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Source: *Biology and Environment: Proceedings of the Royal Irish Academy*, Vol. 104B, No. 1 (May, 2004), pp. 19-42

Published by: Royal Irish Academy

Stable URL: <http://www.jstor.org/stable/20500203>

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LONG-TERM ENVIRONMENTAL MONITORING SHOWS NO IMPACT FROM SALMON CAGE FARMING IN LOUGH ALLEN, AN IRISH FRESHWATER LAKE

Mark J. Costello, Louise Collier, Jennifer Dowse and Declan Quigley

ABSTRACT

Evaluation of the long-term cumulative impacts of fish farming requires long-term data sets. In this study, we analysed water-quality monitoring data collected monthly from 1993 to 1999 and information on phytoplankton available since 1951. Seventeen parameters were measured in water at the farm and control sites: namely oxygen, nitrite, nitrate, ammonia, total phosphorus (TP), soluble reactive phosphate, chlorophyll *a*, suspended solids, pH, biochemical oxygen demand, chemical oxygen demand, alkalinity, silicate, total nitrogen (TN), conductivity, colour and temperature. Lake bed sediments and water at the cages were analysed for TP, TN and organic carbon in 1988, 1990 and 1998.

The phytoplankton indicate that there have been no changes in the oligotrophic status of the lake since 1951. There were no trends in water quality over the past fifteen years. Except for ammonia, no other significant differences in water parameters between farm and control sites were detected. Chemotherapeuticants were not detected. Several parameters were more variable in near-shore water samples than in fish cages and open-water control sites. There was seasonal variation in water parameters and a thermocline in summer. At the fish cages, ammonia was elevated above and oxygen depleted below the thermocline. These levels were not a risk to fish health and were not of environmental concern, nor were they close to permitted levels. In sediments there were no significant correlations between TP and TN and none between lake depth or distance from the cages. Overall, this study shows that salmon culture can be conducted in a freshwater lake for over thirteen years without polluting either the water or lake bed.

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Received 5 April 2001.
Read 25 September
2003. Published 31 May
2004.

INTRODUCTION

Freshwater aquaculture has increased about twenty-fold in Ireland since 1970 (Dempsey *et al.* 2000), creating the opportunity for significant environmental impacts (Institute of Aquaculture *et al.* 1990). A study of farms based along rivers in Ireland found two (11%) were polluting (Costello *et al.* 1995). A parallel study in Ireland by Massik and Costello (1995) did not support the suggestion by Carr and Goulder (1990a) that the release of alkaline phosphatase from farms may increase phosphorus availability and thus promote algal blooms. However, there remains a risk that the culture of fish in cages in lakes may cause nutrient enrichment, with consequent impacts on water quality and the lake bed. This risk may be expected to increase with time if wastes accumulate at the farm site. Assessment of environmental impact therefore requires long-term monitoring. However, no such studies have been published for

salmon farms in Ireland or elsewhere. Indeed, there are no published time-series data on lake water quality and nutrients or on sediment nutrients in Ireland (Irvine *et al.* 2001).

Salmon farming began in Lough Allen in 1987. During the early years after the farm was established, two reports assessed the trophic status and potential impact of waste nutrients from the fish farm on Lough Allen (Champ 1988; Gavine 1992). Both concluded that the farm had no impact on the lake then and that an increase in stock to one million Atlantic salmon smolts would not have an adverse impact on the lake. Gillooly (1989), Maloney (1991) and Maloney *et al.* (1991) could detect effects on water quality within the farm cages and on sediment quality under one fish cage unit. However, they also found no change in lake water quality and no significant environmental impact by the farm. Since 1997, the farm's licensed capacity has been realised, with four additional fish cage units. The present paper analyses the water

and sediment quality data collected as part of the farm monitoring programme to determine if the farm has had any impact on water or sediment quality. This study rejects the hypothesis that long-term fish culture will have a negative impact on freshwater lakes and contributes to the debate about the sustainability of freshwater cage culture.

The present study site, Lough Allen, was omitted from two major studies on lake ecology in Ireland (Irvine *et al.* 2001; McCarthy *et al.* 2001). However, as a result of the farm-monitoring programme, there is more water and sediment quality data available for Lough Allen than any other lake in Ireland. Therefore, another important use for the present data is for comparison with lakes in Ireland and other countries.

STUDY AREA

Lough Allen is the eighth largest lake in Ireland (3500ha, 13km long) (Murray 1996) and is situated in the upper part of the River Shannon catchment. Several studies on lakes in Ireland indicate that Lough Allen is oligotrophic, with a high 'colour' caused by peat staining of the water (Water Pollution Advisory Council 1983; Toner *et al.* 1986; Champ 1988; Environmental Research Unit 1993; Bowman 1998a; 1998b; Environmental Protection Agency 1998; Lucey *et al.* 1999). The underlying geology is shale and sandstone (Charlesworth 1963), and there is little mineral input from the streams entering the lake (Bowman 1998b). The open water phytoplankton of the lake confirms its oligotrophic status (Bowman 1998a; 1998b). The southernmost part of the lake is slightly enriched by sewage from Drumshambo sewage treatment works (Bowman 1998b). Excluding this area, maximum summer chlorophyll levels recorded in the past are 3.0mg m⁻³ in 1983 (Water Pollution Advisory Council 1983), 4.4mg m⁻³ in 1985 (Bowman 1998a; 1998b), 7.0mg m⁻³ (Environmental Research Unit 1993), 6.1mg m⁻³ in 1995, 5.9mg m⁻³ in 1996 and 6.2mg m⁻³ in 1997 (Bowman 1998a; 1998b; Lucey *et al.* 1999), all classifying the lake as oligotrophic (i.e. <8mg m⁻³).

METHODS

Samples were taken from the water surface and at several depths at the cages and at an open water control site from a boat. Additional surface water samples were taken from the shore at Corry Strand, Fahy Shore, Cormongan Shore and from a bridge at the lake outlet following standard methods (American Public Health Association 1985).

WATER

All water samples were collected by Glan-Uisce Teo and brought to the laboratory within six hours of collection (Mary Hensey pers. comm.). Below-surface samples were collected using an 'IOS' water sampling bottle. Water samples were analysed by Glan-Uisce Teo, by the Central Marine Services Unit (CMSU) of the National University of Ireland or by Neptune Labs, Galway, following standard methods (Strickland and Parsons 1972; American Public Health Association 1985) (Table 1). Temperature was measured in the lake, and samples for oxygen analysis were fixed in the field. Samples for ammonia, oxidised nitrogen and soluble reactive phosphorus were filtered on 0.45µm glass fibre cartridge prior to analysis. Methods of analysis have not changed since 1993 (Mary Hensey, pers. comm.). Glan-Uisce Teo and CMSU are accredited and participate in intercalibration exercises for some of these analyses under the Irish Accreditation of Laboratories Scheme (I.S. EN 45001, reg. no. 0047). Glan-Uisce Teo conduct intercalibrations for biochemical oxygen demand (BOD), chemical oxygen demand (COD), soluble reactive phosphorus, suspended solids and total ammonia nitrogen, and CMSU for nitrate, nitrite, total nitrogen and total phosphorus.

Water was analysed for the chemotherapeuticants malachite green, formaldehyde, chloramine-T, oxytetracycline, oxolinic acid and sulphadiazine. This sampling was conducted on the following dates: on 28 June, 26 July, 31 August, 28 September, 27 October, 23 November and 20 December in 1994; on 30 January, 28 February, 29 March, 26 April, 25 May, 28 June, 26 November, 24 August, 21 September, 19 October, 14 July and 7 December in 1995; and on 24 January, 22 February, 21 March and 16 April in 1996. Thus, 138 samples of surface water were analysed for six chemotherapeuticants on 23 days.

SEDIMENT

The same methods for sediment sampling were used in the field and laboratory by Gillooly (1989) and Maloney (1991) and in the present study. Bathymetry ranged from about 10m to 30m within the study area (Fig. 1). Samples were collected by an Ekman grab, and the top 2cm of the sample was collected in a small container. The sample was oven-dried at 100°C and then digested in sulphuric acid with sodium and copper sulphate. This released the nitrogen and phosphorus into solution, and both nutrients were analysed in a Technicon II autoanalyser. Results are expressed per dry weight of sediment.

Gillooly (1989) and Maloney (1991) also estimated the organic matter content by 'loss on ignition' (LOI), namely by placing the dried sample in a furnace at 550°C and determining the loss in weight. The sediment sampling by Aqua-Fact (1994) also used a grab, but organic matter was estimated by wet acid dichromate oxidation. The Aqua-Fact (1994) study provided data on sediment granulometry, which was determined by laser analysis.

In the present study, sediment analyses were conducted by the Aquatic Services Unit, National University of Ireland, Cork. They included three

samples of reference material obtained from the European Commission in their analysis. For total phosphorus, values of 2.30mg P g⁻¹, 2.33mg P g⁻¹ and 2.34mg P g⁻¹ were obtained for a certified value of 2.36mg P g⁻¹. For total nitrogen, 39.4mg N g⁻¹, 40.7mg N g⁻¹ and 40.8mg N g⁻¹ were obtained for a certified value of 34.2mg N g⁻¹. This indicates an underestimate of 3% for total phosphorus and an overestimate of up to 16% for total nitrogen. Analyses by Gillooly (1989) and Maloney (1991) were conducted under supervision in the Environmental Sciences Unit, Trinity College, Dublin.

Table 1—The water quality parameters analysed as part of the Lough Allen fish farm monitoring programme reported by Glan-Uisce Teo. Abbreviation, units and analysis methods are listed.

Parameter	Abbreviation	Units	Laboratory performing analysis	Method of analysis (see APHA 1985)
Temperature	Temp.	°C	Glan-Uisce	Mercury thermometer
Conductivity	Cond.	µS	CMSU	Electrometric
Suspended solids	SS	mg l ⁻¹	Glan-Uisce	Gravimetric (dried at 104°C)
pH	pH	-log [H]	Glan-Uisce, CMSU	Electrometric at 25°C
Alkalinity	Alk.	mg CaCO ₃ l ⁻¹	CMSU	Acid base titration using 0.1 M HCl and methyl red and bromocresol green indicators
Colour	—	OD 100mm	CMSU	Spectrophotometric measurement at integrated UV absorption at 250–350nm
Dissolved oxygen	DO	mg O ₂ l ⁻¹	Glan-Uisce	Winkler
Biochemical oxygen demand	BOD	mg O ₂ l ⁻¹	Glan-Uisce	DO after 5 days incubation at 20°C
Chemical oxygen demand	COD	mg O ₂ l ⁻¹	CMSU	Dichromate reflux (2 hr, mercuric chloride added to eliminate halide interference)
Chlorophyll <i>a</i>	Chl- <i>a</i>	mg l ⁻¹	Glan-Uisce	Acetone extraction, spectrophotometric at absorbance of 665nm
Soluble reactive phosphorus	SRP (Ortho-P)	mg PO ₄ -P l ⁻¹	Glan-Uisce	Molybdate reactive after filtration
Total phosphorus	TP	mg PO ₄ -P l ⁻¹	CMSU	Photo-oxidation ¹ to SRP
Total ammonia nitrogen	TAN	mg NH ₄ -N l ⁻¹	Glan-Uisce	Phenate method
Nitrite	NO ₂	mg NO ₂ -N l ⁻¹	CMSU	Azo dye photometric
Nitrate	NO ₃	mg NO ₃ -N l ⁻¹	CMSU	Cadmium reduction to NO ₂
Total oxidised nitrogen	TON	mg NO _x -N l ⁻¹	CMSU	Cadmium reduction to NO ₂
Total nitrogen	TN	mg N l ⁻¹	CMSU	Photo-oxidation ¹ to NO ₂
Silicate	Si	mg Si l ⁻¹	CMSU	Molybdate reactive after filtration
Malachite Green	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography
Formaldehyde	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography
Chloramine-T	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography
Oxytetracycline	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography
Oxolinic acid	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography
Sulphadiazine	—	mg l ⁻¹	Neptune Labs	High pressure liquid chromatography

CMSU = Central Marine Services Unit, National University of Ireland, Galway.

¹ Strickland and Parsons (1972).

