e.g. sewage treatment works, intensive livestock units and land-based fish farms, although runoff water will affect the resulting concentrations of P in the waters receiving these effluents. **Figure 3** which is based on Scottish studies, distinguishes between the patterns of P concentration resulting from different sources of the nutrient, and highlights the effects of rainfall variation on P levels by taking two contrasting situations well-representative of the vagarious weather regime of Northern Britain.

3.2 Factors controlling the levels of and seasonal fluctuations in lake P ($[P]\lambda$)

Traditionally, $[P]_\lambda$ refers to the concentration in the open water column. It should be noted, however, that in shallow waters at least, this will be generally underestimated since P incorporated into rooted vegetation and bottom-dwelling organisms will be ignored. In almost any water too, the small samples taken for P (and other chemical) analysis will tend to under-represent large, but relative sparse, macro-algae, zooplankton and of course, fish.

Values for $[P]_\lambda$ are controlled to a considerable extent by the factors determining $[P]_j$ discussed above. However, there are additional issues which affect $[P]_\lambda$:

i) *external supplies of nutrients that do not enter the system via the feeder stream network;* good examples are aeolian and rain derived material deposited directly on the lake surface, and inputs from fish cages and from roosting geese (see below), although effluents from sewage treatment works discharging directly into a standing water (as in the case of a works at Loch Leven) are also important.

ii) *fluxes between lake sediments and the overlying water;* this includes the sedimentation onto, or adsorption of material by the bottom deposits, as well as the release of e.g. phosphate from deep sediments particularly under anoxic conditions (see e.g. Mortimer 1942) but also from shallow, littoral deposits due to wind-induced mixing of P-rich pore waters (Drake and Heaney 1988).

The excreta of wildfowl such as over-wintering geese, which feed largely (by day) on land and roost (by night) on the water (Hancock 1982; Bailey-Watts, Sargent, Kirika and Smith 1987; Bailey-Watts and Kirika 1991), comprise an interesting example of a 'flushing-independent' source of nutrients, and one adding virtually nothing by way of water to increase flushing rate. A number of waters including the Loch of Strathbeg and Loch Eye receive a large proportion of their total P input *via* geese. Additional sources of P from in-loch fish cages can be considered as being similar to inputs from roosting birds, although perhaps more of a 'point-source' nature. Phillips, Bailey-Watts *et al.* (1990) and Bailey-Watts, May, Kirika and Lyle (in prep) have identified a number of sites where this source of nutrients is a significant part of the total P burden. P release from sediments is accompanied by no extra water to increase P. Nevertheless, the flushing regime determines very much the extent and duration of P release. When low flushing and warm weather conditions coincide, the release of P from sediments can result in enormous accumulations of the nutrient in the water column - as in (i) Loch Leven where levels exceed 150 mg m^{-3} in certain summers (Bailey-Watts, Kirika, May and Jones 1990; Bailey-Watts, May and Kirika 1991), (ii) Coldingham Loch, with >220 mg m^{-3} (Bailey-Watts, Lyle, Kirika and Wise 1987) and (iii) Yetholm with >400 mg m^{-3}, Kilconquhar with >1g m^{-3}) and Hoselaw with >5g m^{-3} (Bailey-Watts, unpublished observations). Then the OECD models are far from appropriate for explaining the relationships between $[P]_\lambda$ and $[P]_j$.

The report by Bailey-Watts, May, Kirika and Lyle (in prep.) identifies a number of Scottish lochs in which P is likely to be released from the sediments in summer. In the case of a number of the Lunan lochs (Tayside) this is because stratification is intermittent, and by the summer, even the bottom water (and thus surface sediment) temperatures are high. In addition there are shallow waters which do not stratify, but are warm throughout the column by summertime. Contrastingly, in the deeper lochs such as Shiel and St Mary's, bottom water and surface sediment temperatures remain low throughout the year - though not as low as the 'classic' temperature of 4°C. Partly as a result of this, the oxygen levels remain well above zero, but not necessarily above 60-70% saturation.

The factors bringing about increases in $[P]_\lambda$ are thus, numerous, and their individual influences on temporal patterns of $[P]_\lambda$, will differ both within and between lakes. This is because the seasonality of supply differs between the sources. Figure 4 illustrates this and again indicates the likely influence of contrasting flushing regimes.

