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THREATS TO THE OLIGOTROPHIC STATUS OF LOCH SHIEL: A RE-APPRAISAL IN THE LIGHT OF NEW INFORMATION ON FOREST FERTILISATION SCHEDULES

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Report to Scottish Natural Heritage
(March 1994)

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Contract Completion Date: 31 March 1994
TFS Project No: T04051g1

The Institute of Freshwater Ecology is part of the Terrestrial and Freshwater Sciences Directorate of the Natural Environment Research Council.

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Summary

New information on likely schedules of fertiliser application to coniferous forest, results in a desk-generated total phosphorus (P) loadings of 5.35t y^{-1} and 3.25t $y^{-1}$to Loch Shiel; the former figure assumes an average P loss of 1.0kg ha^{-1} y^{-1} from forest within 8 years of fertiliser application while the latter assumes older forestry receiving no fertiliser, and losing P at a rate of 0.1kg ha^{-1} y^{-1}.

The main land-derived sources to the loch would be coniferous forest itself, losing 2.2t or 41% of the higher of the two totals, and rough grazing/moorland releasing 1.2t or 23%. In the situation where all coniferous forest is older than 16 years, forest could contribute as little as 3% of the lower of the two totals. Losses from the smolt cages are estimated at 0.87t - 16% and 26% of the totals.

Expressed as specific areal loadings the total inputs are 0.27g m^{-2} y^{-1} and 0.17g m^{-2} y^{-1}. The higher figure is similar to the OECD limit considered 'permissable' for a waterbody of the mean depth of Loch Shiel i.e. 41m.

Annual mean loch-wide P concentrations ($[P]_\lambda$) of 5.3μg l^{-1} and 3.3μg l^{-1} are calculated, when the total loadings are 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.

Total P levels in the area of the Dalilea fish cages would be expected to indicate the worst case and maximum impact on P levels in the loch, because they 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. Nevertheless, the concentrations measured on 25 September 1992, were very moderate (3.6μg l^{-1} to 7.5μg l^{-1}). Surface water SRP concentrations were also extremely low (0.3μg l^{-1} to 1.6μg l^{-1}). Contrastingly, near-sediment SRP levels ranged from 1.0μg l^{-1} to 203μg l^{-1}, suggesting some remobilisation of phosphate from the sediments; however, a value of 50μg 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, two relatively high organic matter values (27% and 29% of sediment dry weight) were recorded beneath two of the three 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. 1.9μg l^{-1} to 2.4 μg l^{-1}. While oligotrophic desmid, diatom and chrysophyte species were noted, however, there was also a visible though minor bloom of the cyanobacterium (blue-green alga) Anabaena.

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- and P-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 historical data on P, nitrate-N and chlorophyll$_a$ 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, south-west basin 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 $70\mu g$ 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 were significantly and consistently higher in the shallower basin than in the deeper, north-east trench, but even these have never exceeded $3\mu g$ chlorophyll$_a$ l^{-1}. Perhaps somewhat unexpectedly for such a large loch, 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, fish farming, arable agriculture, improved grassland management, and the disposal of domestic waste from houses and hotels, must all be viewed as threats.

An extensive assessment of forest fertilisation schedules in the Loch Shiel catchment is still needed. In contrast to statements in an earlier report, it is now felt that a nutrient loading study is warranted; there are no field data on inputs from the Shiel drainage area. A major focus of such work would be the assessment of loadings of P via the River Polloch. As this drains land containing the majority of the coniferous forest in the catchment, it could be expected to lose P at a rate per hectare in excess of many of the other types of land. However, as this sub-catchment also contains 34 standing waters and large area of marsh, its potential for enriching the loch may have been under-estimated as these waters are likely to retain P.

The effects of nutrient inputs from the Polloch may also be minimised by transport of the majority of the water to the deep, less productive basin of the loch, rather than the shallow basin which would be expected to be more efficient at converting nutrient inputs to plant biomass. 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 of the type maintained by the Institute of Aquaculture and Marine Harvest, is essential for establishing baseline values and detecting any future changes in trophic state. A suite of chemical and biological, trophic monitoring programmes is proposed, and the relative advantages and disadvantages of the various approaches are outlined.The basic 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 the easier option of prevention, would be necessary by the time significant increases in nutrient levels, and certainly biological responses, were evident.

It is concluded that every reasonable effort should be made to minimise nutrient inputs to the loch as far as possible. 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.

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1. INTRODUCTION

1.1 Background

One of the recommendations of the first report on the threats to the oligotrophic status of Loch Shiel was that the phosphorus (P) loss coefficients on which the impact of conifer plantations on the total loading of P to the loch were calculated (by desk analysis), should be checked (Bailey-Watts *et al.*, 1993). The prime reasons for emphasising this, were firstly, that the values used then stemmed from field studies carried out at least a decade ago, and secondly, as those studies were not done in the Loch Shiel catchment, they were unlikely to reflect the actual schedules of fertiliser application applying to this catchment. The recommendation was followed up, and the present report is a re-appraisal of the situation regarding P loadings to Shiel. It is stressed, however, that there are still no more quantitative data on forestry-related P losses, although in consultation with researchers and managers familiar with the Loch Shiel forests, a number of aspects have been identified which suggest that the earlier report over-estimated these losses (equivalent to 4t per year) by a considerable margin. In this connection, while the earlier report considered that an intensive field study to measure P losses would be of debatable value, such a programme is now seen as very appropriate.

The present study has also taken the opportunity to collate the views of foresters and fish farmers on future trends in their practices where these have a bearing on nutrient losses to surface waters. The Forests and Water Guidelines (Forestry Commission, 1991) are of particular relevance as they recommend improved targeting of aerial applications of P to avoid buffer strips and surface waters. In addition, more and more P is now applied in granular form which is a more efficient fertiliser and susceptible to less losses in runoff than unground rock phosphate.The present report thus differs from the previous one, only in the updated sections on P losses, and the interpretation of the figures relating to these.

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 character 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 algal growth, for example, nitrate.

The term 'oligotrophic' generally refers to a waterbody in which the annual mean total P concentration does not exceed $10\mu g\ l^{-1}$ (OECD, 1982). 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. In any event, the concern of this report is not with markedly, even mildly, accelerated eutrophication, but with factors that might lead to rather small increases in trophic status, ones that could still push the loch beyond the upper limit of oligotrophy.

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, 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 ($\bar{z}$, 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). Awareness of 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, 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 used by Bailey-Watts *et al.*, (1992a,b), but taking into account more up-to-date information on forest fertiliser application schedules, and caged fish 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 examination 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. 1960ha *cf* 1330ha. Shiel is thus a very deep waterbody; extending 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 rather 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). As already noted, the area drained by the loch is enormous i.e. 230km^2, although this 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}. This is the flushing rate (ρ) in loch volumes per year.

The water is of low conductivity (*ca* 30-40 μS cm^{-1}) and is 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 20kg t^{-1} fish produced y^{-1} from the cages; inputs from rural communities (on septic tank systems) assumed a *per capita* loss of 1kg 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* 2kg 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 resulted in a longterm (15-y) annual loss rate of 0.4kg ha^{-1} y^{-1}, or half this for forest as a whole. The study by Bailey-Watts *et al.* (1993) which incorporated more information about forest fertilisation schedules and fish farming in particular resulted in a considerably higher total annual P input of 7.39t with some 54% being attributed to forestry and only 12% to smolt rearing cages.

Bailey-Watts *et al.* (1992a, b) pointed out that, as the mean hydraulic retention time of this loch is 1.45y (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 4.9μg l^{-1}. This was considered to be in keeping with the general oligotrophic nature of the system, as were the figures of 7.1μg l^{-1} (north east basin) and 12.4 g l^{-1} (south west basin) measured in summer 1991, although the latter was special in exceeding 10μg l^{-1} (*cf* figures between November 1977 and October 1978 (Bailey-Watts and Duncan, 1981a). The equivalent mean, lake-wide P concentration predicted by Bailey-Watts *et al.* (1993) was 7.3μg l^{-1}, which is even more in line with the measured values quoted above.

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.

The P loadings were estimated as described in by Bailey-Watts *et al.* (1992a). The P loss coefficients employed in that report were used, except for (i) that relating to fish farming which is taken from Phillips *et al.* (1988) as used by Bailey-Watts *et al.* (1993), and (ii) that used for forestry in the light of the following considerations. Literature reviewed by Bailey-Watts *et al.* (1988) and used by Bailey-Watts *et al.* (1992a, 1993) suggested that runoff from fertilised forest is initially high, losing *ca* 2kg P ha^{-1} y^{-1} for the first 3 years, and then decreases to background levels over the next 4 years. These reports also assumed that forests are fertilised (at a rate of 450kg unground rock phosphate P ha^{-1} whether for 'remedial', 'new' or 'restock' purposes according to actual data reviewed) every 7 years of the life of the forest. As a result, a mean annual loss rate of 1kg P ha^{-1} was used to estimate the forestry-derived inputs of this nutrient to Loch Shiel. This rate was also considered to apply over periods of ≥7 years. Recently obtained information, however, suggests that the bulk of a forest area selected for fertiliser treatment will probably receive only two applications - at 0 and 8 years of age during the first rotation of some 55 years. At this stage it still seems reasonable to assume that P is lost at an average rate of 1kg ha^{-1} y^{-1} over the first 16 years, but for the remainder of the rotation period a value of 0.1kg ha^{-1} y^{-1} (as used for deciduous woodland) is probably more appropriate. These values are thus entered in **Table 2** along with the other P loss coefficients used in this study. Other reasons suggesting that current forestry practices will lead to further reductions in P losses are discussed later, as are those supporting the view that the production of salmon smolts is likely to increase in efficiency as regards the use and thus, wastage, of P.

