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Evaluation of Sediment Recovery After Removal of Finfish Cages from Marine Farm Lease No.76 (Gunpowder Jetty), North West Bay.

Catriona K. Macleod, Iona M. Mitchell, Christine M. Crawford and Robert D. Connell

Summary

Sediment condition at Gunpowder Jetty in North West Bay, Tasmania was evaluated to determine both the rate and extent of recovery after removal of finfish cage aquaculture operations in August 1999. Replicate samples were collected over 24 months from stations directly under, adjacent to and 10m, 20m and 35 m from three randomly selected cages. Sediment particle size distribution, organic matter content and sulphide levels were measured and benthic infaunal community structure was determined. Current flows and directions for the lease area were recorded and sediment conditions were also assessed by video for comparison.

The results indicate that the site was highly depositional. Current flows were low and tidally driven, although prevailing weather conditions could significantly influence the hydrodynamics of the system at times. The cage sediment was highly impacted at the time of cage removal; exhibiting high sulphide levels and a community structure clearly indicative of polluted/ hypoxic conditions. The extent of impact diminished rapidly with both time and distance from the cages. The influence of the cages was not generally detectable at 35m and after 2 months transitional conditions were indicated at the cage stations. However, after 24 months, although the sulphide levels at the cage locations had returned to reference conditions, the benthic community structure still differed significantly from that of reference stations.

Several species were identified as indicative of impacted, transitional and recovered conditions. *Capitella capitata* complex and *Malacoceros tripartitus* were clearly dominant under impacted conditions; *Theora fragilis* , *Corbula gibba* and *Euphilomedes* sp. (MoV 18) indicated the transitional community; whilst *Amphiura elandiformis*, *Lysilla jennacubinae* and *Nucula pusilla* characterised unimpacted conditions.

Major impacts were obvious in sulphide levels and with video assessments; shown by the presence of bacterial mats, blackened sediments and gas bubbles. However, the macrobenthic community structure better reflected differences in sediment condition and the rate of recovery.

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1. Introduction

The increased deposition of organic material associated with aquaculture waste, such as excess feed and faeces from the finfish cages, can result in areas of localised organic enrichment (Brown *et al*., 1987; Lumb, 1989; Holmer and Kristensen, 1992; Hargrave *et al*., 1993)). The ability of sediments to assimilate such increases in organic load is variable and depends on both abiotic and biotic factors (Lumb, 1989; Chang and Thonney, 1992; Wu *et al*., 1994). Typically, the flora and fauna of impacted sediments adapt to utilise this new nutrient source, resulting in changes in benthic community structure. However, if the sediment's capacity to assimilate organic inputs is exceeded and the sediment becomes anoxic, the sediment biogeochemistry will be altered towards a system dominated by anaerobic forms of metabolism and toxic degradation products (hydrogen sulphide and ammonia) can be released into the environment affecting farm production and aquatic ecosystem health (Gowen *et al*., 1988; Rosenthal and Rangeley, 1988; Holmer and Kristensen, 1992).

To overcome this it is usual for farmers to leave areas of seabed free from farming activities for a period of time to enable them to recover, a process commonly referred to as fallowing. However, it is currently not clear to what extent recovery of the sediment occurs or to what degree the natural environmental conditions can influence recovery. It is important from the perspective of both farm management and ecosystem protection to have a better understanding of the processes involved in recovery. There have been many studies of the effects of organic enrichment on sediments under cages, (eg Brown *et al*., 1987; Ritz *et al*., 1989; Weston, 1990; Woodward *et al*., 1992; Holmer and Kristensen, 1992; Findlay *et al*.,1995; Cheshire *et al*., 1996; Hargrave, *et al*., 1997). Several recent reviews of the impacts of aquaculture (Iwama, 1991; Gowen and Rosenthal, 1993; Wu, 1995; Black, 2001) have suggested that the magnitude and scale of impact is dependent on both husbandry parameters and the physical, chemical and biological characteristics of the environment. Husbandry practices will vary according to the species farmed, the culture techniques and the type and origin of feed used. However, there have been fewer studies relating to the length of time required for sediment recovery after cessation of farming and the results have differed markedly, with estimates of benthic infaunal recovery ranging from 7 weeks (Ritz *et al*., 1989) to 21 months (Black, 2001) and greater than 23 months (Karakassis *et al*., 1999). The benthic infaunal recovery patterns described in all of these studies appear to show a reversal of the successional pattern originally described by Pearson and Rosenberg (1978) for organic enrichment. Two of these studies examined the effects of Atlantic salmon (*Salmo salar*) culture; Ritz *et al*. (1989) in coastal waters off S.E. Tasmania and Black (2001) from the west of Scotland. However, the results from Karakassis *et al*. (1999) were from an investigation of sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*) culture in Cephalonia Bay, Greece. Consequently, the variability in the estimates of recovery is probably a combination of both ecosystem and farm management differences.

