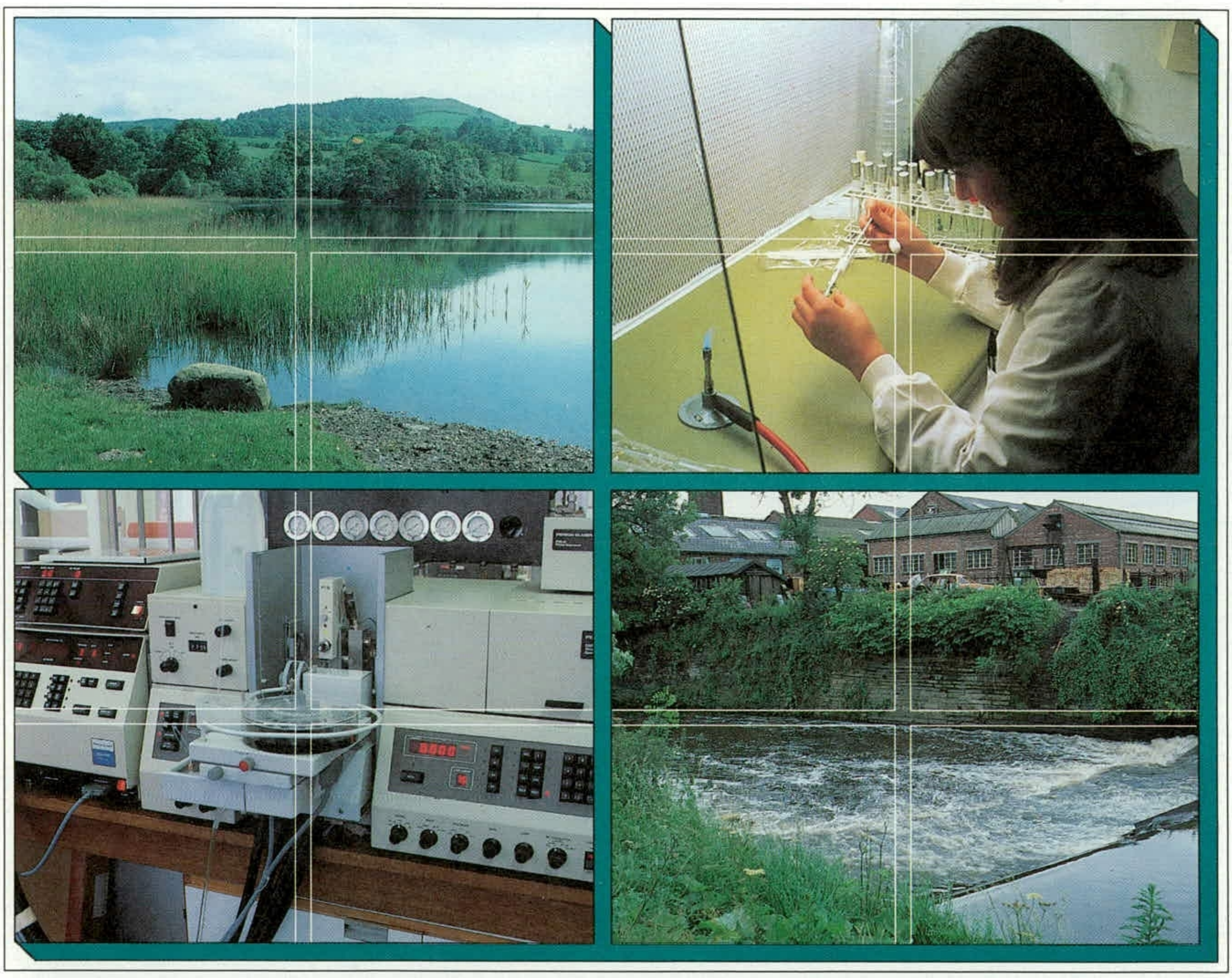
Institute of Freshwater Ecology

Threats to the oligotrophic status of Loch Shiel

Principal investigators:

A E Bailey-Watts BSc, PhD, MIWEM (Project Manager)
I D M Gunn BSc, MSc
A Kirika
A A Lyle

Final Report to Scottish Natural Heritage
(March 1993)



Natural Environment Research Council

THREATS TO THE OLIGOTROPHIC STATUS OF

LOCH SHIEL

Institute of Freshwater Ecology
Edinburgh Laboratory, Bush Estate, Penicuik
Midlothian EH26 0QB, Scotland

Telephone: 031 445 4343; Fax: 031 445 3943

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Project Manager: A E Bailey-Watts

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A E Bailey-Watts, BSc, PhD, MIWEM
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Summary

New information on fertiliser application to coniferous forest in the Shiel catchment, and on feed used in the cage rearing of salmon smolts in the loch, results in a desk-generated total phosphorus (P) loading of 7.39t y^{-1}.

The main land-derived sources are coniferous forest (3.99t or 54% of the total) and rough grazing/moorland (1.2t and 16%). Losses from the smolt cages are estimated at 0.87t (12%).

Expressed as a specific areal load the total input is 0.377 g m^{-2} y^{-1}. This figure comes approximately half-way between the OECD 'permissable' and 'dangerous' limits of 0.25 g m^{-2} and 0.50 g m^{-2} respectively, for a waterbody of the mean depth of Loch Shiel i.e. 41m.

An annual mean loch-wide P concentration ($[P]_\lambda$) of 7.3 $\mu g\ l^{-1}$ is calculated, when the loading is combined with a desk-derived flushing rate (ρ, 0.69 loch volumes y^{-1}), a model-predicted P retention coefficient (0.45), and the measured mean depth quoted above, according to the Dillon and Rigler model.

Surface water TP levels ranged from 3.6 $\mu g\ l^{-1}$ to 7.5 $\mu g\ l^{-1}$ on 25 September 1992 in the area of the Dalilea fish cages which have been in the loch for longer than any of the other cages, and are set in one of the shallowest and most sheltered situations of the various groups of cages. Surface water SRP concentrations were extremely low (0.3 $\mu g\ l^{-1}$ to 1.6 $\mu g\ l^{-1}$) but these represent what is remaining of this bio-available form of P after plant uptake. Contrastingly, near-sediment levels ranged from 1.0 $\mu g\ l^{-1}$ to 203 $\mu g\ l^{-1}$, suggesting some remobilisation of phosphate from the sediments; however, a value of 50 μg SRP l^{-1} was obtained at the 'control' site in the middle of the south-west basin of the loch.

Spatial variations in P and organic matter content of the surface sediments in this area were not strongly correlated, nor did they relate consistently to proximity to a cage; however, 2 relatively high organic matter values (27% and 29% of sediment dry weight) were recorded beneath 2 of the 3 cages examined, and corresponded to near-sediment dissolved oxygen values of 20% - compared to 80% to 90% elsewhere.

Phytoplankton levels expressed as chlorophyll *a* were very low i.e. 1.9 $\mu g\ l^{-1}$ to 2.4 $\mu g\ l^{-1}$. Oligotrophic desmid, diatom and chrysophyte species were noted, but also the cyanobacterium (blue-green alga) *Anabaena*, producing a visible though very minor bloom.

Benthic invertebrates gave the clearest indications of extreme, though localised, effects of the cages; total numbers of invertebrates varied from 10^4 m^{-2} in organic-rich sites under cages, to 10^2 m^{-2} in open water, and the relative abundances of e.g. tubificid worms and larval Chironominae generally increased with increasing levels of organic matter.

A review of available data on P, nitrate and chlorophyll in Loch Shiel, suggests that the loch is oligotrophic and has been for at least the last 15 years. Concentrations have fluctuated, but only in the shallower of the 2 basins (south-west) has P occasionally approached or exceeded 10 $\mu g\ l^{-1}$ which is used here as the limit of 'oligotrophy'. Early spring and winter nitrate concentrations rarely exceed 70μg N l^{-1}, although in 1991 values of 100$\mu g\ l^{-1}$ to 150$\mu g\ l^{-1}$ were recorded. The low-N environment suggests that the production of many planktonic algae would be limited by N shortages if P levels alone were allowed to rise. In some years, phytoplankton levels are significantly and consistently higher in the shallower basin, but have not exceeded 3 μg chlorophyll *a* l^{-1}. The records illustrate inter-annual differences in the timing of maxima and minima and in the values achieved.

In that any nutrient enrichment accelerated by Man constitutes a threat to the oligotrophic status of Loch Shiel, coniferous forestry in particular, but also the albeit relatively minor contributions from fish-farming, arable agriculture, improved grassland management, and the disposal of domestic waste from houses and hotels, must be viewed as threats.

It is essential that the fertilisation data on which the present loading from coniferous forest is based, is checked. New research is proposed to assess the loadings of P *via* the River Polloch; this drains land containing the majority of the coniferous forest in the Shiel catchment. This sub-catchment also contains 34 standing waters which are likely to trap P; if so, the estimate of the forestry contribution to the total P burden on Loch Shiel is too high, as would be the total loading figure. The effects of nutrient inputs from the Polloch may be minimised by transport of water to the deep, north-east basin of the loch, rather than the shallow, south-west basin.

The biological effects of enrichment, not elevated levels of nutrients, are the main cause for concern. Moreover, the overall physical structure of, and the low concentrations of e.g. nitrate in, Loch Shiel, tends to mediate against an efficient conversion of P supplies to e.g. phytoplankton. However, prolonged stratified conditions in a good summer, could lead to more evident algal growths; moreover, N-fixing blue-green species would be expected to predominate in the high phosphate-low nitrate environment that can be envisaged if P enrichment occurred without enhanced inputs of nitrate.

Monitoring is essential for establishing baseline values by which to judge future change. A suite of chemical and biological, trophic monitoring programmes is proposed, and their relative advantages and disadvantages are outlined. The rationale of monitoring Loch Shiel is explored, however. It is likely that conditions recorded in one year may reflect events taking place years previously. A programme of restoration rather than prevention, would be necessary by the time significant increases in nutrient levels, and certainly by the time biological responses, were evident.

It is concluded that every reasonable effort should be made to reduce the rate of enrichment, or at least prevent it proceeding any faster than it is at present. Comments are made on the possible long-term nature of enrichment of Loch Shiel, and whether thought should be given to maintaining the oligotrophic status over geological timescales over which the system would be expected to become richer even under 'natural' conditions.

CONTENTS

1. INTRODUCTION 1
 1.1 Background 1
 1.2 Aims of the project, and the approaches adopted 3

2. GENERAL CHARACTERISTICS OF THE SHIEL SYSTEM WITH A BEARING ON THE TROPHIC STATUS OF THE LOCH 5

3. INVESTIGATIVE METHODS 7
 3.1 Desk studies 7
 3.2 Field investigations 11
  3.2.1 Sampling sites 11
  3.2.2 Probe measurements 12
  3.2.3 Chemical and biological sampling 12
 3.3 Laboratory analyses 13
  3.3.1 Phosphorus and organic matter determinations 13
  3.3.2 Phytoplankton analysis 14
  3.3.3 Invertebrate benthos 14

4. RESULTS AND DISCUSSION 16
 4.1 Desk analyses 16
  4.1.1 An estimate of the inputs of total phosphorus (P) from the catchment 16
  4.1.2 An estimate of the P loading attributable to smolt-rearing in the loch 17
  4.1.3 A desk-derived total P loading to Loch Shiel 18
  4.1.4 Predicted total P concentration in the loch 18
 4.2 Field results from the Dalilea fish cage area - September 1992 18
  4.2.1 General physico-chemical conditions 18
  4.2.2 Phosphorus levels in the water column 19
  4.2.3 Phosphorus and organic matter content of the surface sediments .. 20
  4.2.4 Phytoplankton 21
  4.2.5 The invertebrate benthos 22
 4.3 The trophic status of Loch Shiel - historical context 24

5. CONCLUSIONS - THREATS TO THE OLIGOTROPHIC STATUS OF LOCH SHIEL 26
 5.1 The current situation 26
 5.2 Recommendations for preserving the oligotrophic status of Loch Shiel 27
 5.3 Future work and monitoring 28

6. ACKNOWLEDGEMENTS 31

7. REFERENCES 32

FIGURES 36

1. INTRODUCTION

1.1 Background

There is not an enormous amount of information on Loch Shiel, but it has been the subject of various limnological studies since the turn of the century [Murray and Pullar, 1910] and during the last 15 years in particular [Maitland, 1981; unpublished Highland River Purification Board and Institute of Aquaculture (IoA) reports; Bailey-Watts *et al.*, 1992a,b]. This reflects a growing realisation that, while rich, shallow waterbodies are the most likely to exhibit signs of eutrophication with, for example, dense algal blooms, large, deep lochs may not be immune from the effects of enhanced inputs of nutrients. This is of considerable concern in the case of Loch Shiel, as its designation as an SSSI rests on its oligotrophic status. Indeed, the present study examines the nutrient status of the loch in the context of the term 'oligotrophic'. Any considerations about the trophic status of a waterbody should include nutrients - and especially nitrogen (N) and phosphorus (P) because their availability influences considerably the general levels of plant productivity. However, this report focuses almost entirely on P and the potential threats of the inputs of this element to the oligotrophic status of Loch Shiel. While in such a deep loch, light conditions play an important part in determining phytoplankton production, P is likely to be the main limiting nutrient. This means that enrichment with P would lead to increased production of phytoplankton, although the relative abundance of the component species may change depending on the availability of other elements required for growth, for example, nitrate.

The term 'oligotrophic' means different things to different people, but it generally refers to a waterbody in which the annual mean total P concentration does not exceed 10 $\mu g\ l^{-1}$ [OECD, 1982] i.e. the nutrient status. It is important to remember, however, that even in very rich waters, it is the biological status in the form of e.g. algal blooms and increased invertebrate production, which stem from the eutrophication are the cause of major concern - not the increased nutrient concentrations *per se*. Equally, while it may take a long time to markedly increase the levels of nutrients, let alone the overall biological productivity, of a system as large as Loch Shiel, it would take many years to restore the waterbody, once the effects of nutrient enrichment took hold. And in any event, the concern of this report is not with markedly, even mildly, accelerated eutrophication, but with threats to its oligotrophic status as defined by the OECD P level quoted above.