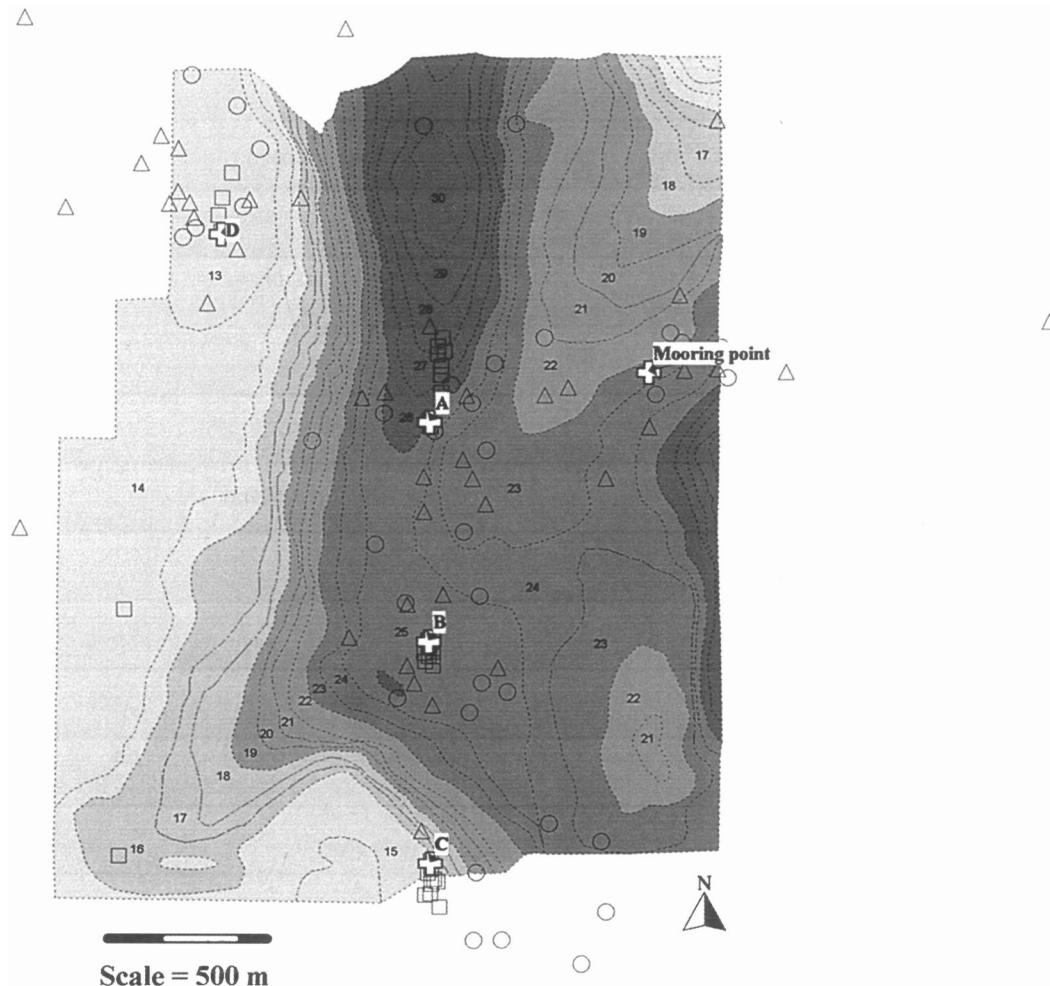


Fig. 1—Plot of the distribution of sediment sampling stations in 1989 (\triangle), 1991 (\circ) and 1998 (\square) in relation to bathymetry (contours numbered in metres) and cage moorings (crosses).

DATA ANALYSES

The distribution of sediment sampling stations in Gillooly (1989), Maloney (1991) and the present study has been mapped in a geographical information system (MapInfo). The previous surveys used a sextant and measured rope to locate their sampling stations and then mapped their sampling stations to scale. Their maps and the bathymetric charts provided by the farm were scanned and imported into the GIS. During the present survey a hand-held Geographical Positioning System (GPS) unit was used to provide a national grid reference for the pivot point of each cage unit and other sampling stations. This allowed all sampling stations to be given a grid reference to be overlaid on the bathymetry and enabled the results to be illustrated on a map.

The basic statistics of minimum, mean (average), median (middle value) and maximum

are available in tables that can be downloaded from the website for Ecological Consultancy Services Ltd (2003). Variability in the data is indicated by the coefficient of variation [standard deviation \times (100/mean)] (Elliott 1983). Statistical significance of comparisons between farm and control sites was tested by one-way analysis of variation (ANOVA) using Minitab version 3, and correlations were tested using the Pearson product moment correlation coefficient (r) using Microsoft Excel version 7.

RESULTS

WATER

For most parameters, there were no environmentally significant differences (ANOVA, $P > 0.05$) in water quality parameters between the farm

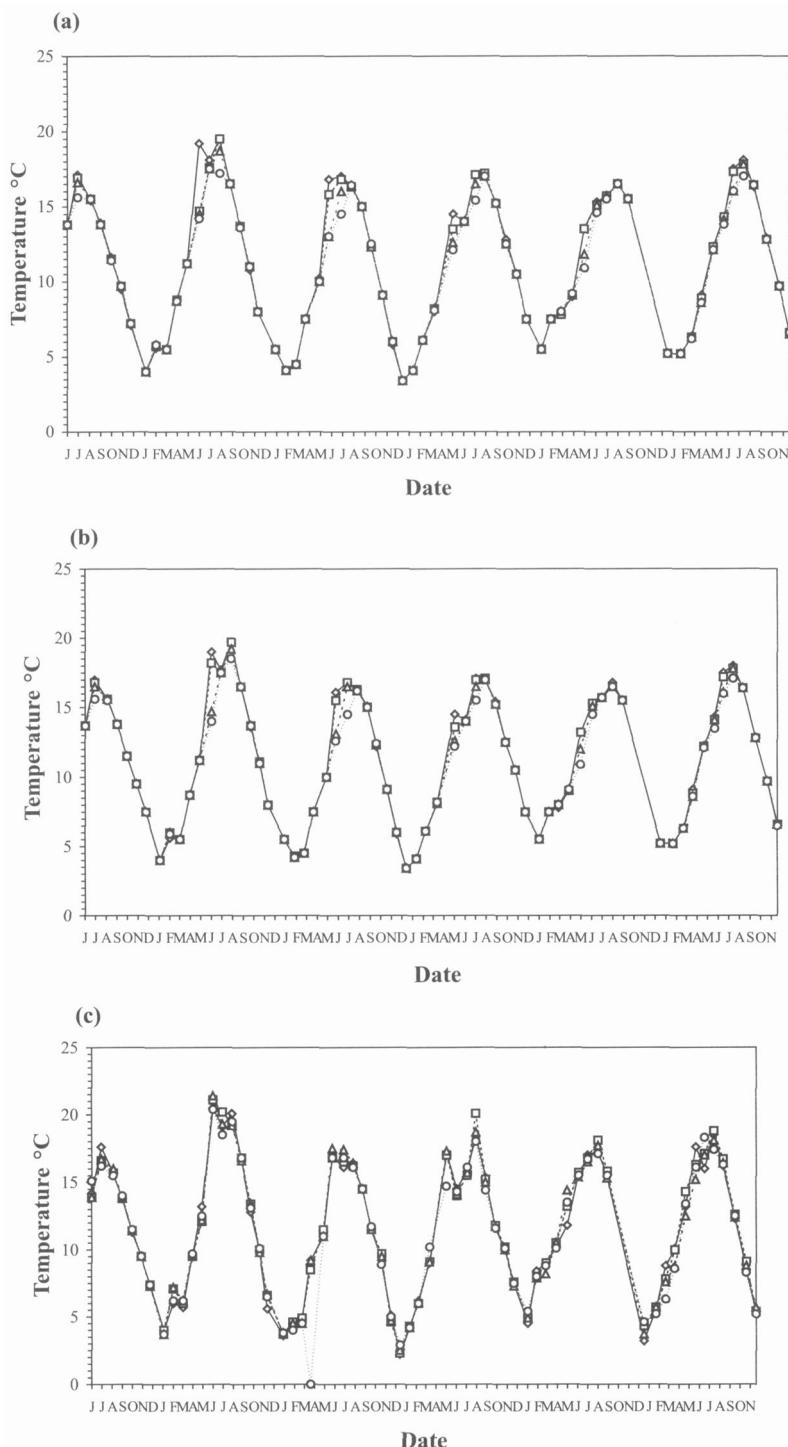


Fig. 2—Changes in water temperature (°C) from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. In (a) and (b), data at 0m deep is indicated by a diamond, data at 6m deep by a square, data at 12m deep by a triangle and data at 18m deep by a circle. In (c) data from Corry Strand is indicated by a diamond, data from Fahy Shore by a square, data from Cormongan by a triangle and data from the lake outlet by a circle.

cages and other control sites (Figs 2–13 and Table 2). Ammonia tended to be higher at the farm cages than control sites (Fig. 6) and was significantly higher ($P = 0.004$) at the cages at a depth of 0m. In contrast, suspended solids, BOD, COD and pH were very variable at the shore sites and tended to be higher there than at both the cages and at the open water control sites (ANOVA, $P < 0.05$) (Figs 10–13). Total phosphorus was significantly higher at the shore sites compared to the open water control site (Fig. 3 and Table 2), probably reflecting phosphorus bound to suspended solids.

All parameters, except oxygen, nitrite, soluble and total phosphorus, and silicate were more variable and usually had higher minima and higher maxima at the shore sites than at the farm cages and open water control site (Table 3), and variation was significant ($P = 0.045$) for COD at shore sites.

On certain occasions, some parameters had unusually high or low levels at all sampling sites: nitrite (Fig. 4) (July 1998), nitrate (Fig. 5) (November 1997), BOD (Fig. 11) (January 1996) and ammonia (Fig. 6) (January 1999). pH results were low and variable from June to September 1993, but far less variable thereafter, being generally between 6.8 to 7.3 (Fig. 13). Phosphorus (SRP and TP) was unusually high at both cages and open water control sites in January 1995 (Figs 7 and 8). No environmental significance is attached to these anomalous results, which probably indicate sample contamination or analytical error.

Water temperature, chlorophyll *a* and pH were lower and oxygen concentration was higher during the winter months (Figs 2, 3, 9 and 13). Although BOD was usually close to its detection limit of $2\text{mg O}_2 \text{ l}^{-1}$, elevated levels were most frequent during the winter (Fig. 11).

Only ammonia differed significantly with water depth at the cage site (Fig. 6 and Table 2). Water temperature, oxygen concentration, chlorophyll, and phosphorus (SRP and TP) also varied with water depth, but by such a small amount that it was not significant. The results indicated a seasonal temperature gradient of up to 4°C between the bottom and surface of the lake from May to August (Fig. 2). The highest levels of ammonia occurred at the lake surface or at 6m depth on most sampling days (Fig. 6). Dissolved oxygen concentration was lower in deeper water in August 1995 and August 1996 (Fig. 3). Both soluble reactive and total phosphorus tended to be higher in surface samples than in deep-water samples (Figs 7 and 8). Chlorophyll was measured at different depths in 1993–4 and was consistently lower in the deeper water.

The chemicals malachite green, formaldehyde, chloramine-T, oxytetracycline, oxolinic acid and

sulphadiazine were not detectable in the surface water at levels of 1mg l^{-1} at any sites on any of the dates sampled.

The discharge licence from Leitrim County Council stipulated permitted levels for eleven parameters. Since 1993, samples were analysed for 23 chemotherapeutics and 849 parameters were measured; a total of 6076 sample analyses were performed. Five samples ($<0.08\%$) at the cage sites exceeded discharge limits, and twelve samples at control sites exceeded these levels (Table 4). Of these, it has earlier been noted that pH readings in 1994 were unusually variable. The single value of phosphorus exceeding the limits was three times higher than the next highest value and cannot be considered representative of phosphorus levels. It may reflect sample contamination by organic matter.

SEDIMENTS

The available sediment quality data comprises 42 samples from 1988 (Gillooly 1989), 43 from 1990 (Maloney 1991), 4 from 1994 (Aqua-Fact 1994), and 30 from 1998 (present study). Additional samples were not analysed in the present study because strong water currents meant that only 30 of about 50 attempts to take a grab sample were successful. In this study, but not previous surveys, sampling stations were concentrated within the areas covered by the rotation of the cages so as to increase the likelihood of detecting elevated levels. The stations extended to 390m (Gillooly 1989), over 500m (Maloney 1991) and 260m (present study) from the cages and are thus considered to include samples reflecting background conditions (Fig. 1).

All sediment samples taken in 1998 had a fine surface layer of pale brown silt (about 1mm) overlying a grey-black mud. Previous granulometric analysis has determined that the sediments are almost entirely mud (Aqua-Fact 1994). The mud typically had a faint odour of hydrogen sulphide, as is typical of fine unaerated muddy sediments. From the mud colour and odour, it is considered that the mud surface was oxygenated and deeper mud was hypoxic. No invertebrates or signs of invertebrates (e.g. shells, tubes) were visible in the samples in this and previous surveys.

