Recovery of the sedimentary environment from disturbance from a variety of sources including industrial effluents (Ueda et al., 1994), trawling (Tuck et al., 1998), dredging (Kenny and Rees, 1996), colliery waste dumping (Johnson and Frid, 1995), shellfish harvesting (Hall and Harding, 1997), copper contamination (Hall and Frid, 1995) and iceberg scouring (Gutt et al., 1996) have been studied. In addition the response of individual species to disturbance have been examined eg. Capitella capitata (Bonsdorff and Pearson, 1997) and two epibenthic gastropod species, Hydrobia ulvae and Littorina littorea (Chandrasekara and Frid, 1998). Apart from the present paper and a separate work from the same group (Pereira et al., in prep.), no other study of the recovery of the marine benthic environment after cage farming has been found.

By examining changes in benthic communities along with sedimentary biogeochemical parameters over 2 years at 3 Scottish fish farm sites where farming had been recently discontinued, we have attempted to determine the important factors which relate community response to the sedimentary environment and to determine the rate at which impacted sediment returns to more normal conditions. This information can be used to inform management options on the sustainable usage of the sea-bed for intensive culture. This is of interest both to the regulator, who may wish to determine the appropriate site rotation period for the minimisation of environmental impact and to the farming industry, who require information on the most appropriate management strategies which maximise fish health.

In this paper some of the primary biological and biogeochemical data from this study are presented. The very large data set output from this work, which has temporal, spatial and depth resolution for biological and biogeochemical parameters will be presented in detail elsewhere. Here we present temporal biological data from the most impacted stations only and chemical data, except redox, from the top 3 cm of sediment at these stations. Analyses of the spatial and temporal variations down-core and across the transects using multi-variate statistical techniques are in preparation.

MATERIALS AND METHODS

Three Scottish Atlantic salmon Salmo salar farm sites, located near Cairndow, Loch Fyne (site A), Kingairloch (site B) and in Glenmore Bay, Loch Sunart (site C) were selected as field study sites, representing highly, moderately and lightly enriched conditions respectively. Each site had been operated using several separate cage groups and, at the beginning of this study, farming was discontinued at one of these groups and the cage structures removed. Summary information for each of the sites is given in Table 1.

The recovery of the sea-bed after the cessation of fish farming: benthos and biogeochemistry

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KEY WORDS: Sediment recovery · Benthos · Fish farming · Biogeochemistry · Site rotation

ABSTRACT: The recovery of the seabed after intensive fish farming has been studied for 2 years at 3 sites by following macrofaunal succession and sedimentary biogeochemical parameters such as redox potential, pore-water ammonia, organic nitrogen and organic carbon. Sediments at each of the sites were dominated by very small animals and abundances were very high. Over the course of the study, improvements in terms of increased numbers of species and increased redox potential were recorded together with decreases in organic carbon, nitrogen and pore-water ammonia. The process of recovery was, however, incomplete at the end of the study and it is argued that recovering sediments are highly vulnerable to additional inputs. This calls into question the practice of site rotation to allow sediments to recover.

INTRODUCTION

The gross effects of particulate wastes from intensive cage culture on marine benthic habitats and processes in northern European and other cool-temperate regions are now fairly well established (reviewed by Black, 1998) but very few detailed studies appear in the primary literature (Brown et al., 1987; Holmer and Kristensen, 1992; Hargrave et al., 1993; Wildish et al., 1993; Findlay et al., 1995). The observed effects follow a similar pattern to other organic pollutant sources (Pearson and Rosenberg, 1978) including reducing sediments, hypoxia in the water overlying the sediment, increased sulphate reduction, and marked changes in benthic faunal and meiofaunal (Duplisea and Hargrave, 1996) communities in terms of species number, diversity, abundance and biomass.

Table 1. Summary information for each of the study sites.

Location	Depth range (m, LW)	Mean mid-water current speed cm s^{-1}	Date farming ceased	Approx. cage group biomass in the preceding year	Major antibiotic used
Loch Fyne (A)	30-35	2.7	1/92	200 t	sulphadiazine, trimethoprim
Kingairloch (B)	30	6.1	1/92	200 t	amoxycilin
Loch Sunart (C)	25	4.2	12/91	50 t	oxytetracycline

Preliminary benthic surveys at the vacant sites were performed visually and on the basis of redox potential (Pearson and Stanley, 1979; Zobell, 1946). Stations at each site were established along a gradient of decreasing impact outward from the centre of the fallowed cage site (stations 1, 2 and 3 respectively). Permanent marker buoys were deployed at the stations to enable repeated sampling from the same area. At site C, where sampling was by diving, a marked bottom swim line was laid to enable divers to complete the entire sampling operation without having to surface. Divers were used in preference to remote coring at site C as the study area was relatively small. Sediment samples were obtained using a Craib corer (Craib, 1965) at Sites A and B. Standard Perspex cores (Ø 57 mm, overall length 240 mm) were used for both diving and ship-borne sediment coring. Attempts were made to ensure cores were undisturbed at the sediment/water interface and at least 100 mm deep. Each site was sampled on a quarterly basis over 2 years. Samples were either processed immediately (benthic fauna, redox profile, pH profile) or returned to the laboratory (< 3 h) and kept chilled overnight (under oxygenated seawater taken from near the seabed at the sampling site) prior to processing.