The depth of Loch Shiel (mean value 41m) would tend to generally mediate against the production of high plant biomass whatever the nutrient availability. However, such deep waters often stratify in summer, such that planktonic algae can be entrained in a generally well-illuminated, upper mixed layer (the epilimnion) of relatively few metres depth, and can build up their populations more effectively. Large waters are often relatively poorly flushed, so that there is plenty of time for planktonic organisms to increase in numbers before they are washed out of the basin. The physics of the loch thus determines what biology will result, for a given chemical situation - something generally appreciated many years ago by e.g. [Rawson, 1953], and reflected in the incorporation of mean depth (**z**, in metres) and flushing rate (ρ, in loch volumes y^{-1}) or its reciprocal, the mean hydraulic retention time ($\tau_{(\omega)}$, in years or days) in classic eutrophication models [Dillon and Rigler, 1974; Vollenweider, 1975, 1976]. The degree to which material entering the basin is retained e.g. in the sediments, is also determined largely by physical factors, and the model of [Kirchner and Dillon, 1975] is based on a correlation between P retention and a quotient relating loch area and annual water input (see below). The attention to physical aspects in the case of Loch Shiel is particularly important, in view of the contrasting depths, and probable differences in flushing and P retention capacities, of its two basins.

As an exercise in national limnology in respect of eutrophication, the study by Bailey-Watts *et al.* [1992a] could not cover in detail any of the 31 systems examined, let alone the attention warranted for Shiel, which is one of Scotland's largest lochs [Maitland, 1981]. The present work aims to improve on this.

Although lying in an area of outstanding - albeit not natural - beauty, Loch Shiel is likely to receive significant amounts of nutrients from its catchment (forest, rough grazing land, for example), even if losses per hectare are low. This is simply because the catchment is so enormous i.e. 230km^2. There are also fish (smolt-rearing) cages in the loch itself, and because they contrast with the catchment-derived sources, in the following respects they are likely to have different impacts:

(i) the seasonal pattern of nutrient loading, which relates to fish growth, production and feeding schedules, rather than runoff in the manner of rain- or snowmelt-related inputs.

(ii) the negligible volume of additional (diluting, flushing) water introduced, *cf* runoff materials.

(iii) the accompanying chemical 'mix' (relatively high organic matter content).

(iv) the form of the P (significant soluble and bio-available components).

1.2 Aims of the project, and the approaches adopted

The aims of the present study were:

- to assess the degree of threat to the oligotrophic status of this loch, and distinguish between the likely extent and impacts of the P coming from external sources, and those relating to the cages.
- to predict future changes in the trophic status of the loch assuming for example, that smolt production was significantly increased.
- to suggest monitoring procedures for detecting such changes.

There were 4 main components of the work:

- a desk study of the type developed by Bailey-Watts *et al.* [1992a,b], but taking into account more detailed information on forest fertiliser application schedules, and fish cage production to improve the P loading estimate, and thereby provide a better prediction of P levels in the loch itself.
- an assessment of the current trophic status of the loch and a review of past nutrient and chlorophyll records; enabling actual P levels to be compared with the model-predicted figures.
- an examinantion of aspects of the chemistry and invertebrate benthos associated with the sediments in the Dalilea fish cage area; this was done (in September 1992) on the premise that fish cages are likely to impact more

immediately on the sediments of the loch, than on the general levels of nutrients or organisms in the water column.

- an outline of strategies for monitoring the condition of Loch Shiel; these range from the 'ideal' year-round, temporally and spatially intensive sampling programmes, to the more feasible (likely?) occasional, surveys of trophic indicator organisms; the rationale of such exercises is briefly discussed.

2. GENERAL CHARACTERISTICS OF THE SHIEL SYSTEM WITH A BEARING ON THE TROPHIC STATUS OF THE LOCH

Loch Shiel is a substantial waterbody of *ca* 80 x 10^7 m^3. This is some 16 times the volume of Loch Leven. Its surface area, however, is less than 1.5 times that of the Kinross loch i.e. 1960 ha *cf* 1330 ha. Shiel is thus a very deep waterbody; it extends to 130m in the eastern, deeper trench (lying along a NW-SE axis), and to 30m in the western basin (lying E-W). The mean depth of the loch, at *ca* 41m is thus very unrepresentative of the system as a whole.

Such a large body of water acts as a depository of a significant proportion of the material entering from its catchment - and probably *ca* 45% of the P (see below). The area drained by the loch is enormous i.e. 230 km^2, although it is only *ca* 12 times that of the loch itself. The climate in the area is characterised by very high rainfall - *ca* 2600 mm y^{-1} which equates to a net input of some 2200 mm y^{-1}, when corrected for evaporation. Because, the catchment-to-loch area ratio is so moderate, however, annual water runoff, at *ca* 51 x 10^7 m^3, is considerably less than the volume of the loch, and together with 4.2 x 10^7 m^3 y^{-1} entering in rain directly over the surface of the loch itself, amounts to only 0.69 loch volumes y^{-1} (ρ).

The water is of low conductivity (*ca* 30-40 μS cm^{-1}) and of slightly acid pH (5.5-6.0 units) [Bailey-Watts and Duncan 1981a; unpublished IoA reports]. The part of north-west Scotland in which this system is situated, is characterised by large tracts of rough grazing land. Bailey-Watts *et al.* [1992a] estimated that some 75% of the catchment consists of this type of cover while the corresponding figures for forest, improved grassland and arable agriculture are approximately 20%, 4% and <1% respectively. Other lochs cover about 0.5% of the catchment area. The fact that Loch Shiel and its catchment comprise an intrinsically pristine setting, may explain its attraction to the forestry and fish-farming industries.

Bailey-Watts *et al.* [1992a, b] estimated a total annual P loading of 4.8 t, of which some 33% was attributable to the fish cages, and 23% to forestry. These calculations used the following P loss coefficients (in kg P ha^{-1} y^{-1} unless otherwise stated): 0.2 for forest, 0.25 for arable land, 0.07 for rough grazing areas, 0.4 for improved grassland, and 20 kg t^{-1} fish produced y^{-1} from the cages; inputs from rural communities (on septic tank systems) assumed a *per capita* loss of 1 kg P y^{-1}. The forestry-related loss rate assumed that all forest was coniferous, and that half of the total area had been fertilised in the last 15 years, and had lost *ca* 2 kg P ha^{-1} y^{-1} in the first 3 years after fertiliser application [see Harriman, 1978; Malcolm and Cuttle, 1983], but virtually no further P thereafter. This results in a longterm (15-y) annual loss rate of 0.4 kg ha^{-1} y^{-1}, or half this for forest as a whole. The present study has allowed considerably more time to evaluate these preliminary estimates, and indeed, gather more information about forest fertilisation schedules and fish feeding in particular. The new results (see below) come to a quite different conclusion as regards the relative contributions of forestry and fish-farming to the total P supply to Loch Shiel, and this total is also considerably higher than the 4.8t estimated earlier.

Bailey-Watts *et al.* [1992a, b] pointed out that, as the mean hydraulic retention time of this loch is 1.45 y (1/0.69), 1.45 times the estimated annual loading of P will enter the loch in the time a volume of water equivalent to that of the loch has been renewed. The same study looked at the relationship between predicted P load, P retention, flushing rate and mean depth according to the model of [Dillon and Rigler, 1974 - see Section 3.1), and predicted a lake-wide P concentration (strictly, the 'spring overturn' but taken here as an approximate annual mean), of 4.9 $\mu g\ l^{-1}$. This was considered to be in keeping with the general oligotrophic nature of the system, as were the figures of 7.1 $\mu g\ l^{-1}$ (north east basin) and 12.4 $\mu g\ l^{-1}$ (south west basin) recorded in summer 1991, although the latter was special in exceeding 10$\mu g\ l^{-1}$ (*cf* figures between November 1977 and October 1978 [Bailey-Watts and Duncan, 1981a].

3. INVESTIGATIVE METHODS

3.1 Desk studies

There were two elements to the desk assessments of the trophic status of Loch Shiel. One of these reviewed data on nutrient and chlorophyll levels; the extent and various sources of these data are indicated in **Table 1**. The other part of the desk study firstly estimated the P loading to the loch and the contributions of the various sources to the total load, and secondly, predicted an in-lake P level that would result from this input.

For the loading estimation, the system described in detail by Bailey-Watts *et al.* [1992a] was used; here, different P loss coefficients from ones employed earlier were used for forestry and fish farming, in the light of the considerations outlined above, and the provision of new information.

First, discussions with Scottish Natural Heritage and Marine Harvest, and a review of the literature covered by Bailey-Watts, Kirika and Howell [1988] suggested that runoff from fertilised forest is initially high (i.e. losing *ca* 2 kg P ha^{-1} y^{-1} for the first 3 years, as assumed in the earlier study), but that it takes a further 4 years to decrease to background levels. What is more, this schedule is apparently reflected in the 7-year interval between (re-) fertilisation programmes practiced by forestry managers. (Even this is presumably not immutable, however, as, in some situations at least, the P status of the trees is checked, to determine whether fertilisation is necessary). The new data suggested that between 1985 and 1990, a total of *ca* 1520 ha (approximately 25% of the total area of forest estimated from maps) had been treated - for 'remedial', 'new' or 'restock' purposes - with unground phosphate rock almost exclusively by aerial application, at a rate of 450 kg ha^{-1} (equivalent to 60 kg P ha^{-1} y^{-1}):

Table 1: Water sampling programme at Loch Shiel for information on nutrient and chlorophyll levels in period 1979-1992. (CHL*a* = chlorophyll *a*; TP = total phosphorus; SRP = soluble reactive phosphorus; NO_3 = nitrate)

		1979	1980-1984	1985	1986	1987	1988	1989	1990	1991	1992
S.W. BASIN	No. sample visits	--	--	--	2 (17 for CHL *a*)	2 (46 for CHL *a*)	2 (26 for CHL *a*)	2 (48 for CHL *a*)	2 (9 for CHL *a*)	3 (21 for CHL *a*)+	1
	Parameters measured	--	--	--	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*
N.E. BASIN	No. sample visits	--	--	--	2 (17 for CHL *a*)	2 (46 for CHL *a*)	2 (26 for CHL *a*)	2 (48 for CHL *a*)	2 (9 for CHL *a*)	3 (21 for CHL *a*)+	1
	Parameters measured	--	--	--	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*	SRP,TP, NO_3, CHL *a*
INFLOW	No.sample visits	--	--	--	1 @	--	--	--	--	--	--
	Parameters measured	--	--	--	SRP	--	--	--	--	--	--
OUT-FLOW	No.sample visits	2 *	4 (y^{-1}) *	2 *	1 @	--	--	--	--	1 +	--
	Parameters measured	SRP	SRP	SRP	SRP	--	--	--	--	SRP	--

Notes;

(1) All samples taken by Marine Harvest/Institute of Aquaculture unless indicated otherwise: * Highland Purification Board; +Institute of Freshwater Ecology, 1 set of samples; @ Institute of Terrestrial Ecology; ITE measured SRP in the N.E. & S.W. Basins.
(2) S.W.Basin includes Marine Harvest/IoA sample sites 1 & 2.
(3) N.E.Basin includes Marine Harvest/IoA sample sites 3, 4, 5 & 6.

102 ha in June 1985

211 ha in August-September 1986

252 ha in September 1987

508 ha in September 1988

288 ha in September 1989

and 158 ha in September 1990

Some 940 ha (62%) of the fertilised area lies within 3.6 km of the loch edge, while the rest lies at a distance of >5 km from the loch. Assuming that P levels over forest areas are 'topped up' every 7 years, an average yearly loss of 1 kg P ha^{-1} is used, and this is included with the other loss rates in **Table 2**. Much of the coniferous forest around Loch Shiel, is situated in the area drained by the River Polloch. This sub-catchment contains 34 standing waters including Loch Doilet and a marshes between it and Loch Shiel itself. These features could have a considerable effect on P losses (see section 5.1).

The second factor to be modified, was the P loss figure for the fish cages. Now, a value of 7.85kg t^{-1} fish produced y^{-1} [Phillips, Mowat and Clarke, 1990], is used. (**Table 2**). It results from an analysis of data supplied by Marine Harvest on fish production, feeding schedules and the P content of the feeds and is only *ca*. one-third of the figure assumed by Bailey-Watts *et al.* [1992a] and taken from the literature collated by [Institute of Aquaculture *et al.*, 1990].

The third piece of new information, concerned deciduous woodland. This was not included in the earlier study and amounts to only 2.5% of the catchment area. Literature reviewed by Bailey-Watts *et al* [1992a] suggests that a loss rate of 0.1 kg P ha^{-1} y^{-1} is appropriate for this type of land cover.

Table 2: **Phosphorus loss coefficients used for estimating the loadings to Loch Shiel from the catchment, the smolt-rearing cages and the rural population; all values refer to annual losses of total P.**

Source	**Loss coefficient**
Land:	
Rough grazing/moorland	0.07 kg ha^{-1}
Coniferous forest	1.00 kg ha^{-1}
Improved grassland	0.40 kg ha^{-1}
Deciduous woodland	0.10 kg ha^{-1}
Arable agriculture	0.25 kg ha^{-1}
The rural community:	1.00 kg $person^{-1}$
Smolt-rearing cages:	7.85 kg t^{-1} fish produced

The P loadings i.e. the products of these coefficients and the areas of land, numbers of people etc, to which they apply, define the main components of the eutrophication pressures on, and thus threats to, the loch and the potential effects on the P content of the water. An idea of actual in-loch concentrations, is gained by dividing this loading by the annual amount of water flowing into the loch, to give a mean influent concentration, $\mathbf{[TP]_I}$ in the OECD [1982] eutrophication models.