Considering the entire data sets, differences between years in minimum, mean and median values for total phosphorus and nitrogen were within analytical error. The minimum values were marginally lower in 1998 than previous years, but maximum values were much higher. These high values occurred under cage unit B (samples 11–13). Another sample exceeding previous years' maxima occurred under cage unit C (sample 18).

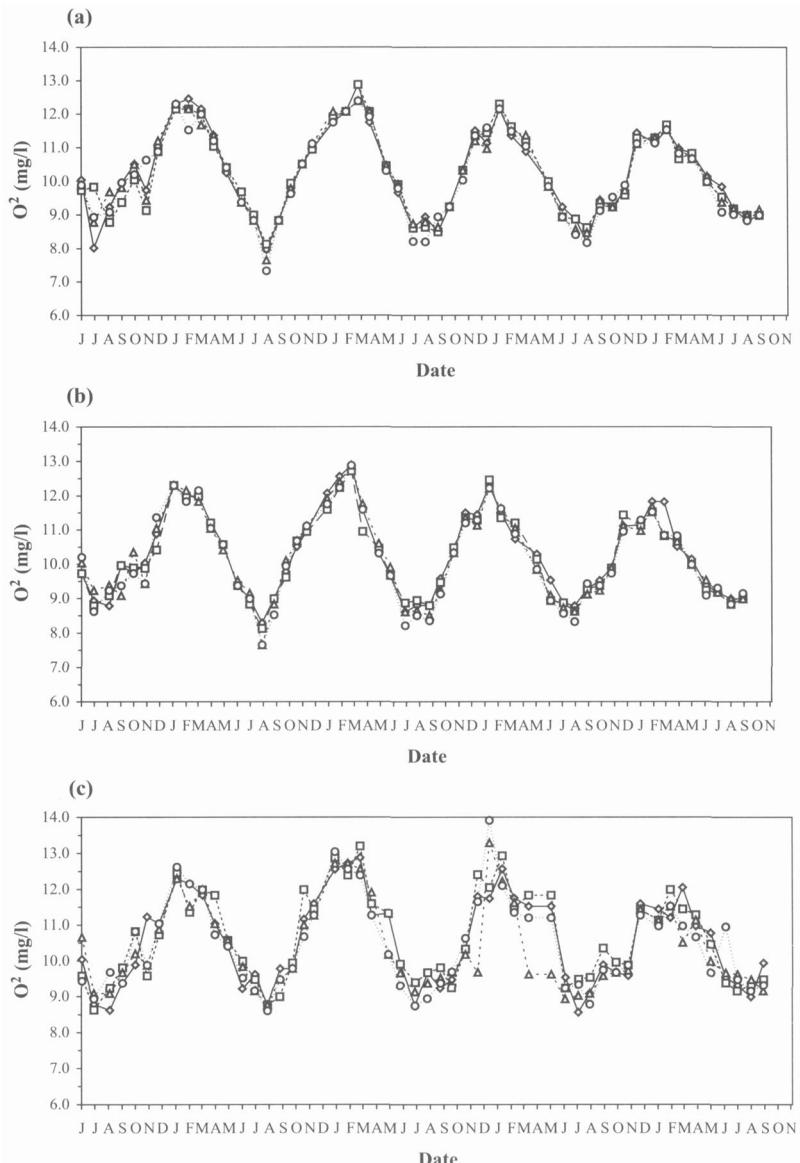


Fig. 3—Changes in oxygen concentration (mg l^{-1}) from 28 June 1994 to 24 September 1998 at (a) cages, (b) control sites and (c) shore sites. Symbols as in Fig. 2.

The 1998 data was more variable than previous surveys: seven times more for phosphorus, and three times more for nitrogen (Table 5).

Organic matter levels were not estimated in 1998 because surveys in 1988 and 1990 found them to be a less sensitive indicator of fish farm impacts compared to phosphorus. The four samples analysed in 1994 produced much lower estimates of organic matter than both previous studies (Figs 14–16). These differences between studies are attributed to the different methods used to estimate organic matter.

LONG-TERM IMPACT OF FRESHWATER SALMON CAGES

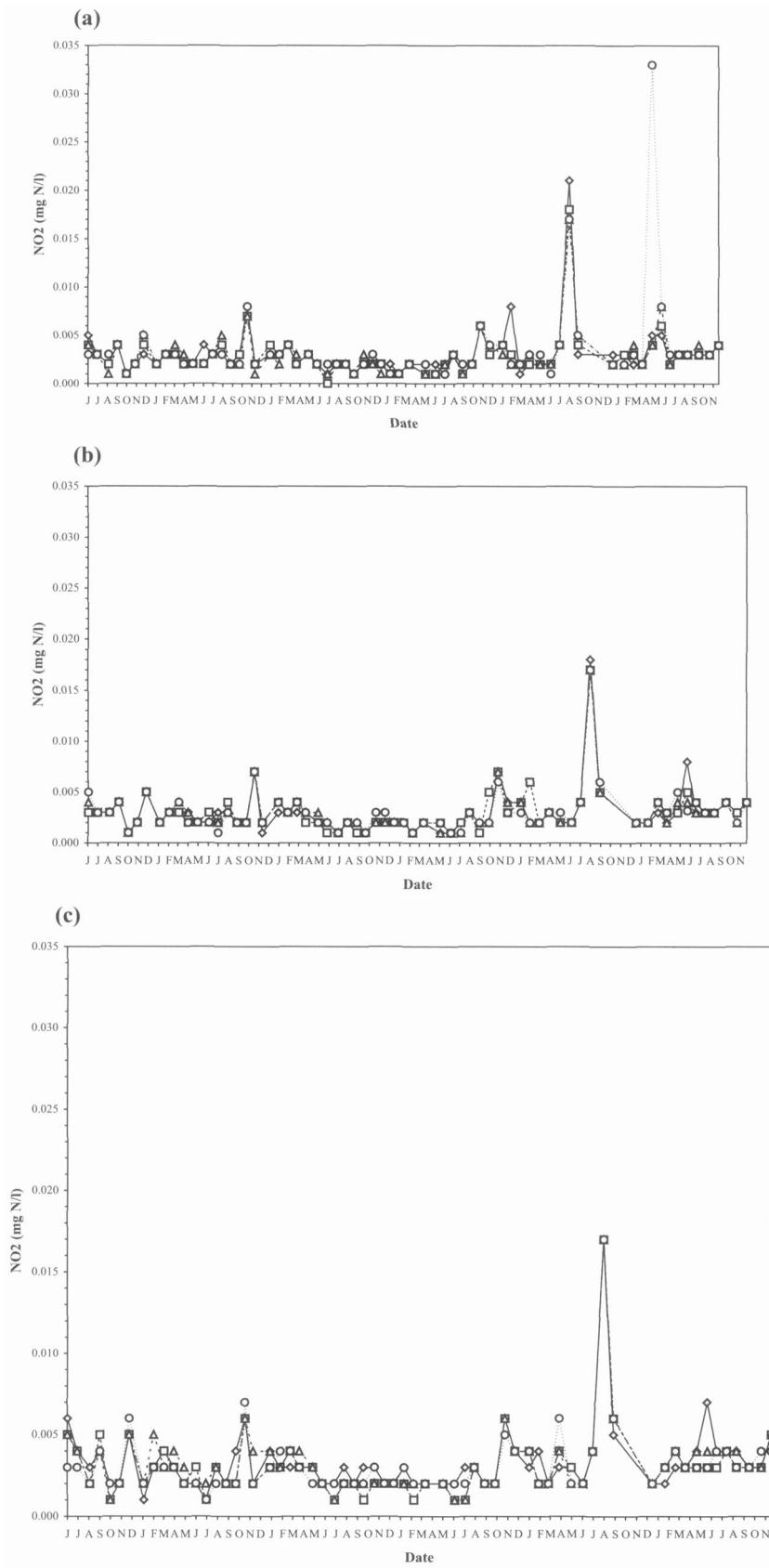


Fig. 4—Changes in nitrite (NO_2 , mg N l^{-1}) from 28 June 1994 to 15 December 1999 at (a) cages, (b) control sites and (c) shore sites. Symbols as in Fig. 2.

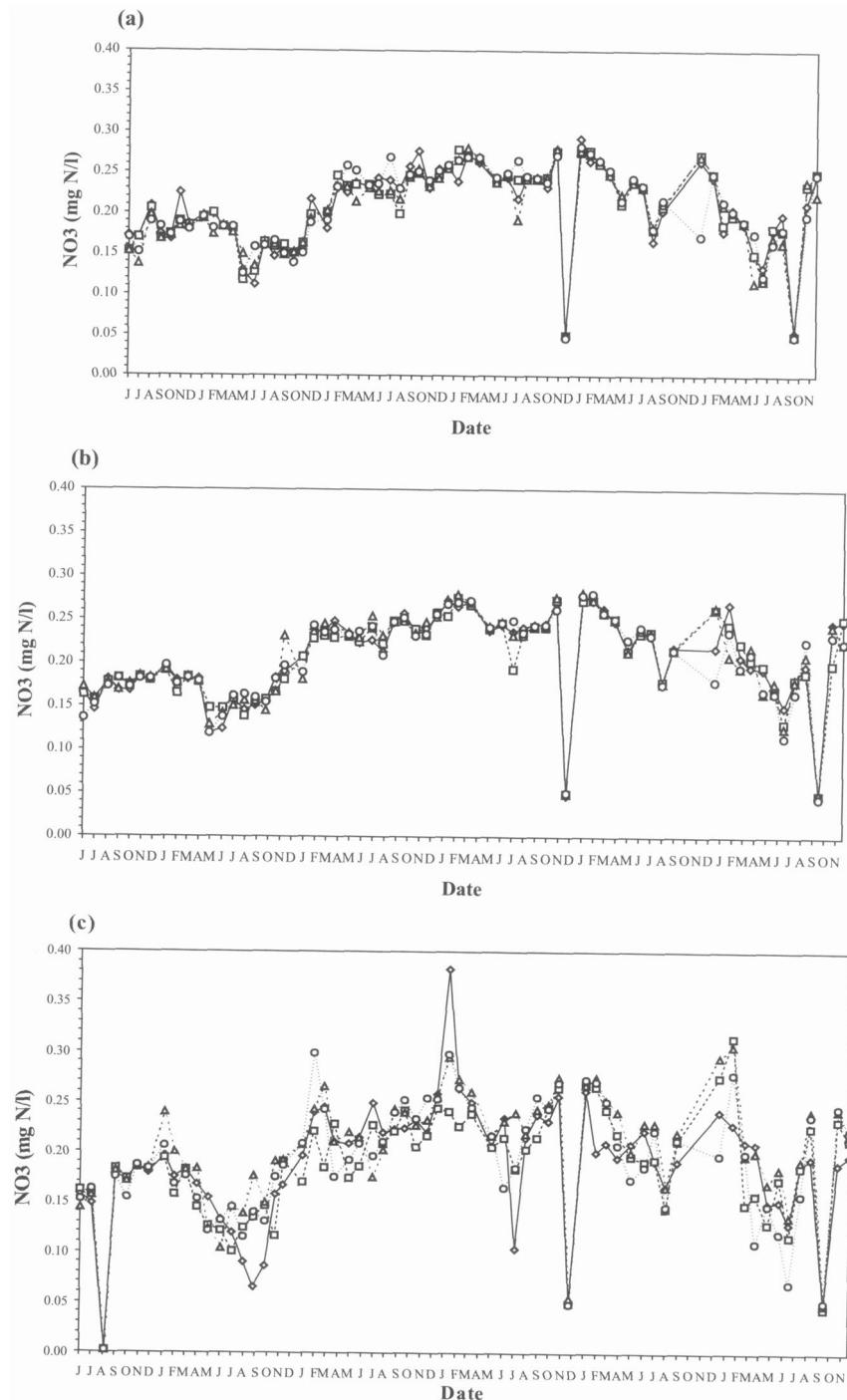


Fig. 5—Changes in nitrate ($\text{NO}_3 \text{ mg N l}^{-1}$) from 28 June 1994 to 15 December 1999 at (a) cages, (b) control sites and (c) shore sites. Symbols as in Fig. 2.

There was no significant correlation between total phosphorus or nitrogen and lake depth or distance from the cages in the 1998 sediment data. In 1990, total phosphorus significantly decreased with

distance from the cages (Table 6). Total phosphorus concentrations were significantly correlated with total nitrogen in 1998 but not in previous surveys.