Sediment pore water ammonia was measured (DML autoanalyser, after Grasshoff, 1983, modified by B. Grantham) with increasing sediment depth by slicing duplicate sediment cores into 1 cm sections under nitrogen and centrifuging to remove pore water. Data given here are the means of the top 3 slices (0-3 cm) for duplicate cores. Sedimentary redox potential levels with depth were determined in duplicate from separate cores by macro-electrode immediately after sample recovery. Data reported here are the redox potential at 4 cm depth, Eh_4. Sediment organic carbon and nitrogen were determined by Perkin Elmer 2400 elemental autoanalyser in triplicate from 3 cm core sections which had been freeze dried and ground. Inorganic carbonates were removed from a sub-sample by exposure to hydrochloric acid vapour. Antibiotic residues were measured from these same sections but the results of these analyses will be reported elsewhere. Complete methodologies for HPLC determination of antibiotics are given in Provost (1996) and Provost *et al.* (1997). Measurements of fluxes of

nutrients and hydrogen sulphide from replicate cores from each site, station and sampling event were made from cores incubated in the laboratory. These data will be presented elsewhere.

To quantify recovery by the macrobenthos, triplicate cores were taken at each station and immediately sliced into 3 cm sections to 9 cm sediment depth and were fixed in buffered formo-saline and rose bengal. Samples were returned to the laboratory where they were sieved onto 0.5 mm mesh, sorted into major phyla and identified to lowest taxonomic group, usually to species.

Non-metric multidimensional scaling (MDS) was performed on square-root transformed species data using the PRIMER program (©Plymouth Marine Laboratory, 1994). Combined data from triplicate cores were entered from all sampling events as a time series for each site. Ordinations for sites A and C yielded stress values of 0.02, indicating excellent 2-d representation; stress equalled 0.06 for the site B ordination, indicating good 2-d representation (Clarke and Warwick, 1994).

RESULTS

The taxa comprising the top 90% of the total abundance (or the top 5 taxa, whichever is greater) from each X1 station at each sampling event, together with total abundance (*A* converted to 1000s m^{-2}) and total numbers of taxa (*S*, for the three cores), are listed in Table 2 together with the total biomass (*B* m^{-2}).

Table 2. The taxa comprising at least 90% of total A (or top 5 taxa), the total abundance, A (1000s m^{-2}) and total number of taxa, S (for 3 core tubes of area 2.55 x 10^{-3} m^{-2}) and total biomass, B (g m^{-2}) from each sampling event for combined depths and cores. For polychaetes, trophic guild nomenclature follows Fauchald and Jumars (1979): "in position 1 B=sub-surface deposit feeder, S=surface deposit feeder, C=carnivore, F=filter feeder, H=herbivore; in position 2 D=discretely motile, M=motile; in position 3 J=jawed, T=tentaculate, X=other structures". For others, F=filter feeder, O=omnivore.

A1 1/9/92 Taxon	%	Trophic Guild	A1 16/11/92 Taxon	%	Trophic Guild	A1 9/3/93 Taxon	%	Trophic Guild	A1 27/8/93 Taxon	%	Trophic Guild
Capitella sp.	65	SMX, BMX	Nematoda	59	O	Nematoda	57	BMX	*Mediomastus fragilis*	32	SMX, BMX
Nematoda	31	O	Capitellidae	18	SMX, BMX	*Capitella capitata*	21	SMX, BMX	Nematoda	28	O
Paraonga caeca	1	C	*Ophryotrocha* sp.	15	HMJ, CMJ, SMJ	*Parathalestris harpacticoides*	18	H	*Protodorvillea kefersteini*	18	CMJ
Ostracoda	1	FM	*Protodorvillea kefersteini*	3	CMJ	*Ophryotrocha* sp	2	HMJ, CMJ, SMJ	*Micruspio* sp	3	FDT, SDT
Rhodine gracilior	1	BSX	*Raricirrus beryli*	2	SMX	*Protodorvillea kefersteini*	1	CMJ	*Phaloe inornata*	2	CMJ
Total %	99		Total %	97		Total %	99		*Thyasira flexuosa*	2	BH
S	10		S	19		S	8		*Macrophthalmus* sp	2	HMJ, CMJ
A (k)	129		A (k)	312		A (k)	312		*Scalibregma inflatum*	1	BMX
B	810		B	77		B	386		*Prionospio fallax*	1	FDT, SDT
									Total %	90	
									S	42	
									A (k)	136	
									B	150	

A1 18/10/93 Taxon	%	Trophic Guild	A1 24/1/94 Taxon	%	Trophic Guild	A1 11/4/94 Taxon	%	Trophic Guild	A1 27/6/94 Taxon	%	Trophic Guild
Nematoda	67	O	Nematoda	65	O	Nematoda	59	O	Mytilidae (juv)	34	F
Ophryotrocha sp.	12	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	14	HMJ, CMJ, SMJ	*Mediomastus fragilis*	18	SMX, BMX	Nematoda	22	O
Protodorvillea kefersteini	8	CMJ	*Capitella capitata*	7	SMX, BMX	*Protodorvillea kefersteini*	16	CMJ	Maldanidae	16	BSX
Capitella capitata	7	SMX, BMX	*Bradya B.typica*	5	H	*Pseudopolydora paucibranchiata*	1	FDT, SDT	*Protodorvillea kefersteini*	8	CMJ
Mediomastus fragilis	2	SMX, BMX	*Mediomastus fragilis*	4	SMX, BMX	*Chaetozone setosa*	1	SMT	*Mediomastus fragilis*	6	SMX, BMX
Total %	96		Total %	95		Total %	95		*Prionospio fallax*	2	FDT, SDT
S	10		S	20		S	28		*Scalibregma inflatum*	2	BMX
A (k)	129		A (k)	383		A (k)	188		Total %	86	
B	21		B	54		B	68		S	38	
									A (k)	151	
									B	82	