In contrast to the factors that raise $[P]_\lambda$ by supplying the nutrient to the loch, there are physical, chemical and biological processes that remove it from the column.

physical factors - flushing: in situations where release of P from sediments and ingress of material *via* wildfowl or cage-reared fish are important, incoming water may be relatively dilute with regard to P; then, the throughflow of water will effect a lowering of the concentration in the lake.

sedimentation: in addition to the fact that considerable amounts of P arriving at a loch in particulate form may sink rapidly onto the deposits, losses are incurred by 'heavy' planktonic algae such as diatoms, through cells sedimenting out of the column; a comparison of weekly changes of dissolved silica concentrations with those in the silica likely to be incorporated into diatoms, suggested that during some weeks over the winter 1988-1989 in Loch Leven, the observed increases in diatom biomass were equivalent to less than one-fifth of the actual increases in these populations (Bailey-Watts 1991).

adsorption onto surfaces: the affinity of phosphate ions for surfaces of sediments and aquatic plants, leads to a removal of the nutrient from the water column (see e.g. Brook and Holden 1957; Bailey-Watts, Kirika and Howell 1988).

chemical factors: complexing of P with inorganic compounds such as iron and manganese hydroxides, and calcium compounds is well-known; these processes may bring about a lowering of water column P levels.

[Dissolved P may also complex with humic materials (Jones, Salonen and de Hann 1988), but these would still be included in the open water analysis.]

biological factors: here, the concern is with plants which bring about a reduction in TP concentrations in the water column. These include planktonic diatoms, which - as already indicated - incur heavy losses through sinking and remove P by that mechanism. In addition, there are rooted hydrophytes and attached algal communities which sequester P and remove it from the water *per se,* but not actually from the column.

Figure 5 summarises the processes that are important in reducing $[P]_\lambda$ and indicates the likely seasonal patterns in the relative importance of the various factors.

3.3 Factors affecting algal development

The concern here is with factors that determine how many algae are produced - and then subsequently recorded - per unit of P supplied. These factors are of major importance in defining the sensitivity of a waterbody to its particular burden of P. There are four issues that need to be considered in developing conceptual models of the relationship between [P] and [chl].

i) *the fraction of P loading that may not be available to algae or other plants, regardless of the sufficiency of their other environmental requirements*; included here will be certain particulate and organic constituents of the incoming P.

ii) *the abundance and status of potential competitors with plankton, for the P resource*, i.e. aquatic hydrophytes and algal communities associated with shallow muds (the epipelic forms or epipelon), stony surfaces (epilithon), sand grains (epipsammon) and other plants (epiphyton for closely attached species, periphyton for loosely attached swathes of e.g. *Cladophora, Spirogyra*).

iii) *environmental factors determining the production of algal cells*;

    (a) the light regime, as affected by incoming radiation, its attenuation due to dissolved colour and particulate matter, and the depth of mixing (z_{mix}) the mean depth (z) and the z_{mix} : z ratio; all of these affect net photosynthesis, and influence the balance between 'shade' and 'sun' species.

(b) the stocks and rates of supply of nutrients, including N as well as P, through their effect on general biomass maxima.

(c) the concentrations and rates of supply of other elements e.g. silica, major ions and trace elements, through their effect on species composition.

(d) flushing rate, by controlling the time during which cells can grow and divide.

iv) *environmental factors determining the numbers of algal cells recorded/observed*: these are the loss factors including the physical processes of sinking (sedimentation) and flushing mentioned above, and biological factors such as (a) parasitism by fungi - especially *Chytridiales* and (b) grazing by Protozoa, Rotifers and micro-Crustacea. Both planktonic and benthic (sediment-associated) interactions are important in impacting considerably on the sequence of appearances of different types of phytoplankton, since many parasites and grazers target particular species.