LONG-TERM IMPACT OF FRESHWATER SALMON CAGES

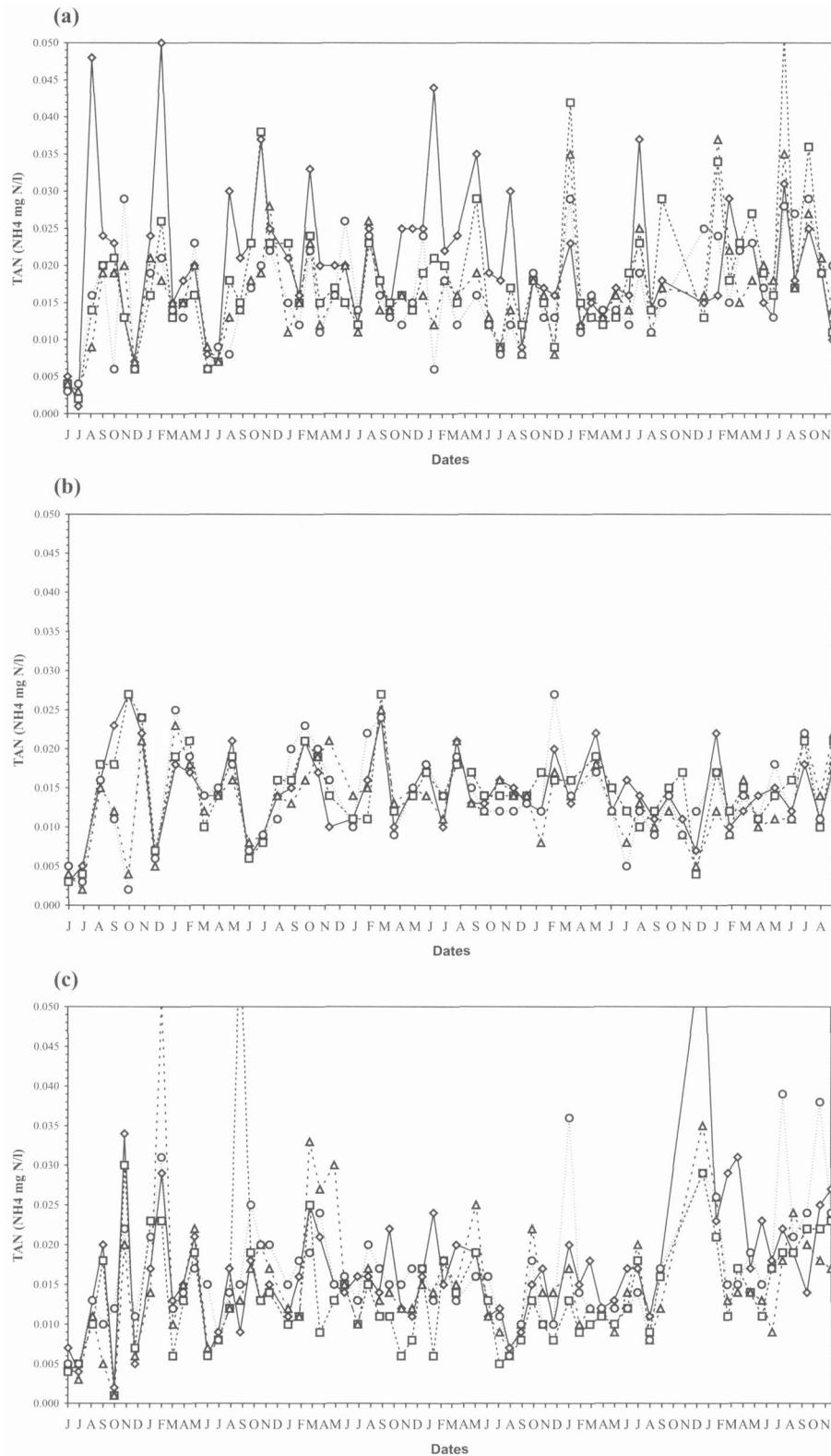


Fig. 6—Changes in ammonia ($\text{NH}_4 \text{ mg N l}^{-1}$) concentration from 28 June 1994 to 15 December 1999 at (a) cages, (b) control sites and (c) shore sites. The values for Cormongan shore on 28 February 1995, Fahy shore on 21 September 1995 and Corry Strand on 12 January 1999 are $0.051 \text{ mg N l}^{-1}$, $0.060 \text{ mg N l}^{-1}$ and $0.059 \text{ mg N l}^{-1}$ respectively and are not shown on the graph for clarity of presentation. Symbols as in Fig. 2.

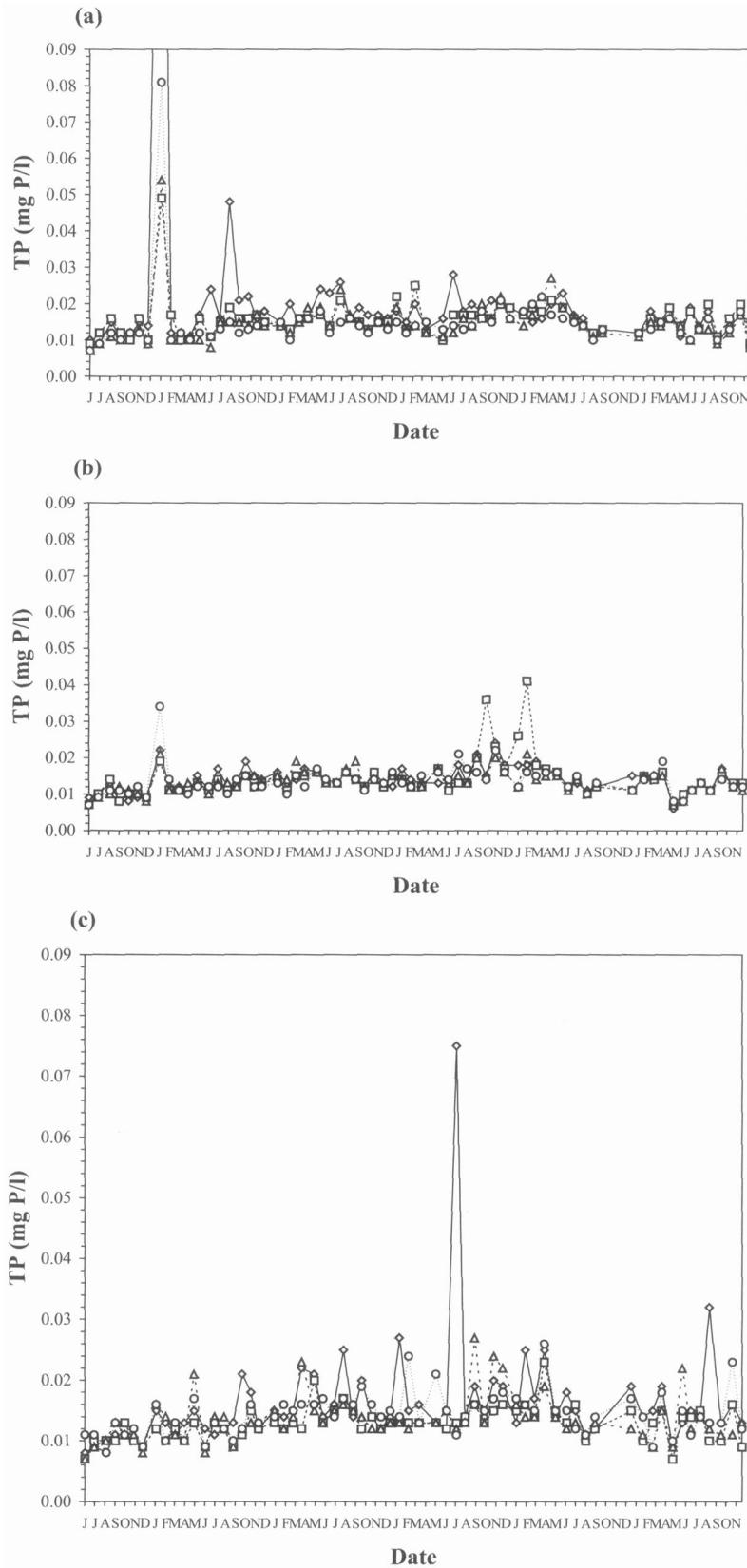


Fig. 7—Changes in total phosphorus (mg P l^{-1}) from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. The value for the cage at a depth of 0m on 30 January 1995 is $0.238 \text{ mg P l}^{-1}$: this is not shown on the graph. Symbols as in Fig. 2.

LONG-TERM IMPACT OF FRESHWATER SALMON CAGES

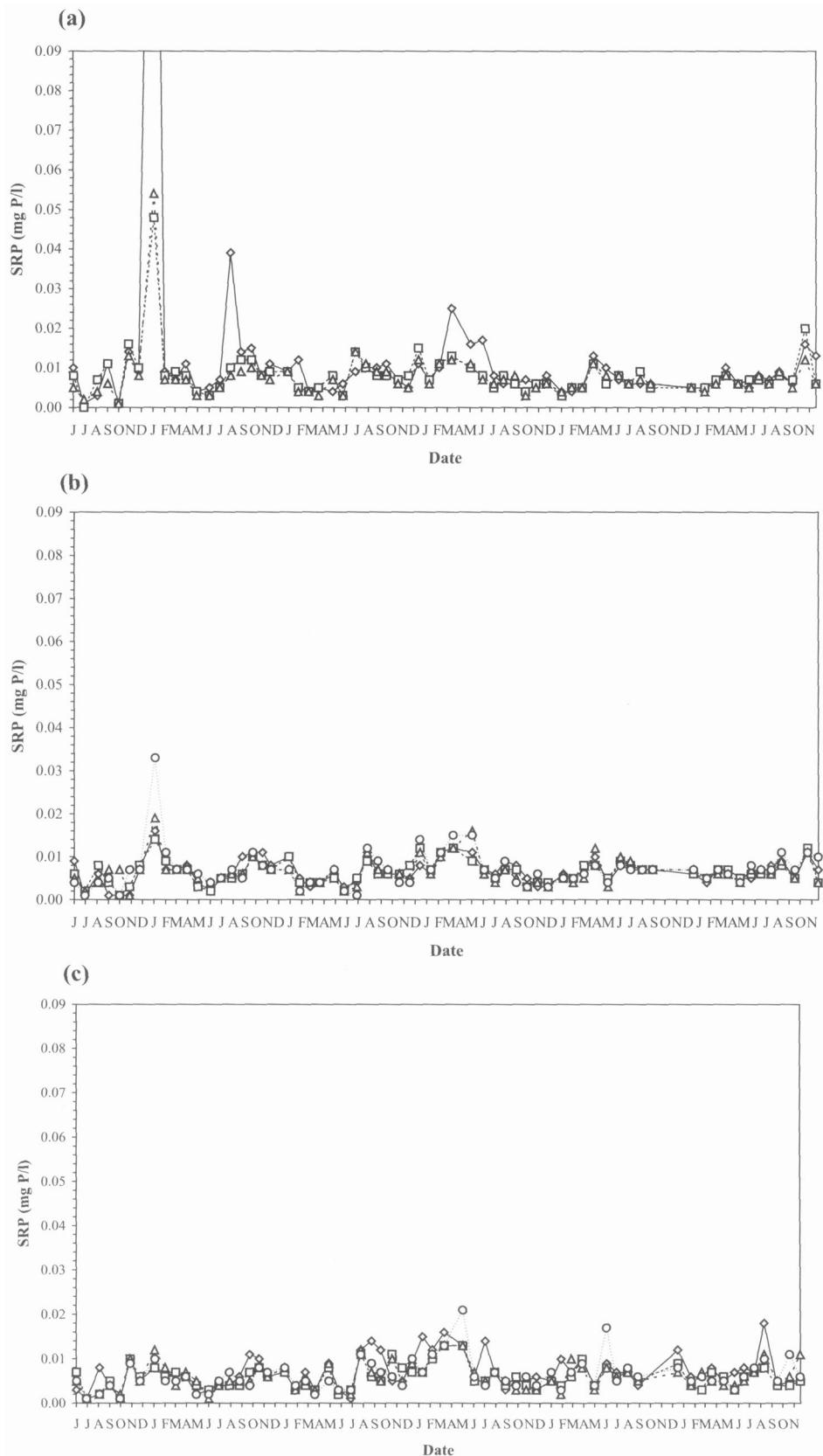


Fig. 8—Changes in soluble reactive phosphate (mg P l^{-1}) from 28 June 1994 to 15 December 1999 at (a) cages, (b) control sites and (c) shore sites. The value for the cage at a depth of 0m on 30 January 1995 is $0.238 \text{ mg P l}^{-1}$: this is not shown on the graph. Symbols as in Fig. 2.