If the water passed through the system with no losses of particulate material, for example, to the sediments, $\mathbf{[TP]_I}$ would also be the mean in-lake concentration - $\mathbf{[TP]_\lambda}$ in the OECD, and the Dillon and Rigler models. However, a proportion of the P is lost to the sediments. This proportion relates closely to the P retention coefficient ($\mathbf{R_P}$) i.e. the difference between the annual input of P, and the amount passing out of a lake (in the same year), expressed as a fraction of the loading. The empirical model developed by [Kirchner and Dillon, 1975] predicts ($\mathbf{R_P}$) from a term $\mathbf{q_s}$, 'the areal water loading', which is a measure of the rate at which water is passing through a lake (i.e. m y^{-1}), and obtained by dividing the volume of water entering the system (V_{in}, in $m^3\ y^{-1}$) by the surface area of the lake (A_l, in m^2). Then:.

$$R_P = 0.426e^{(-0.271qs)} + 0.574^{(-0.00949qs)}$$

A value for $[TP]_\lambda$ can be predicted by relating the P loading, **L** (expressed as a specific areal burden - in mg m^{-2} lake surface), and R_P, **z** and ρ as already defined, according to the equation of Dillon and Rigler [1974]:

$$[TP]_\lambda = L\ (1-R_P)/(z \times \rho)$$

3.2 Field investigations

3.2.1 Sampling sites

The original programme which envisaged sampling sediments and water near fish cages and at control sites over the length of the loch was abandoned because of bad weather. A revised programme was adopted which sampled 15 sites within, and at various distances from, 3 of the 4 groups of smolt-rearing cages at Dalilea (towards the southern end of the loch), including a 'control' station midway across the loch from these cages (**Figure 1**). The Dalilea location was chosen primarily on the basis of ease of access, shallowness, and to some extent, shelter. As a consequence of these attributes, however, build up of waste material on the sediment beneath these particular cages is likely to be considerably greater (per unit fish production) than under cages set in deeper water, where the longer column will allow greater dispersal of faeces and uneaten food. The situation is all the more compounded by the fact that the Dalilea cages have been in Loch Shiel for longer than any of the other units; in this connection, it should be noted that the cages are anchored to allow limited movement - not to swing freely throughout 360°. While these cages can thus be expected to represent 'worst cases' in regard to the P, organic matter and invertebrate content of the sediments in their vicinity, the results do not have any effect on conclusions about the P loading to the loch from fish-farming.

The locations of the 15 sampling sites, and their relation to the fish cages and to each other, were determined by measuring their distance and direction from a known point on the shore. To achieve a distance measure, a 4-m pole with two cross marks 2.5m apart, was erected on the shore edge and its position noted. From the boat, at each sampling site, a 35mm colour slide photograph of the pole was taken. The space between the marks on the slide was related, after calibration, to the distance from the boat to the pole, which was calculated for each site from the field photographs. To determine direction, a second pole was erected 5.6m inshore from the first pole. A line of sight through both poles was identified on the far shore. As each field photograph of the distance target i.e. the first pole, included the second pole, the apparent gap between them could be measured; this was done by examining the transparencies under a microscope. Using appropriate trigonometric functions, an angle between the line of sight referred to above, and a line from the boat to the first pole, was deduced. Thus, the distance and direction of each sampling site from a known point, and the location, size and orientation of the cages could be plotted with reasonably accuracy (**Figure 1**).

3.2.2 Probe measurements

A Corning Check Mate® probe system was used to measure the temperature, dissolved oxygen content, pH and conductivity of surface and near-sediment water (the latter above intact cores collected with a Jenkin Surface Mud Sampler).

3.2.3 Chemical and biological sampling

a. surface water: surface water samples were taken in duplicate using a 2-m length of 6-cm diameter, Marley® plastic drainpipe, to obtain an integrated sample of water over this depth. After lowering the pipe vertically into the water until its end was just under the surface, a tight-fitting rubber bung was pushed into the top end; the whole tube was then raised until the lower end was just below the water surface and could be closed by hand. The water was then transferred to a plastic bucket, before subsampling into polyethylene bottles which had been acid-washed and repeatedly rinsed in hot tapwater and, just prior to taking the samples, rinsed in the loch water. The samples were kept cool and dark until further processing (filtering, fixation etc.) which was done within 15 minutes of collection, at the loch side. Unfiltered aliquots of water were stored for the analysis of total phosphorus (TP), while subsamples filtered under partial vacuum through Whatman GF/C discs were saved for the determinations of total soluble P (TSP), soluble reactive P (SRP), nitrate ($NO_3.N$), dissolved silica (SiO_2) - although the silica and nitrate analyses are not reported here. For the chlorophyll *a* determinations, Whatman filters with the trapped particulate material were stored immediately in ice-boxes, and transferred to a freezer on the day of sampling and kept frozen and in the dark until further analysis.

b. sediment, and near-sediment water: The sediment material used for chemical analysis was collected with a Jenkin Surface Mud Sampler. Some of the water overlying each of the 2 Jenkin cores from each site, was siphoned off and stored for later chemical analysis - as for the surface water, except that no samples for total P and chlorophyll were taken. The top 5cm of mud from each core was extruded and put into a polythene bag; a representative subsample was transferred to a pre-weighed 50-ml polypropylene centrifuge tube with a screw-cap. The bags and tubes were sealed in an attempt to minimise oxygenation of the material and chemical changes in e.g. the phosphate adsorption status. It is recognised that such changes could still take place especially in smelly samples, even at the moderate temperatures prevailing at the time; it was impossible, however, during this campaign to use 'glove-box' techniques and process the muds under an atmosphere of e.g. nitrogen.

For the invertebrate work, 30 sediment samples (2 per site) were collected with an Ekman grab. The samples were preserved with formalin in the field before being transferred to the Edinburgh laboratory for further analysis.

3.3 Laboratory analyses

3.3.1 Phosphorus and organic matter determinations

Phosphorus fractions were determined using absorption spectroscopy. SRP was determined using the ammonium molybdate-ascorbic acid method of Murphy and Riley [1962] on the GF/C-filtered water. TP and TSP were analysed as above, but on unfiltered water, and following sulphuric acid-potassium persulphate digestion to convert all the P to the soluble reactive form.

Phosphorus in each of the sediment cores was measured by diluting a small, known weight (~ 0.2g) of the mud with 50ml distilled water. This was then mixed using a magnetic stirrer, and with the mixing in progress, a measured aliquot was removed and treated as for TP above. The rest of the sediment was transferred to aluminium foil trays for oven-drying at 60°C over several days. After grinding the dried sediment using a pestle and mortar, a small, known weight of the material was transferred to a 500-ml conical flask for the determination of readily oxidisable organic carbon using the modified Walkley-Black method [Gaudette *et al*, 1974]; this involved digestion of the sample by a mixture of potassium dichromate and sulphuric acid, followed by back-titration of the unused dichromate with ferrous ammonium sulphate. Carbon figures were converted to organic matter by multiplying them by 1.72.

3.3.2 Phytoplankton analysis

Apart from a brief microscopic examination of a pair of 18-μm mesh net tow samples collected from near the cages and near the control site, the phytoplankton studies were limited to the determination of chlorophyll *a*. The GF/C discs referred to above, were steeped in 90% methanol and stored in the dark, overnight at *ca* 4°C. The resulting extracts were centrifuge-cleared and their optical densities measured using a Philips PU8670 spectrophotometer fitted with 4-cm path-length cuvettes. The equation derived by Talling and Driver [1963] was used to convert optical density readings at 665 nm (minus those read at 750nm for turbidity correction) to concentrations of chlorophyll *a*.

3.3.3 Invertebrate benthos

The 30 samples were washed to separate the benthos from the sediment. The benthos was sorted into taxonomic groups prior to identification. The oligochaete worms were mounted in polyvinyl lactophenol before being identified using Brinkhurst's [1971] key. Immature oligochaetes could not be identified to species level as the above key relies on characteristics found only in mature specimens.

Chironomid larvae were initially placed in a concentrated solution of potassium hydroxide (to clear the internal tissues allowing a better view of the surface structures important in identification), followed by the dissection of the head capsule from the body, prior to mounting in polyvinyl lactophenol. Identification of the Chironomidae was based on Wiederholm [1983] and was taken to genera level. Unfortunately, the shells of a number of the *Pisidium* specimens, had become decalcified, preventing identification. The Glossiphonidae, Leptoceridae, Sericostomatidae and Polycentropidae were identified using the relevant FBA keys [Elliot and Mann, 1979; Wallace *et al.*, 1990; Edington and Hildrew, 1981].

4. RESULTS AND DISCUSSION

4.1 Desk analyses

4.1.1 An estimate of the inputs of total phosphorus (P) from the catchment

The areas of the different types of land cover in the catchment and the estimated numbers of people, are indicated with the corresponding predicted P losses in

Table 3: Components of the catchment-derived loading of total P to Loch Shiel.

Source	Area covered (ha)	Predicted annual P loading (t)
Rough grazing/moorland	17221	1.20
Coniferous forest	3992	3.99
Improved grassland	918	0.37
Deciduous woodland	600	0.06
Arable agriculture	58	0.015
Rural community (*ca* 630 people)	-	0.63
	Estimated total	**6.27**

Table 3. This suggests that forestry (equivalent to *ca* 64% of the total catchment-derived loading), and rough grazing land/moorland (19%) could account for the vast majority of the P coming into the loch from the catchment. The forestry contribution reflects the combination of a relatively small area, and a fairly high P export coefficient, while the rough grazing input results from the large area of land losing P at a very low rate per hectare. By comparison, the contributions from improved grassland, deciduous woodland, arable agriculture and people, are minor, together amounting to <20% of the total. More than half of this is attributed to the rural community, but it is worth pointing out that most of the people live near the western end of the loch close to the outflow. This could minimise the effects of any sewage-

derived P that is likely to pass into the loch, by it being flushed out of the system relatively rapidly.

If the total, catchment-derived loading of P (6.27t y^{-1}) is divided by the annual inflowing water volume i.e. 50.7 x 10^7 m^3 (**V_{in}** as defined above) excluding rain falling on the loch surface, and 54.9 x 10^7 m^3 with direct rain, values of 12.4μg l^{-1} and 11.4μg l^{-1} for **$[TP]_{in}$** are obtained. The likely extra input of P in rain falling on the loch was not taken into account by Bailey-Watts *et al.* [1992a,b]. However, if the average concentration in rain was 5μg l^{-1}, this would amount to 0.25t y^{-1}. Added to the 6.27t calculated above, this gives a new annual total of 6.52t, which results in a mean influent P concentration (the load divided by 54.9 x 10^7 m^3) of 11.9μg l^{-1} *cf* the 11.4μg l^{-1} calcuated above. These values are not so different from those of *ca* 9μg l^{-1} measured by Bailey-Watts *et al.* [1992a] in 4 of the inflows in summer 1991, but they exceed the values of 5-6μg l^{-1} in another 4 streams sampled at that time, but are considerably less than the figures reported for two streams near Acharacle in which total P levels of 135μg l^{-1} and 94μg $^{-1}$ (each with a large proportion of SRP) were measured in 1991.

4.1.2 An estimate of the P loading attributable to smolt-rearing in the loch

Strictly confidential data of Marine Harvest on the P content of the freshwater diets presently used, feed inputs and fish production for 1991 and 1992, suggest that a total of 0.87 t P is currently added to the loch from this industry each year. This takes account of uneaten feed, and the probable losses of soluble and particulate components of P *via* excretion. The figure is equivalent to 14% of the P load estimated to be derived from the land. The smolt-rearing activities introduce considerably less P to the loch than that reckoned to be derived from either coniferous forest or the rough ground. This is even allowing for errors of (i) *ca* 30% in the choice of export coefficients and estimates the size of the rural community, and (ii) in the smaller estimated percentages of various land areas. Apart from the fact that the P loss coefficients assigned here to coniferous forest land need checking, there are reasons related to the standing waters in the Polloch sub-catchment, to suggest that not all of the nutrient leaving the forests actually reaches the loch. Meanwhile, smolt-rearing appears to add more P than land-based agriculture, deciduous woodland, the rural community, and rain falling on the loch, put together - particularly if, as suspected, the numbers of people in the catchment have been over-estimated rather than under-estimated.

If the annual loading attributed to smolt-rearing (0.87 t P) was distributed throughout the water column of Loch Shiel i.e. with no losses to the sediments, it would *raise* the P concentration by just 1.1 $\mu g\ l^{-1}$ if the loch volume is taken, or 1.6 $\mu g\ l^{-1}$ if the annually-renewed volume is taken. Strictly speaking the smolt-rearing contribution does not affect the influent P level i.e. $[\mathbf{TP}]_{\mathbf{in}}$ as defined above, but it would be classed as doing so when the data are considered in relation to the eutrophication models incorporating this term. If the total loading is divided by $\mathbf{V_{in}}$ as defined above, $[\mathbf{TP}]_{\mathbf{in}}$ becomes 13.5$\mu g\ l^{-1}$ *cf* 11.9$\mu g\ l^{-1}$ quoted above.

4.1.3 A desk-derived total P loading to Loch Shiel

The estimated total loading is 7.39t P y^{-1} to Loch Shiel i.e. 6.52t from the land and in direct rain, plus 0.87t from the fish cages (12% of the overall total). Expressed as a specific areal load i.e. per m^2 of loch surface (1960 ha), this is 0.377g y^{-1}. Interestingly, this value comes approxiamately half-way between what Vollenweider [1968] considered the 'permissible' and 'dangerous' values of 0.25g m^{-2} and 0.50g m^{-2}, for a lake of the mean depth of Shiel (41m).

4.1.4 Predicted total P concentration in the loch

Insertion of the values for predicted loading (0.377g $m^{-2}\ y^{-1}$), P retention coefficient (0.45), flushing rate (0.69 loch volumes y^{-1}) and loch mean depth (41m), into the Dillon and Rigler model, results in an annual mean, loch-wide P concentration of 7.33$\mu g\ l^{-1}$.