B1 8/7/92 Taxon	%	Trophic Guild	B1 26/10/92 Taxon	%	Trophic Guild	B1 22/3/93 Taxon	%	Trophic Guild	B1 14/6/93 Taxon	%	Trophic Guild
Nematoda	74	O	Nematoda	80	O	Nematoda	54	O	Nematoda	53	O
Capitella sp.	11	SMX, BMX	*Capitella* sp.	10	SMX, BMX	*Ophryotrocha* sp.	20	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	35	HMJ, CMJ, SMJ
Ophryotrocha sp.	10	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	4	HMJ, CMJ, SMJ	*Capitella capitata*	7	SMX, BMX	*Capitella capitata*	3	SMX, BMX
Phyllodoce mucosa	2	CMS	*Prionospio fallax*	4	FDT, SDT	*Parathalestris harpacticoides*	4	H	*Abra* sp.	2	S, F
Abra alba	1	S, F	*Mediomastus fragilis*	0.5	SMX, BMX	*Mediomastus fragilis*	3	SMX, BMX	*Protodorvillea kefersteini*	1	CMJ
Total %	99		Total %	98.5		*Prionospio fallax*	3	FDT, SDT	Total %	94	
S	16		S	19		Total %	91		S	27	
A (k)	109		A (k)	210		S	34		A (k)	189	
B	9		B	25		A (k)	194		B	64	
						B	89				

5

Table 2 (continued)

B1 13/9/93 Taxon	%	Trophic Guild	B1 17/1/94 Taxon	%	Trophic Guild	B1 25/4/94 Taxon	%	Trophic Guild	B1 18/7/94 Taxon	%	Trophic Guild
Nematoda	71	O	Nematoda	91	O	Nematoda	82	O	Nematoda	88	O
Ophryotrocha sp.	10	HMJ, CMJ, SMJ	*Prionospio fallax*	3	FDT, SDT	*Ophryotrocha* sp.	7	HMJ, CMJ, SMJ	*Prionospio fallax*	4	FDT, SDT
Prionospio fallax	9	FDT, SDT	*Mediomastus fragilis*	2	SMX, BMX	*Prionospio fallax*	4	FDT, SDT	*Mediomastus fragilis*	2	SMX, BMX
Mediomastus fragilis	3	SMX, BMX	*Ophryotrocha* sp	1	HMJ, CMJ, SMJ	*Capitella capitata*	2	SMX, BMX	*Leptognathia breviremis*	1	O
Thyasira flexuosa	1	BH	*Thyasira flexuosa*	1	BH	*Mediomastus fragilis*	2	SMX, BMX	*Abra* sp.	1	S, F
Total %	94		Total %	98		Total %	97		Total %	96	
S	34		S	33		S	24		S	38	
A (k)	137		A (k)	312		A (k)	181		A (k)	216	
B	168		B	52		B	34		B	42	

C1 22/7/92 Taxon	%	Trophic Guild	C1 9/11/92 Taxon	%	Trophic Guild	C1 1/3/93 Taxon	%	Trophic Guild	C1 21/6/93 Taxon	%	Trophic Guild
Capitella sp.	59	SMX, BMX	*Capitella* sp.	58	SMX, BMX	*Ophryotrocha* sp.	40	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	26	HMJ, CMJ, SMJ
Ophryotrocha sp.	36	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	36	HMJ, CMJ, SMJ	*Capitella capitata*	35	SMX, BMX	*Longipedia minor*	19	H
Nematoda	2	O	Nematoda	2	O	*Parathalestris harpacticoides*	10	H	Nematoda	9	O
Spio decorata	2	FDT, SDT	*Spio decorata*	2	FDT, SDT	Nematoda	6	O	*Sphaerosyllis* sp	9	CMJ
Malacoceros fuliginosus	1	FDT, SDT	*Malacoceros fuliginosus*	1	FDT, SDT	*Abra alba*	3	S, F	*Capitella capitata*	8	SMX, BMX
Total %	100		Total %	99		Total %	94		*Sphaerosyllis tetralix*	5	CMJ
S	8		S	12		S	17		*Prionospio fallax*	5	FDT, SDT
A (k)	182		A (k)	183		A (k)	37		*Abra alba*	4	S, F
B	136		B	101		B	3		*Bradya B.typica*	3	H
									Ophelina acuminata	2	BMX
									Exogone naidina	1	CMJ
									Mediomastus fragilis	1	SMX, BMX
									Total %	92	
									S	26	
									A (k)	25	
									B	14	