There have been significant changes in aquaculture farming practices in Tasmanian since the study by Ritz *et al*. (1989) and it is therefore unlikely that the rates observed at that time are still representative. Although the culture species differed in the more recent studies (of Karakassis *et al*., 1999; Black, 2001) husbandry practices are likely to be more comparable. Black (2001) contrasted his results with those of Karakassis *et al*. (1999) and noted that the rates seemed to be very much higher in the warmer waters but cautioned that local hydrographic conditions will have a considerable influence on recovery rates, noting that in quiescent areas recovery may take much longer than in more hydrodynamically energetic areas.

Against this background, the present study was a joint initiative of the aquaculture industry (Aquatas Tasmania Australia Pty Ltd) and the Department of Primary Industries, Water and Environment (DPIWE) and was developed with the aim of assessing the rate of sediment recovery after removal of all salmon cages from a lease located in North West Bay, South-East Tasmania. It is clear that there is a very broad range of factors which may influence the sediments recovery rate, but one of the principal objectives of this research was to identify the changes/stages in both the benthic faunal community structure and the physical/ chemical status of the sediments over time associated with long term fallowing of an intensively farmed marine finfish cage site in the cool temperate waters of Tasmania.

This report presents the results from the benthic, video and physical-chemical assessments over twenty-four months of sampling (August 1999 – September 2001). It was envisaged that these results would provide useful information on the rate of sediment recovery and the benthic changes involved, which would be incorporated into farm management protocols to facilitate the optimal management of lease areas and ensure the sustainability of ongoing operations.

2. Methods

2.1 Sample Site Location

The Gunpowder lease (lease 76) located towards the eastern shore of North-West Bay in the D'Entrecasteaux Channel was first granted in 1985 to Aquatas Pty Ltd and operated as a commercial salmon farming operation until August 1999. When in operation, the farm was a smolt ongrowing site, occupying an area of 3.12ha, with up to 16 production pens including both 60 and 80m circumference polar circles. The lease area is relatively shallow (14-20m) and can be subject to elevated summer temperatures (DPIF, 1997). In the year prior to closure the farm stocked approximately 2-300 tonnes of fish; however stocking levels were markedly reduced in the 3-4 months prior to the site's closure as stock were transferred from the site (S.Percival (Aquatas), pers. comm.). Over the preceding four years this site had been stocked more or less continually with little or no fallowing, adjustment of stocked biomass according to the time of year (water temperatures) was the only response to environmental conditions (S.Percival (Aquatas), pers. comm.). Relatively good fish performance was achieved at the site and farm management considered the growing conditions of this lease to be "average". Current fallowing and feed management practices at Aquatas are reported to have changed significantly from those reported in the five year period prior to closure of the Gunpowder lease (S.Percival (Aquatas), pers.comm.).

Prior to the removal of the cages, in late August 1999, three cages were selected at random for this study (Fig. 1). At each of these cage locations fixed transects were positioned on the seabed running from directly beneath the cage (-10m) to a distance 35m from the cage boundary (Fig. 2). Sample stations were established at –10m, 0m (cage edge), 10m, 20m and 35m. The –10m station was not established until the second sample visit in early September 1999 (0.5 months) when the empty cages had been removed. The positioning of the reference stations was based on the results of both overseas and local experience. Although the actual extent of impact will vary at each site depending on its specific stocking density, feed input levels and current flow characteristics, benthic impacts from caged salmonid culture have generally been found to be restricted to between 25m (Brown *et al*., 1987) and 35m (Gowen *et al*., 1988; Macleod, 2001; Crawford *et al*., 2002), although subtle effects have been detected up to 100m from operational cages (Weston, 1990). The studies by Macleod (2001) in Port Esperance and on the Tasman Peninsula and by Crawford *et al.* (2002) in the Huon estuary both investigated the spatial impacts of cage farming under operating conditions very similar to those practiced at the Gunpowder Jetty lease prior to its closure. In consideration of these findings, references for each transect were located 150m from the cages, directly in line with the fixed transect (Fig. 1 and 2) and from similar depths. The positions of the 0m, 35m and reference stations at each transect were fixed using a differential global positioning system (DGPS).