DISCUSSION

WATER QUALITY

Differences between the farm cage sampling site and the open water control site were limited to occasionally elevated levels of ammonia. These levels were neither of environmental concern nor close to permitted levels. The rare occurrences of water quality values exceeding permitted levels were generally not unique to the farm site but also occurred at control sites. The lack of significant differences between fish cage sites and controls is typical of cage farming in freshwater (e.g. Cornell and Whoriskey 1993) and marine (Gowen 1990; author's unpublished data) situations. This has led to increased emphasis on monitoring of sediment conditions at fish cages and less emphasis on water quality (e.g. Scottish Environmental Protection Agency 1998). Indeed, the monitoring data demonstrate that future sampling of chemotherapeuticants, COD, BOD, soluble reactive phosphorus, ammonia, pH, suspended solids, nitrite and nitrate is not necessary. However, in future monitoring monthly sampling of chlorophyll, total phosphorus, soluble reactive phosphorus and total oxidised nitrogen monthly in surface waters should be continued. Should these values change significantly, monitoring should be reviewed and extended if necessary.

The shore control sites at Corry Strand, Fahy Shore, Cormongan and the lake outlet were not comparable with the farm site. Most parameters were more variable at these sites than either the cage or open water control site. This is probably a consequence of the shore sites being shallower and thus more affected by wind disturbance of bottom sediments and less buffered against changes in air temperature. Monitoring of previous data revealed a marked increase in open water total oxidised nitrogen after heavy rain (Champ 1988). Although the lake outlet is not comparable with the farm site, it may provide a useful indication of the nutrient status of the lake as a whole. In future, monitoring should continue at the cage, open water and lake outlet sites but it is not necessary at the shore sites.

Variations in water quality with depth reflected natural conditions and fish activity at the farm cages. Summer warming caused water to be warmer near the surface, and chlorophyll levels were higher in water less than 10m deep, reflecting the availability of sufficient light for photosynthesis. Higher levels of ammonia and phosphorus in surface samples compared to deeper samples may reflect excretion by the fish or greater plankton activity in the surface waters. Previous farm and open-water monitoring data occasionally showed elevated levels of total phosphorus and

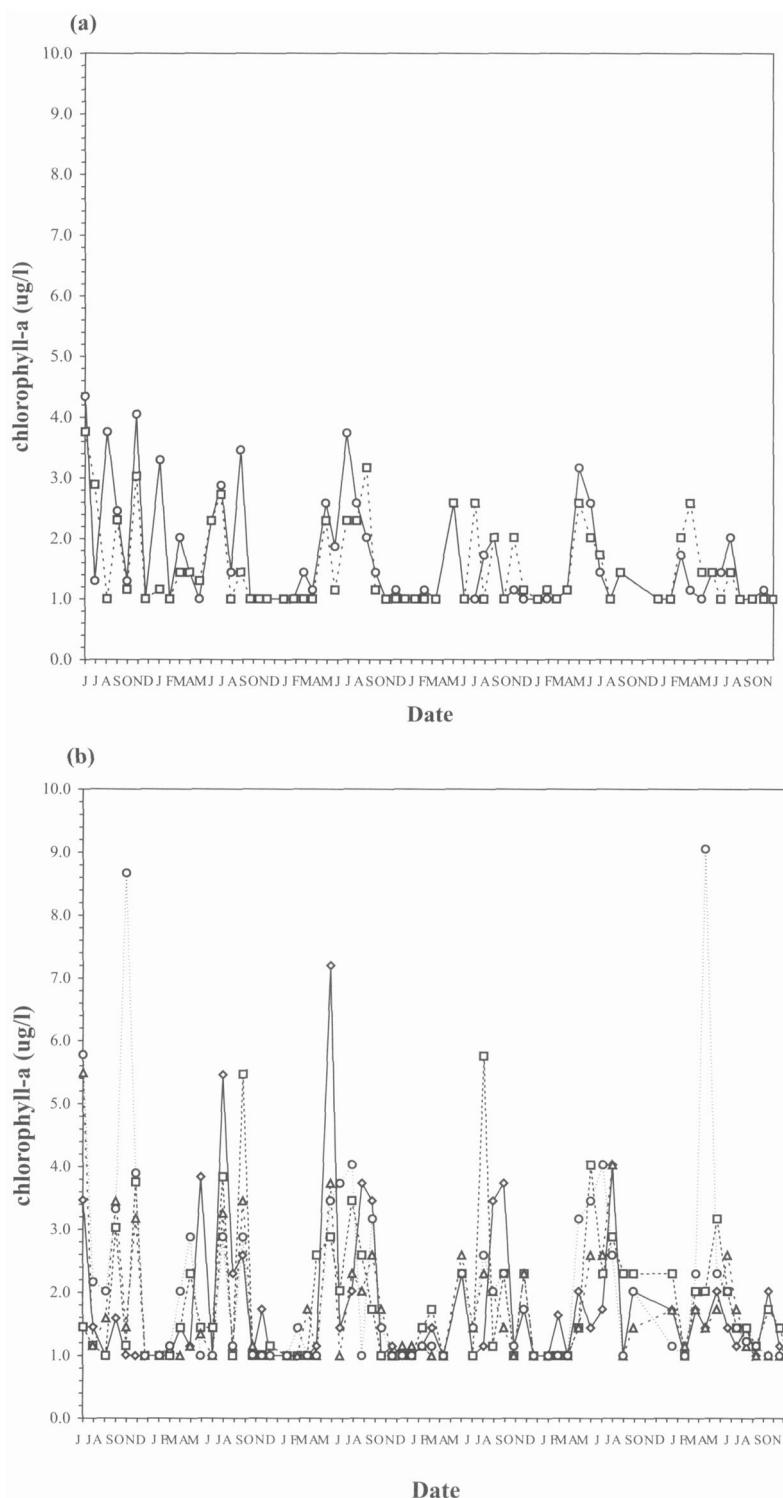
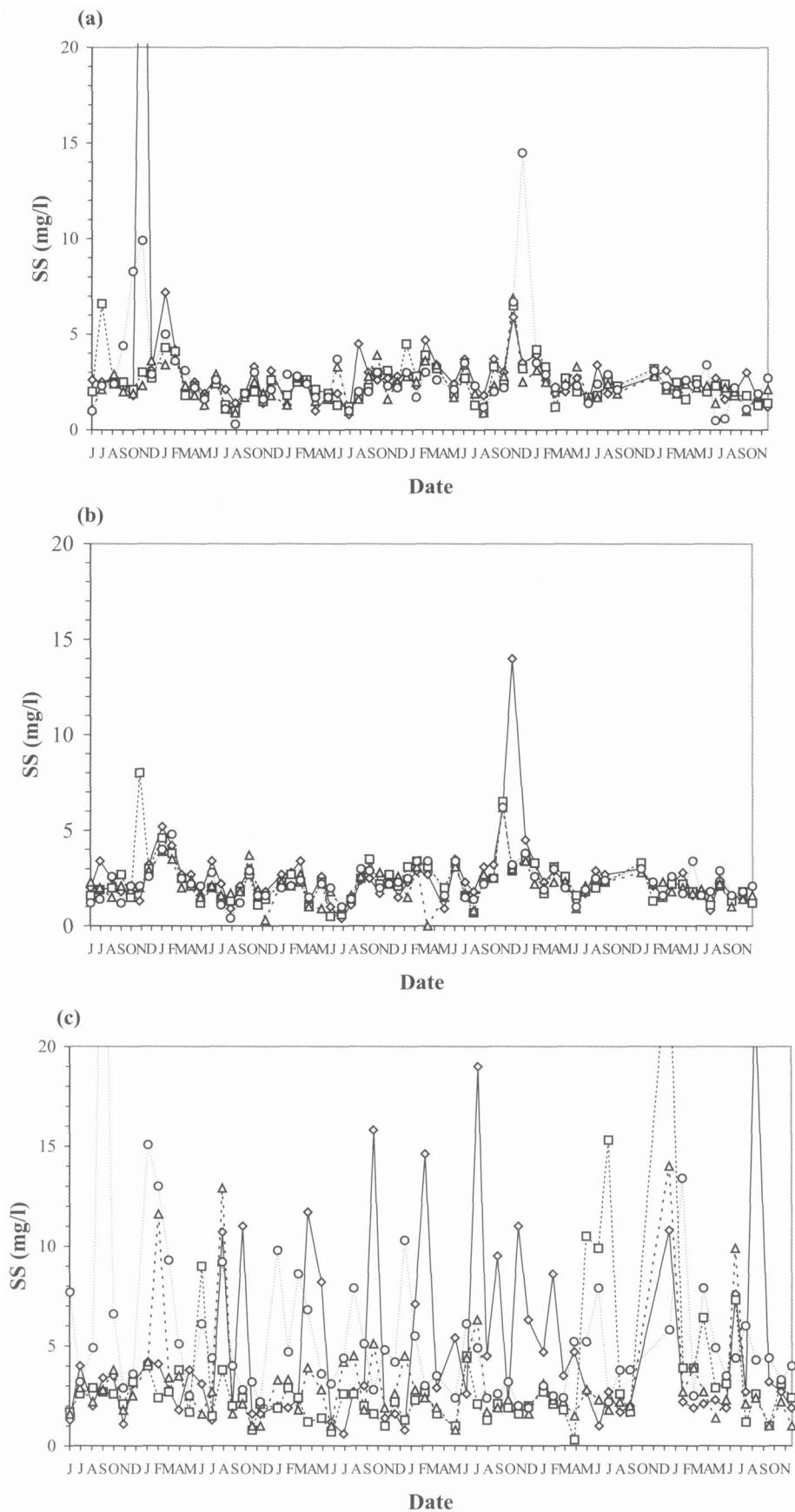


Fig. 9—Changes in chlorophyll a ($\mu\text{g l}^{-1}$) from 28 June 1994 to 15 December 1999 at (a) farm cages (circles) and open water control (squares) and (b) shore sites. Symbols as in Fig. 2.

Fig. 10 (opposite)—Changes in suspended solids (mg l^{-1}) from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. Values in (a) are excluded for cages at 0m depth, on 23 November 1994 (39.6 mg l^{-1}), in (b) for the lake outlet on 28 September 1994 (33.5 mg l^{-1}) and in (c) for Fahy Shore on 12 January 1999 (25.4 mg l^{-1}) and for Corry Strand on 7 September 1999 (22.6 mg l^{-1}). Symbols as in Fig. 2.

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ammonia and decreased oxygen below the thermocline (Champ 1988). At both the farm and control sites, Gillooly (1990) observed elevated phosphate at the surface in three sampling occasions in summer. He suggested this may be released on the death of plankton. Further studies at other open water control sites would be necessary to clarify this. However, levels were still low and not of environmental concern.

Oxygen concentration was lower in deeper water in August 1995 and August 1996 and was probably caused by a seasonal thermocline. The Environmental Protection Agency (EPA) found lower oxygen levels (74–76% saturation) at mid-lake sites surveyed below a thermocline in Lough Allen in July 1995 and July 1996, indicating that this natural phenomenon probably occurs throughout the lake (Environmental Protection Agency 1998; Bowman 1998b). Such conditions did not persist, and there is no suggestion that they may be a threat to the fish farm or local environment. Prolonged warm and calm weather in the future may lead to a thermocline persisting for long enough to cause hypoxia below the thermocline. However, Maloney (1991) found that the thermocline developed at 8–12 m depth, which is well below the fish cages. At the farm sites, the most vulnerable biological indicator of any significant release of anoxic water and associated hydrogen sulphide from below the thermocline would be the fish in the farm cages. Thus monitoring of water quality at different depths should continue, but it should be limited to temperature and oxygen during the summer.

The phytoplankton of Lough Allen has been sampled in 1951, 1973–4, 1986–8, 1995 and 1997 (Champ 1988; Bowman 1998a; 1998b). It is similar to that recorded from other Irish lakes (Southern and Gardiner 1938; Round and Brook 1959). Bowman (1998b) provides a list of 34 species of phytoplankton collected from Lough Allen in 1995–7, with 10 to 23 species found in any one sample. He concluded that this diverse and diatom-dominated community was indicative of oligotrophic conditions.

The EPA sampled Lough Allen at seven stations in 1985 and 1995–7, on three to four sampling dates per year (Bowman 1998b). With the exception of a sewage-enriched sampling station at the southernmost station near Drumshambo, the water quality data collected by the fish farm were within the range of the values recorded by the EPA. Neither the EPA (Bowman 1998b; Lucey *et al.* 1999) nor the fish farm data show any upward or downward trend in nutrients over the years.