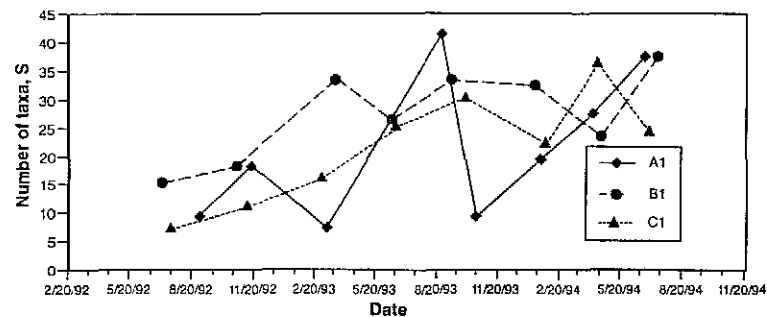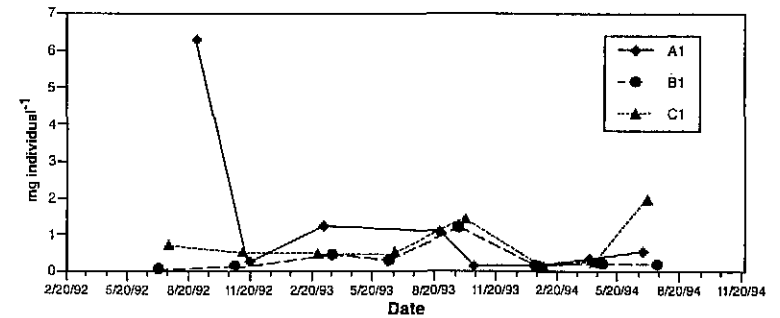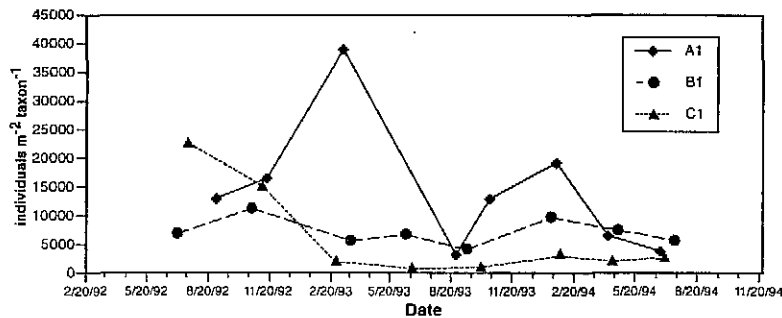| C1 4/10/93 Taxon	%	Trophic Guild	C1 31/1/94 Taxon	%	Trophic Guild	C1 18/4/94 Taxon	%	Trophic Guild	C1 4/7/94 Taxon	%	Trophic Guild
Nematoda	51	O	Nematoda	32	O	Nematoda	57	O	Nematoda	64	O
Ophryotrocha sp.	18	HMJ, CMJ, SMJ	*Ophryotrocha* sp.	29	HMJ, CMJ, SMJ	*Mysella bidentata*	8	F	*Abra alba*	7	S, F
Capitella capitata	7	SMX, BMX	*Capitella capitata*	23	SMX, BMX	*Ophryotrocha* sp.	8	HMJ, CMJ, SMJ	*Sphaerosyllis* sp.	6	CMJ
Prionospio fallax	5	FDT, SDT	*Abra* sp.	3	S, F	*Sphaerosyllis tetralix*	7	CMJ	*Longipedia minor*	4	H
Mediomastus fragilis	2	SMX, BMX	*Sphaerosyllis tetralix*	3	CMJ	*Capitella capitata*	6	SMX, BMX	*Ophryotrocha* sp.	3	HMJ, CMJ, SMJ
Ampelisca spinipes	1	F	Total %	90		*Sphaerosyllis* sp.	4	CMJ	*Sphaerosyllis tetralix*	3	CMJ
Mysella bidentata	1	F	S	23		*Exogone naidina*	1	CMJ	*Prionospio fallax*	2	FDT, SDT
Abra alba	1	S, F	A (k)	81		Total %	90		*Thyasira flexuosa*	2	BH
Sphaerosyllis sp.	1	CMJ	B	9		S	37		Total %	92	
Spionidae	1	FDT, SDT				A (k)	87		S	25	
Total %	90					B	25		A (k)	73	
S	31								B	144	
A (k)	36										
B	52										

Fig. 1. Number of benthic taxa *S* from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

Although the sites were chosen in order of decreasing organic enrichment from the initial surveys, all the sites had low numbers of taxa (8-16) at the beginning of the survey (Fig. 1), with site C, supposedly the least impacted, having the least. Two years later, sites A and B had 38 taxa but site C was still impoverished with only 25 taxa identified. Site A fluctuated wildly with an initial increase in *S* followed by a decrease to below the initial level. This decrease was followed by a massive increase in *S* to 42 in the summer of 1993 followed by a second crash in the autumn; *S* then steadily increased until the end of the study. The decrease in species number was accompanied by an increase in the relative dominance of nematodes or the opportunistic polychaete *Capitella capitata* on both occasions whereas the subsequent recovery periods were dominated by the polychaetes *Mediomastus fragilis* and *Protodorvillea kefersteini*. While the first crash may have been a consequence of seasonality, the authors believe that the second crash is a direct consequence of a secondary input of organic material to the site. In the late summer of 1993 a group of cages were moored close to the site for several weeks prior to their fish being harvested. It was not intended that these fish be fed but there would have been some faecal input to the sediment. Data from the 2 more remote stations at site A are not presented here but the same crash did not occur at these stations, indicating that this phenomenon was local to station A1, further supporting the argument that some organic input had taken place at this time to retard the recovery process.

The number of taxa at Site B also fluctuated over the studied period but less wildly than at site A, showing a relatively steady upward trend interrupted by potentially seasonal reductions in the spring of both 1993 and 1994. Site C showed the most steady upward trend in *S* with some variability in the 1994 samples. At site B nematodes predominated throughout the survey but the polychaetes *C. capitata* and *Ophryotrocha* sp., which

were secondarily dominant during the first year, were replaced by other small polychaetes and bivalves during 1993-94. A similar dominance progression was observed at site C.