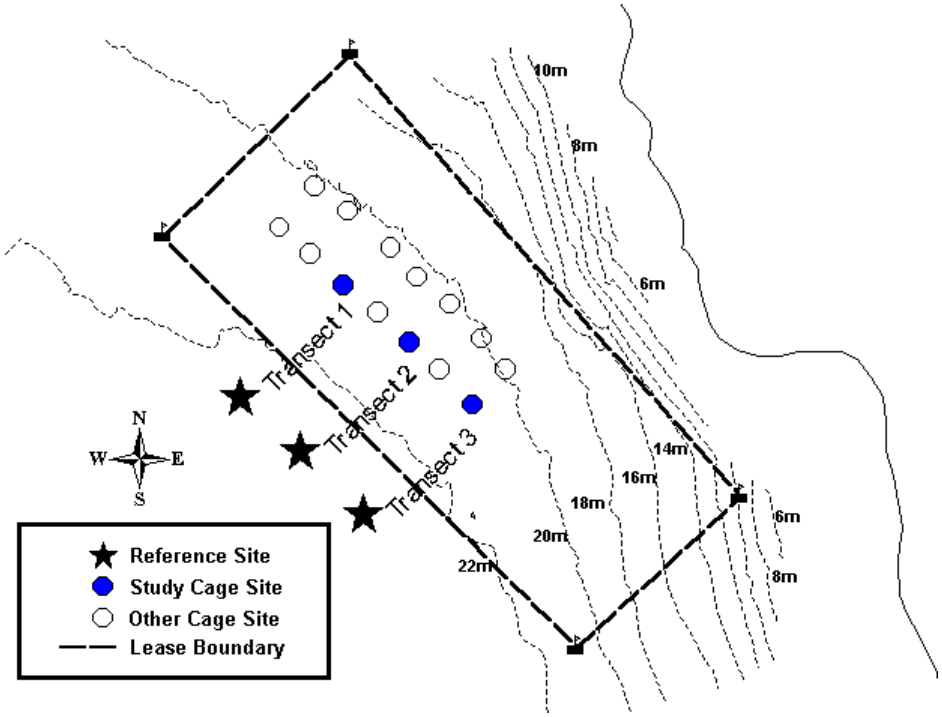


Fig. 1. Location of cage study sites and reference positions in relation to the Gunpowder Jetty lease.

The initial sampling was undertaken approximately one week after removal of the cages and samples were then collected after two weeks and after 1, 2, 4, 6, 8, 10, 12, 15, 18, 21 and 24 months. Samples were initially collected more frequently as it was anticipated that the initial recovery might be rapid.

Samples were collected by diver from each of the sample stations for macrofaunal and physical / chemical analyses. Video footage was collected along the transect line and from an area within a 2m radius of the reference station.

Fig. 2. Schematic of sampling positions along transects.

The farm provided daily feed input figures for each of the study cages for the 7 months prior to the cage removal.

2.2 Current flow data

An acoustic doppler current profiling (ADCP) meter was deployed at the site over a period of 34 days from 30/11/1999 until 3/1/2000 to measure current flow characteristics (magnitude and direction) throughout the water column. Data was collected hourly.

Analysis was undertaken on the original data, with no modification or transformations applied (i.e. the current velocity was not resolved into components of across-shore or long shore, tidal influences were not removed and the data were not adjusted for the influence of coastal geomorphology).

2.3 Physical / Chemical Analyses

At each sample station three replicate core samples were collected using perspex tubes 250mm long and 45mm internal diameter. Two holes/ports were located on opposite sides of the core tubes, 6cm from the top. This allowed measurements of redox and sulphide to be made at equivalent depths in the sediment. These holes were sealed with duct tape during deployment. In the laboratory the bottom bung was removed from the core tube and a purpose built plunger inserted, allowing the sediment within the tube to be pushed upwards whilst still maintaining a watertight seal. The sediment sample was positioned within the tube so that the holes/ports were at the appropriate depth for sediment chemistry sampling (1cm and 4cm). After the redox and sulphide measurements were obtained the sediment sample was extruded and sectioned for analysis of grain size and organic matter.