4. OVERVIEW OF FACTORS CAUSING SIGNIFICANT DEVIATION FROM MODEL PREDICTIONS.

4.1 *General considerations*

Much of this section concerns the factors that result in a $[P]_j.[P]_\lambda$ co-ordinate which is significantly distant from the line described by the model equations. It is worth pointing out firstly however, that close-time interval chemical sampling, backed up by, preferably continuous, flow-gauging is vital. Inadequate sampling will produce misleading results. For example, sampling restricted to the 'growing season' (whatever that is) is likely to over-estimate the influence of the internal loading of P due to the release of phosphate from muds. Understanding of the system would also be impaired, by over-estimating the importance of bloom-forming algae and, in this connection the influence of filter-feeding micro-Crustacea in enhancing the success of the larger algae (Bailey-Watts 1986). By the same token, dense crops of algae, and the standing stocks of P incorporated into these cells, may be missed by sampling only in spring and summer; yet, these could affect considerably the mean annual values for $[P]_\lambda$. Indeed, late winter diatom populations in Loch Leven, often attain a considerably higher biomass than the traditionally more troublesome cyanobacteria in summer, when expressed as concentrations of P or chlorophyll per unit area of lake surface (Bailey-Watts 1988; in press). Also, with flushing being such a cardinal feature of eutrophication research, and a major determinant of $[P]_j$, $[P]_\lambda$ and the succession and biomass of phytoplankton, it needs to be measured as accurately as possible. This presents a major challenge because of the highly irregular, episodic nature of rainfall, and thus the delivery of water, to Scottish lochs. Intensive work at Loch Leven suggests that in some situations, more than 50% of the annual supply of TP from a stream may be transported to the lake in a single storm surge lasting but a few hours or days (Bailey-Watts and Kirika 1987). This was due to the effect on particulate P, and has parallels with some of the results obtained for the Lough Neagh catchment.

Another factor to be considered where a significant uncoupling of $[P]_i$ and $[P]_\lambda$ is found, is the time taken for a standing water to respond to its loading. Indeed, a 'new equilibrium' might be attained only after a prolonged period of stable loads. Moreover changes wrought through inter-annual variation in the abundance of fish or, as in the case documented for Loch Leven, zooplankton (Bailey-Watts, Kirika, May and Jones 1990), may be considerable, even when loads are relatively constant.

4.2 *Factors resulting in a high $[P]_\lambda : [P]_i$ ratio*

A higher than predicted ratio of in-lake to inflow P concentration will arise when significant contributions to the total loading of P to the water column originate from:

i) within the lake itself i.e.:

  - by the circulation of previously laid-down P in the sediments, through re-suspension of particulate matter and mixing of pore waters
  - by the release of phosphate ions from their normal chemical binding sites
  - from in-lake fish-rearing cages.

ii) outside the lake, but transported to the water by means other than the feeder streams: here, under water springs, aeolian sources and birds are the major possibilities that come to mind.

4.3 *Factors resulting in a low $[P]_\lambda : [P]_j$ ratio*

The factors affecting a real in-lake to inflow P concentration ratio that is considerably, lower than the existing models would predict, are not so conceivable, although the situation could arise where an incoming P load is dominated by heavy, particulate components which are resistant to bacterial and/or chemical re-mineralisation, as these may sediment out as soon as they reach the lake. However $[P]_j$ may appear to exceed $[P]_\lambda$ where the P supplies are sequestered by rooted or attached organisms. This is because this P, is likely to have passed undetected, since analyses of the phosphorus contents of macrophytic vegetation and algal mats, for example, are rarely carried out. Shallow 'weedy' waters would be expected to exhibit this sort of deviation from the model predictions and example of this type of loch is Linlithgow (Lothian Region). In spite of high P loadings the water is very clear in summer (*ca* 6m Secchi disc transparency), and *Potamogeton* species, *Elodea* and large aggregations of filamentous green algae dominate the underwater scene.

5. RECOMMENDATIONS FOR FUTURE WORK

The albeit few, intensively studied sites have provided a good base from which to approach further catchments, and assess their status and the main features of the way they function as regards eutrophication. The earlier work has also identified a number of major gaps in knowledge, however, and more quantitative studies are needed to assess the influence of:

- the widespread, 'creeping' increase in the number of septic tank units over the country,
- the spreading of dung and organic slurries to improve grassland,
- forestry fertilisation programmes.