4.2 Field results from the Dalilea fish cage area - September 1992

4.2.1 General physico-chemical conditions

The temperature and conductivity differed little between the surface and near-sediment waters i.e. 11.3°C to 11.5°C, and 40μS cm^{-1} to 43μS cm^{-1}, excluding a suspicious outlier of 56μS cm^{-1} at the surface at site 6. Dissolved oxygen levels were generally high i.e. 80-89% saturation, except in the near-sediment waters at site 13 (52%) and the cage sites 5 and 10 (both 20%). As often found in our experience, pH was the most variable of these factors i.e. 5.6 to 6.7 units, with no consistent relationship with site location or the other measurements.

4.2.2 Phosphorus levels in the water column

Surface water P levels were very moderate, but varied significantly (over 2-fold) from 3.6μg TP l^{-1} to 7.5μg TP l^{-1}, but without any particular association with site (**Figure 2**). Thus, the control station was as rich as the cage sites 10 and 15, although the maximum value of 7.5μg l^{-1} relates to cage site 5. These concentrations can only be viewed as a broad indication of the P status of the loch because they refer to a single occasion. It is unlikely, however, that a waterbody as 'physically-buffered' as Loch Shiel will exhibit more than a 4-fold variation in P over the year. It is encouraging that they tally reasonably well with the predicted annual mean, lake-wide level of 7.33μg l^{-1}.

The corresponding SRP values are very low - and at 0.3 μg l^{-1} to 1.6μg l^{-1}, near the limits of detection and of statistically significant difference. These values, however represent, at the time of sampling, what remains of this bio-available form of P, after plant uptake. The situation regarding SRP (**Figure 3**) in the near-bottom waters, however, is quite different. Firstly, the values are, with one exception (1μg l^{-1} at site 1) all significantly higher than the surface set, with a range of 4μg l^{-1} to 203μg l^{-1}. Secondly, the control site gave a value of 50μg l^{-1}, which ranks approximately in the middle of the range, and exceeds the figure of 29μg l^{-1} recorded under one of the cages; however, values of 80μg l^{-1} and 203μg l^{-1} were recorded under the other 2 cages. Thirdly, the results suggest considerable patchiness; for example, the control site with 50μg l^{-1} is relatively close to site 6 with 4μg l^{-1}; and site 2 which is in-shore and dominated by Quillwort (*Isoetes lacustris*), gave a value of only 4.7μg l^{-1}, while a pair of richer sites nearby i.e. 3 and 14 at which plants were not noticeable, gave values of 46μg l^{-1} and 70μg l^{-1} respectively.

These results suggest that some of the P in this sediment is potentially recyclable, but the immediate effect is on concentrations in the overlying water column; the effect on the loch as a whole is minor - probably amounting to a small fraction of a microgramme per litre over the year.

The present study cannot contribute much to seasonal aspects, but it is worth noting that feed input rates vary over the year, according to e.g. water temperature and size of fish. With minor variations between years, the percentage of the total annual food given, varies from 6 or 7% per month from April to June in year 1, and from November in year 1 to March in year 2, to *ca* 13% in July and August, and 15% in September and October. Fairly high rates of feed input thus occur in summer when conditions such as the water temperature, and the degree of stratification, are most favourable for algal growth. The higher feeding rates were being used during the period of this study's field visit.

4.2.3 Phosphorus and organic matter content of the surface sediments

The P content in the uppermost 5 cm of sediment beneath and near the cages, ranged from 0.1 to 2.0% on a dry weight basis (**Figure 4**). For comparison, sediment P levels measured in 29 loch basins by Bailey-Watts *et al.* [1992a] in summer 1991, ranged from 0.06% to 0.48%. In September 1992, the open water 'control' site in Loch Shiel (south-west basin), gave a value of 0.34% which is significantly higher than the figure of 0.12% from somewhat deeper water in this basin in July 1991, and the value of 0.17% obtained from the considerably deeper north-east basin at that time [Bailey-Watts *et al.*, 1992a]. Values >0.5% are probably due to the fish cages, but as already asserted, they are to be expected at such a shallow, sheltered and long-used site.

Some of the concentrations exceed the nominal figure of *ca* 1.2% for the mean P content of fish feed *presently* used. This could be due to a number of factors such as the likely high content of P-rich bone tissue in faeces, and the fact that the standard 5-cm cores probably contain material from before the introduction of low-P diets. In this connection, it should be stressed that the results for each core are not comparable. Rates of sediment production would be higher near cages, even if water depth, currents, particle settling velocities, sediment physics (e.g. compaction, particle size) and chemistry, were uniform over the sampling area. A 5-cm core from beneath a cage (rapid build-up) thus represents a shorter period of sedimentation, than that represented by a core of the same length taken from an open water station (lower sedimentation rate).

Organic matter content varied from 7% to 38% with no consistent relationship with proximity to a cage (**Figure 5**). Thus, while cage sites 10 and 15 gave relatively high values (29% and 27% respectively), cage site 5 produced one of the lowest figures (11%), and the highest value of all (38% quoted above) corresponds to site 6 which is nearest the open water control station. Also, while the organic-rich deposits at cage sites 10 and 15 tally with the dissolved oxygen levels of 20%, the richest site 6 gave an oxygen figure of 84%.

4.2.4 Phytoplankton

From the above data on P, and knowledge about the general relationship between P levels and phytoplankton, very low algal concentrations would be expected. That this expectation is realised, is evident from the pigment values obtained in September 1992. Apart from the samples collected at cage site 5 which happened to produce the highest mean figure of 2.4μg chlorophyll *a* l^{-1}, the overall range is 1.9μg l^{-1} to 2.2μg l^{-1}. At these levels, the differences between sites cannot be considered statistically significant.

The component species and population densities have not been quantitatively assessed, but 'oligotrophic' species of the following were noted: desmids (*Staurastrum*, *Staurodesmus*), diatoms (R*hizosolenia*, *Tabellaria*), colonial chrysophytes (*Dinobryon*, *Mallomonas*). These are all large algae, the small forms (nanoplankton and picoplankton) having not been examined on this occasion. In addition, the cyanobacterium *Anabaena*, was recorded, and indeed colonies were visible to the unaided eye. This underlines the fact that potentially toxic, bloom-forming blue-green algae are not the preserve of rich shallow waters [see also Bailey-Watts *et al.*, 1992a], although they are unlikely to achieve - on an lake-wide basis - the densities characteristic of eutrophic waters.

4.2.5 The invertebrate benthos

The benthic invertebrates are listed in **Table 4** where the sample sites are separated into five groups according to their position relative to the cages. **Figure 6** relates the number of invertebrate taxa to different sample sites, and shows that the higher numbers of taxa were present at the inshore sites (i.e. 2, 3 and 14). However, rather than indicating better water quality, these results probably reflect the more diverse habitat of this sub-littoral zone which had clumps of the submerged water plant *Isoetes lacustris* and littoral fauna such as the caddis species *Mystacides azurea, Cyrnus trimaculatus* and *Sericostoma personatum.* In contrast to these edge sites, the other 12 sample points had invertebrate communities of a more simplified structure, dominated by Chironomidae, Oligochaeta and the bivalve mollusc *Pisidium* - all characteristic of deeper, profundal areas of large Scottish lochs [Smith, Cuttle and Maitland, 1981].

Despite the paucity of invertebrate taxa away from the loch shore, the benthos below the fish cages and in the adjoining sediments had generally a much higher invertebrate population relative to the inshore, the offshore and the control stations **(Figure 7)**. This suggests a strong link between the fish cages and the greatly increased density of benthos recorded there.

A more detailed examination of the distribution and composition of the Oligochaeta and Chironomidae populations, which constituted the bulk of the benthos, is interesting. Mature specimens of two oligochaete species were found in the survey, *Lumbriculus variegatus* (Family Lumbriculidae) and *Limnodrilus hoffmeistrei* (Family Tubificidae). In addition, indeterminate, immature specimens of two groups of Tubificidae (*Potamothrix/Tubifex sp* and *Limnodrilus/Potamothrix sp*) were found. **Figure 8** indicates the large numbers of oligochaete worms present in and around the fish cages. These were comprised entirely of Tubificidae. Contrastingly the other sediments supported much lower numbers of oligochaetes, consisting predominantly of Lumbriculidae plus a few Tubificidae. **Figure 9** shows that the distribution of the Chironomidae is similar to that described above for the Oligochaeta i.e. there is a large increase in numbers of individuals beneath the cages and in the nearby sediments. This is accompanied by a change in population structure from one comprising Tanypodinae with additional members of the sub-families Prodiamesinae, Diamesinae and Chironominae, to one dominated by Chironominae, particularly the genus *Chironomus*. In the limited sampling of the profundal zoobenthos carried out by Smith, Cuttle and Maitland [1981], three species of oligochaetes were found in Loch Shiel, namely *Limnodrilus hoffmeistrei*, *Tubifex tubifex* and *Peloscolex ferox* (all Tubificidae) and two chironomid genera *Metriocnemus* (Orthocladinae) and *Polypedium* (Chironominae).

Tubificidae and Chironominae are regarded as being tolerant of organic enrichment from point sources such as fish cages, because these taxa have the ability to withstand lower oxygen levels such as would be associated with the sediment environment beneath the cages. A recent internal Marine Harvest report on the sedimentary environment around the Loch Shiel fish cages, including Dalilea, found depressed redox potentials in the sediments in the vicinity of the cages compared to other parts of the loch, thereby inferring reduced oxygen levels due to organic enrichment (see sections 4.2.1 & 4.2.3). Tanypodinae and Lumbriculidae are thought to be less tolerant of organic enrichment although Wilson and McGill [1982] indicate that *Macropelopia* and *Procladius* are relatively tolerant, as is *Monodiamesa*; contrastingly, *Protanypus* is considered to be relatively intolerant.

A review of studies on the environmental effects of freshwater farming on invertebrate benthos [Institute of Aquaculture *et al.*, 1990], also found increased populations of 'organic pollution-tolerant' invertebrates such as Oligochaeta and Chironomidae beneath fish cages, though the specific taxa are not indicated in this particular review. The benthos below the cage at site 15 does not follow the pattern described above. Here, although *Pisidium* was recorded, no chironomids were found and oligochaete numbers were much reduced compared to the other cage sites (i.e. 5 and 10). It may be significant that this cage has been in its present position longer than the other two cages. The grab samples were particularly smelly and contained large amounts of solid organic matter.

The bivalve mollusc *Pisidium* was recorded at a number of sampling stations with no explicable pattern. The majority of the identified specimens were *Pisidium* *hibernicum*, a species with no clear habitat preferences (J. Bass *pers comm.*). One specimen of *P. personatum* was identified, the sole species recorded in Loch Shiel during a survey of the profundal zoobenthos in 1980 [Smith, Cuttle and Maitland, 1981].

4.3 The trophic status of Loch Shiel - historical context

Surface and deep water TP levels recorded by the Institute of Aquaculture (IoA) in their twice yearly sampling programme from 1986-1992 (March/April and November each year), suggest that the loch has remained oligotrophic over that period. Although Bailey-Watts *et al.* [1992a] measured 12.3μg l^{-1} in the south-west basin on 25 July 1991, a signicantly lower concentration of 7.4μg l^{-1} was obtained for the north-east basin. Almost without exception since 1986, and then only when values from all 6 sites sampled were fairly similar, the IoA records also suggest the shallower of the 2 basins (i.e. the south-west basin) is the richer in P.

The values for annual maxima (normally November) in **Figure 10** increased from 1986 to 1990, but decrease thereafter with the exception of the high value of 12.3μg l^{-1} in July 1991 (see above). **Figure 10** plots surface values only, as variation with depth is minor. However, the average levels over the loch as a whole appear to have remained below 10μg l^{-1} - as were the surface water values reported for the Dalilea fish cage area in September 1992, and all values given for 1978, albeit for a (single) sampling point well towards the north-eastern end of the loch [Bailey-Watts and Duncan, 1981a]. Nitrate concentrations corresponding to the TP data rarely exceed 70μg N l^{-1}, although in April 1991 all 6 stations sampled (4 along the deep basin, and 2 in the shallow basin) gave higher values i.e. 110μg N l^{-1} to 150μg N l^{-1}.

Primarily as a consequence of SRP levels rarely exceeding 1μg l^{-1}, nitrate-N to phosphate-P weight ratios are nearly always higher than 50:1, and often greater than 100:1. This tends to support the view that P is the major limiting nutrient, but also that even without further N enrichment, addition of more P would lead to more algae. As the nitrate levels in Loch Shiel are low (compared to the milligrammes per litre recorded in many lowland catchments), P levels would not need to increase by very much (on their own), to reduce the N to P ratio in favour of N-fixing blue-green algae.

The IoA database on chlorophyll (**Figure 11**) is much more extensive than, and gives different information from, the TP records, but confirms the oligotrophic nature of this waterbody. Values corresponding to the TP points in **Figure 10**, do not show such clear trends. When all of the data in **Figure 11** are considered, at least three very intersting and related features are evident, and these have a considerable bearing on thoughts about monitoring the condition of large waterbodies (see Section 5.3). The first point concerns the basic nature of the data. In spite of the majority of the values being $<1.5 \mu g\ l^{-1}$ which is itself low, weekly increases or decreases of usually only a fraction of a microgramme per litre, are often maintained for many weeks, even months. Second, the phytoplankton can vary significantly between years (see especially 1987 and 1989 data) in terms of (i) overall abundance, (ii) the timing of maxima and minima, and (iii) the sizes of the maxima. From its large size, Loch Shiel might have been expected to exhibit less variability. Whether the inter-annual differences are accompanied by shifts in species composition is not known. The third feature identified by these data, is the lack of a consistent trend in annual maxima. For example, even if the highest chlorophyll concentrations shown for 1988 (north-east basin, July-August) were not the actual maxima for that year - there being no data for the previous 3 or 4 months - they exceed the maxima recorded in 1989 during which samples were collected every week or so. Moreover, none of the figures plotted for the deep basin in **Figure 11** exceed to any significant extent the $1.5 \mu g\ l^{-1}$ recorded as the annual maximum for that basin in 1978 [Bailey-Watts and Duncan, 1981b] - although the latter stems from monthly sampling only. In summary therefore, there is no evidence of 'eutrophication' over some 15 years, to a point where either open water P or chlorophyll levels exceed those characteristic of oligotrophic waters.