2.3.1 Granulometry

Half of the top 4cm from two cores was collected for sediment particle size analysis. A sub-sample of each was passed wet through a graded series of sieves (4mm, 2mm, 1mm, 500µm, 250µm, 125µm and 63µm). The sediment retained on each sieve was dried and weighed and the percentage of the total sample weight calculated. The fraction smaller than 63µm was determined by calculation of the difference between the initial sample weight and the combined weight of the retained fractions.

2.3.2 Organic Matter Determination

Total organic matter was determined by a modification of the loss on ignition technique (Greiser and Faubel, 1988). Samples collected from the top 4 cm of each core were homogenised and a sub-sample of approximately 2-5 grams taken. In order to remove excess carbonate from the samples, samples were sieved to remove large shell fragments and any remaining carbonate was neutralised by acidification with 1N HCl. The samples were oven dried for 24 hours at 60° before being transferred to a muffle furnace for 4 hours at 500°C. The weight of organic material was calculated as the difference between the oven dried and final furnace ashed weights.

2.3.3 Sulphide

Core samples were transported to the laboratory in a specially modified insulated container. This contained a sponge insert with positions for the core tubes. In order to maintain the samples at their ambient temperature the sponge was soaked in seawater. In the laboratory samples were processed as soon as possible and in the order in which they had been collected. Sub-samples (4ml) were taken from each replicate core at depths of 1cm and 4cm using a cut-off 5ml syringe. Sulphide was measured using a Cole-Parmer 27502-40 silver/sulfide electrode following the technique described by Wildish *et al*. (1999). Sulphide standards were prepared before each sampling event and electrode calibration curves were determined.

2.3.4 Redox

Redox potential was measured using a WTW Microprocessor pH Meter with a combination Mettler Toledo pH/redox probe at 1 and 4cm. The probe was calibrated between core readings using Zobells ferro/ferricyanide solution and allowed to equilibrate for 10 seconds before taking each reading. Results were corrected to the hydrogen reference electrode. This data was discarded because contamination of the redox probe occurred several times during this study. This can result in falsely elevated measures in samples with low redox potential and consequently differences between sites appear reduced. Redox data obtained prior to detection of the problem may also be in doubt and therefore results for redox have not been included in this report.

2.4 Macrofaunal assessment

At all sample sites five replicate samples were collected for assessment of the benthic macrofaunal community structure using hand held 150 mm diameter PVC pipe corers to a depth of 100mm (sampling area of 0.0177m^2). Samples were collected by diver and transferred immediately to mesh bags (0.875mm), on the boat the bags were rinsed and transferred to containers of 4% formalin for fixation of the fauna. In the laboratory each sample was sieved to 1mm, sorted and the animals retained were identified to the lowest possible taxonomic level and enumerated.

2.5 Video

Video footage was obtained using a Hi-8 underwater colour video camera. Video recordings were assessed at each of the sample stations and environmental variables were scored as an average value for all frames observed 2 m either side of the sample stations. Videos were scored according to the criteria described by Crawford *et al*. (2001).

2.6 Statistical Analysis

Univariate data were analysed by Analysis of Variance (ANOVA) with homogeneity of variances being checked using box-plots. Data were untransformed. A two-way fixed effects model ANOVA with factors station and time was used to assess variations in particle size, macroinvertebrate abundance, number of species and diversity. Tukey's post-hoc test was undertaken for subsequent pairwise comparisons to determine which sites/groups were significantly different.

Multivariate analyses were conducted on the community assessment results to identify patterns within the data and where possible quantify and describe these patterns. Replicates were combined and the data was subjected to a square root transformation in order to adjust the importance of species dominants. Two-way crossed analysis of similarities (ANOSIM) was conducted on the *a priori* groupings of cage (-10m and 0m stations), farm (10m and 20m stations), boundary (35m station) and reference station at each sample time to test for the null hypotheses H1 - that there are no differences in community composition among the *a priori* groups within each time and H2 – that there are no differences over time allowing for the fact that there may be differences between the *a priori* groups. Patterns in the data were identified using agglomerative hierarchial cluster analysis and the results displayed as dendrograms and ordination plots using multi dimensional scaling (MDS). SIMPER analysis was used to determine if any particular species were indicative of these patterns (Clarke and Warwick, 2001). In order to more clearly identify patterns in the community data, each of the *a priori* station groupings were also analysed independently in relation to the reference conditions. Video and biotic data sets were compared using RELATE, which evaluates the rankings in each of the underlying similarity matrices through a rank correlation co-efficient and applies a simple permutation test of the null hypothesis that there is no relation between the two matrices. All the multivariate analysis techniques were included in the Plymouth Routines in Multivariate Ecological Research (PRIMER) software package.