Maloney (1991) and Gavine (1992) repeated Gillooly's (1989) suggestion that Lough Allen may be nitrogen-limited. This would be unusual for an oligotrophic or mesotrophic freshwater lake. In

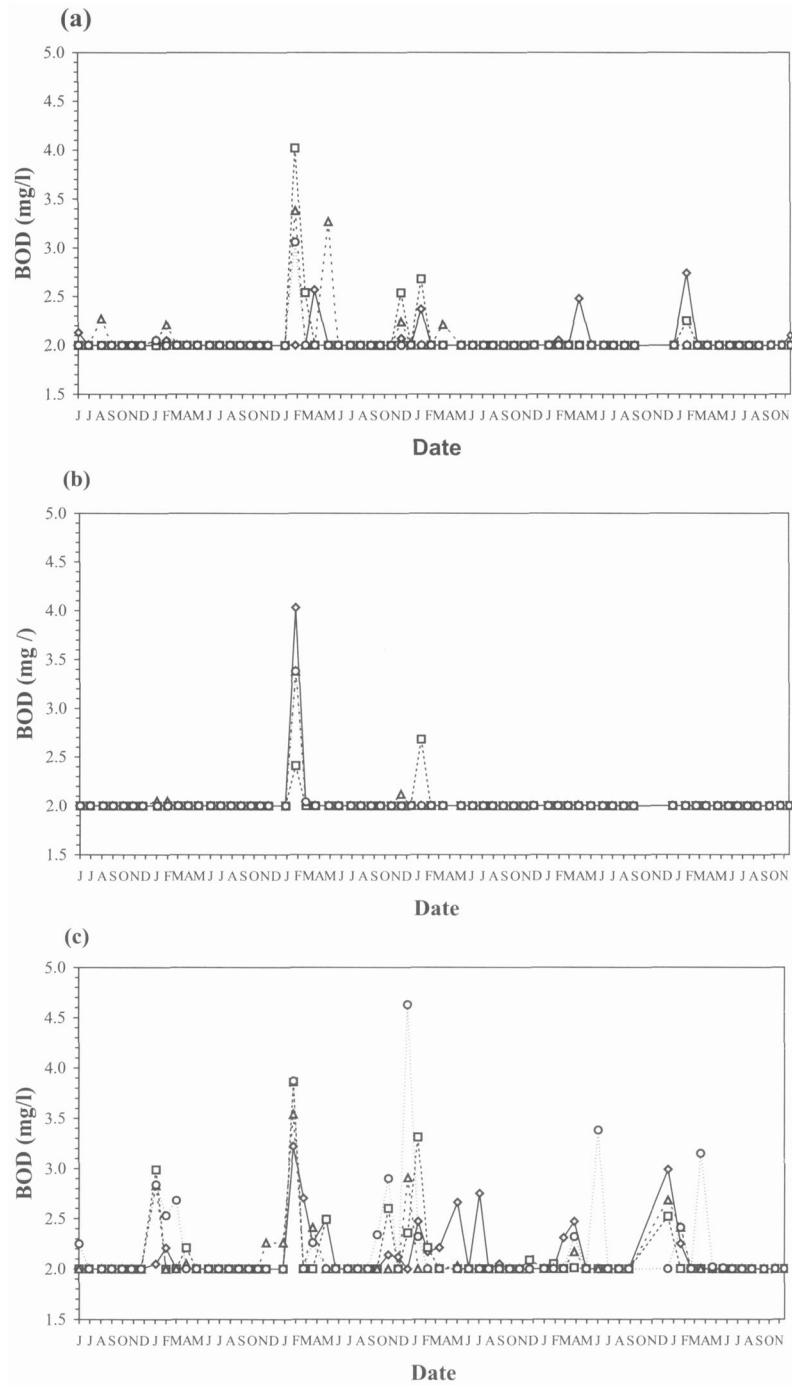


Fig. 11—Changes in biochemical oxygen demand (mg l^{-1}) from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. Symbols as in Fig. 2.

contrast, Champ (1988) concluded it was phosphorus- and perhaps light-limited. Should any changes in water quality occur, the environmental conditions and nutrient budgets of Lough Allen should be reviewed and modelled more thoroughly.

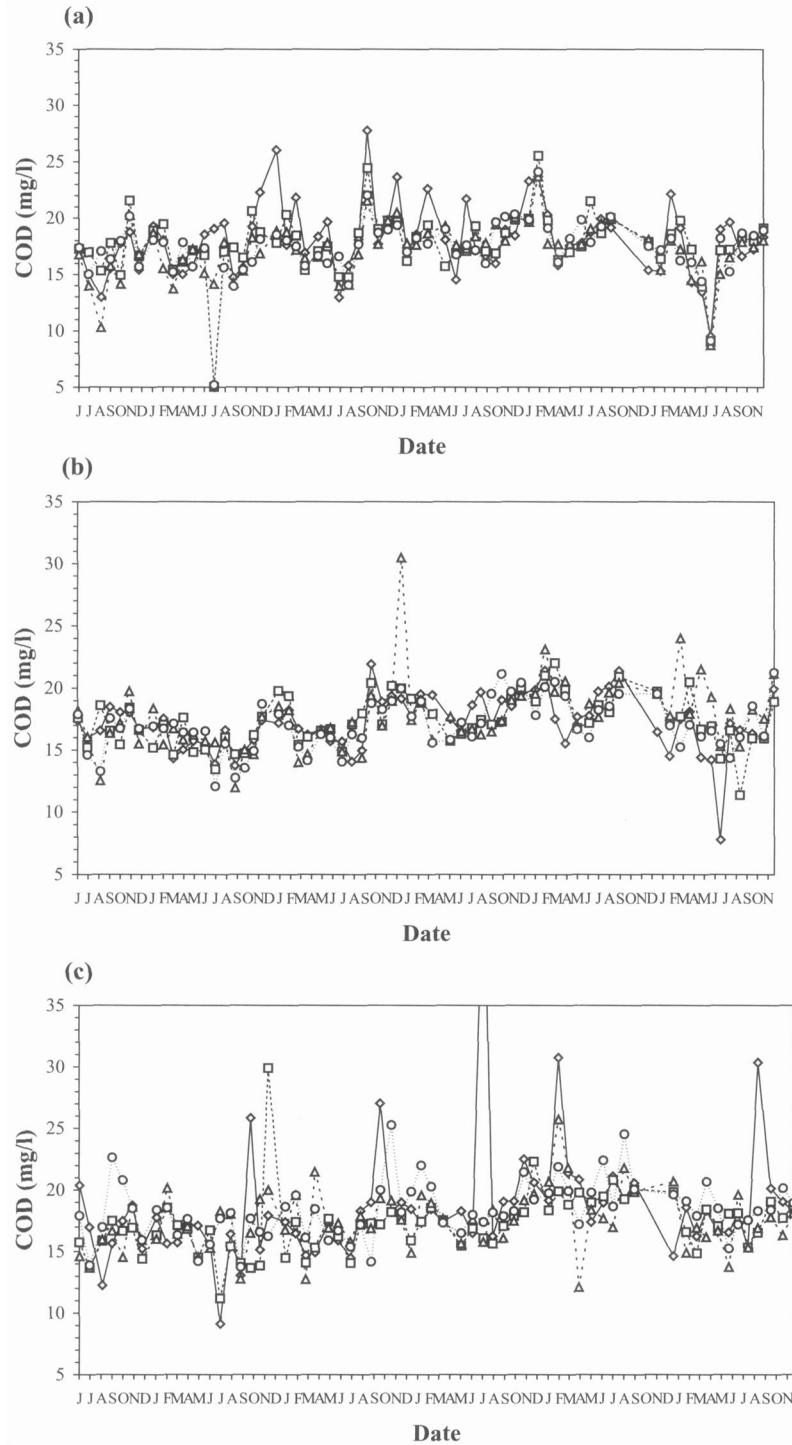


Fig. 12—Changes in chemical oxygen demand (mg l^{-1}) from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. The value for Corry Strand on 30 July 1997 (46.98 mg l^{-1}) is excluded for clarity of presentation. Symbols as in Fig. 2.

Lough Allen has a notably low water transparency for an oligotrophic lake. The EPA recorded Secchi transparency readings of 0.8m in 1995, 1.6m in 1996, 1.6m in 1997 (Lucey *et al.* 1999), 1.5m in 1998, 1.8m in 1999 and 1.6m in 2000 (McGarrigle *et al.* 2002). The averages of

recent monthly Secchi transparency readings were similar, namely 1.4m in 2001 and 1.2m in 2002 (Mary Hensey, pers. comm.). However, zebra mussels, which were discovered in Lough Allen in 1998 (Electricity Supply Board 1999), may result in increased transparency (Lucy and Sullivan 2001).

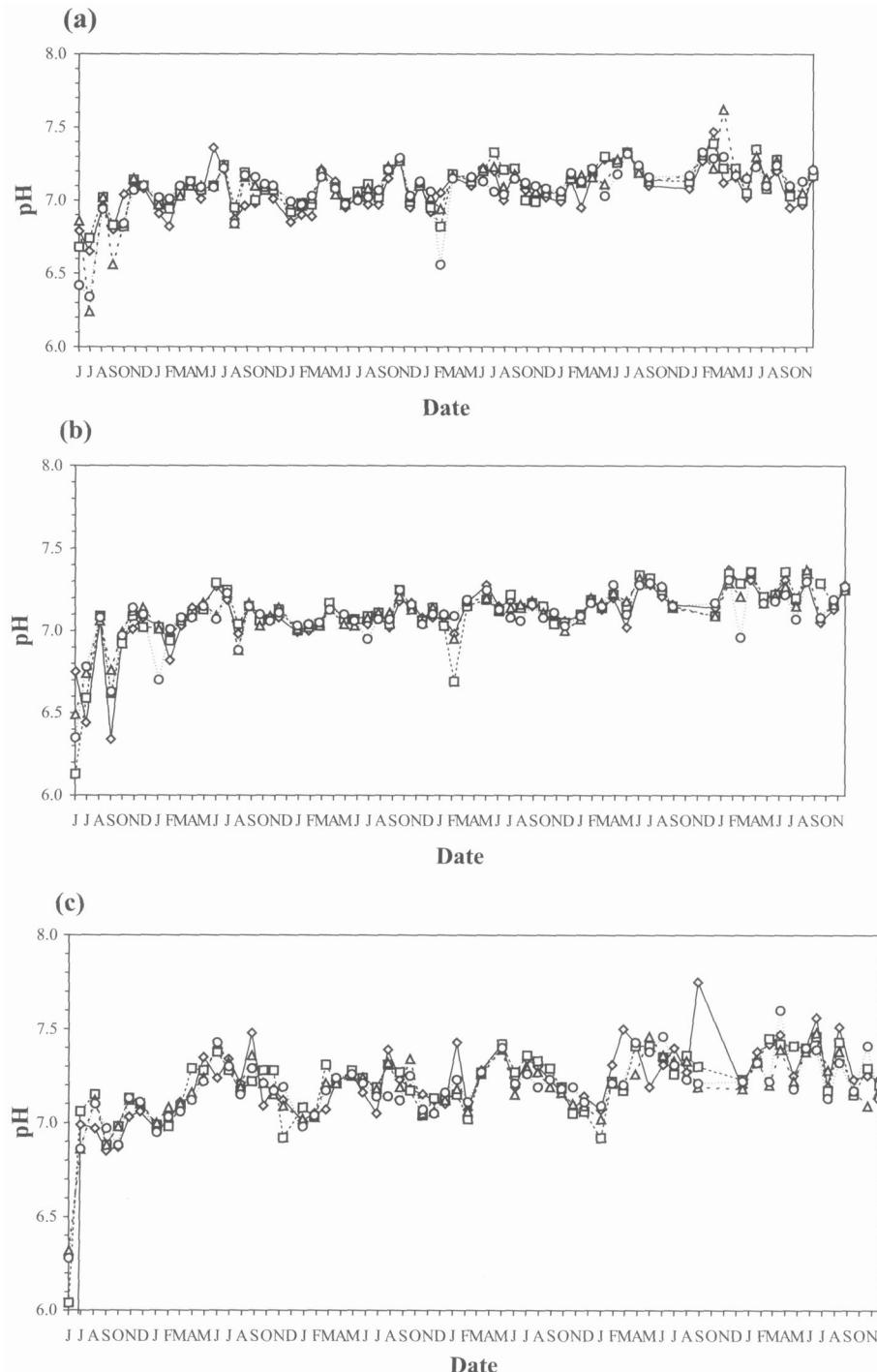


Fig. 13—Changes in pH from 28 June 1994 to 15 December 1999 at (a) salmon cages, (b) control sites and (c) shore sites. Symbols as in Fig. 2.

Simple model calculations based on phosphorus and lake flushing rate may not accurately account for chlorophyll production should light levels be limiting phytoplankton growth. Background and monitoring information on transparency is thus desirable, as transparency may be a critical factor limiting phytoplankton growth in Lough Allen.

The results of the water quality monitoring reported here indicate that the farm has not had an impact on water quality in Lough Allen. This conclusion is supported by recent independent assessments of the quality of Lough Allen. Based on water quality, chlorophyll and shore line observations, they concluded that the lake was not

Table 2—Results of one-way analysis of variance on the water quality data collected from 28 June 1994 to 15 December 1999.