5. CONCLUSIONS - THREATS TO THE OLIGOTROPHIC STATUS OF LOCH SHIEL

5.1 The current situation

The present results suggest that the main threats to the oligotrophic status of Loch Shiel stem currently from forestry and rough upland ground. The former is as a result of a relatively small area losing P at a high rate per hectare, while the latter results from the combination of a low P loss rate and an enormous area. In that any of Man's activities enhancing the rates of input of this nutrient to the loch, comprises a threat, fish farming, septic tank discharges, arable farming and improved grass management must all be implicated. The present estimates, however, suggest that even fish-farming which probably introduces as much P to the loch as all of the other minor sources together, contributes less than one-eighth to the total input. From the figures given in section 4.1.2, it can be calculated that an increase of say, 25% in smolt production, would add the equivalent of probably <0.5 μg TP l^{-1} to the loch and this does not take into account of a likely loss of nearly half of this to the sediments. Interestingly, the chemical and biological impacts attributable to the fish cages are undoubtedly more easily identified than those due to even the massive diffuse inputs (e.g. forestry). Reasons have been given already, however, for suspecting that organic matter accumulation, increased abundance and decreased diversity of benthic invertebrates, observed in the Dalilea cage area, represent a worse case.

It is unlikely that even a much-needed scrutiny of forestry management practices in the Loch Shiel catchment, or a thorough P loading study (see below), would alter the main conclusion regarding the sources of P to the loch i.e. that forestry-related inputs outweigh the fish cage loadings. Nevertheless, there are a number of other considerations suggesting that the ratio of forest P losses to fish cage inputs may not be as high as suggested so far i.e. 4.6:1 (from **Table 3**), and that in any event, the impact of the forestry P may not be as great as that of the cages *per unit of P supplied.* The arguments, all of which need validation by scientific measurement and experimentation, are as follows. First, 'fish cage P' is probably considerably richer in potentially bioavailable P, than 'forest P'. Second, while the rate at which P emanates from the cages varies with the seasonal schedule of feeding (see above), the material is coming into the loch more or less continuously; by contrast, runoff from forests are largely determined by rainfall patterns, in addition to the fact that the main inputs occur (at most?) every 7 years. The third point relates to the fact that much of the P assumed to originate from forestry land, comes from the subcatchment drained by the River Polloch [see Maitland, 1981]. This enters the loch near the confluence of the two basins. Little is known about water circulation patterns, but while the general flow of Polloch water is likely to be towards the outflow (Shiel Bridge), the sill between the two basins may impede this movement, and cause back-up and transport north-east into the deeper basin. In addition, as shown by studies on Loch Lomond [e.g. Smith, Lyle and Rosie, 1981] quite enormous volumes of water can be shifted by the wind. Certainly, the impacts of P loadings on P and algal levels are likely to be less if the deep basin is the recipient. Fourthly, the Polloch catchment contains a large number of standing waters i.e. 34 [Maitland, 1981]. These may be significant in 'trapping' a certain amount of material, including P, entering them (in the same way as Loch Shiel itself retains some of its P load). Prominent among these waterbodies is likely to be Doilet (Doilate) and the marshy area occupying much of the space between Doilet and Shiel.

Data on nutrient levels in the loch itself support the view that P is the major limiting nutrient, and further inputs of P would lead to increases in biological productivity. For the loch to be no longer classified as oligotrophic, total P levels would not have to increase by very much. Bio-available P (SRP) concentrations are very low, and in spite of generally low nitrate values, would need to be raised by nearly 10-fold, for nitrate-N to become limiting, and N-fixing blue-green algae to be favoured.

5.2 Recommendations for preserving the oligotrophic status of Loch Shiel

Regardless of the execution and outcome of work proposed below, every effort should be made to prevent the eutrophication of Loch Shiel proceeding at anything faster than it is assumed to be doing at present. The statutory authorities have also to decide whether it is necessary to stem threats effecting the 'natural' evolution and enrichment over geological time spans. The technology and knowledge regarding good catchment management in this connection, exist. Certainly, ways of eradicating as far as possible, the inputs of nutrients from houses and hotels should be given priority attention. As indicated above, our predictions of the P contributed by forestry, may well be an overestimate (see also below), but the fact remains that some of the values measured in the deep basin approach 10μg l^{-1} and a number in the shallow basin exceed this level. Indeed, P levels in Loch Shiel probably reflect earlier loadings rather than those of the current year. If this is so, and if loadings have generally increased rather than decreased in the last decade, for example, higher P concentrations may be recorded in the future, for a while at least, even if P loadings are reduced immediately.

5.3 Future work and monitoring

Statements about the P loading rest entirely on desk-generated figures, and as such, must be validated. This is even bearing in mind that in-loch concentrations predicted from them, relate reasonably closely to measured values. Some of the uncertainties could be cleared up by discussions with forest managers. Otherwise, field measurements are necessary. An intensive programme of chemical sampling coupled with hydrological monitoring of a number of major feeder waters would be ideal. In any event, consideration should be given to assessing the transport of total P and the main soluble and particulate fractions, from the Pollach sub-catchment in which much of the coniferous forest is situated. As well as enhancing the scientific base on which to manage Loch Shiel, such a programme would be of general limnological value.

The data reported here illustrate the value of monitoring chemical and biological features of trophic status. However, for such a large and special waterbody such as Loch Shiel, the basic rationale of monitoring, as well as the details of any recording programmes need to be reviewed.
The recommendations of section 5.2 reflect the view that all reasonable action to preserve the oligotrophic condition of the loch should be taken now. By the time P concentrations have exceeded say, 10μg l^{-1} - if this is used as the gauge of trophic

state - a programme of restoring the loch to lower nutrient levels might be necessary, rather than the easier option of preventing the levels rising in the first place.

This is not to suggest that monitoring be abandoned. Without maintaining the recording already in progress, and initiating programmes on factors that have received little or no attention so far, there would be no benchmarks for judging whether conditions have changed. Moreover, as illustrated again by the existing data, considerable knowledge has been gained about the dynamics of the system. There are arguments for doing more than hitherto, on organisms - especially bearing in mind that the biological consequences of elevated P levels, rather than the higher concentrations themselves, are the greater concern. Whether oligotrophic indicator species would literally disappear if, and as soon as, lake-wide P levels exceeded $10\mu g\ l^{-1}$, could form an hypothesis to be tested by much-needed research. In similar vein, it is not by any means entirely clear whether phytoplankton species characteristic of richer waters would start to appear as soon as P levels rise above $10\mu g\ l^{-1}$; even if they did, the physical make-up of the system is likely to mediate against them building up biomass characteristic of richer waters. In this connection, however, the south-west basin is likely to manifest any widespread increases in nutrient content and biological productivity, earlier than the rest of the loch.

Even if a monitoring scheme aims to detect change in trophic status of the loch as a whole, this does not necessarily imply lake-wide sampling. For example, the structure and abundance of invertebrate communities in the profundal zones, and summer oxygen levels in the deeper layers of water, may suffice as measures of whole-lake 'condition'.

With organisms higher up the food chains being the more extensive integrators of environmental information, fish or invertebrates could be chosen in preference to e.g. phytoplankton, even though algae are of concern *vis a vis* issues over toxins and bloom production. There are certain logistic advantages to be gained by concentrating on organisms such as fish and invertebrates, in that they have longer life cycles and exhibit somewhat more predictable, seasonally regimented population changes. The estimation of the lake-wide distributions and abundances of populations of fish and invertebrates demand not inconsiderable resources, however. Contrastingly, planktonic algae and zooplankton have great potential as indicators of nutrient status, but require short-interval sampling to assess the rapid changes in species abundance. Benthic algae - both living and as persistant remains in sediments - also provide information on trophic status, and like their planktonic counterparts, are likely to respond the swiftest to changes in P levels. They require a programme of sampling somewhat similar to that appropriate for zoobenthos in terms of spatial coverage, and plankton in terms of temporal intensity.

Elements of monitoring programmes which could be executed in various combinations, and which vary in intensity and take account of the points above, are listed below for consideration:

(i) a single, summertime limnological survey to include the vertical profiling of temperature, dissolved oxygen, pH, conductivity, nutrients and plankton, in the deepest parts of each basin, and a survey of the profundal zoobenthos.

(ii) 4-monthly surveys of the littoral algal and invertebrate benthos; this would be valuable in highlighting 'hot-spots' of nutrient enrichment.

(iii) assessments of the phytoplankton status as executed by IoA during 1987 and 1989 i.e. at more or less weekly intervals, but backed up by phytoplankton population assessments (not just chlorophyll *a* concentration as then), and nutrient, especially P analyses, and with the addition of two extra sampling sites - at Glenfinnan and Shiel Bridge which could be sampled even when weather prevented open water work.

(iv) quarterly assessments of major ions.

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FIGURES

Figure 1. Location of sample sites at Dalilea, Loch Shiel.

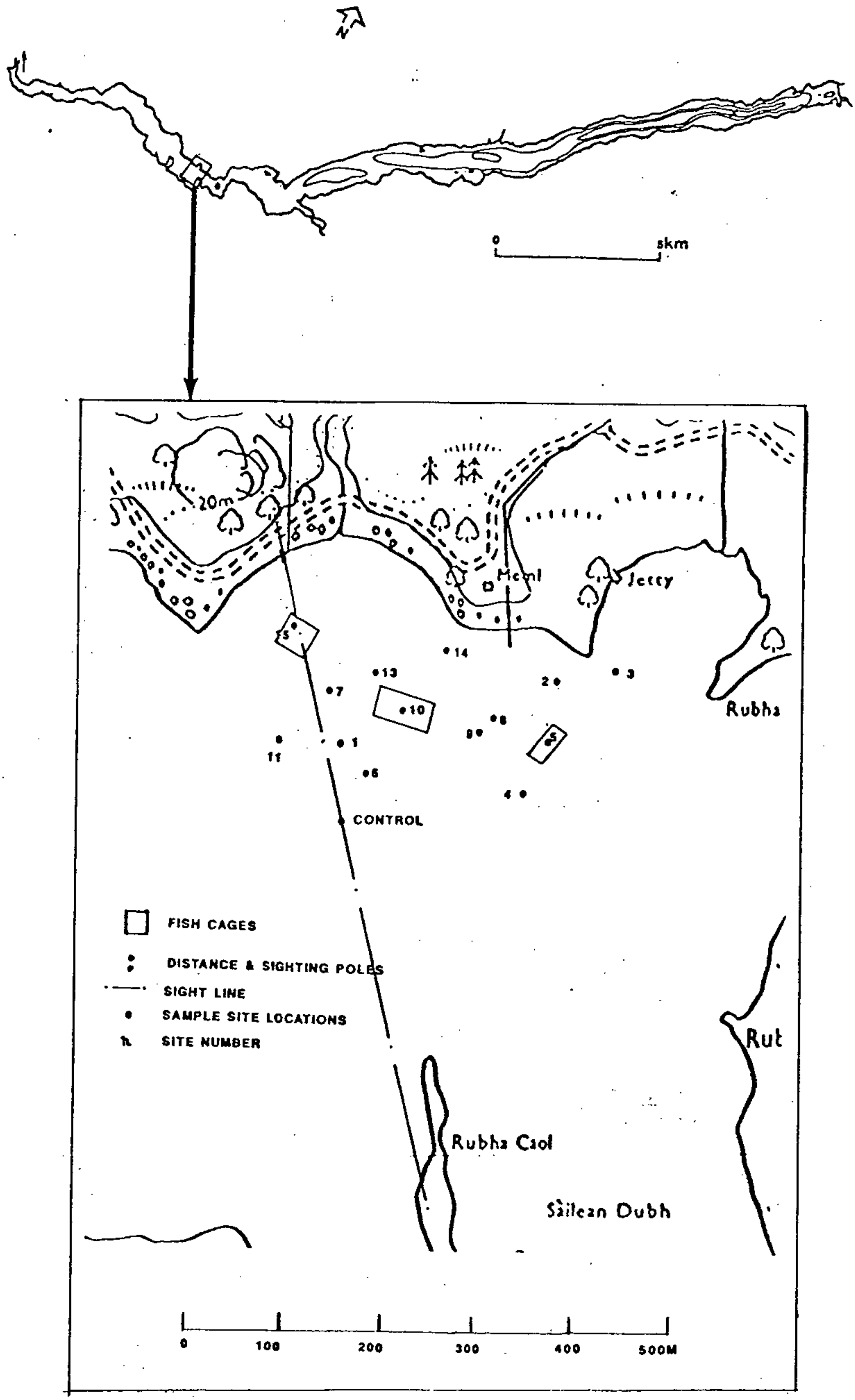


Figure 2. Total Phosphorus in surface water ($\mu g\ l^{-1}$)