Table 1: Water sampling programme at Loch Shiel for information on nutrient and chlorophyll levels in period 1978-1993. (chl = chlorophyll$_a$; TP = total phosphorus; SRP = soluble reactive phosphorus; N = nitrate-nitrogen)

Sampling Site		1978-1979	1980-1984	1985	1986	1987	1988	1989	1990	1991	1992	1993
S.W. BASIN	No.sample visits	--	--	--	2 (17 for chl)	2 (46 for chl)	2 (26 for chl)	2 (48 for chl)	2 (9 for chl)	3 (21 for chl)	2	2
	Parameters measured	--	--	--	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl
N.E. BASIN	No.sample visits	7 @	--	--	2 (17 for chl)	2 (46 for chl)	2 (26 for chl)	2 (48 for chl)	2 (9 for chl)	3 (21 for chl)	2	2
	Parameters measured	SRP,N, chl	--	--	SRP,TP, N, chl	SRP,TP, N chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl	SRP,TP, N, chl
INFLOW	No.sample visits	--	--	--	1 @	--	--	--	--	--	--	--
	Parameters measured	--	--	--	SRP	--	--	--	--	--	--	--
OUT-FLOW	No.sample visits	9 * @	4 (y^{-1}) *	2 *	1 @	--	--	--	--	1 +	--	--
	Parameters measured	SRP,N	SRP	SRP	SRP	--	--	--	--	SRP	--	--

Notes;

(1) All samples taken by Marine Harvest/Institute of Aquaculture unless indicated otherwise: * Highland Purification Board; @ Institute of Terrestrial Ecology; + Institute of Freshwater Ecology. (2) ITE also measured SRP in N.E. & S.W. Basins in 1986. (3) 1991 includes 1 set of parameters measured by IFE. (4) Limits of detection of N & SRP in 1978 ITE samples were 20μg l^{-1} & 10μg l^{-1} respectively (Bailey-Watts & Duncan, 1981a).

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.

Land:

Rough grazing/moorland - 0.07 kg ha^{-1}

Coniferous forest:

average for first 16 years of age - 1.00 kg ha^{-1}

average for 16 years plus - 0.10 kg ha^{-1}

Improved grassland - 0.40 kg ha^{-1}

Deciduous woodland - 0.10 kg ha^{-1}

Arable agriculture - 0.25kg 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 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 ($\mathbf{V_{in}}$, in m^3 y^{-1}) by the surface area of the lake ($\mathbf{A_l}$, in m^2). Then:

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

A value for $[\mathbf{TP}]_\lambda$ can be predicted by relating the P loading, **L** (expressed as a specific areal burden - in mg m^{-2} lake surface), and $\mathbf{R_P}$, 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 tap water 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 sub-sample was transferred to a pre-weighed 50ml 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 mud 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 500ml 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 665nm (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. Identification was based on Brinkhurst (1971). Immature oligochaetes could not be identified to species level as the above key relies on characteristics found only in mature specimens. Chironomidae 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**. Two P loading values are listed for

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	2.20* 0.10**
Improved grassland	918	0.37
Deciduous woodland	600	0.06
Arable agriculture	58	0.015
Rural community (*ca* 630 people)	-	0.63
	Estimated totals	**4.475* 2.375***

* and ** - see text

coniferous forest, and these are the averages for a 32-year period. They are calculated for two contrasting situations. The larger value assumes that 2000ha (equivalent to 50% of the total covered by conifers in the Shiel catchment) was planted (year 0) and fertilised in years 0 and 8, and subject to no further fertiliser applications thereafter, while the other 2000ha was not planted until year 16, but was fertilised then and at year 24. The lower figure assumes the whole area (4000ha) is covered in forest older than 16 years and receives no remedial P applications over the 32-

year period. As two, what are considered extreme, values have been calculated for the losses of P from coniferous forest (**Table 2**), the remainder of this report gives two values for the total burden of this nutrient on the loch, for the percentage contributions of the various sources to these totals, and for predicted mean influent and in-loch P concentrations. In each case, the first value will relate to the higher loss value in **Table 2**, while the second value will relate to the lower loss figure.

The two total P loadings from the land, that is, excluding P derived from smolt-rearing cages in the loch itself (see below), are approximately 4.5t y^{-1} and 2.4t y^{-1} (**Table 3**). In relative terms, and regardless of the value for coniferous forestry, arable land contributes <1% and deciduous woodland no more than 3%. Moreover, while the potential contributions from improved grassland are higher than these, they are still relatively moderate at 8% and 16%. Where the first of the two situations regarding coniferous afforestation prevails, the rural community would contribute *ca* 14%. If, as envisaged in the second situation, all of the trees were more than 16 years of age, the value would very nearly double i.e. to 27%. Some 27% of the total inputs of P to the loch would be derived from the large area of rough grazing/moorland, even during the period where the coniferous forest is being fertilised; again, a value of approximately double this (51%) is considered to be the more likely where the coniferous forest is 'old' in the terms defined above. These percentages differ by approximately two-fold, because (i) the desk estimates for the loss of P from the area of coniferous forest itself, amount to very nearly 50% (actually 49%) where two consecutive 16-year periods following tree planting are envisaged, and (ii) under the second forest situation, a value of only 4% is calculated.

If the two total, catchment-derived loadings are divided by $\mathbf{V_{in}}$ as defined above i.e. 50.7 x 10^7 m^3 excluding rain falling on the loch surface, values of 8.8μg l^{-1} and 4.7μg l^{-1} for $\mathbf{[TP]_{in}}$ are obtained. The figures do not differ in any material way if the extra input of water and P in rain falling on the loch surface, is taken into account (Bailey-Watts *et al.*, 1992a,b). The higher of the two concentrations does not differ significantly from those of *ca* 9μg l^{-1} measured by Bailey-Watts *et al.* (1992a) in 4 of the inflows in summer 1991. However, it exceeds the values of 5-6μg l^{-1} measured in another 4 streams sampled at that time, and this is considerably less than the figures of 135μg l^{-1} and 94μg l^{-1} reported for two streams near Acharacle; these latter contained a large proportion of SRP, perhaps indicating the influence of domestic drainage (Bailey-Watts *et al.*, 1992b).

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

Commercially 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 0.87t 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 19% and 37% of the two P loads estimated to be derived from the catchment. The smolt-rearing activities can be considered to introduce considerably less P to the loch than that reckoned to be derived from young coniferous forest, but much more than older forest - even allowing for errors of *ca* 30% in the choice of export coefficients and in estimating of the size of the rural community, and errors of approximately 10% in assessing various land areas.

If the annual loading attributed to smolt-rearing (0.87t P) was distributed throughout Loch Shiel i.e. with no losses to the sediments, it would *raise* the P concentration by only 1.1μg l^{-1} if the loch volume is taken, or 1.6μg l^{-1} if the annually-renewed volume is taken.

4.1.3 A desk-derived total P loading to Loch Shiel

The estimated total loadings of P taking account of the two situations as regards coniferous forest fertilisation are 5.35t P y^{-1} and 3.25t P y^{-1}, as against the earlier estimates of 4.8t (Bailey-Watts *et al* 1992a, b) and 7.39t (Bailey-Watts *et al* 1993). It should be noted that an estimated total P loading assuming the whole Loch Shiel catchment was 'natural' would be 2.3t P y^{-1} i.e. catchment area of 230km^2, combined with a P loss coefficient of 0.1kg ha^{-1} y^{-1}.

The contribution from what is assumed here to be a constant yearly input from the fish cages amounts to 16% and 26% of the overall totals. The loadings due to afforestation itself could reach 41% under the first of the two fertilisation situations envisaged, but as little as 3% if the second situation as defined above prevails.

Expressed as a specific areal loadings i.e. grams per square metre of loch surface per year, the total tonnages are 0.27 and 0.17. The higher value does not differ significantly from that of 0.25g m^{-2} y^{-1} which Vollenweider (1968) considered the upper limit 'permissible' for a lake of the mean depth of Shiel (41m).

4.1.4 Predicted total P concentration in the loch

Insertion into the Dillon and Rigler equation of the predicted specific areal loadings (270 and 170mg P m^{-2} y^{-1}), the P retention coefficient (0.45), flushing rate (0.69 loch volumes y^{-1}) and loch mean depth (41m), results in annual mean, loch-wide P concentrations of 5.3μg l^{-1} and 3.3μg l^{-1}. Both values are in close proximity to measured concentrations (see Sections 4.2.2 and 4.3).

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. Nevertheless, they 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 levels of 5.3μg l^{-1} and 3.3μg 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 $50\mu g\ l^{-1}$, which ranks approximately in the middle of the range, and exceeds the figure of $29\mu g\ l^{-1}$ recorded under one of the cages; however, values of $80\mu g\ l^{-1}$ and $203\mu g\ l^{-1}$ were recorded under the other 2 cages. Thirdly, the results suggest considerable patchiness; for example, the control site with $50\mu g\ l^{-1}$ is relatively close to site 6 with $4\mu g\ l^{-1}$; and site 2 which is in-shore and dominated by Quillwort (*Isoetes lacustris*), gave a value of only $4.7\mu g\ l^{-1}$, while a pair of richer sites nearby i.e. 3 and 14 at which plants were not noticeable, gave values of $46\mu g\ l^{-1}$ and $70\mu g\ l^{-1}$ respectively.

These results suggest that some of the P in this sediment is potentially recyclable, but the main 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 assessed in any detail, 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 *et al.*, 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 three oligochaete species were found in the survey, *Limnodrilus hoffmeistrei, Tubifex tubifex* (Family Tubificidae) and *Lumbriculus variegatus* (Family Lumbriculidae). 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 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 *et al.* (1981), three species of oligochaetes were found in Loch Shiel, namely *L. hoffmeistrei, T. tubifex* and *Peloscolex ferox* (all Tubificidae) and two chironomids *Metriocnemus* (Orthocladinae) and *Polypedium* (Chironominae).

Table 4: List of Invertebrate Benthos recorded at Dalilea, Loch Shiel.