The macrofaunal data were also compared using several univariate indices (richness - number of taxa, abundance - total number of individuals and the Shannon index (Shannon and Weaver, 1963)) which have been generally applied in environmental impact assessments (Warwick, 1993) and which have been particularly useful in determining aquaculture impacts (Brown *et al*., 1987; Lim, 1991; Wu *et al*., 1994).

3. Results

On several occasions it was found that transect 3 had been moved between sampling trips. On each occasion the line was re-positioned and samples were collected, but as there was some uncertainty regarding the exact position of the samples it was decided not to include the results from transect 3 in the final analysis.

3.1 Farm Production Information

Early in 1999 cage 3 was fed more than the other two cages however, feed input was markedly reduced at all cages from May and only relatively low amounts of feed were administered over the last 3 months before cage removal (approx. 75kg / cage / day) with very little difference in the input amounts between the cages (Fig. 3).

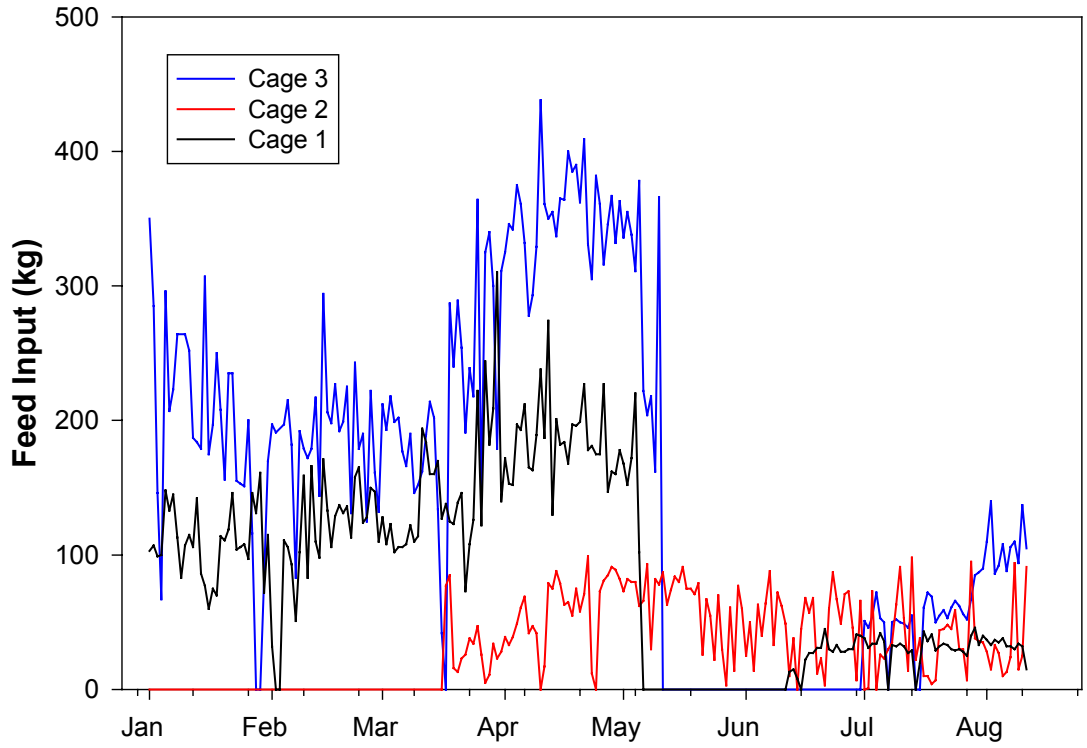


Fig. 3. Daily feed input (kg) for each of the study cages over the 7 months prior to cage removal.

3.2 Current flow

Current flow results are presented in 2m depth intervals from 3 m above the seabed to 17 m above the seabed (approximately 4m below the sea surface. Generally current flows were low, less than 10 cms^{-1} (Table 1). The average current velocity throughout the water column was approximately 3.4 – 4.3 cms^{-1}. However, near the seabed flows were further reduced with flows less than 5 cms^{-1} recorded for approximately 80% of the time. The maximum flow near the seabed was 10.9 cms^{-1} whilst near the surface the maximum flow increased to around 20 cms^{-1}.