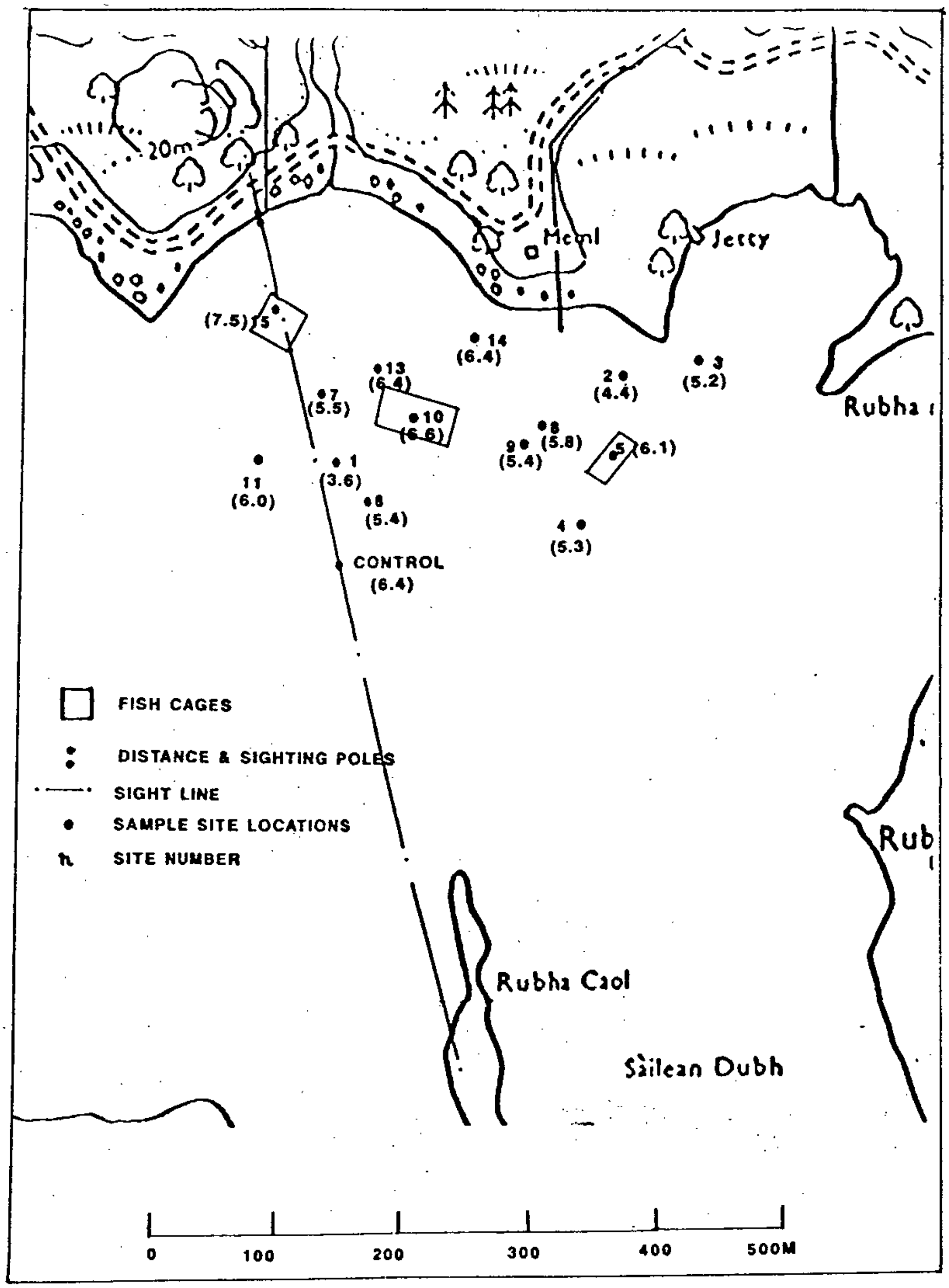


Figure 3. Soluble Reactive Phosphorus in water overlying sediment ($\mu g\ l^{-1}$)

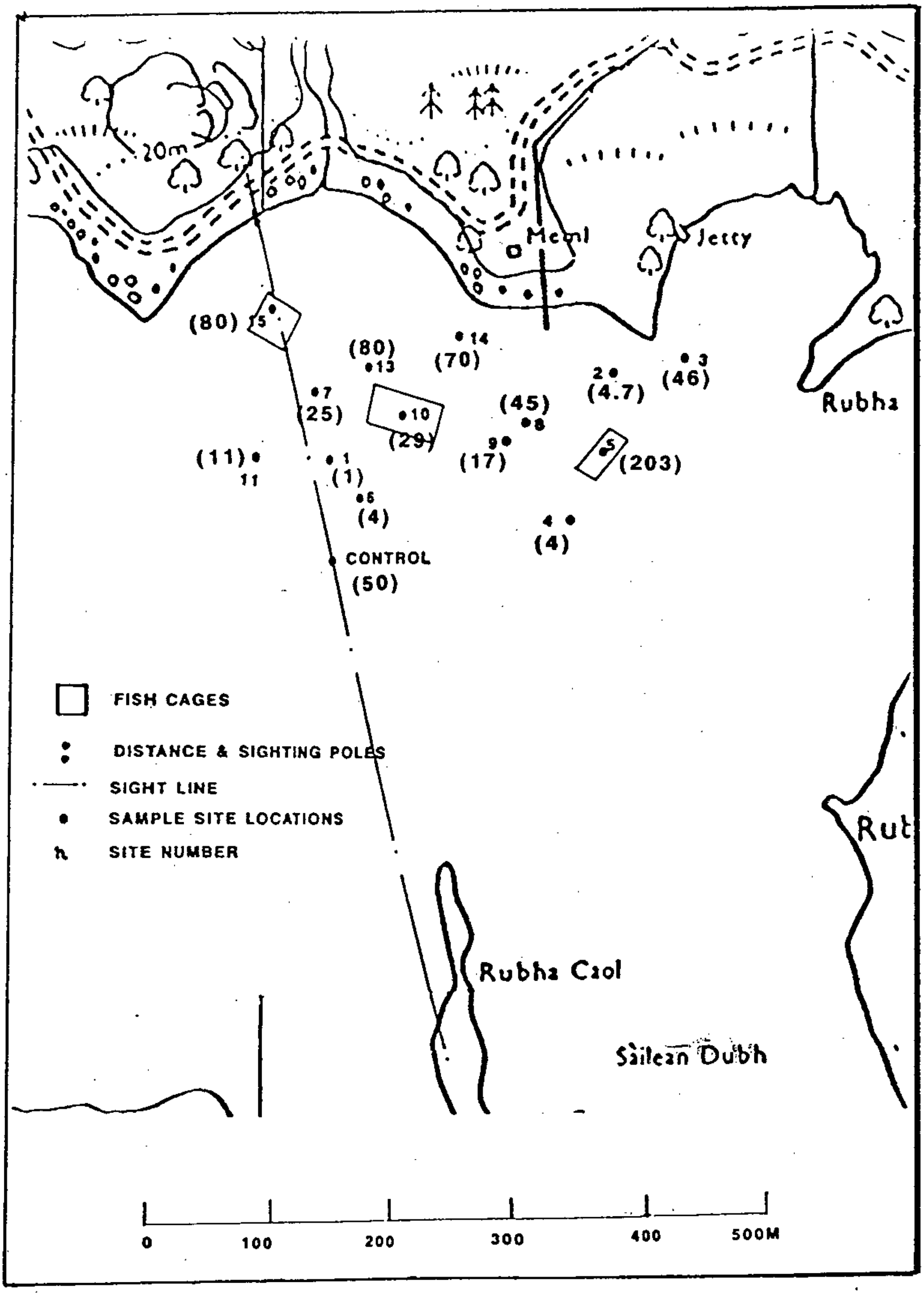


Figure 4. % P of sediment dry weight.