To advance knowledge on these aspects, such that we materially improve our ability to predict and prevent, or at least control, their impacts, requires close attention to seasonality. Indeed, seasonality is a major feature of:

- the sampling schedules operated by the significant programmes on eutrophication assessment
- newer thinking on classifying nutrient inputs and distinguishing their main effects
- current trends in modelling the eutrophication process, i.e. loadings *per se*, as well as the responses of the receiving waters.

Intensive studies of specific areas, sub-catchments or even fields, put over to a restricted range of activities leading to nutrient enrichment, are urgently needed. Only through such studies can our *repertoire* of nutrient export coefficients (see below) be increased. To address the requirement for knowledge on the current and likely future state of freshwaters nationwide, the Institute of Freshwater Ecology is developing another approach, which has so far shown considerable promise (Bailey-Watts, May, Kirika and Lyle, in prep.). The programme for each lake catchment is comprised of 2 main elements:

i) desk analysis of catchment characteristics (e.g. topography, geology, aspect, slope, landuse and rainfall), combined with information on nutrient export coefficients: this allows the eutrophication pressures on a system to be estimated and certain sensitivity factors (particularly those relating to lake form and flushing rate) to be assessed.

ii) Summer limnological reconnaissance: a summertime sampling (and preferably a winter time field programme, too) provides information on the chemical and biological status of the system; this is not only of value in its own right, but critical for validating the predictions about eutrophication and its likely consequences, based on (i) above.

6. ACKNOWLEDGEMENTS

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Figures

Figure 1

Lines of best fit to the data points relating $[P]_\lambda$ and $[P]_j$ from (a) the Shallow Lakes and Reservoirs programme, and (b) the combined OECD study sites.