The polar scatter plots for all depths show a tighter clustering of measures in the deeper waters signifying lower flows near the seabed (Fig. 4). The predominant water movement directions were north west and south east, reflecting tidal influences and running parallel with the depth contours. In the upper water column (i.e. 13-17 m above the seabed) the current velocity was overall slightly greater than at depth, but this increase was particularly evident when flows were in a north-westerly direction (Fig. 4). Most of the stronger flows towards the north west were a result of a three day period of strong winds between 28 –31 December 1999 (Table 2).

Table 1. Average, minimum, maximum current speeds (cms^{-1}) and percentage frequency of recorded flows < 5 cms^{-1}, < 10 cms^{-1} and > 10 cms^{-1} for 2m depth intervals (n = 829).

Depth range above seabed	Average (cms^{-1})	Min (cms^{-1})	Max (cms^{-1})	%< 5 cms^{-1}	%< 10 cms^{-1}	%> 10 cms^{-1}
3 - 5 m	3.5	0.1	10.9	79.9	99.3	0.6
5 - 7 m	3.4	0.0	13.0	80.0	99.2	0.6
7 - 9 m	3.4	0.1	13.0	80.9	99.0	1.0
9 - 11 m	3.4	0.0	15.9	79.7	98.7	1.3
11 - 13 m	3.6	0.0	19.7	78.5	97.1	2.8
13 - 15 m	3.9	0.1	18.2	76.0	95.4	4.1
15 - 17 m	4.3	0.1	20.1	66.9	94.3	5.3

The percentage frequency histogram plots of current direction show a bimodal pattern of flow (tidal) with the primary flows being towards the north west (arc range ~ 280^o to 340^o clockwise) and south east (arc range ~ 110^o to 170^o clockwise) (Fig. 5). However, overall the greatest flow was generally in a south easterly direction, i.e. outwards from the bay. Flows were recorded in all other directions but for the most part these were at considerably reduced and variable frequencies.

Table 2. Average, minimum, maximum current speeds (cms^{-1}) and percentage frequency of recorded flows < 5 cms^{-1}, < 10 cms^{-1} and > 10 cms^{-1} for 2m depth intervals (n = 78) during a three day strong wind period (28-31 December 2000)

Depth range above seabed	Average (cms^{-1})	Min (cms^{-1})	Max (cms^{-1})	%< 5 cms^{-1}	%< 10 cms^{-1}	%> 10 cms^{-1}
3 - 5 m	4.8	0.4	10.9	59.0	96.2	3.8
5 - 7 m	4.7	0.2	10.5	51.3	98.7	1.3
7 - 9 m	5.2	0.3	10.9	51.3	97.4	2.6
9 - 11 m	5.6	0.4	11.0	37.2	98.7	1.3
11 - 13 m	6.2	0.3	11.8	38.5	85.9	12.8
13 - 15 m	7.2	1.2	14.1	35.9	70.5	26.9
15 - 17 m	7.9	0.5	16.0	28.2	65.4	33.3