	Inshore			Cages			Near Cages				Offshore				C
	2	3	14	5	10	15	7	8	9	13	1	4	6	11	12
Sphaeridae *Pisidium* sp.+		✓				✓					✓	✓			✓
Pisidium hibernicum		✓	✓			✓									✓
Pisidium personatum							✓								
Glossiphonidae *Helobdella stagnalis*	✓	✓	✓												
Nematoda	✓	✓	✓		✓										
Lumbriculidae *Lumbriculus variegatus*	✓	✓	✓				✓				✓				
Tubificidae *Limnodrilus hoffmeistrei*			✓	✓	✓	✓		✓	✓	✓			✓		
Limnodrilus/ Potamothrix sp.*				✓			✓							✓	
Tubifex tubifex				✓	✓			✓							
Potamothrix/ Tubifex sp.*				✓	✓			✓							
Chironomidae *Chironomus* sp.				✓	✓		✓	✓	✓	✓				✓	
Endochironomus sp.									✓						
Protanypus sp.													✓		
Monodiamesa sp.													✓		
Macropelopia sp.	✓	✓													
Procladius sp.		✓	✓								✓			✓	
Leptoceridae *Mystacides azurea*	✓														
Polycentropidae *Cyrnus trimaculatus*		✓													
Sericostomatidae *Sericostoma personatum*		✓													

N.B. + - Decalcified *Pisidium* sp.   C - Control   * - Immature Oligochaeta

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 Chironomidae 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 *et al.*, 1981).

4.3 The trophic status of Loch Shiel - historical context

Surface and deep water TP levels recorded twice (in March/April and November) by the Institute of Aquaculture (IoA) from 1986-1993, suggest that the loch has remained oligotrophic over that period. Although Bailey-Watts *et al.* (1992a) measured $12.3\mu g\ l^{-1}$ in the south-west basin on 25 July 1991, a significantly lower concentration of $7.4\mu g\ l^{-1}$ 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 that the south-west basin, which is the shallower of the two basins, is the richer in P. **Figure 10** plots surface values only, as variation with depth is minor.

The values for annual maxima (normally November) in increased from 1986 to 1990, but decreased thereafter with the exception of the high value of $12.3\mu g\ l^{-1}$ in July 1991 (see above). However, the average levels over the loch as a whole appear to have remained well below $10\mu g\ l^{-1}$ - 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). While the mean loch-wide concentrations of $5.3\mu g\ l^{-1}$ and $3.3\mu g\ l^{-1}$ predicted in Section 4.1.4 come well within the range of values shown in **Figure 10**, the higher value is closer to the mean of the run of points up to autumn 1991, while the lower figure is more in keeping with the values recorded in autumn 1991 and thereafter apart from the two points plotted for the shallow, south-western part of the loch in autumn 1993.

Nitrate concentrations corresponding to the TP data rarely exceed $70\mu g\ N\ l^{-1}$. However, in April 1991 all 6 stations sampled (4 along the deep basin, and 2 in the shallow basin) gave higher values i.e. $110\mu g\ N\ l^{-1}$ to $150\mu g\ N\ l^{-1}$.

Primarily as a consequence of SRP levels rarely exceeding $1\mu g\ l^{-1}$, 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. Nevertheless, while even without further N enrichment, addition of more P would lead to more algae, because 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$_a$ (**Figure 11**) is much more extensive than that on TP, and confirms the oligotrophic nature of this waterbody. At least three very interesting 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 in overall phytoplankton biomass 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$_a$ concentrations shown - those for the shallower south-west basin July-August 1988 (**Figure 11a**) - 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, few of the values plotted for the deep north-east basin in **Figure 11b** 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 stemmed from monthly sampling only. In summary therefore, there is no evidence of significant enrichment over some 15 years, that is, to a point where either open water P or chlorophyll$_a$ levels exceed those characteristic of oligotrophic waters.

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

5.1 The current situation

Long-term variation in annual inputs of P to Loch Shiel are likely to be considerable. In the present desk study, this is attributed to shifts of some 10-fold in the amounts of P lost from coniferous forest. One of the situations envisaged by this study includes a period during which young forest land is fertilised; then, the main threats to the oligotrophic status of Loch Shiel will stem from that activity (41% of a predicted total P input of 5.35t y^{-1}). Under the same conditions, upland would contribute some 23%, while smolt-rearing at the current production rate would contribute some 16%. A quite different set of figures results from the other situation envisaged i.e. where the forest is more than 16 years old and receives no fertiliser, and so loses P at a rate of only 0.1kg ha^{-1} y^{-1}. Then, the major components of a predicted total loading of 3.25t y^{-1} are upland (37%), fish farming (19%) and the rural community (19%).

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 grassland management must all be implicated. Indeed, the higher of the two total loadings considered here, equates to figure of 0.27g P m^{-2} loch surface per year, which does not differ significantly from a value considered to represent the maximum 'permissible' for a waterbody of the Loch Shiel type. The chemical and biological impacts attributable to the fish cages are undoubtedly more easily identified than those due to even the highest, diffuse inputs. 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.

The earlier report suggested that 'It is unlikely that even a much-needed scrutiny of forestry management practices in the Loch Shiel catchment .... would alter the main conclusion regarding the sources of P to the loch i.e. that forestry-related inputs outweigh the fish cage loadings'. However, the present report suggests that this is far from the case, bearing in mind the latest views of foresters on fertiliser schedules. Since, as already emphasised, there are no actual data on P losses from the Shiel catchment, however, it is probable that another statement in the previous report should also be revised ; this suggested that 'a thorough P loading study' would also not change the original view regarding the relative importance of afforestation and fish farming. Now, such a study based on Loch Shiel appears crucial if the uncertainties expressed in the previous report are to be examined i.e. on (i) the relative bio-availability of P from forestry and fish farming, and (ii) the potential ameliorating nature of the marsh and Loch Doilet which may trap material draining the Polloch catchment wherein much of the coniferous forestry is situated. In addition, Marine Harvest anticipate an increase in production tonnage in the next few years due to improvements in stock performance and feeding techniques. Fewer, larger fry would be utilised to grow a similar number of smolts to greater size than those currently produced. There would be significant improvements in Food Conversion Ratios as a result, causing minimal additional P input (G. Willoughby *pers comm.*).

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, assuming the first of the two situations envisaged here regarding coniferous forestry. Nevertheless, bio-available P (SRP) concentrations, representing what has not been utilised by plants at the time of sampling, are very low. In spite of generally low nitrate values, SRP concentrations would need to be raised 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

In-loch nutrient chemistry and chlorophyll$_a$ data suggest that in spite of increases in forestry- and fish farm-related nutrient inputs, the water has not become significantly richer. This does not mean, however, that further increase in loadings would not result in increased algal levels. Indeed 'eutrophic indicator' species of algae such as *Anabaena flos-aquae* have already been recorded. Regardless of the execution and outcome of work proposed below, every effort should thus be made to minimise as far as is possible and reasonable, further inputs of nutrients. The statutory authorities have also to decide whether it is necessary to stem threats affecting the 'natural' evolution and enrichment over geological time spans. The technology and knowledge regarding good catchment management in this connection, exist. In this context, it is important to ensure that the Forests and Water Guidelines are adhered to and that Marine Harvest are capable of demonstrating improvements in feed use and food conversion ratios. Certainly, ways of eradicating as far as possible, the inputs of nutrients from houses and hotels should be given priority attention, even though most of the people live near the western end of the loch close to the outflow, where the effects of any sewage-derived P passing into the loch is likely to be limited by being flushed out of the system relatively rapidly. 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 $10\mu g\ l^{-1}$ and a number in the shallow basin exceed this level.

5.3 Future work and monitoring

Statements about the P loading continue to 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 further discussions with forest managers. However, more field measurements are crucial. Even after three desk analyses, we are no nearer to the 'actual' situation. Actual field data are required, and an intensive programme of chemical sampling coupled with hydrological monitoring of a number of major feeder waters is recommended. Prime consideration should be given to assessing the transport of total P and the main soluble and particulate fractions, from the Polloch 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, $10\mu g\ l^{-1}$ - 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, in the future, 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 markedly 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 persistent remains in sediments - also provide information on trophic status, and like their planktonic counterparts, are likely to respond most rapidly to changes in P levels. They require a programme of sampling somewhat similar to those appropriate for zoobenthos in terms of spatial coverage, and those used for 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.

6. ACKNOWLEDGEMENTS

We are especially grateful to Graham Willoughby and his colleagues at Marine Harvest, and Ian Strachan (Scottish Natural Heritage) for the time they spent with us in Lochailort and their help with our preparations for the practical field work, and to Brian Bellwood (Highland River Purification Board), Paul Biggin (Forest Enterprise) and Miller Harris (Tilhill Economic Forestry) and Tom Nisbet (The Forest Authority Research Division) for offering, and discussing with us, their views on future trends in the industries as regards phosphorus losses. We also thank them and David Howell (SNH) for valuable comments on an earlier draft of this report. Miss Andrea Meachan helped with data analysis, and Jon Bass and Rick Gunn helped with invertebrate identification. The Glenfinnan House Hotel is thanked for general hospitality and for allowing us to store samples and carry out 'laboratory' work.