<i>Parameter</i>	<i>Testing differences between...</i>	<i>No. samples</i>	<i>d.f.</i>	<i>F</i>	<i>P</i>
Temperature	Depths at cages	64	3	0.11	0.957
	Depths at open water control sites	64	3	0.10	0.962
	Shore sites	64	3	0.01	0.998
	Cages (depth mean), open water controls and shore sites	64	2	0.12	0.877
Oxygen	Depth at cages	52	3	0.06	0.982
	Depth at open water control sites	52	3	0.06	0.963
	Shore sites	52	3	0.47	0.707
	Cages (depth mean), open water controls and shore sites	52	2	0.82	0.441
Nitrite	Depth at cages	64	3	0.40	0.752
	Depth at open water control sites	64	3	0.02	0.996
	Shore sites	64	3	0.17	0.915
	Cages (depth mean), open water controls and shore sites	64	2	0.13	0.879
Nitrate	Depth at cages	64	3	0.05	0.984
	Depth at open water control sites	64	3	0.04	0.989
	Shore sites	64	3	1.13	0.339
	Cages (depth mean), open water controls and shore sites	64	2	1.68	0.189
Ammonia	Depth at cages	64	3	4.47	0.004
	Depth at open water control sites	64	3	0.88	0.451
	Shore sites	64	3	1.73	0.161
	Cages (depth mean), open water controls and shore sites	64	2	2.02	0.136
Total phosphorus	Depth at cages	64	3	2.08	0.104
	Depth at open water control sites	64	3	0.43	0.735
	Shore sites	64	3	4.66	0.003
	Cages (depth mean), open water controls and shore sites	64	2	2.61	0.076
Soluble reactive phosphate	Depth at cages	64	3	1.25	0.291
	Depth at open water control sites	64	3	0.39	0.764
	Shore sites	64	3	1.70	0.167
	Cages (depth mean), open water controls and shore sites	64	2	2.52	0.083
Chlorophyll <i>a</i>	Controls, cages at 0m, and mean of shore sites	64	2	2.34	0.100
	Shore sites	64	3	1.18	0.316
	Depth at cages	64	3	1.37	0.254
	Depth at open water control sites	64	3	1.01	0.388
Suspended solids	Shore sites	64	3	5.33	0.001
	Cages (depth mean), open water controls and shore sites	64	2	19.64	0.000
	Depth at cages	64	3	0.33	0.806
	Depth at open water control sites	64	3	0.02	0.950
pH	Shore sites	64	3	0.62	0.601
	Cages (depth mean), open water controls and shore sites	64	2	7.64	0.001
	Depth at cages	64	3	0.62	0.600
	Depth at open water control sites	64	3	0.08	0.971
BOD	Shore sites	64	3	1.04	0.377
	Cages (depth mean), open water controls and shore sites	64	2	4.46	0.013
	Depth at cages	64	3	1.35	0.258
	Depth at open water control sites	64	3	0.85	0.470
COD	Shore sites	64	3	2.72	0.045
	Cages (depth mean), open water controls and shore sites	64	2	1.23	0.295
	Depth at cages	64	3	0.13	0.881
	Depth at open water control sites	64	3	0.29	0.753
Alkalinity	Shore sites	64	3	0.04	0.957
	Cages (depth mean), open water controls and shore sites	64	2	0.16	0.852
	Depth upstream	9	2	0.20	0.990
	Depth at cages	9	2	0.17	0.994
Silicate	Depth downstream	9	2	7.30	0.000
	Mean of cages, upstream & downstream sites	9	2	0.20	0.990
	Upstream, cages and downstream at 3 depths	9	8	0.17	0.994
	Upstream, cages and downstream at 3 depths	9	8	0.63	0.747
Total nitrogen	Upstream, cages and downstream at 3 depths	9	8	0.17	0.994
	Upstream, cages and downstream at 3 depths	9	8	7.30	0.000
	Upstream, cages and downstream at 3 depths	9	8	0.63	0.747
	Upstream, cages and downstream at 3 depths	9	8	0.63	0.747

Numbers in bold statistically significant at the $P<0.05$ level.

Table 3—Variation (%) between sampling dates for water quality parameters at the farm cages, open water control and shore control sites. Variation is indicated by the coefficient of variation (%). The lowest and highest values with depth for the cages and open water control sampling stations and for the four shore sites are shown.

Parameter	In cages		Open water control		Shore control sites	
	Min.	Max.	Min.	Max.	Min.	Max.
Temperature	39	45	39	44	42	45
Conductivity	6	9	9	28	9	11
Suspended solids	41	149	41	70	65	184
pH	1	3	1	3	1	3
Alkalinity	11	15	10	11	10	15
Colour	10	12	10	10	27	27
Dissolved oxygen	11	13	11	13	11	12
Biochemical oxygen demand	11	23	10	24	15	27
Chemical oxygen demand	16	18	12	16	13	27
Chlorophyll a	18	51	39	55	29	62
Soluble reactive phosphorus	37	236	41	114	45	60
Total phosphorus	23	137	23	98	22	58
Total ammonia nitrogen	28	48	35	74	38	60
Nitrite	30	86	32	80	24	71
Nitrate	7	25	7	24	5	32
Total nitrogen	16	18	16	22	18	20
Silicate	18	22	16	24	17	19

Table 4—Instances where water quality parameters exceeded permitted levels at the farm cages and control sites.

Parameter	Permitted level (mg l^{-1})	No. of samples	Values greater than permitted level	
			At cages	At other sites
BOD	<10	849	—	—
COD	<80	849	—	—
SS	<30	848	53.8 (28 April 1994)	33.5 (28 September 1994) 39.6 (23 November 1994)
SRP (ortho-P)	<0.2	847	0.238 (30 January 1995)	—
TAN	<0.1	849	—	0.106 (12 January 1999)
TON ($\text{NO}_2 + \text{NO}_3$)	<1.5	849	—	—
Malachite green	<0.01	23	—	—
Formaldehyde	<1.0	23	—	—
Chloramine-T	<1.0	23	—	—
Other therapeutants	<1.0	69	—	—
pH	6.5–8.0	847	6.42, (28 June 1994) 6.24, 6.34, (26 July 1994)	6.13, 6.49, 6.35, 6.04, 6.32, 6.28 (28 June 1994) 6.44, 6.59 (26 July 1994) 6.34 (28 September 1994)

enriched (Toner and Sullivan 1977; Clabby *et al.* 1992; Bowman 1998b; Environmental Protection Agency 1998; Lucey *et al.* 1999).

CHEMOTHERAPEUTANTS

No chemotherapeuticants were detected in the 138 analyses conducted. Chemotherapeuticants are used in concentrations of a few parts per million (mg l^{-1}) and are generally diluted to below chemically detectable levels within hours after release. A dye release was conducted at the cage site during a moderate to fresh wind ($5\text{--}8 \text{ m s}^{-1}$) on 19 October 1993 (Marine Computational Services 1993). Measurements of dye dilution found good dispersion to at least 4m depth, and dilutions of one in ten million within twenty minutes of dye release. It is thus extremely unlikely that chemotherapeuticants would be detected in water samples unless sampling is conducted near the point of release and within minutes to hours of release. The absence of chemotherapeuticants at a detection limit of 1ppm in the present monitoring data (138 samples) is thus expected. Sampling for chemotherapeuticants may be useful in risk assessment if it were conducted at the fish cages within a hour or two of use, so as to confirm that dilution rates were such to not have an impact

beyond the immediate cage environment (Scottish Environmental Protection Agency 1998).

SEDIMENT QUALITY

The minimum and median levels of total phosphorus and nitrogen in sediments were very similar in 1988, 1990 and 1998. These values are considered to be conservative estimates of the natural background levels of sediment phosphorus and nitrogen around the cages. The mean value was very similar to the median in each survey, with the exception of phosphorus in this survey (1998). The higher mean in the present survey was due to high values of phosphorus in four of the 30 samples taken directly under the farm cages.

Three published studies have comparable data on sediment conditions under fish cages in freshwater lakes. Loch Tralaig in Scotland was similar to Lough Allen in pH, conductivity, phosphorus levels and turbidity and in being oligotrophic (Kelly 1992). The sediments at control sites and under the fish cages had phosphorus levels of 12.7 mg P g^{-1} and 16.5 mg P g^{-1} respectively, considerably higher than in Lough Allen. Phosphorus concentrations and LOI under cages from three other Scottish lakes

Table 5—Statistics summarising the concentrations of total phosphorus, total nitrogen, and organic matter (as loss on ignition) found around the fish cages in Lough Allen in 1988, 1990, 1994 and 1998. For comparison, the minimum and maximum values found in a study of three freshwater lakes in Scotland (Kelly 1993) are shown.

	Total phosphorus (mg P g^{-1})			Total nitrogen (mg N g^{-1})			Organic matter (% LOI)		
	1998 ¹ (n = 30)	1990 ² (n = 43)	1988 ³ (n = 42)	1998 ¹ (n = 30)	1990 ² (n = 43)	1988 ³ (n = 42)	1990 ² (n = 43)	1994 ⁴ (n = 4)	1988 ³ (n = 42)
Minimum	0.68	0.92	0.96	3.28	2.83	3.34	8.11	3.91	11.44
Mean	3.82	1.28	1.31	4.31	3.81	4.23	12.49	4.08	13.18
Median	0.99	1.16	1.26	4.10	3.91	4.09	12.54	4.11	13.16
Maximum	51.45	2.47	3.31	10.98	4.36	6.42	15.08	4.20	15.00
Standard deviation	9.54	0.37	0.46	1.37	0.26	0.42	1.01	—	0.88
Coefficient of variation%	249.41	28.81	34.79	31.86	6.86	10.05	8.11	—	6.65

¹present study.

²Maloney 1991.

³Gillooly 1989.

⁴Aqua-Fact 1994.

	<i>Loch 1</i>	<i>Loch 2</i>	<i>Loch 3</i>			
	Min.	Max.	Min.	Max.	Min.	Max.
Control	0.57	0.87	—	—	12	16
Under salmon cages	0.38	6.42	—	—	12	40

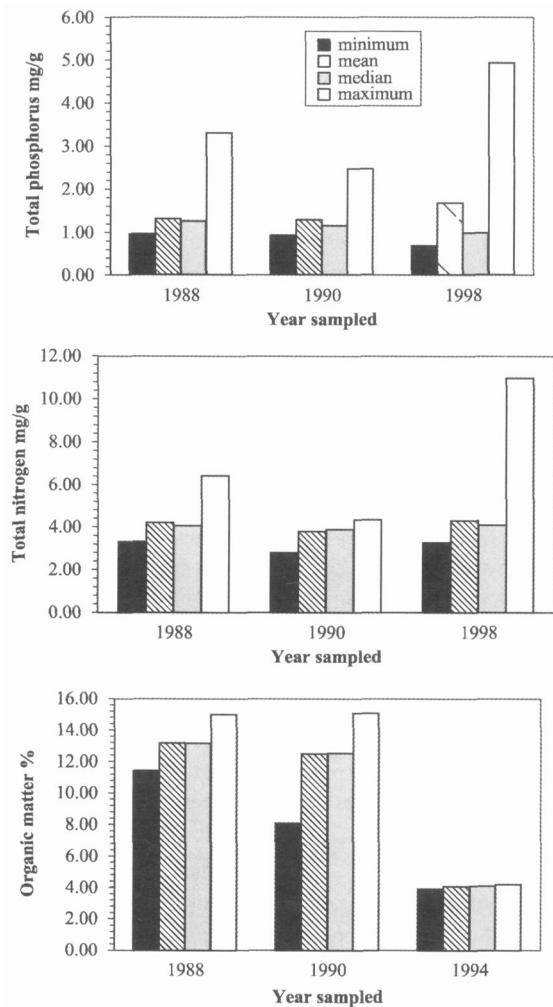


Fig. 14—Sediment quality around the farm cages in 1988 (Gillooly 1989), 1990 (Maloney 1991), 1994 (Aqua-Fact 1994), and 1998 (present study). Minimum, mean, median and maximum values are shown for total phosphorus, total nitrogen and organic matter. Organic matter was estimated as loss on ignition in 1988 and 1990 and as chromic acid oxidation value in 1994.