The abundance data (Fig. 2) were characterised by very large numbers with the maximum recorded being nearly 400,000 ind. m^{-2}. The abundance data showed an increase followed by a decrease at site A. Again the steady decrease was interrupted by a large increase at the beginning of 1994. This may also have been a delayed consequence of the putative fresh input of organic material although it is interesting to note that during the period in summer 1993 when *S* crashed from 42 to 10, abundance remained relatively constant due to the massive increase in nematode numbers, which again may be associated with the fresh input. Site B had low initial abundance but showed a general rise in abundance until the beginning of 1994 when a steady fall might have been expected. Site C showed a high initial abundance dropping after the end of 1992 to a relatively stable level for the remainder of the study.
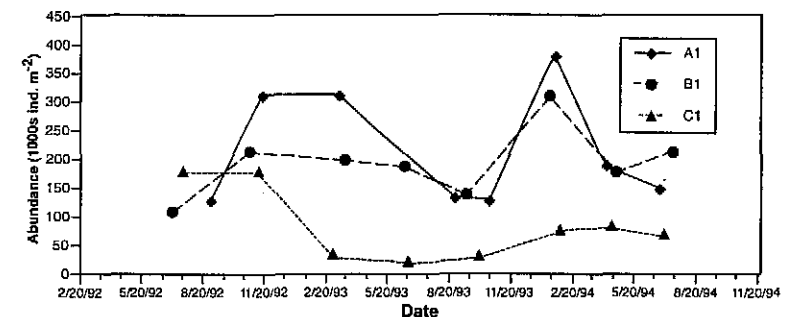
Fig. 2. Abundance of benthic individuals *A* from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

The biomass data for site A (Fig. 3) showed a massive initial level of over 800 g m^{-2} crashing to below 100 g m^{-2} in under 10 weeks as polychaetes were replaced by nematodes as the dominant fauna. Polychaete numbers recovered considerably by March of 1993 with a consequent jump in biomass which thereafter remained below 100 g m^{-2}. There is no evidence in the biomass data of a similar fluctuation in the summer of 1993 as was observed for *S* and later for abundance which is consistent with the small size of the taxa involved in these large swings. The biomass at site B followed the rise seen in abundance from a very low initial level. Similarly at site C, biomass to some extent followed the same pattern of decline then stability that was seen for *S* at the same site.
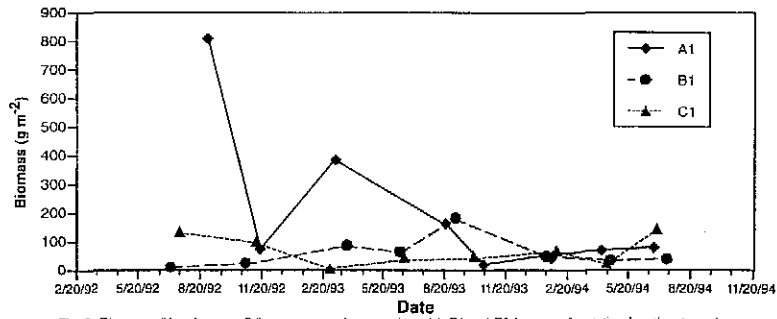
Fig. 3 Biomass of benthic taxa *B* from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

The abundance ratio *A/S* (Fig. 4), which is the mean number of individuals per taxon and therefore an indicator of dominance or evenness, showed a large peak for site A at the start of 1993 which was a consequence of the high abundance and very impoverished community with only 8 taxa identified. A second peak in early 1994 was a consequence of a trebling of abundance with only a doubling in *S*. By the end of the study the abundance ratio for site A was considerably lower than at the start. This was not the case for site B which showed broadly similar abundance ratios throughout the study. At site C a more expected pattern of very high initial values for abundance ratio quickly decreasing to the rather low values was observed.

Fig. 4. Abundance ratio of benthic taxa *A/S* from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

The size ratio *B/A* (Fig. 5), which is the mean weight per individual, was initially high at site A indicating that the capitellids which dominated the community at this time were relatively large in relation to the nematodes which dominated most of the later samples. Only at the last sampling date did the biomass ratio begin to rise

again due to the presence of considerable numbers of juvenile Mytilidae. The size ratio at site B remained very low throughout the sampling period due to the dominance of nematodes. At site C there was a peak in the size ratio in October 1993 which is difficult to explain without a discussion of the biomasses of the individual taxa as, at that time, the station became dominated by nematodes which remained dominant until the end of the study.

Fig. 5. Size ratio of benthic taxa *B/A* from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

At the beginning of the study, organic carbon levels at station A1 (Fig. 6) were exceptionally high (> 10%) and were still elevated at the end of the study (3.8 %). The increase in organic carbon observed at the end of 1993 and the beginning of 1994 is further evidence of additional input at this time although this was observed after the drop in *S* (Fig. 1) but consistent with the increase in abundance (Fig. 2). Organic carbon levels at site B (5.6%) were considerably lower than at site A at the start of the study and fell relatively smoothly to < 2%. Site C showed a relatively steady level of organic carbon of ~ 2%.

Fig. 6. Organic carbon content of surficial (0-3 cm depth) sediment from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

Sedimentary nitrogen levels (Fig. 7) were very closely correlated with organic carbon levels ($\sqrt{}$ transformed N, log transformed C) ($r^2 = 0.79$, $P<0.01$) for all stations indicating that C/N ratios remained broadly constant throughout the study period (Fig. 8).
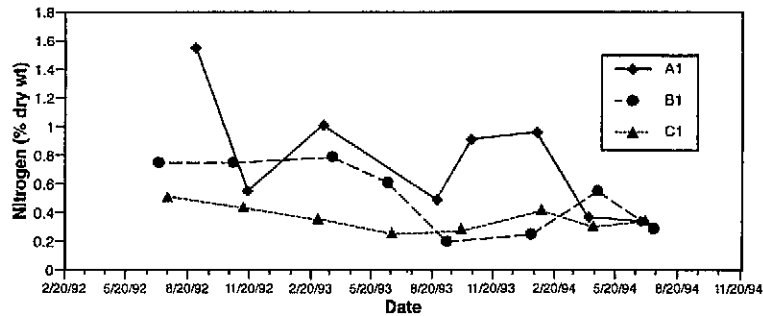
Fig. 7. Organic nitrogen content of surficial (0-3 cm depth) sediment from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