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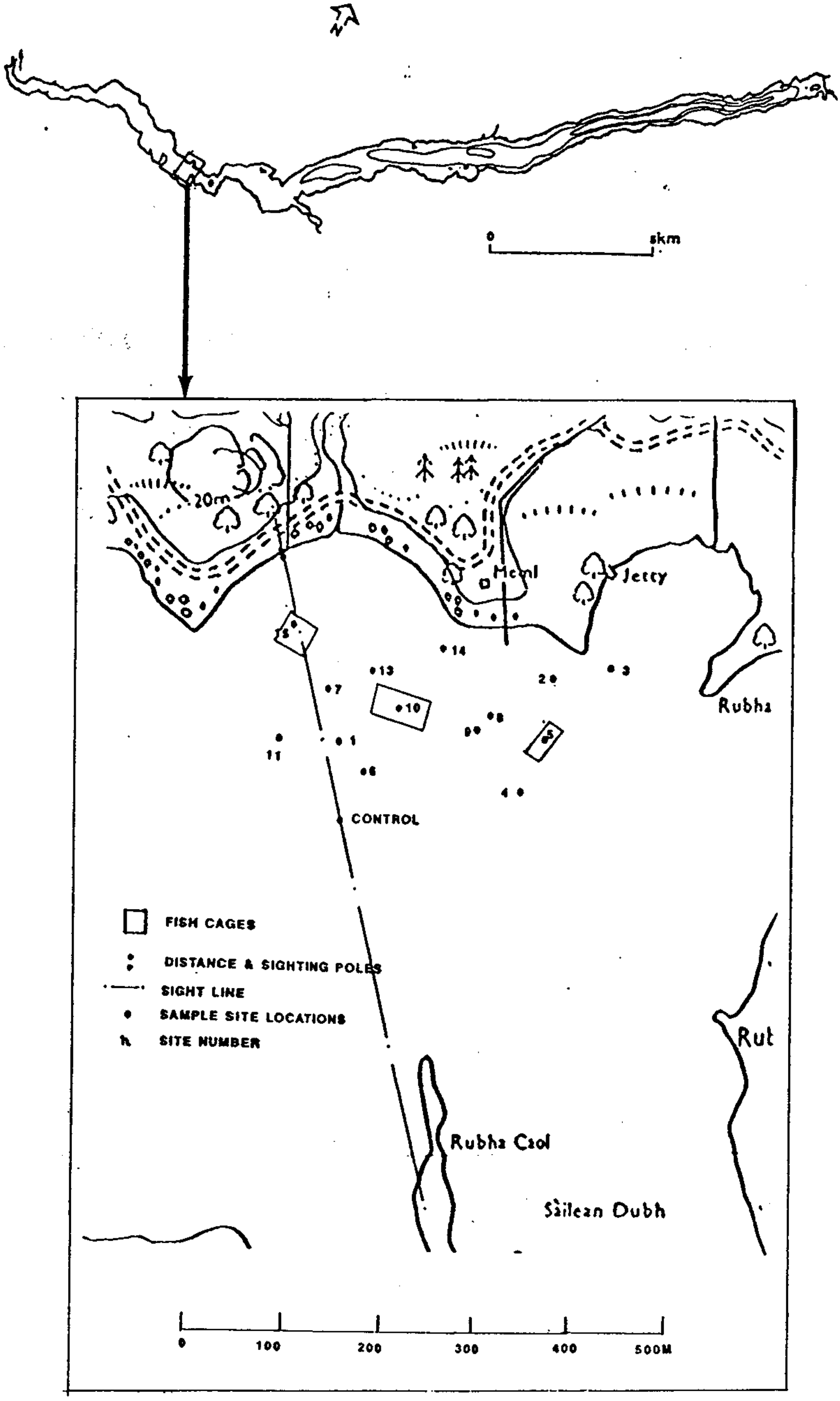
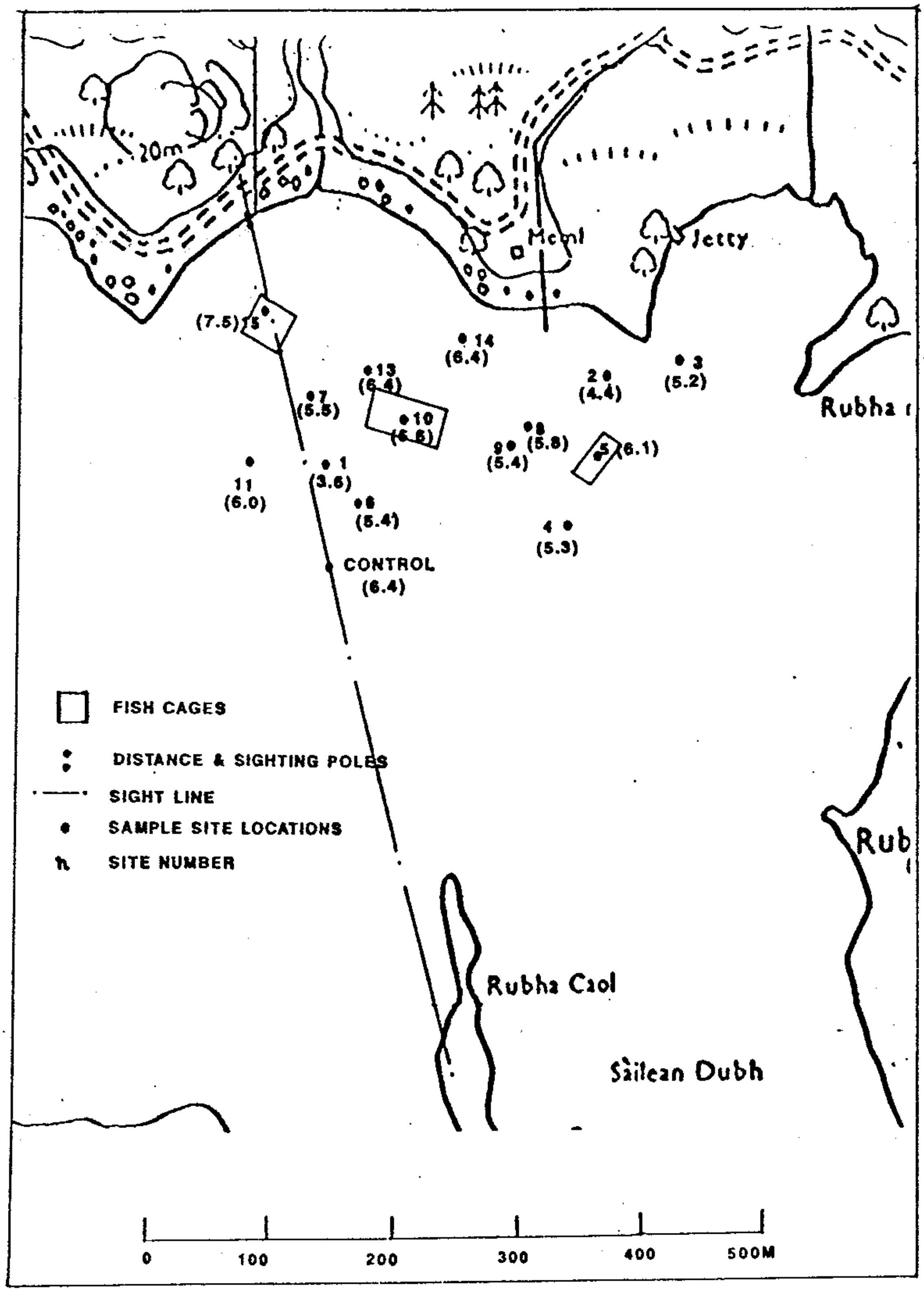
FIGURES

Figure 1. Location of the sample sites in the Dalilea fish cage area



0
5km
20m
Meml
Jetty
Rubha
14
13
3
2
7
10
8
9
5
5
11
1
6
4
CONTROL
FISH CAGES
DISTANCE & SIGHTING POLES
SIGHT LINE
SAMPLE SITE LOCATIONS
SITE NUMBER
Rut
Rubha Caol
Sàilean Dubh
0
100
200
300
400
500M

Figure 2. Total phosphorus concentrations in surface water (μg P l^{-1}) in and around the Dalilea fish cage area-September 1992



20m
Meml
Jetty
Rubha
(7.5) 15
14
(6.4)
13
(6.4)
7
(5.5)
10
(6.6)
2
(4.4)
3
(5.2)
8
(5.8)
9
(5.4)
5
(6.1)
11
(6.0)
1
(3.6)
6
(5.4)
4
(5.3)
CONTROL
(6.4)
FISH CAGES
DISTANCE & SIGHTING POLES
SIGHT LINE
SAMPLE SITE LOCATIONS
n SITE NUMBER
Rub
Rubha Caol
Sàilean Dubh
0
100
200
300
400
500M

Figure 3. As Figure 2 for soluble reactive phosphorus levels in water overlying the sediment (μg P l^{-1})