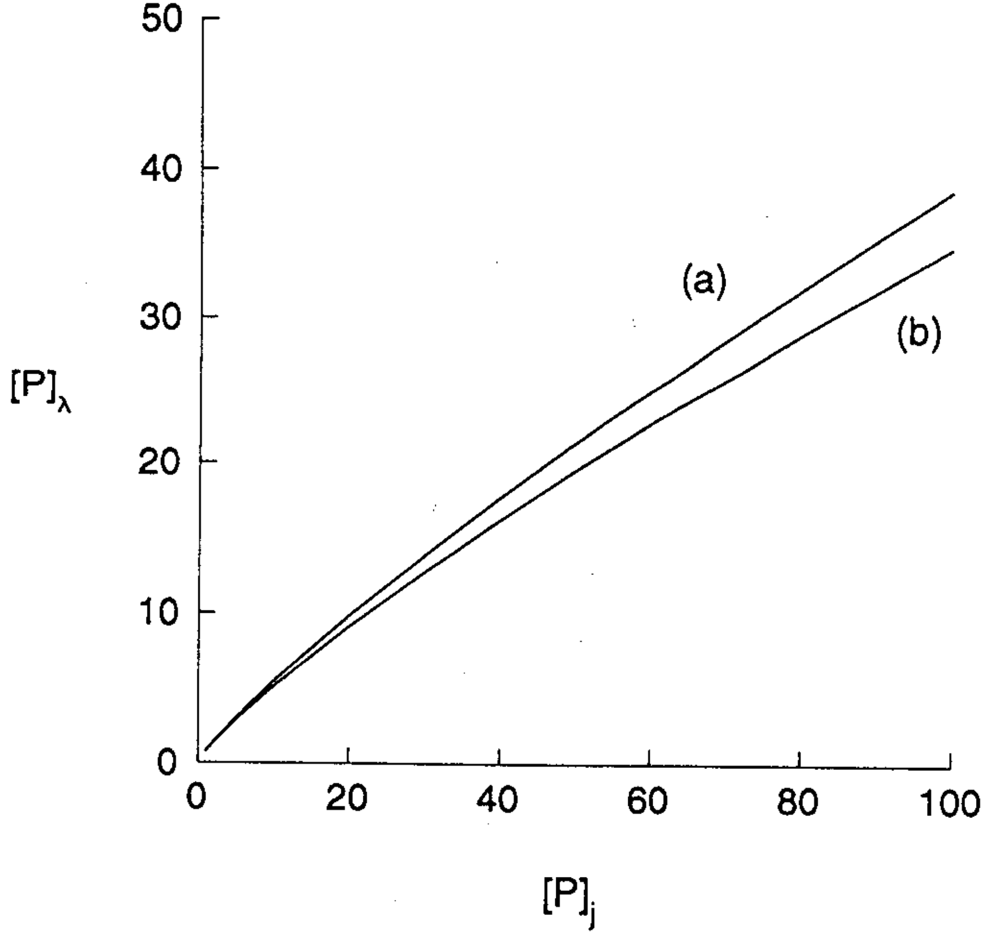


Figure 2

Annual mean in-lake phosphorus concentrations ($[P]_\lambda$) in relation to flushing corrected inflow concentrations for the OECD Shallow Lakes and Reservoirs model ($[P]_\lambda = 1.02\,X^{0.88}$, where $X = [P]_j / (1+ \sqrt{T_{(w)}})$), (a) for situations where $[P]_j$ is 1, 25, 50 or 100 mg m^{-3} and $T_{(w)}$ varies from 0.3 to 3.0 years, and (b) for situations where $T_{(w)}$ is 0.3, 1.0 or 3.0 years and $[P]_j$ varies from 1 to 100 mg m^{-3}.