(Table 5) spanned the range found here for Lough Allen and had higher maximum LOI (Kelly 1993). Analysis of a core of sediment from Lough Leane, Killarney, showed typical LOI of about 20% and an elevated level of LOI near 60% following the burst of a bog and discharge of large amounts of peat into the lake (Murray 1998). In an acid oligotrophic lake in Norway, Skogheim *et al.* (1982) found sediment phosphorus concentrations ranging from natural levels of about 1 mg P g^{-1} to over 3 mg P g^{-1} under fish cages. In this instance fish food and faeces were visible under the cages. Under fish cages in the Mediterranean, total phosphorus concentrations of $3\text{--}9\text{ mg P g}^{-1}$ and LOI of 16–45% were found, with $0.5\text{--}1.0\text{ mg P g}^{-1}$ and 7–10% LOI at control sites (Karakassis *et al.* 1998). The latter study also cites studies from the western Mediterranean, Red Sea

and Norway that recorded LOI of 10–47% under marine fish cages. A eutrophic lake in England polluted by sewage, agricultural runoff and land-based fish farm outputs had an average ($\pm 95\%$ confidence limits) of $4.15 \pm 1.81\text{ mg P g}^{-1}$ (Heaney *et al.* 1992), but maximum values were not given. It would appear that Lough Allen sediments are similar to those of other freshwater lakes and are at the lower end of the range that would be expected under fish cages.

Skogheim *et al.* (1982) and Kelly (1992, 1993) found significant releases of phosphorus from lake sediments, with higher release under fish cages. However, these releases were not ecologically significant because only the immediate area (within 10m) under the fish cages was impacted. Were such a release to occur in Lough Allen, then the impact would be similarly insignificant to the lake ecology.

It is possible that the samples of phosphorus exceeding 10 mg P g^{-1} in the present study contained remains of feed pellets rather than just fish faeces. This was the case under rainbow trout cages in a freshwater lake in Quebec, where two large peaks in sediment phosphorus were related to overfeeding, as indicated by observations of food pellets in the samples (Cornell and Whoriskey 1993). The sediments recovered within six months during which farming continued (Cornell and Whoriskey 1993).

Lake depth can be significantly correlated to sediment phosphorus and nitrogen concentrations (e.g. Heaney *et al.* 1992). This is expected due to increased sedimentation and decreased resuspension of sediments by wave action in deeper waters. However, there was no indication of such an effect around the fish cages in Lough Allen. This suggests that, if any settlement of wastes from the fish cages does occur, the wastes are not accumulating in the deeper parts of the lake.

The sediment conditions in 1998 were the same as described for 1988 (Gillooly 1989) and 1990 (Maloney 1991): namely a brown oxygenated layer of silt, overlying dark anoxic mud that smells of hydrogen sulphide. In contrast, the 1994 survey (Aqua-Fact 1994) did not find any odour of hydrogen sulphide and described the sediment as grey. However, the photographs in the Aqua-Fact (1994) report suggest the sediment could be considered at least partly black. There was no indication of any sediment-living fauna in any of the surveys. Thus, there is no sign of any gross change in sediment quality since 1988. Maloney (1991) found a significant correlation between distance from the cage pivot units and sediment phosphorus concentrations, reflecting some localised effects of wastes from the fish cages. The present study did not find a correlation with distance from the cage units, but did find elevated levels under one cage unit. How persistent these effects will be is not known. At present, they are too limited in extent to be of concern.

LONG-TERM IMPACT OF FRESHWATER SALMON CAGES

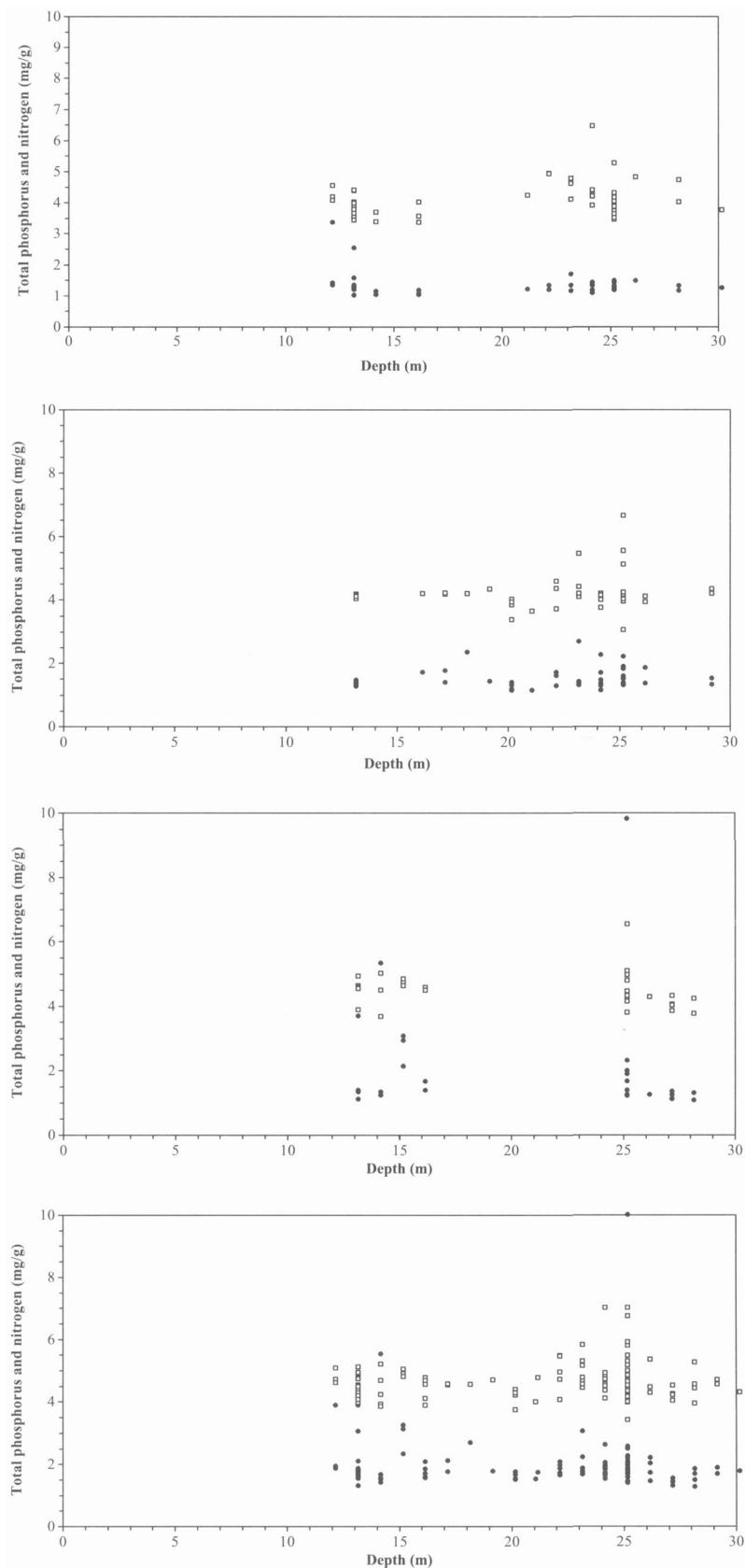


Fig. 15—Total phosphorus (squares) and nitrogen (dots) concentrations (mg g^{-1}) in sediment with depth in 1988, 1990 and 1998 and the combined data sets.

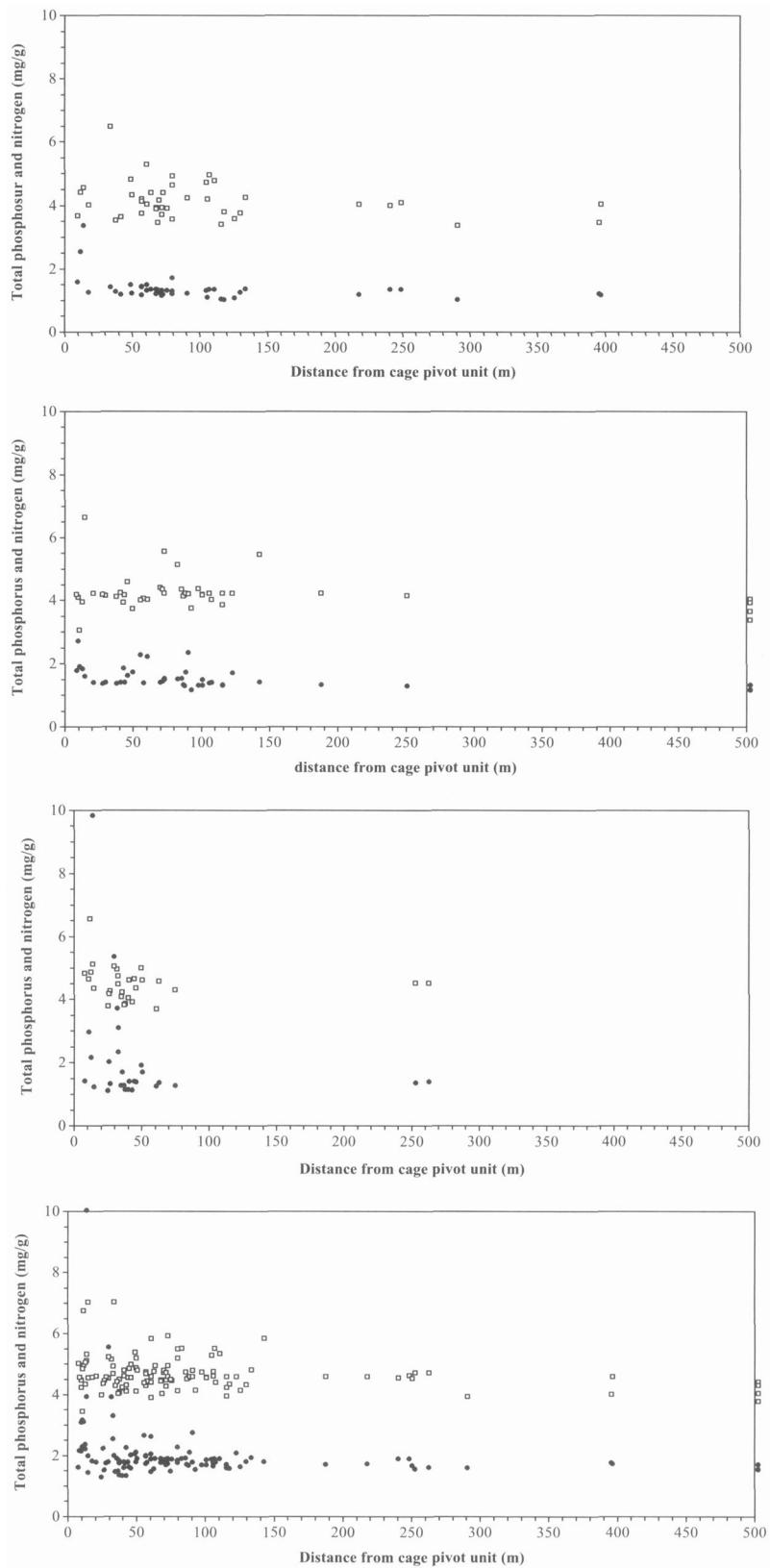


Fig. 16—Total phosphorus (squares) and nitrogen (dots) concentrations (mg g^{-1}) in sediment with distance from the cages in 1988, 1990 and 1998 and the combined data sets.

Table 6—Correlation coefficients (Pearson product moment) for relationships between sediment total phosphorus (TP), total nitrogen (TN), depth and distance of sampling station from cage pivot unit for each and all years data.

	1988	1990	1998	All
No. of samples	42	43	30	115
TP of depth	-0.226	0.150	0.148	0.077
TN of depth	0.267	0.156	0.061	0.125
TP of TN	0.276	-0.016	0.973	0.797
TP of distance	-0.319	-0.422	-0.190	-0.130
TN of distance	-0.266	-0.244	-0.154	-0.204

Significant values ($P < 0.05$) are in bold.

The comparisons between years and with published data indicate that the fish farm is not having a significant impact on the lake sediments. Future sampling of sediments should be conducted within two years to assess the longer-term significance of these impacts. Such a sampling programme would involve (a) the analysis of total phosphorus and nitrogen, (b) at least 30 and preferably 40 sediment samples and (c) the location of sampling stations to account for depth and distance from the cage unit effects.

The present data analyses show that there has been no change in water quality in Lough Allen since the late 1980s, and no other change that could be attributed to the fish farm. There have been increased levels of phosphorus and nitrogen in the lake sediments under one of the four existing cage units. However, these are too localised to be of environmental concern. The results show that salmon cage farming can be sustained in a freshwater lake without polluting the water and the lake bed.

ACKNOWLEDGEMENTS

We thank Mary Hensey (Glan-Uisce Teo) and Ger Morgan (Aquatic Services Unit) for their cooperation in this study. The contribution of farm staff, in particular Myles Woods and Fergal Downes, is also acknowledged. April Stevens, Chris Emblow and Brian Beckett also provided helpful assistance with the paper.

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