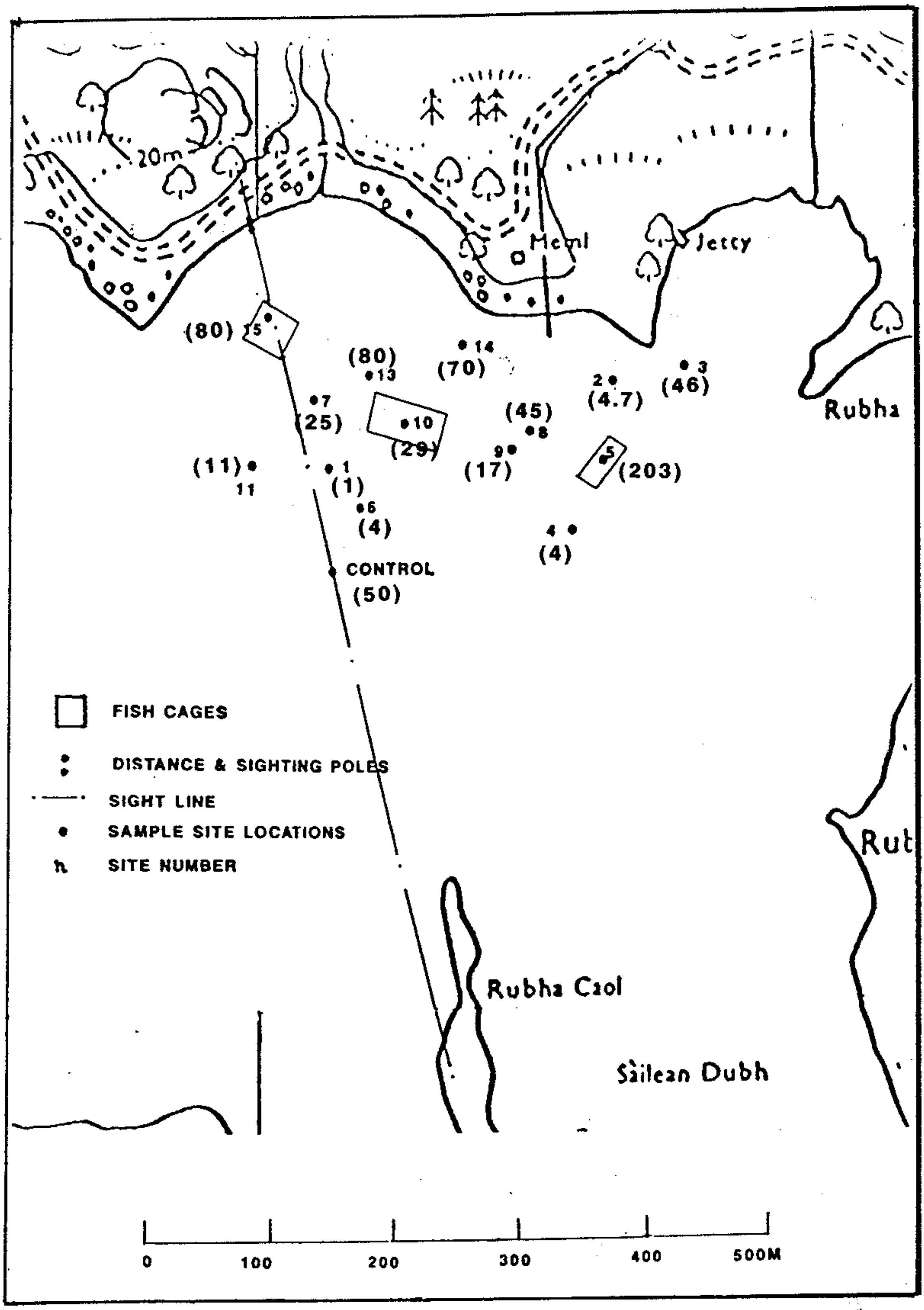

20m
Meml
Jetty
Rubha
(80) 15
(80)
13
14
(70)
2
(4.7)
3
(46)
7
(25)
10
(29)
(45)
8
9
(17)
5
(203)
(11)
11
1
(1)
6
(4)
4
(4)
CONTROL
(50)
FISH CAGES
DISTANCE & SIGHTING POLES
SIGHT LINE
SAMPLE SITE LOCATIONS
n SITE NUMBER
Rut
Rubha Caol
Sàilean Dubh
0
100
200
300
400
500M

Figure 4. As Figure 2 for% P of sediment dry weight

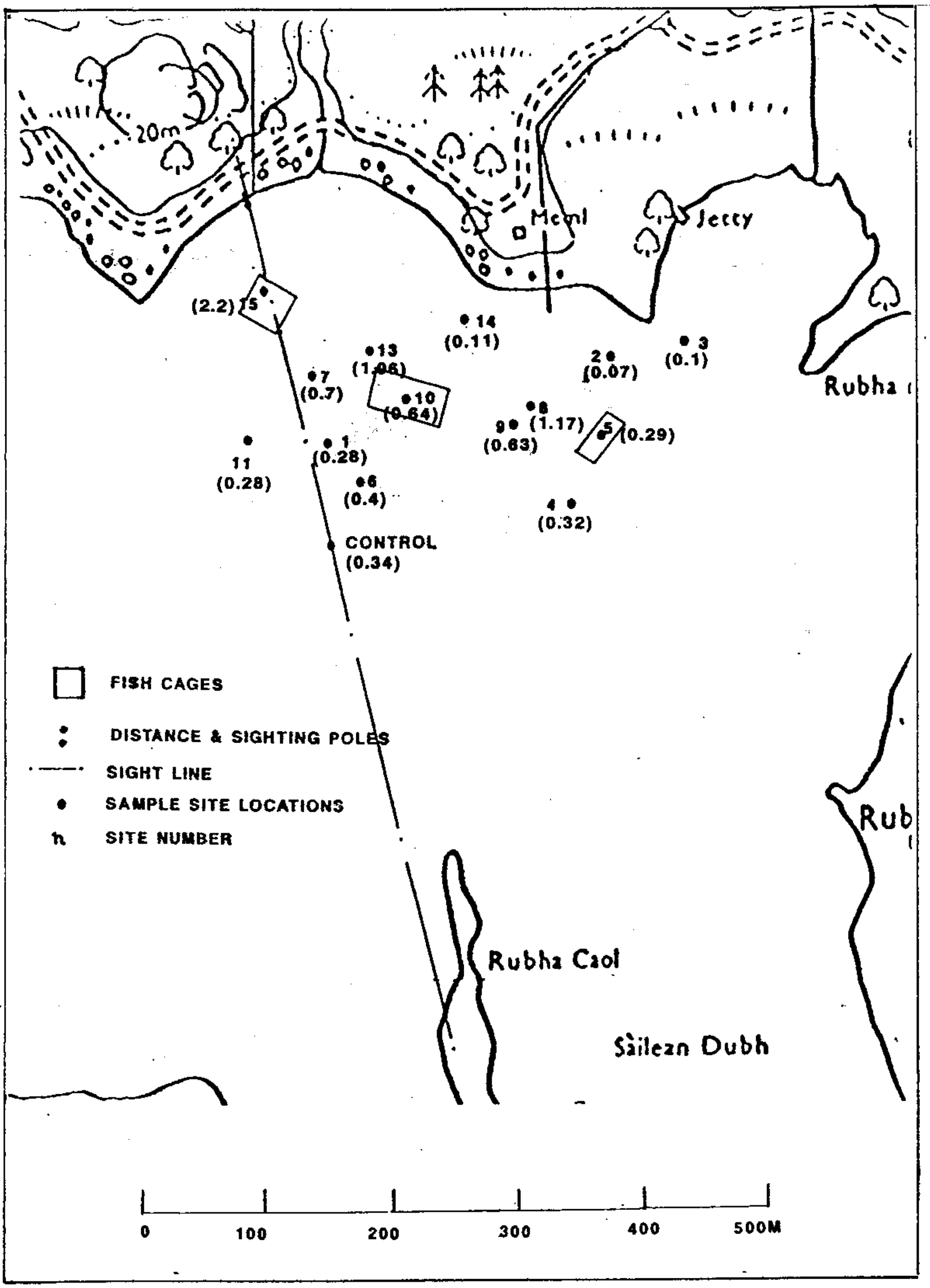
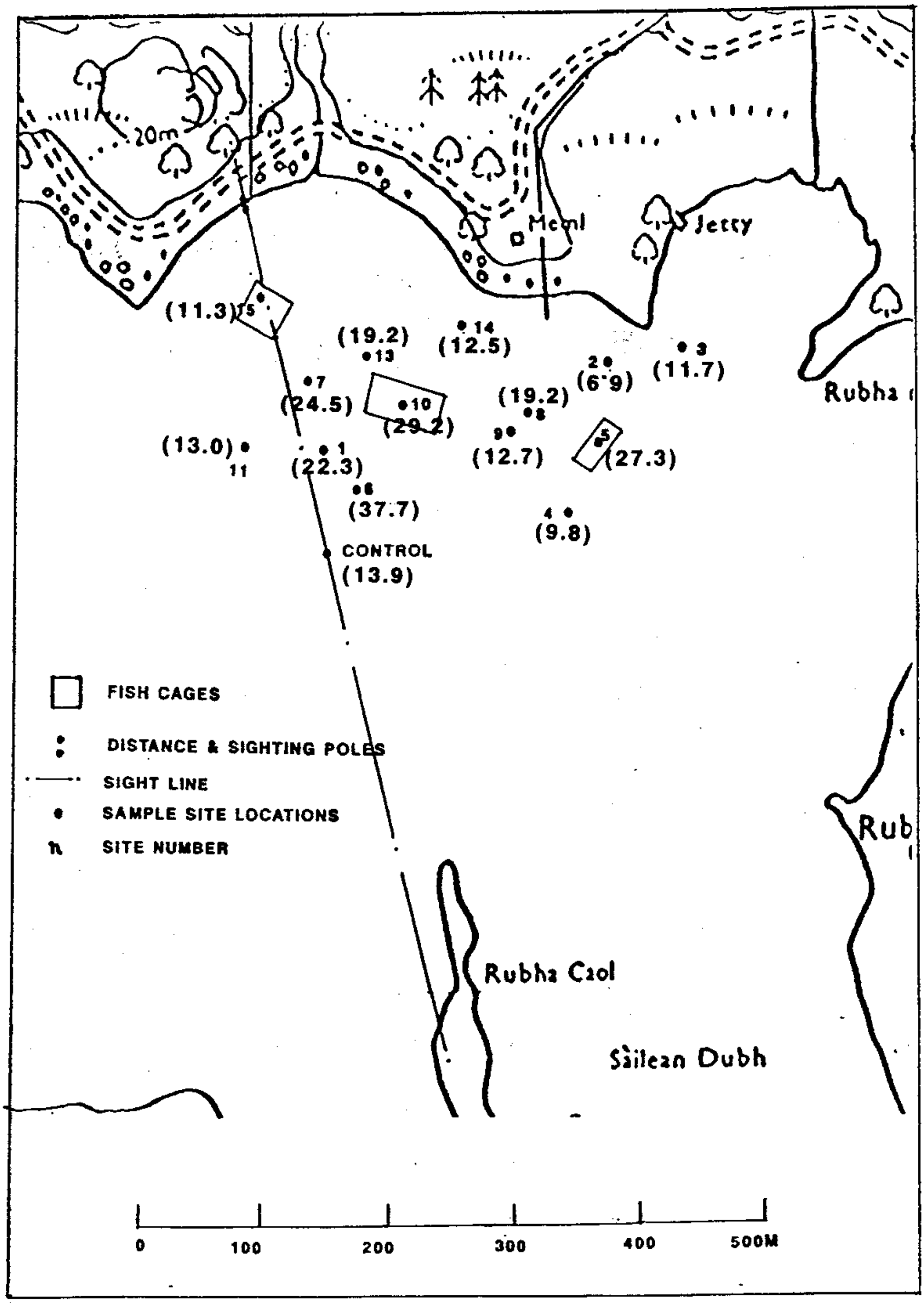
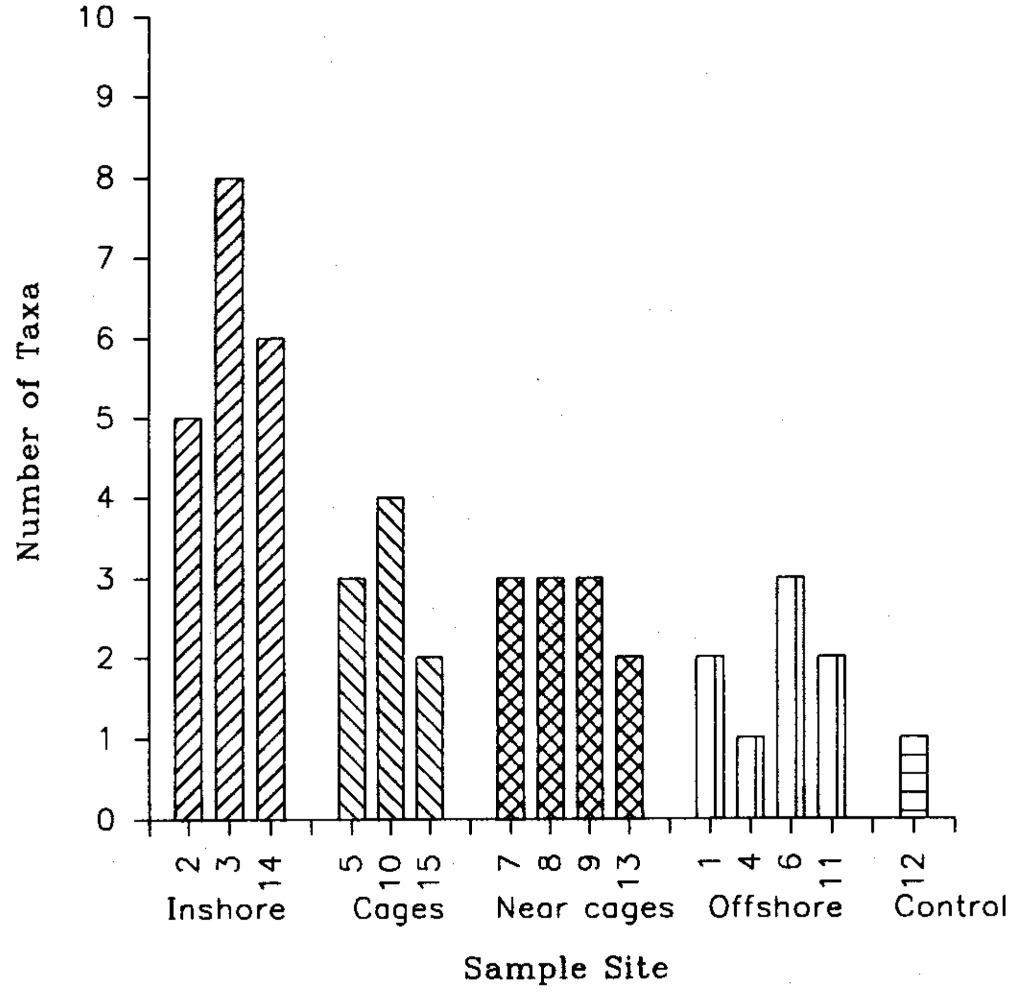