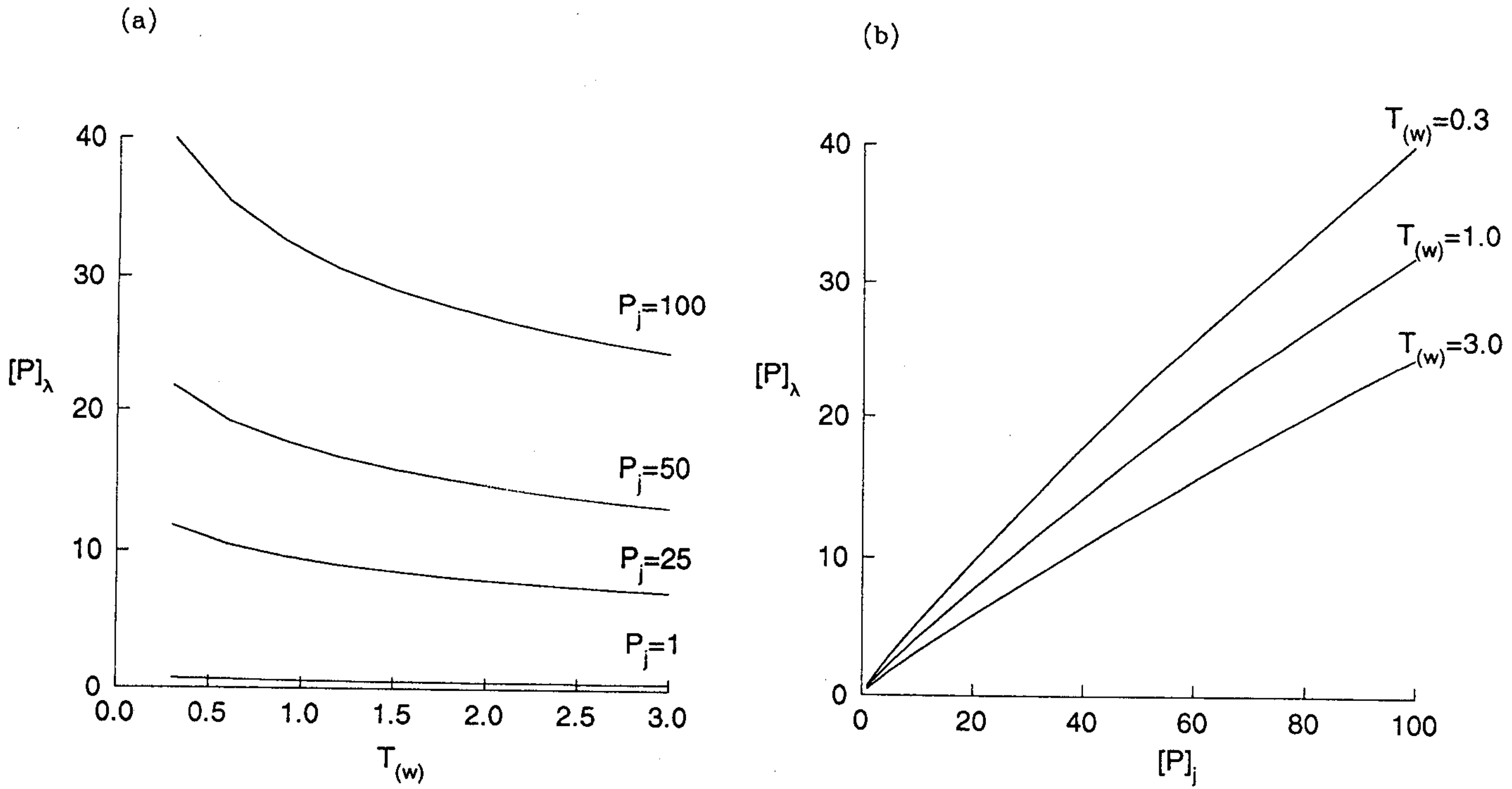


Figure 3

Seasonal patterns in the loadings of P (PL) from 3 different sources (I–III), and the likely fluctuations in P levels [P] in a receiving stream, under 2 contrasting flow (weather) regimes (Q).