This is contrary to expectations as N is mostly associated with labile proteins and therefore should have degraded more quickly than bulk carbon which contains less degradable organic carbon components, thus yielding an increasing C/N ratio with time. The C/N ratio at site A shows a peak during late autumn 1992, which might be due to an increased input of leaf litter at that time and a dip the following winter which may be a consequence of the input of fresh protein-rich material.
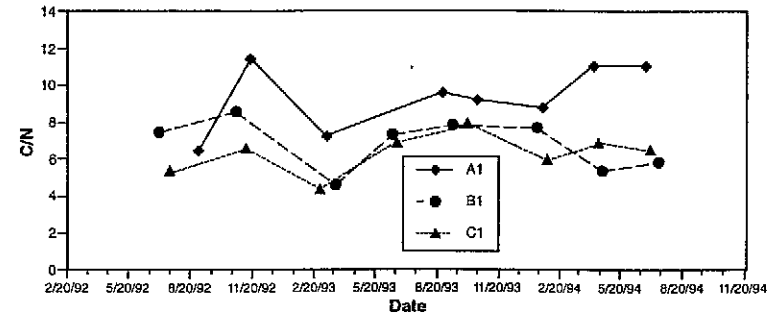
Fig. 8. Organic carbon/nitrogen ratio in surficial (0-3 cm depth) sediment from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

Free ammonia levels at site A (Fig. 9) are relatively stable for the first year then show a massive rise during winter 1993/4 which could be a consequence of this same input event as the ammonium ion is a product of protein deamination. Ammonia concentrations at sites B and C fell over the course of the study and are perhaps better proxies of sedimentary protein than N itself.
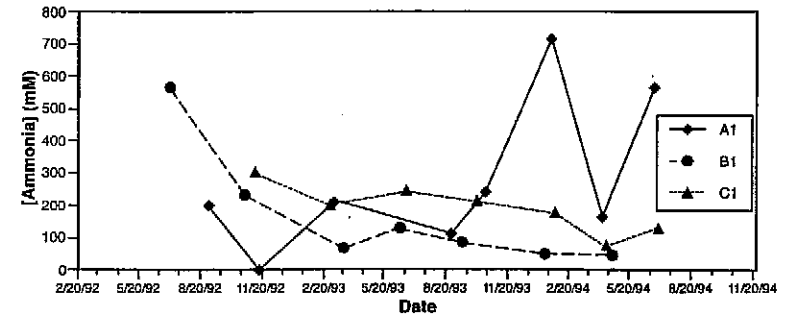
Fig. 9. Pore-water ammonia concentration in surficial (0-3 cm depth) sediment from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

$Eh_{.4}$ showed a marked decrease for station A1 during the autumn of October 1993 consistent with a fresh organic input (Fig. 10). Station B1 exhibited a general trend towards more oxidising conditions following an initial drop in Eh after the first sampling event. Station C1 appeared to recover quickly from very reducing conditions at the beginning of the survey prior to dropping slightly towards the end of the survey.
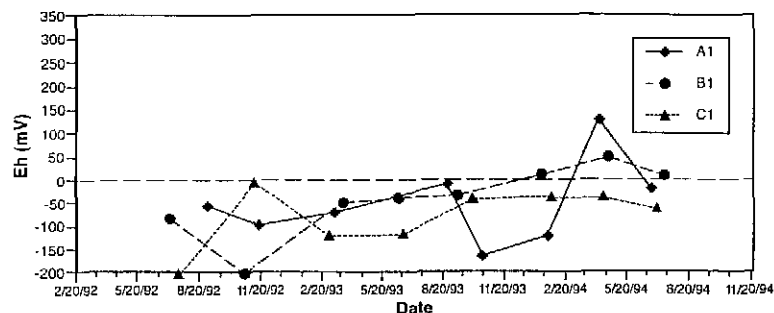
Fig. 10. Redox values (mV, corrected) at 4 cm sediment depth from core samples at stations A1, B1 and C1 (see text for station locations) varying with time.

DISCUSSION

Study methodology

The three sites which were chosen for this study were non-ideal for an experimental study in a number of ways: their management was outside of scientific control; they were within areas which had different histories of fish farming activity; they had differing natural inputs and active farms continued in the immediate vicinities of all sites giving unknown inputs in terms of resuspended materials. Accepting these points, the chosen sites were the best available for this study and allowed the examination of the recovery process in situations typical of this industry.

Three stations were chosen at each site reflecting a transect of sediment enrichment from the former site. The data from the medium and lightly enriched stations (X2 and X3, X = A, B, C) are not presented here as they did not act as true reference sites and so it is not possible to compare the changes taking place at each site with changes in the local environment. The lack of true reference sites is unlikely to prove problematic as the sites are so enriched that changes occurring at reference stations, particularly regarding benthos, are likely to be of a different order of magnitude. In fact, in the semi-enclosed areas in which each site was located, it would have been difficult to have found true reference stations i.e. stations with similar depths and sediment types but with no influence from the farms. Thus the assumption is made at all three sites that changes towards benthos typical of Scottish sea lochs is recovery from the impact of the farm. In a major sea loch study by Pearson *et al.* (1982), the mean number of species *S* in a control sea loch sediment was 37, but only 14 at the station with most organic input, values similar to those found in the present study. The abundances found by Pearson *et al.* were,

however, lower by 3 orders of magnitude and the mean biomass per individual much larger than in this present study, reflecting the smaller size of mesh used here and the resultant small animal size.

The small sample size – cores rather than grabs – is justified by the enormous abundances found in these enriched sediments. Cores were selected as it was thought that grabbing at each station 8 times over 2 years would add another disturbance variable. To compensate for the small sample volume the samples were sieved over a 0.5 mm mesh instead of the more usual 1 mm. It is likely that the majority of the small animals which were retained for analysis would have been lost from a 1 mm sieve.