20m
Meml
Jetty
Rubha
(2.2) 15
14
(0.11)
13
(1.06)
7
(0.7)
10
(0.64)
2
(0.07)
3
(0.1)
8
(1.17)
9
(0.63)
5
(0.29)
11
(0.28)
1
(0.28)
6
(0.4)
4
(0.32)
CONTROL
(0.34)
FISH CAGES
DISTANCE & SIGHTING POLES
SIGHT LINE
SAMPLE SITE LOCATIONS
n SITE NUMBER
Rub
Rubha Caol
Sàilean Dubh
0
100
200
300
400
500M

Figure 5. As Figure 2 for % organic matter of sediment dry weight



20m
Meal
Jetty
Rubha
(11.3) 15
(19.2) 13
14 (12.5)
2 (6.9)
3 (11.7)
7 (24.5)
10 (29.2)
(19.2) 8
9 (12.7)
5 (27.3)
(13.0) 11
1 (22.3)
6 (37.7)
4 (9.8)
CONTROL (13.9)
FISH CAGES
DISTANCE & SIGHTING POLES
SIGHT LINE
SAMPLE SITE LOCATIONS
n SITE NUMBER
Rub
Rubha Caol
Sàilean Dubh
0
100
200
300
400
500M

Figure 6. As Figure 2 for the number of invertebrate taxa



10
9
8
7
6
5
4
3
2
1
0
Number of Taxa
2
3
14
5
10
15
7
8
9
13
1
4
6
11
12
Inshore
Cages
Near cages
Offshore
Control
Sample Site

Figure 7. As Figure 2 for the number of invertebrates (n m^{-2})

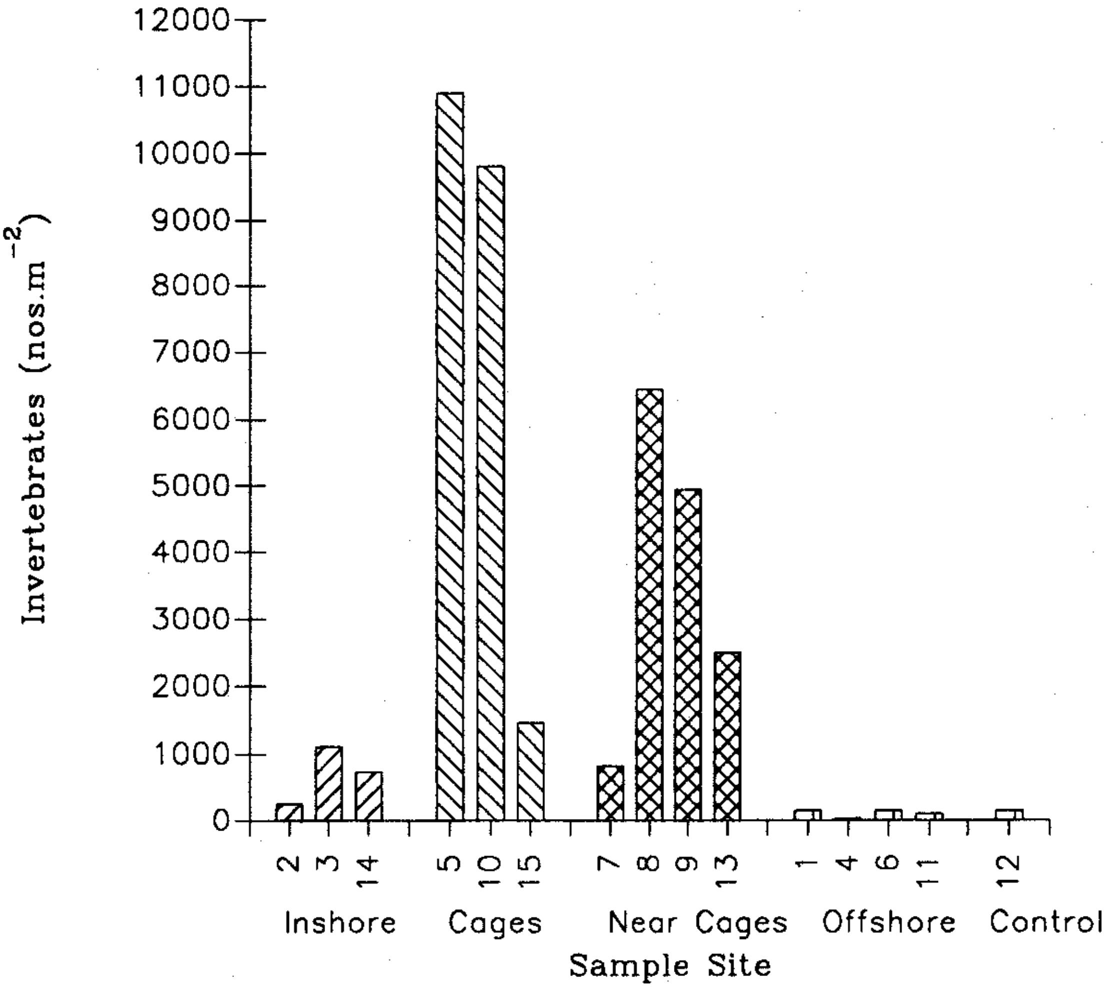

12000
11000
10000
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
Invertebrates (nos.m^{-2})
2
3
14
5
10
15
7
8
9
13
1
4
6
11
12
Inshore
Cages
Near Cages
Offshore
Control
Sample Site

Figure 8. As Figure 2 for the numbers of Oligochaeta (n m^{-2}) and the taxonomic composition