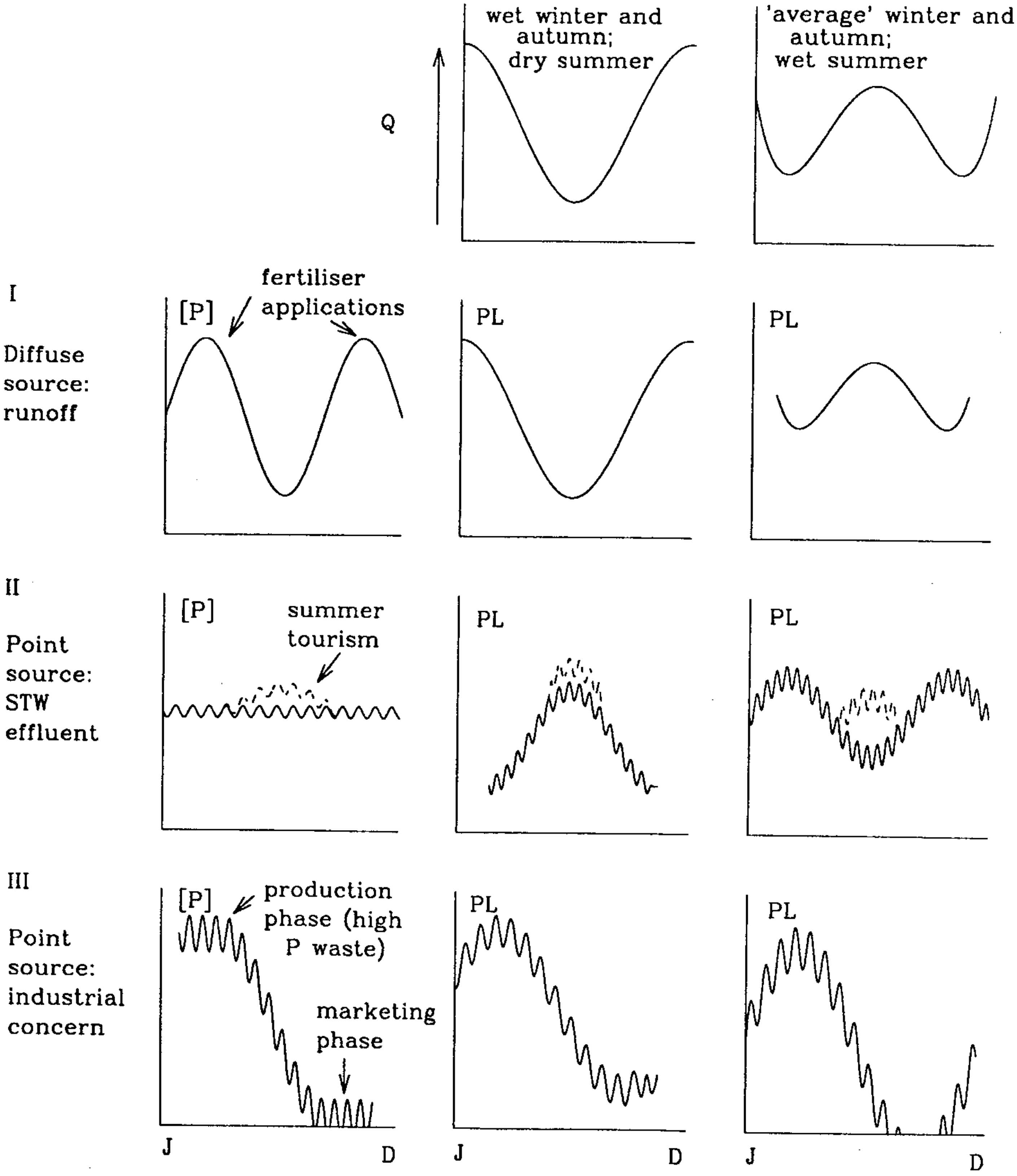


Figure 4

Seasonal patterns of in-lake phosporus concentration ($[P]_\lambda$) resulting from different types of P supply, and under 2 contrasting flushing regimes (ρ).