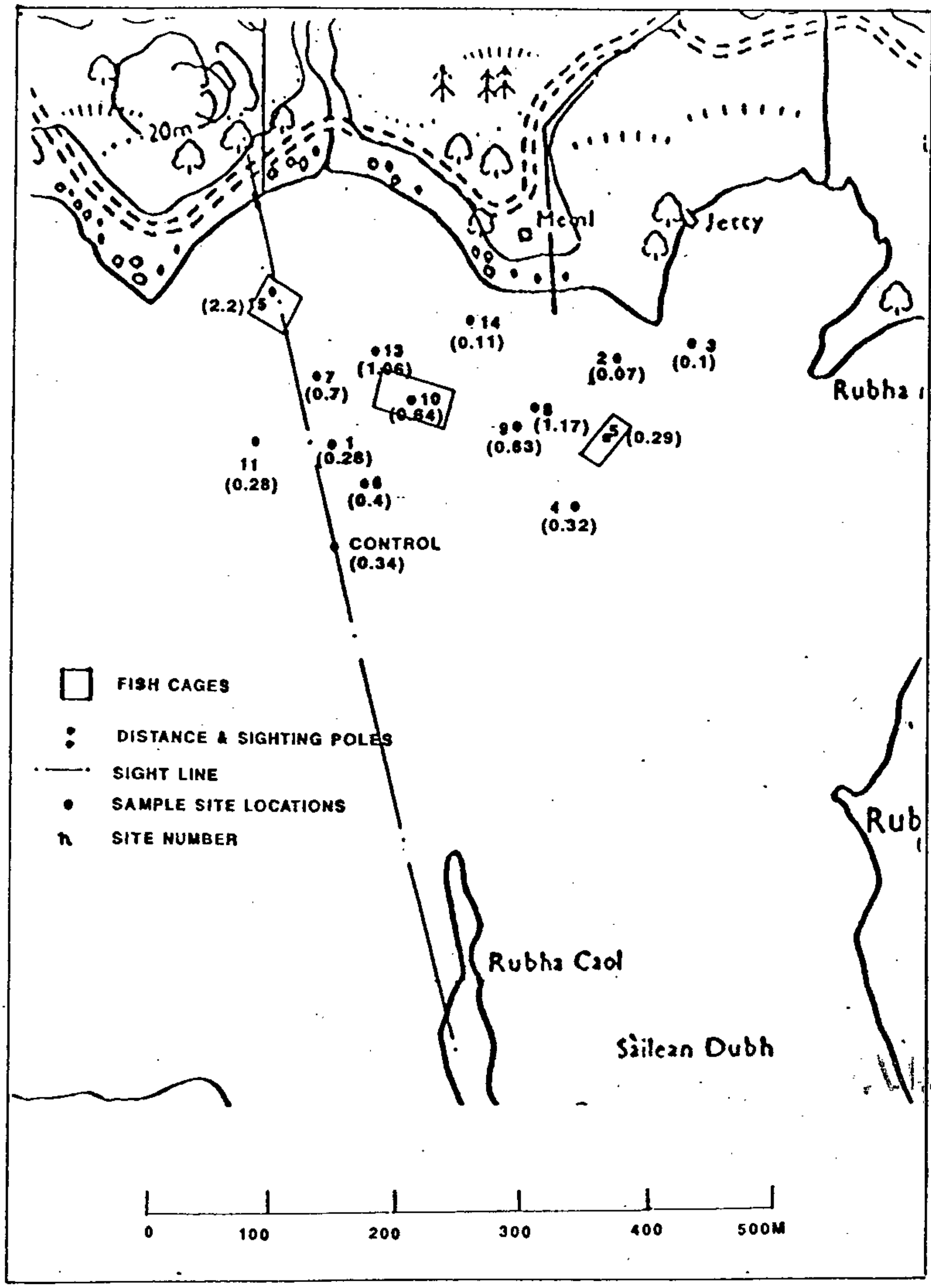


Figure 5. % Organic Matter of sediment dry weight

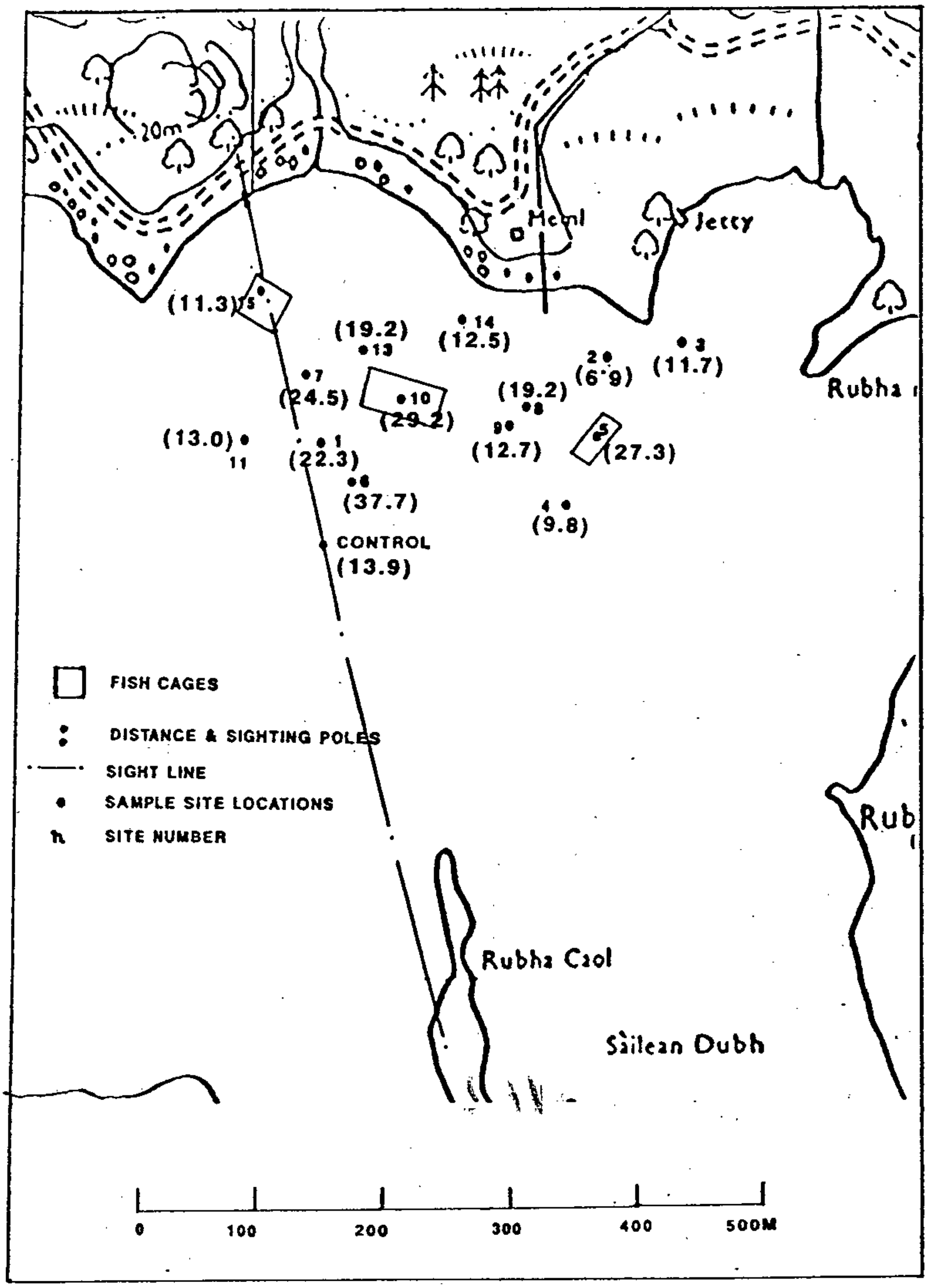


Figure 6. Number of invertebrate taxa at each sampling site

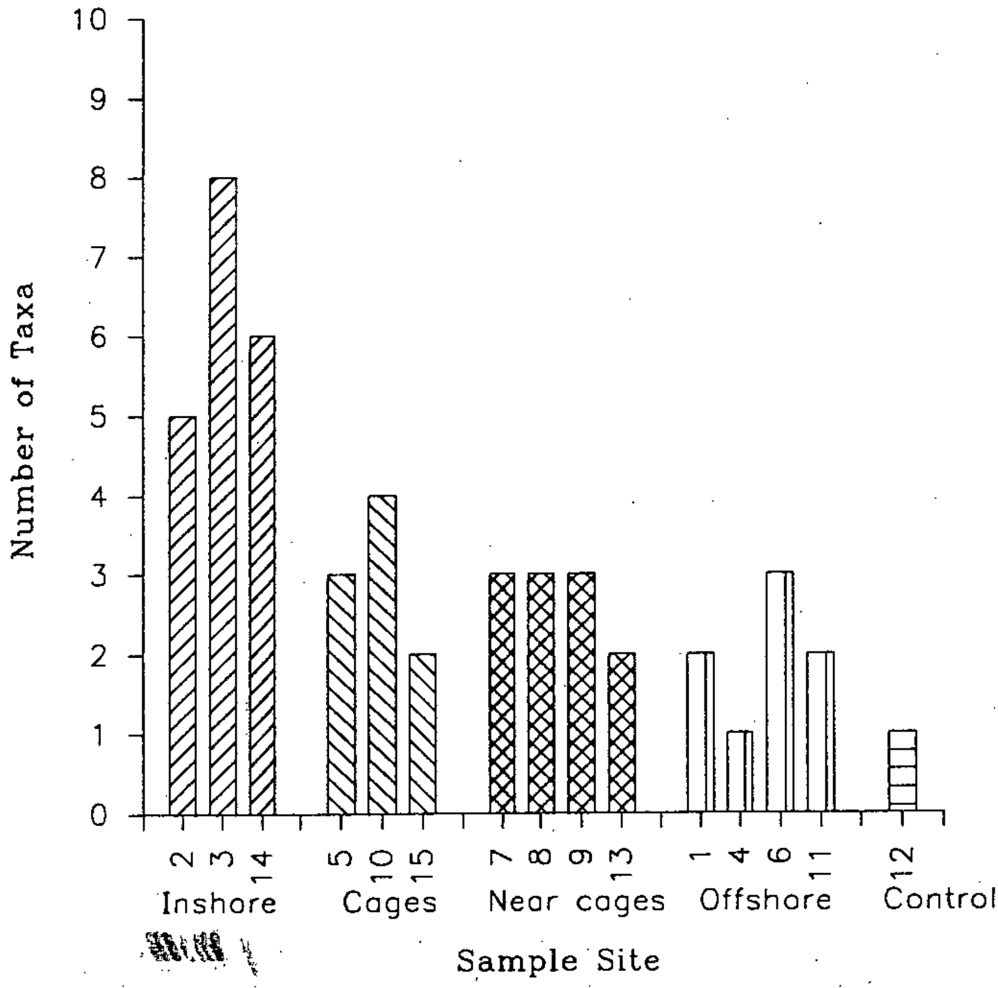


Figure 7. Number of invertebrates at each sampling site

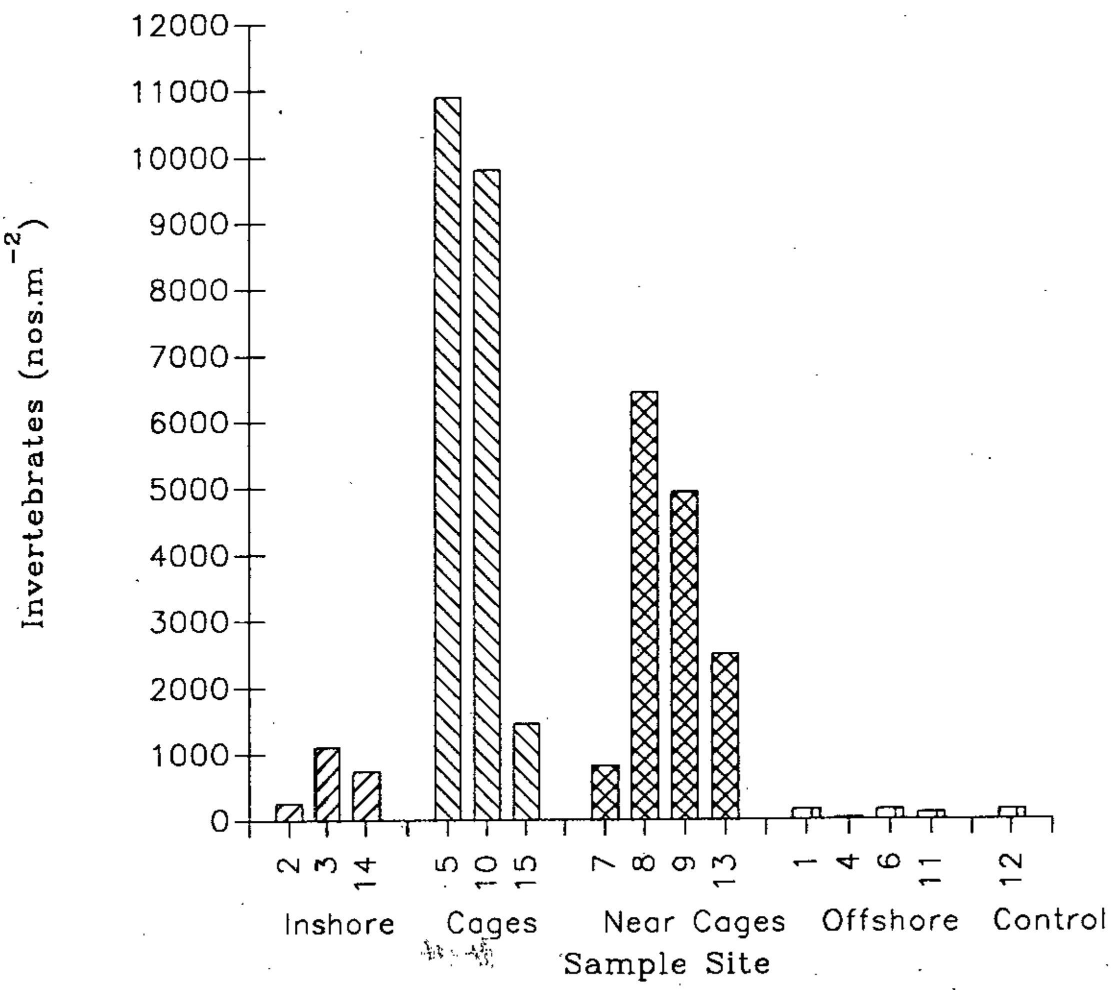


Figure 8. Oligochaeta nos. & taxonomic composition at each sampling site

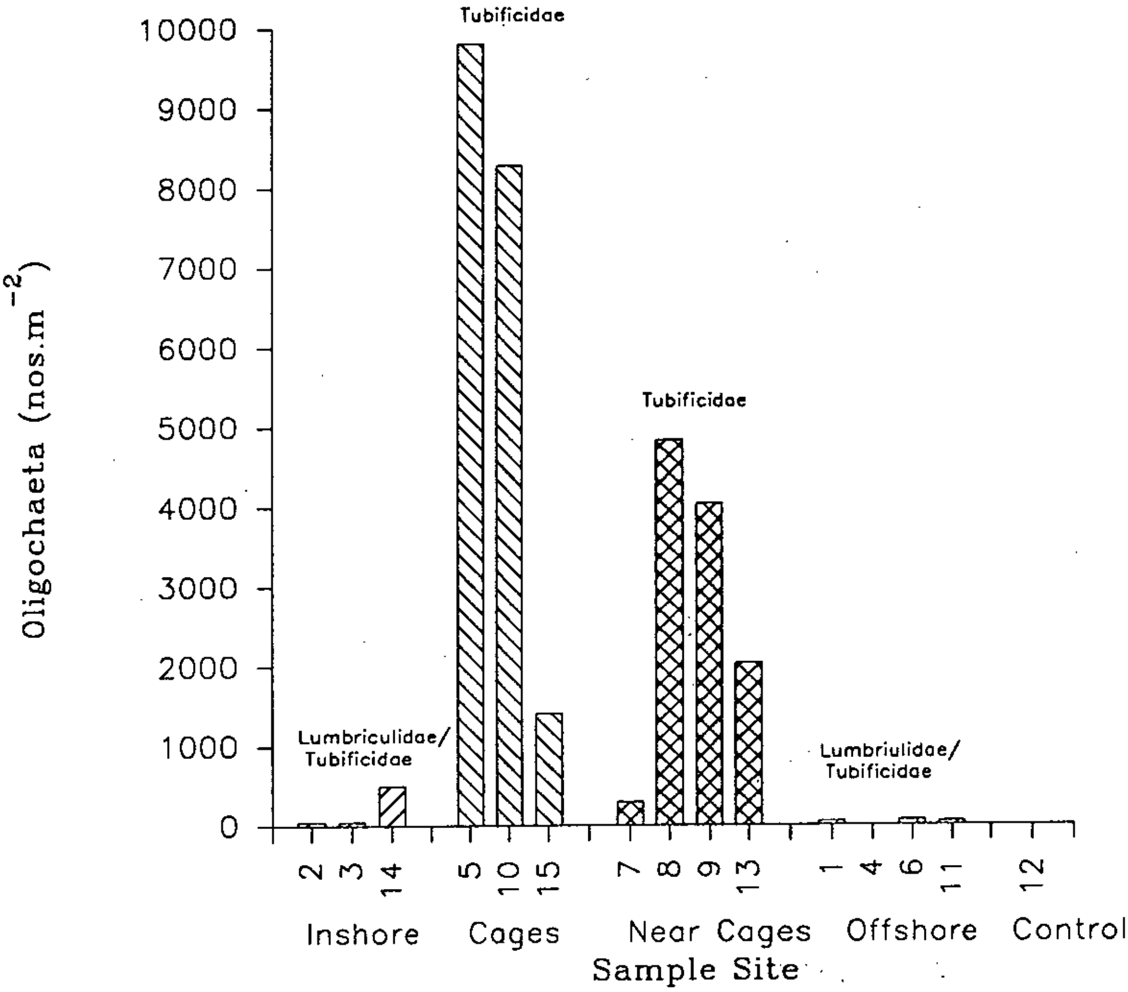


Figure 9. Chironomidae nos. & taxonomic composition at each sampling site

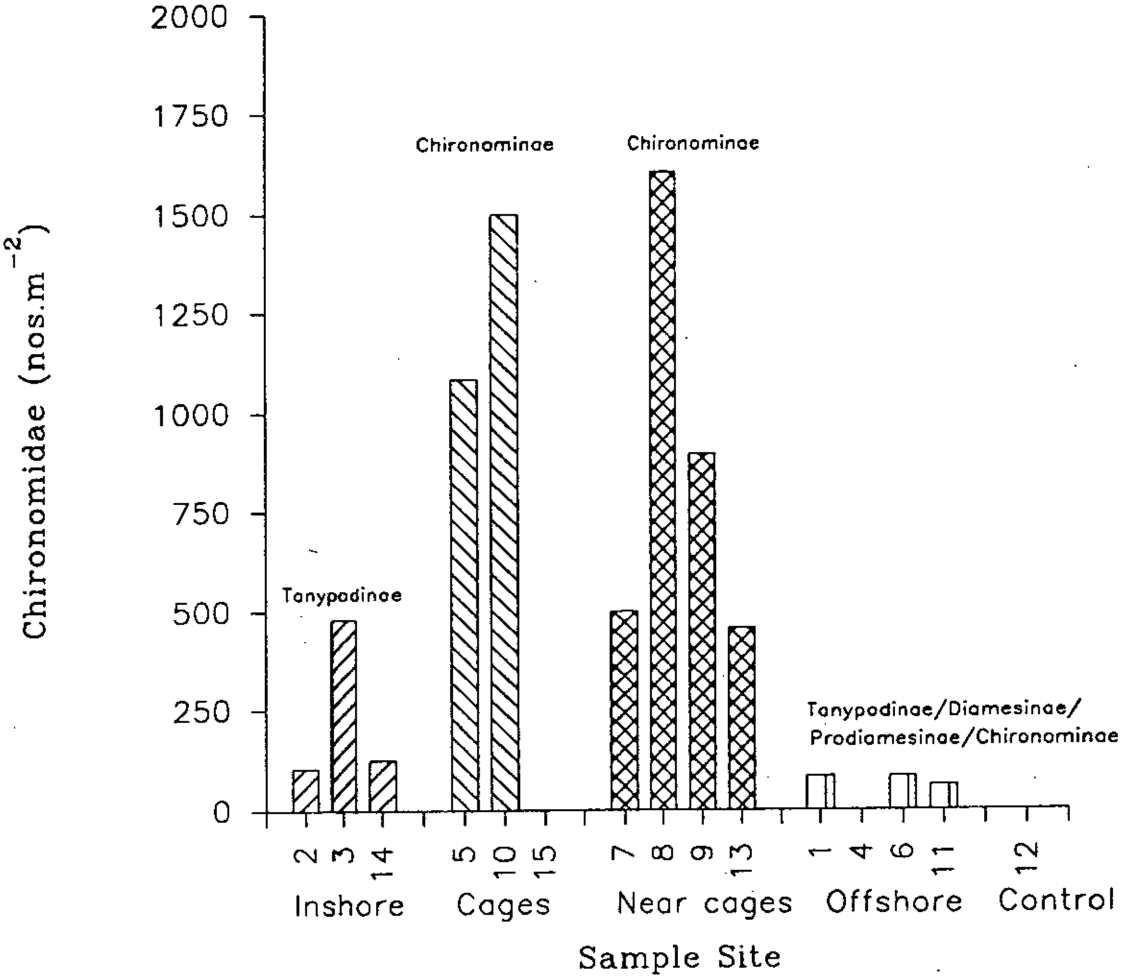


Figure 10. Total phosphorus in Loch Shiel 1986–1992

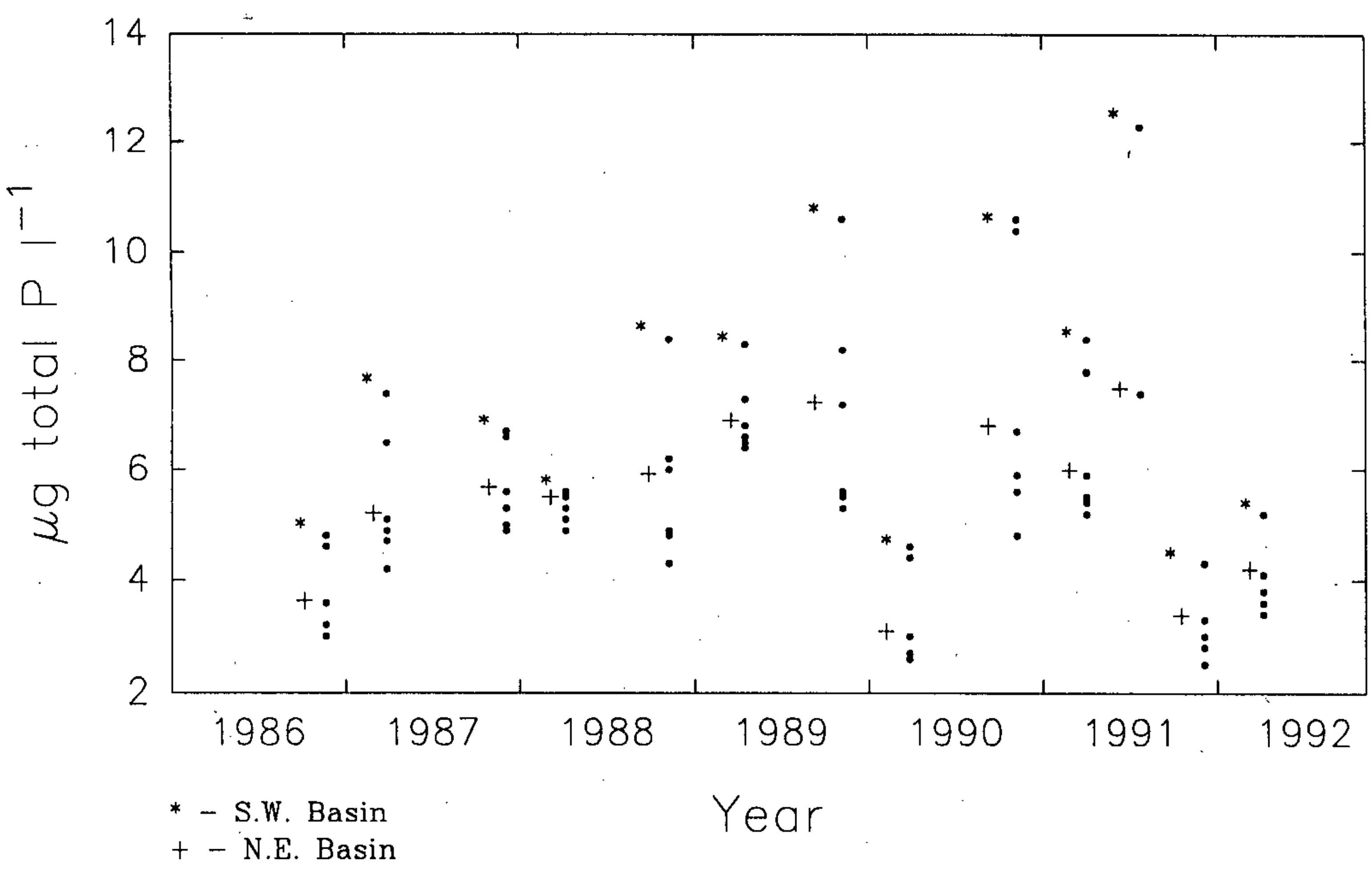
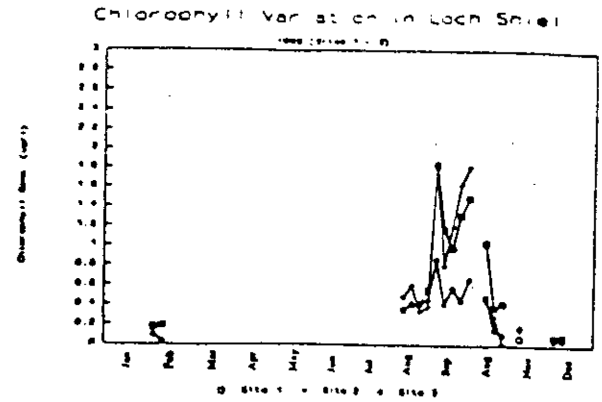


Figure 11. Chlorophyll *a* variation in Loch Shiel (1986-1991).
[IoA data]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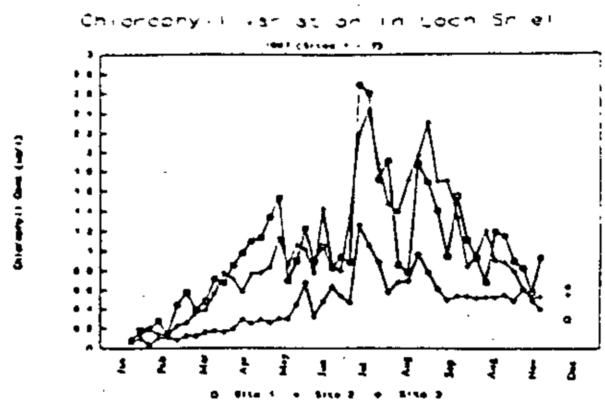