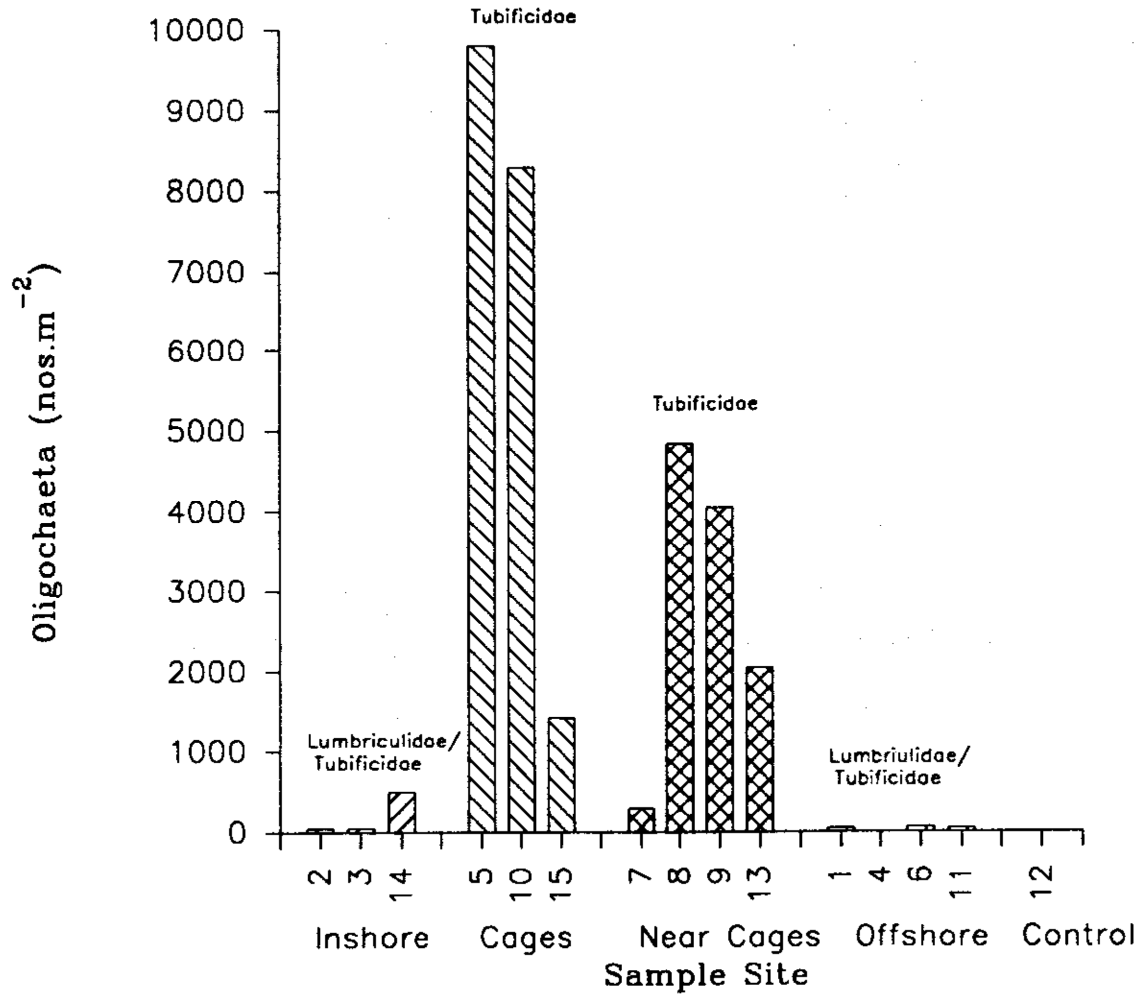

Oligochaeta (nos.m^{-2})
10000
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
Tubificidae
Tubificidae
Lumbriculidae/
Tubificidae
Lumbriulidae/
Tubificidae
2
3
14
5
10
15
7
8
9
13
1
4
6
11
12
Inshore
Cages
Near Cages
Offshore
Control
Sample Site

Figure 9. As Figure 2 for the numbers of Chironomidae (n m^{-2}) and the taxonomic composition

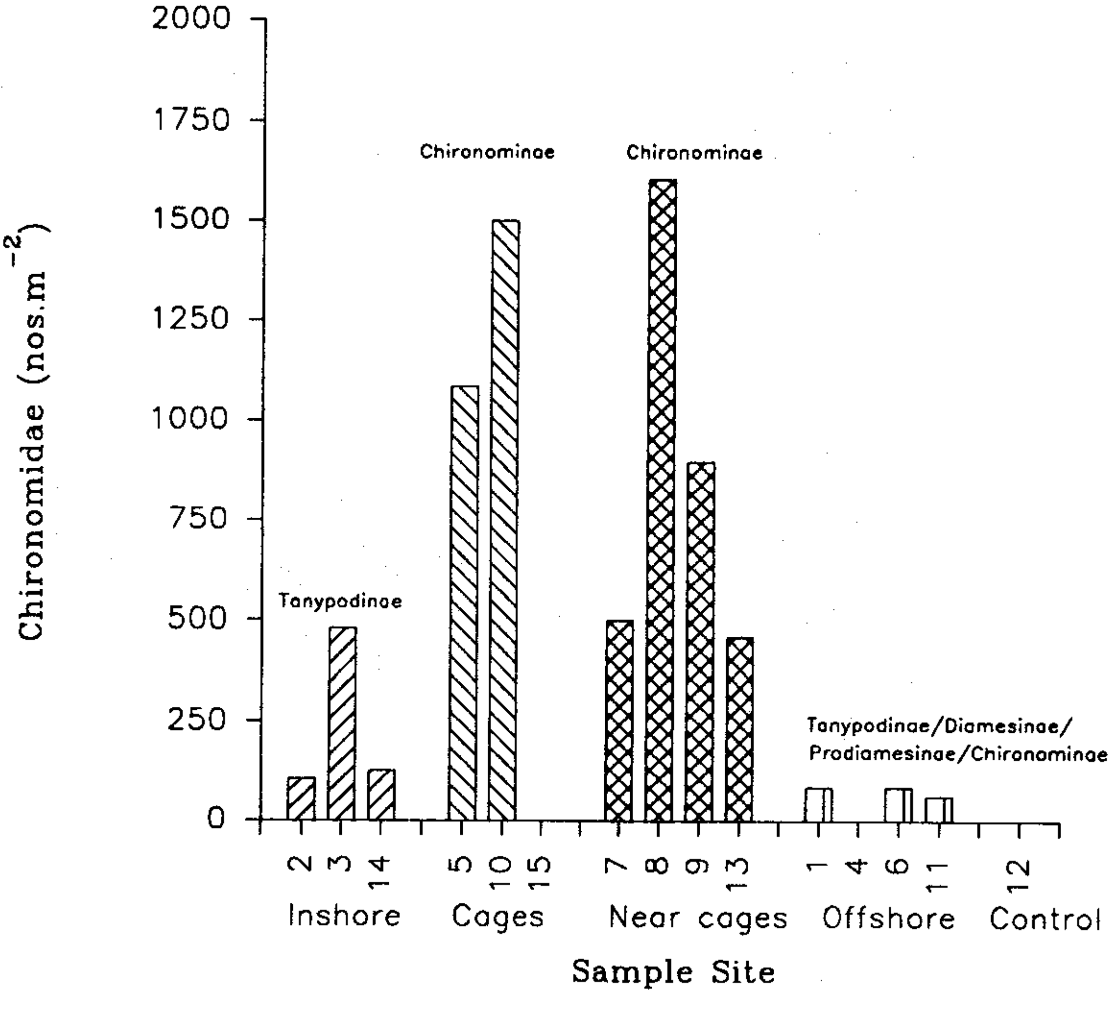

Chironomidae (nos.m^{-2})
2000
1750
1500
1250
1000
750
500
250
0
Tanypodinae
Chironominae
Chironominae
Tanypodinae/Diamesinae/
Prodiamesinae/Chironominae
2 3 14
5 10 15
7 8 9 13
1 4 6 11
12
Inshore
Cages
Near cages
Offshore
Control
Sample Site

Figure 10. Surface water concentrations of total phosphorus (μg P l^{-1}) in Loch Shiel 1986-1993 (data from the Institute of Aquaculture)