Recovery at site A

Several of the results presented support the view that there was a brief fresh input of labile organic material to Loch Fyne in the second half of 1993. The biogeochemical and benthic parameters which indicated gradual recovery until this time were set back by at least 6 months indicating that recovering sediments are rather vulnerable to additional inputs. The Loch Fyne site is also predisposed to large inputs of leaf litter in the autumn, being adjacent to the mouths of the River Fyne and Kinglas Water. While these riverine inputs are likely to be obscured by the input of more labile fish farm derived material in 1993, there is some evidence of this effect in 1992 when autumnal species numbers dropped and the C/N ratio increased. The sediments and their biota in this area may therefore be inherently unstable due to the large, but variable, amount of natural organic input. An examination of the MDS plot for the site (Fig. 11) shows a gradual trend to the right over time with 2 'setback loops' thought to correspond to the 2 processes mentioned here.
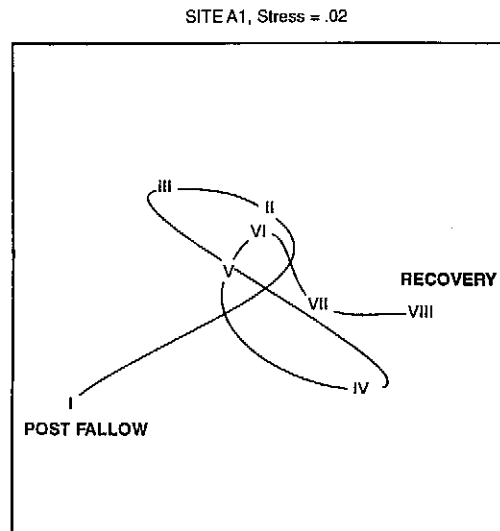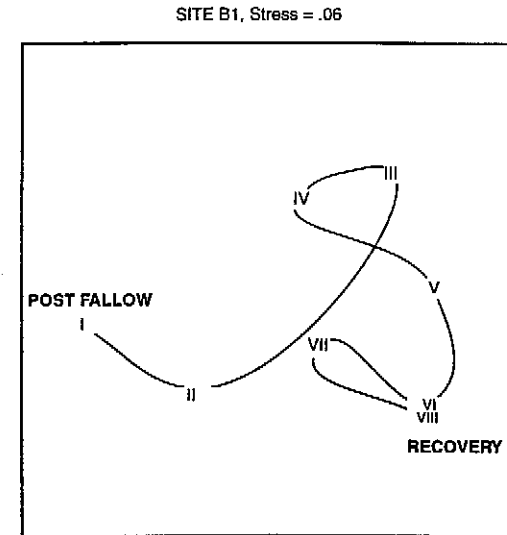
SITE A1, Stress = .02

Fig. 11. MDS plot of Site A time series (square root transformed) species data. Roman numerals refer to sampling occasion (see text for dates).

In terms of the benthos present there was a trend from taxa associated with highly enriched sediments to those species preferring moderate enrichment. Thus initially all of these sites were dominated by nematodes and the polychaetes *C. capitata* and *Ophryotrocha* sp. These taxa are well known to favour highly enriched sediments in many areas (Pearson and Rosenberg, 1978; Pearson *et al.* 1982). During the course of the survey these were gradually replaced as dominants at sites A and B initially by other small polychaetes, e.g. *M. fragilis and P. kefersteini*, associated elsewhere with moderately enriched sediments. By the end of the survey these were themselves being replaced by other small polychaetes and bivalve molluscs characteristic of normal sediments in the area. Species numbers were much higher at the end of the survey but abundances remained high. The initial colossal biomass at site A dropped quickly and at the end of the study was of the same order as that at the other sites.

Recovery at site B

Recovery at site B was less erratic than at site A. The two drops in *S* (Fig. 1) in the spring of 1992 and 1993 may have been the result of smaller larval forms escaping the sieve which later were retained, although this phenomenon was not observed at site C. In terms of abundance and biomass, site B did not appear to conform to normal succession models with both of these parameters peaking in the second year of the study. Both the abundance and size ratios appeared curiously constant throughout the survey. The biogeochemical parameters

appeared more as expected with relatively steady falls in pore-water ammonia, nitrogen and carbon and with a steady rise in redox potential. The MDS plot for site B (Fig. 12), like site A, shows 2 'setback loops' reflecting the winter decreases in *S*.

SITE B1, Stress = .06

Fig. 12. MDS plot of Site B time series (square root transformed) species data. Roman numerals refer to sampling occasion (see text for dates).

Recovery at Site C

Site C differed from the other sites in that farming only occurred for a short period and left a small, localised but intense impact. This site started off with the lowest *S* and redox, but highest *A* of the 3 sites and with a much higher biomass than site B. In terms of the benthic and biogeochemical descriptors, recovery appeared relatively rapid with the major improvements within the first 6 months, although the taxa present showed evidence of perturbation throughout the study. The MDS plot for site C (Fig. 13) shows only one 'setback loop' which is consistent with the less tortuous recovery trajectory of all the parameters measured compared to the other sites. This loop at sampling event 5 (autumn 1993), corresponds to the massive change from polychaete to nematode dominance which occurred at this time.
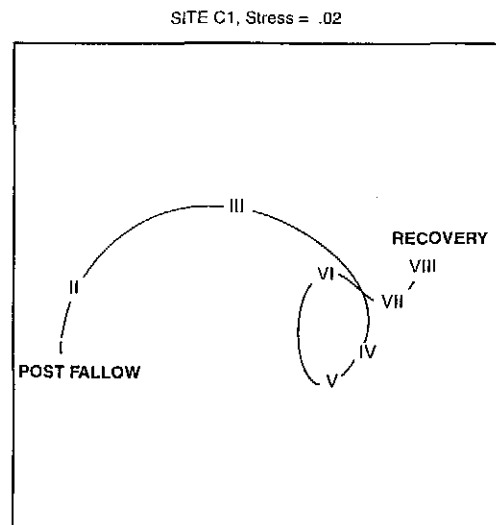
SITE C1, Stress = .02

Fig. 13. MDS plot of Site C time series (square root transformed) species data. Roman numerals refer to sampling occasion (see text for dates).