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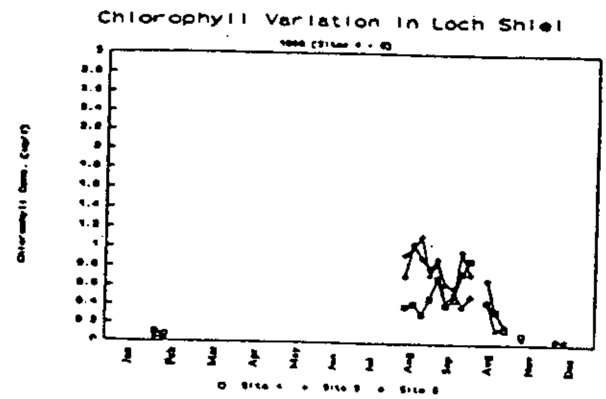
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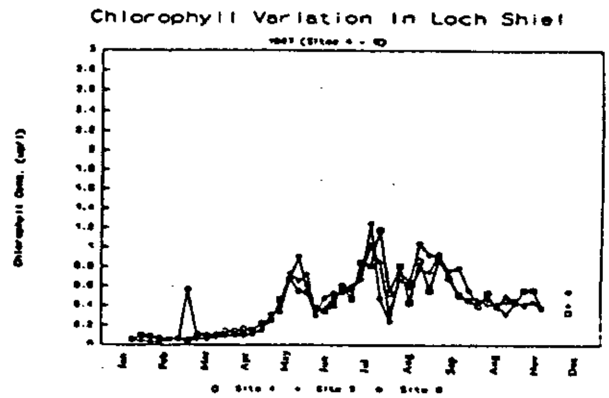
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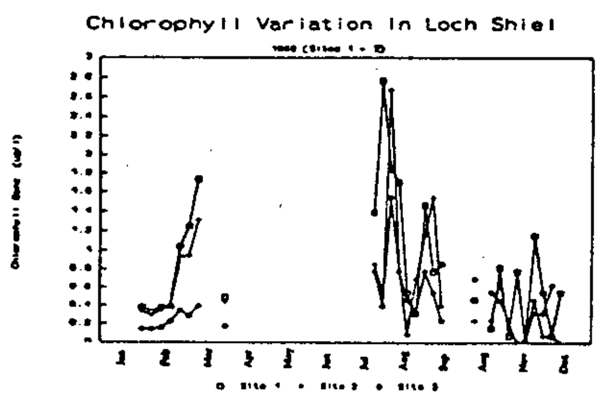
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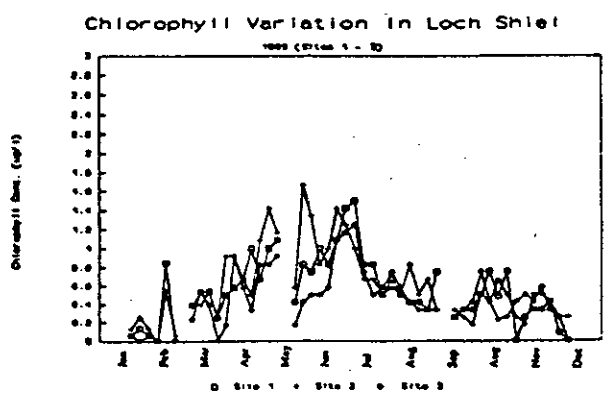
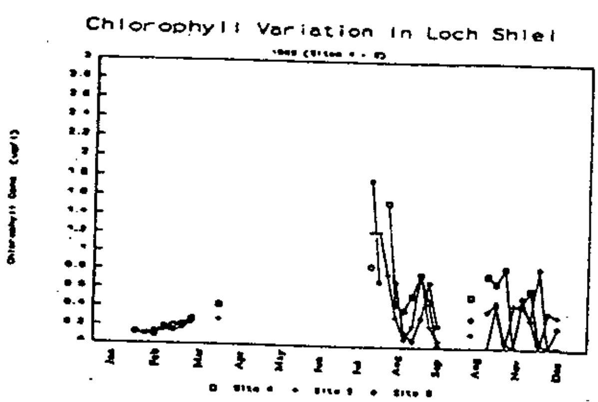

Chlorophyll Variation in Loch Shiel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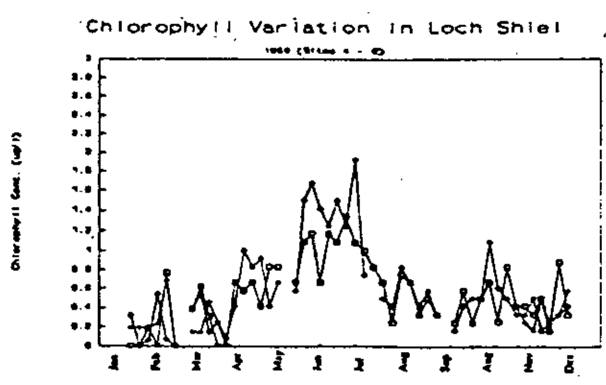
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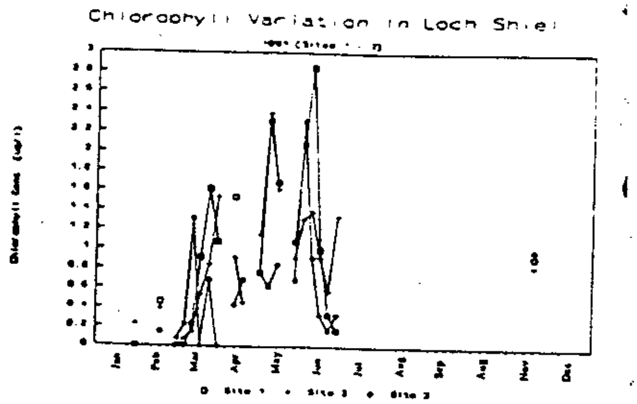
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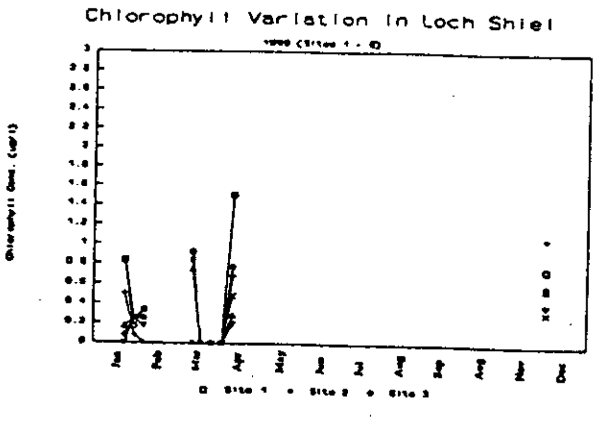
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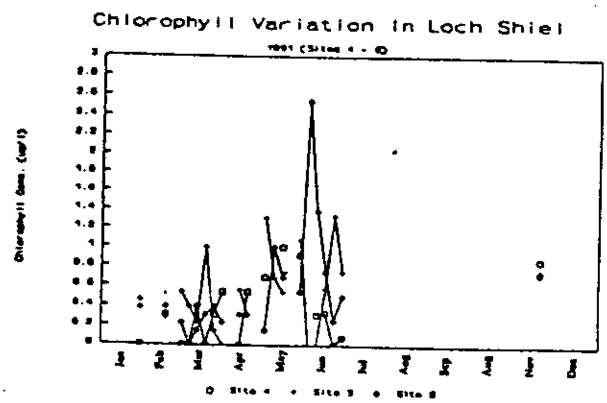
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