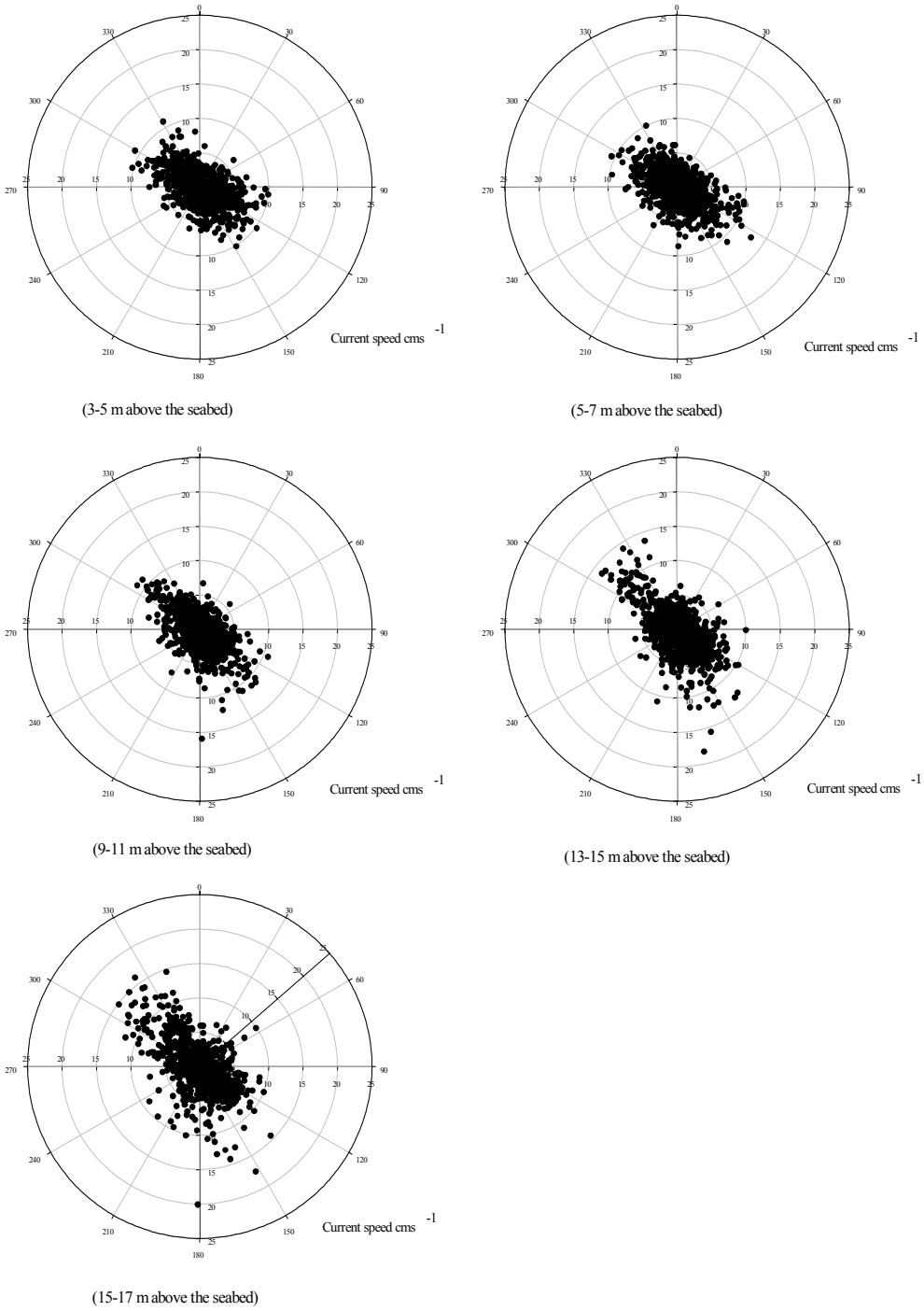


Fig. 4. Polar scatter plots of current speed (cms^{-1}) and direction (degrees) for 2m depth intervals (water profile range 3 – 17 m above the seabed).