Benthic recovery assessment

No single benthic or biogeochemical parameter can be used to assess habitat change in this environment. Species number S and redox potential give good measures of sedimentary recovery although organic carbon and nitrogen were of some utility for sites A and B but not for site C. From Table 2 the shift in trophic guilds with time, is also of utility in detecting recovery at sites A and B. At site A post fallowing, the benthos was dominated by surface and sub-surface deposit feeders, either nematodes or capitellids, until autumn 1993. At this time omnivorous nematodes still dominated, but the motile omnivore *Ophryotrocha* sp. became the second most numerically dominant taxon. As filter feeders appear to be displaced when organic enrichment is highest, the emergence of this guild may be argued to represent recovery, thus the presence of several filter feeders in the summer of 1993 heralded an ephemeral recovery. This was followed by the crash described above, when filter feeders disappeared from site A, only reappearing in the last two samples.

Site B is legitimately described as less impacted than site A, as the presence of filter feeders throughout the sample period demonstrates (Table 2). Although omnivorous nematodes dominated throughout, the second most numerically dominant taxon varied from sub-surface deposit feeders initially, to eventually filter feeders.

As discussed previously, analysis of trends in recovery at site C proved problematic. Curiously, Site C was only dominated by nematodes from the second year of the study, with deposit feeding capitellids followed by omnivorous *Ophryotrocha* sp. dominating at the beginning of the study. Site C also had filter feeders present throughout, with five taxa of this guild present in the dominants (Table 2) during autumn 1993, and three present at the last sampling event. The presence of five filter feeding taxa in the dominants in the autumn 1993 sample is somewhat surprising, given that this was during a 'setback loop' discussed above, when recovery was seen to be arrested, given the assumption relating filter feeders and recovery made above.

Comparisons of physical sedimentary data with other fish farm studies are limited, as there have been few published works on farms in production, and none to date on recovering sediments. The variability in minimum Eh_{4} values reported above, however, (-42 mV at A1, -64 mV at B1 and -148 mV at C1) compare with those values found under salmon cages reported elsewhere. In their study of Scottish salmon farms, Brown *et al.* (1987) reported a minimum Eh_{4} of -186 mV; Weston (1990) reported a Eh_{4} of +120 mV immediately adjacent to a functional cage in Puget Sound, Washington USA, and one of the present authors measured Eh_{4} at -197 mV directly beneath a Scottish fish farm site (T.D. Nickell, unpublished data). Karakassis *et al.* (1998) showed Eh_{4} values (taken from their graphs) of ~ -200 to -150 mV. Iwama (1991) considered Eh values of < -150 mV to indicate anoxic sediments, thus the present redox conditions do not fall at the extreme end of the enrichment scale.

Conversely, present organic carbon values exceed by an order of magnitude those reported by Weston (1990), who, using similar methodology, found a maximum surficial value of 1.26%, compared to the initial A1 value of 10.09% reported here. Background organic carbon values in naturally enriched Scottish sea lochs are highly variable. In comparison with areas not associated with fish farms, Nickell *et al.* (1995) found Loch Sween organic carbon values to range from 3.6 - 7.8%, and Ridgway and Price (1987) observed surface sediment levels in Loch Etive outer basin to be 5.2%, whilst inner basin levels were as high as 7.4%, thus the elevated levels presented here are perhaps not surprising, albeit entirely at odds with the value of ~ 21% reported by Karakassis *et al.* (1998) in their study of a hydrographically quiescent site in Cephalonia. Present organic N values (A1, 1.55%; B1, 0.80%; C1, 0.44%) were far in excess of the 0.16% reported by Weston (1990) at his 0 m station, corresponding to the elevated carbon levels reported above, whilst far less than the ~ 2.6% found by Karakassis *et al.* (1998) in the Mediterranean, further reinforcing the site specificity encountered in this study.

Maximum sediment pore water ammonia levels are also difficult to place in a wider context (A1, 714.40 μM; B1, 564.57 μM; C1, 302.52 μM); Wildish *et al.* (1993) measured peak ammonia values from the overlaying water of 14 μM, so there are no direct nutrient comparisons available at this time.

Site rotation and recovery

The practice of site rotation to allow the benthic environment to recover appears superficially to make sense in terms of sustainable development of fish farming. We have shown that recovery processes can be measured at fish farm sites after the cessation of farming but it is hard to put exact rates on this recovery given the differing site histories, hydrographic regimes and neighbouring communities. It does however appear that all three sites showed recovery of *S* after approximately one year but disturbed communities in terms of indicator taxa after 2 years. The rapid relapse of site A after the putative fresh input in 1993 indicates that recovering sediments are very sensitive to additional stress and sediment conditions quickly revert to their former impacted state. This brings the rationale behind site rotation into some question as gains are hard won and quickly lost. Since site rotation implies that an additional site is being used, the total area impacted will be greater than if rotation had not taken place. A more rational policy might be to maintain inputs such as to maximise benthic biomass, thus maximising the sedimentary organic material processing capacity and minimising the area of seabed disturbed.

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