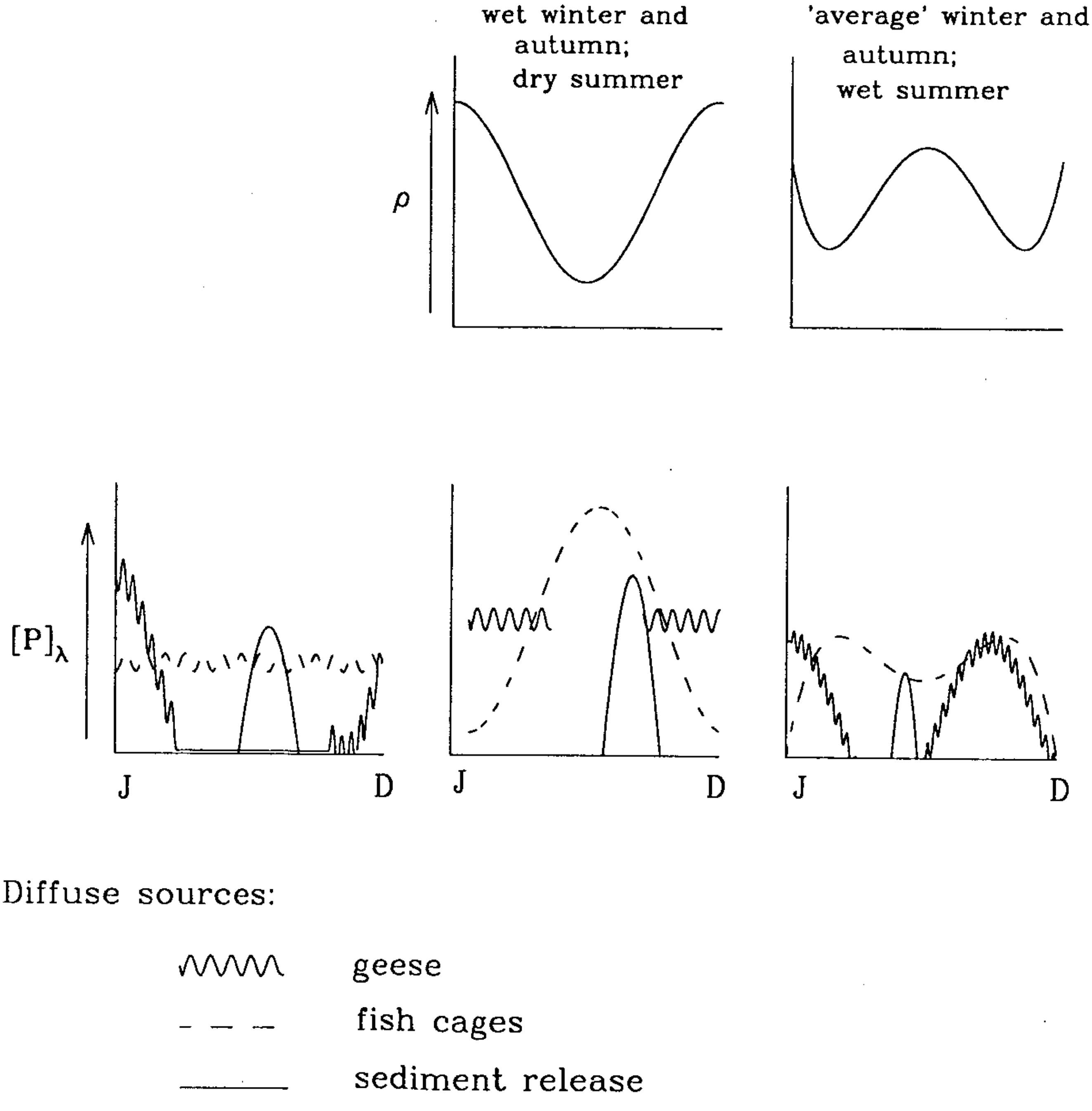
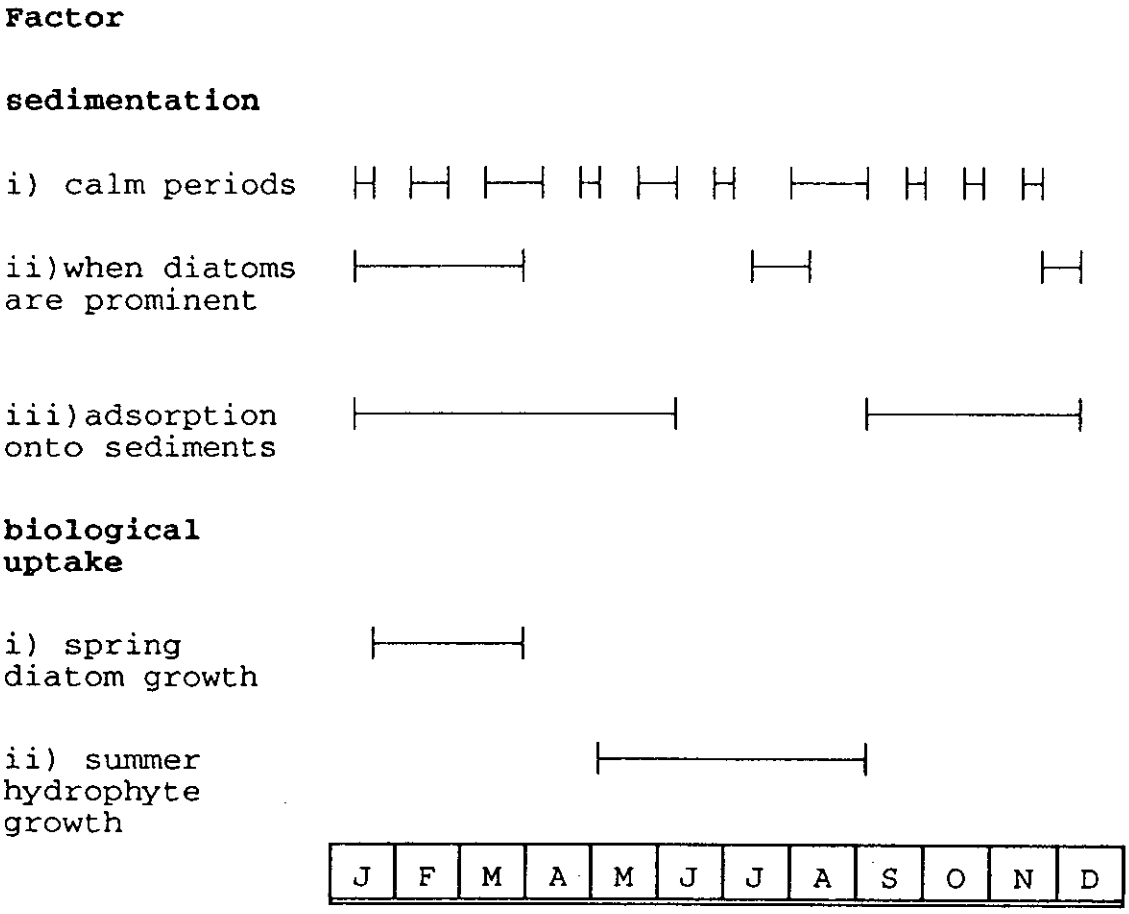


Figure 5

Seasonality of factors/processes removing P from the water column and, in so doing, leading to an increase in the inflow P to in-lake P ratio, and an indication of the situations under which, and times of year, these factors are likely to be most important.



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