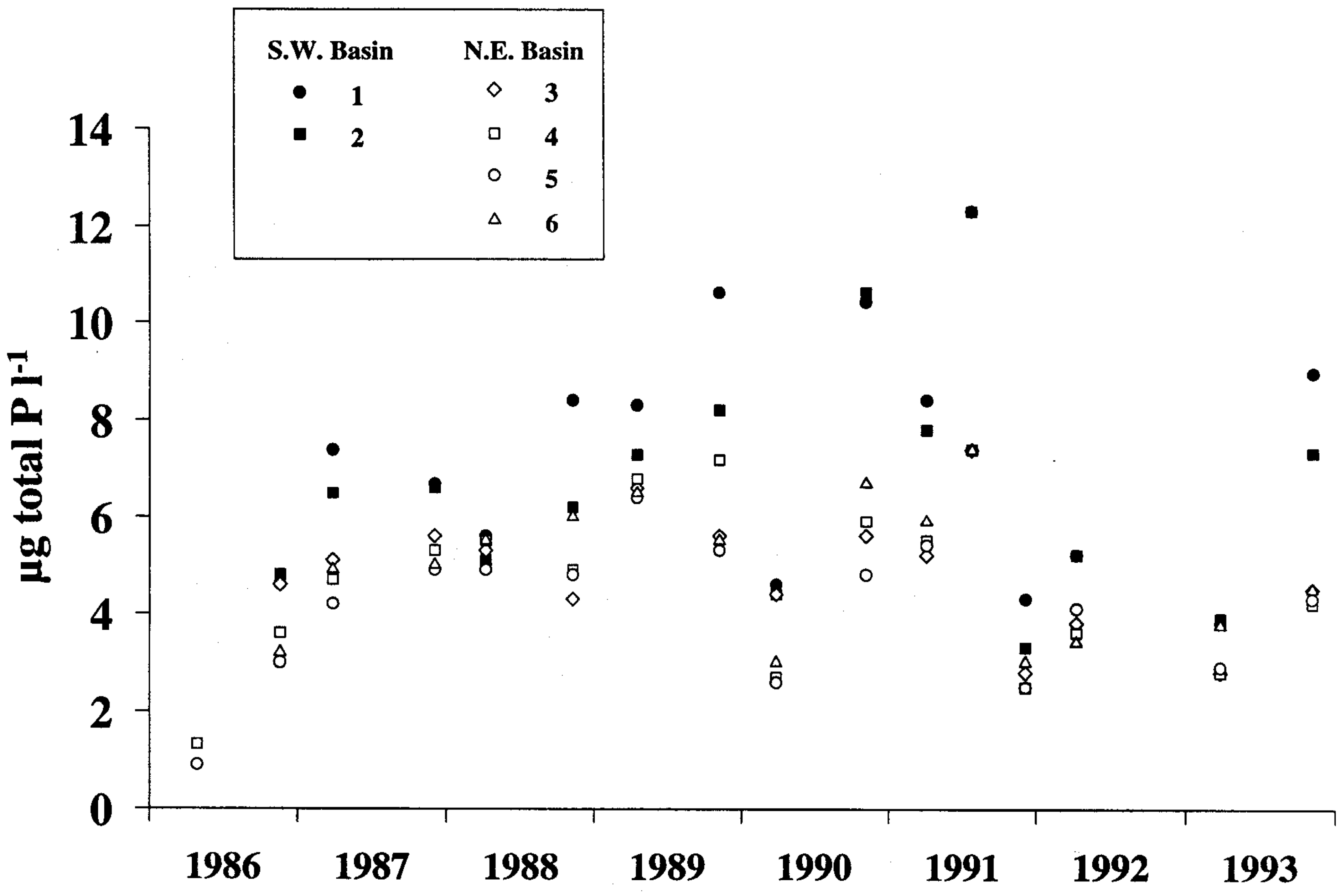

S.W. Basin
1
2
N.E. Basin
3
4
5
6
µg total P l-1
14
12
10
8
6
4
2
0
1986
1987
1988
1989
1990
1991
1992
1993

Figure 11a. Chlorophyll$_a$ levels ($\mu g\ l^{-1}$) in the south-west basin of Loch Shiel 1986-1991 (data from the Institute of Aquaculture)

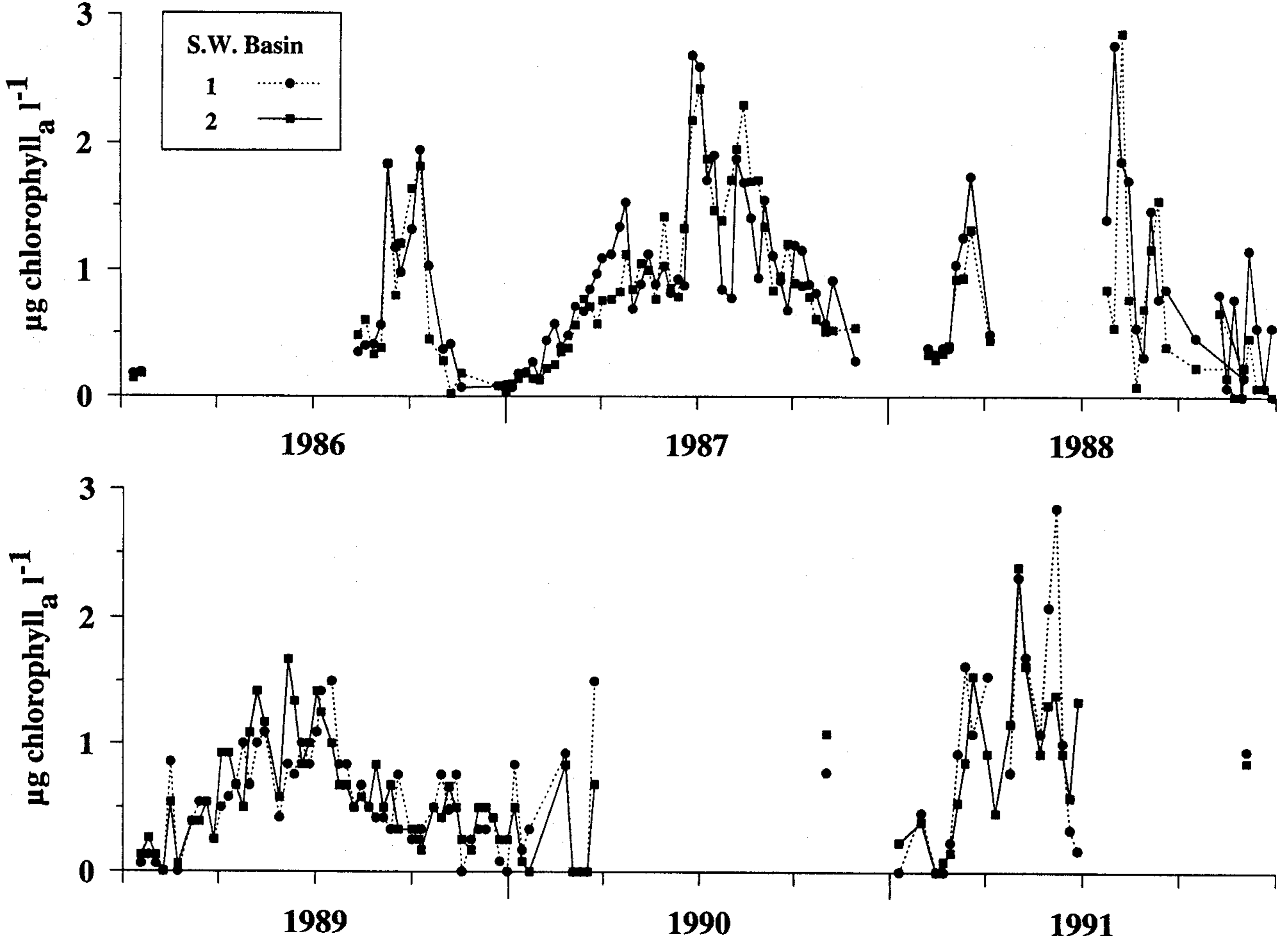
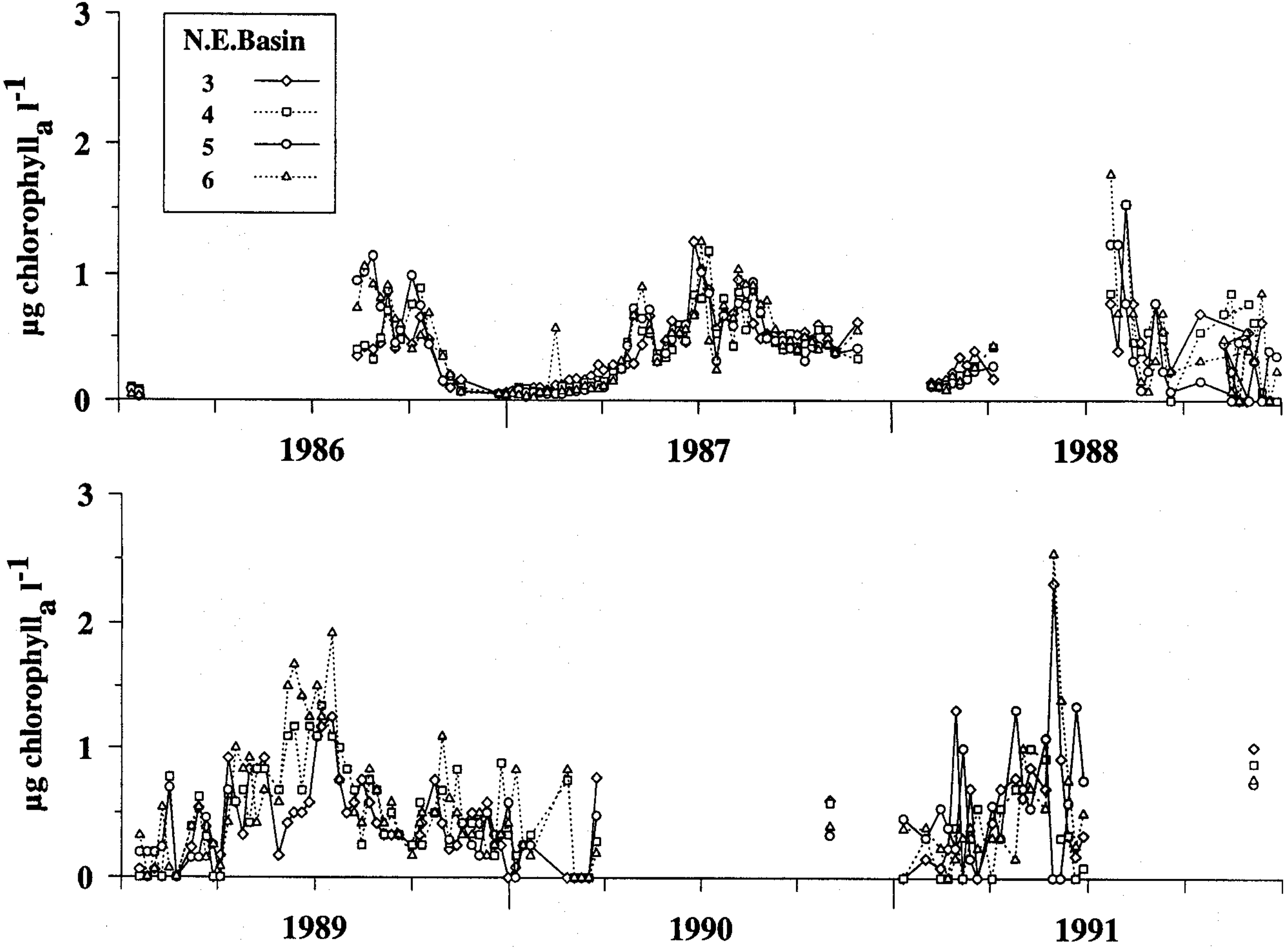

S.W. Basin
1
2
µg chlorophyll$_a$ l^{-1}
0
1
2
3
1986
1987
1988
µg chlorophyll$_a$ l^{-1}
0
1
2
3
1989
1990
1991

Figure 11b. As Figure 11a for the north-east basin of Loch Shiel


N.E.Basin
3
4
5
6
µg chlorophyll$_a$ l^{-1}
0
1
2
3
1986
1987
1988
µg chlorophyll$_a$ l^{-1}
0
1
2
3
1989
1990
1991