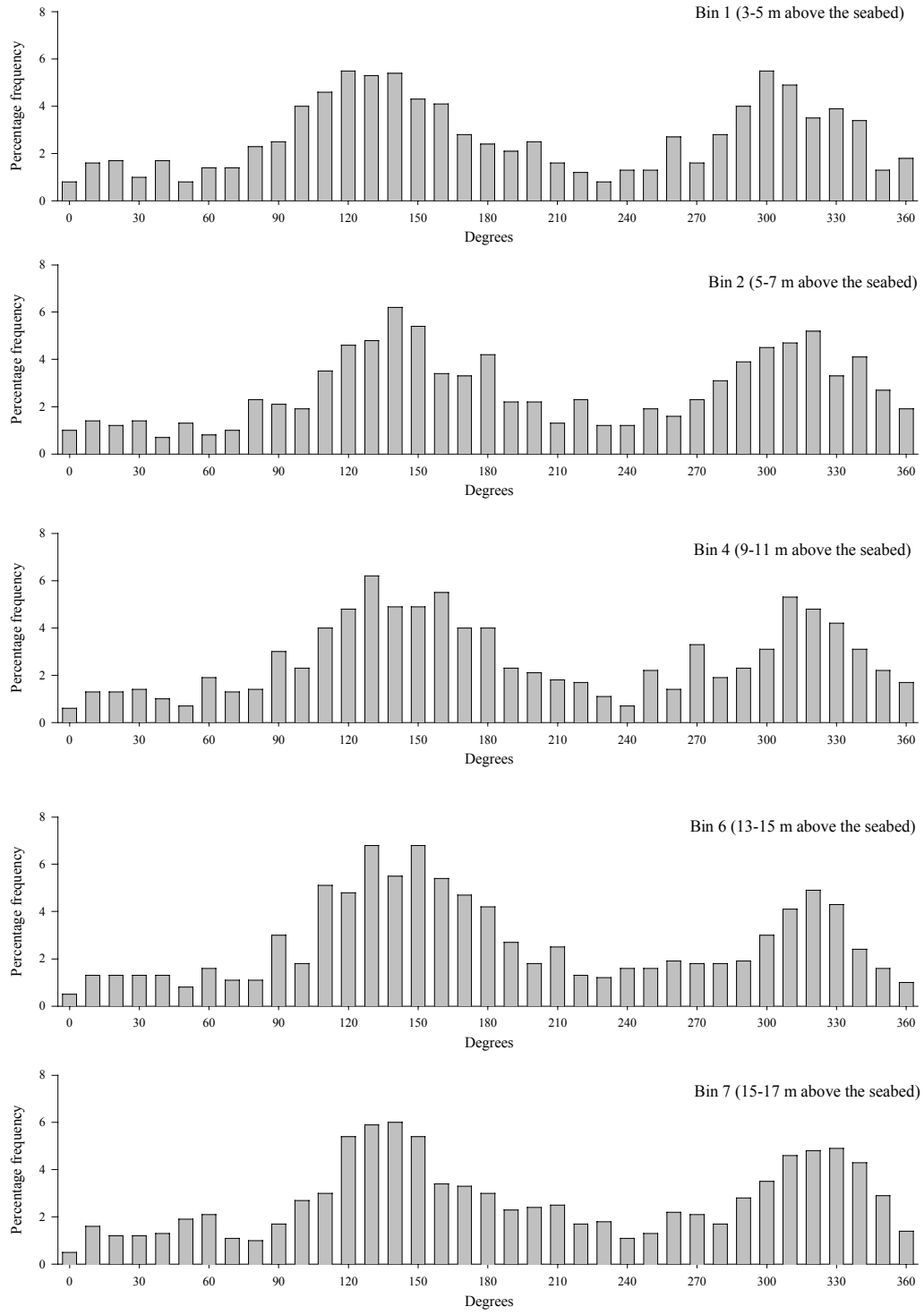


Fig. 5. Frequency histogram plots of current flow direction distribution (10^{o} increments, 0 – 360^{o}) for 2m depth intervals (water profile range 3 – 17 m above the seabed).

Towards the end of the deployment period very strong winds were encountered over a three day period (28 –31 December 1999). These had a considerable influence on current speed and direction throughout the entire water column. Flows near the seabed increased markedly (3.8 % > 10 cms^{-1}) but the effect was even greater in the upper water column (11-17 m above the seabed) where the frequency of flows greater than 10 cms^{-1} increased from around 4% to 30% (Table 2). This storm also affected the direction of water flow, changing the predominant flow to north westerly in the mid-upper water column (9-17 m above the seabed) and west to north-westerly near the seabed (3-7 m above the seabed), i.e. back into the bay.

3.3 Particle Size Distribution

Overall, there was little difference in the particle size distribution over time (Fig. 6). The predominant sediment type at all stations was silt/clay (<0.063 mm). However, in the first 12 months there was a greater proportion of particles in the >0.125 mm and >0.063 mm size classes at the –10m station. The 0m station also showed a slight increase in particles from these size classes at later sample times (15-18 months). At 8 months the 10m samples contained a higher proportion (11%) of particles in the largest size class (>4.0mm) than were recorded at any other time, mainly as a result of samples from transect 1 (data not shown). Similarly at 10 months the 0m station also recorded an increase in particles in the > 4.0mm size class.

The silt clay (<0.063mm) fraction was significantly different between stations (Fig. 7) (2-FACTOR ANOVA: N=207, df=5, F=72.279, p<0.001). Post-hoc comparisons showed that at all sample times the proportion of silt/clay was significantly lower at the –10m and 0m stations than at any of the other stations.

3.4 Organic Matter Measurement

Initial organic matter levels were high at all stations (generally around 20%), especially at the –10m station (Fig. 8). At the –10m station the initial level (0.5 months) was significantly higher than at the 0m, 20m and 35m stations but did not differ significantly from the reference station (ANOVA: N=24, df=5, F=3.681, p=0.018). Levels declined at all stations over time; a reduction of between 60-70% from the start to end of study. After 24 months there was no significant difference in organic matter content at any of the sample stations (ANOVA: N=24, df=5, F=1.792, p=0.165).

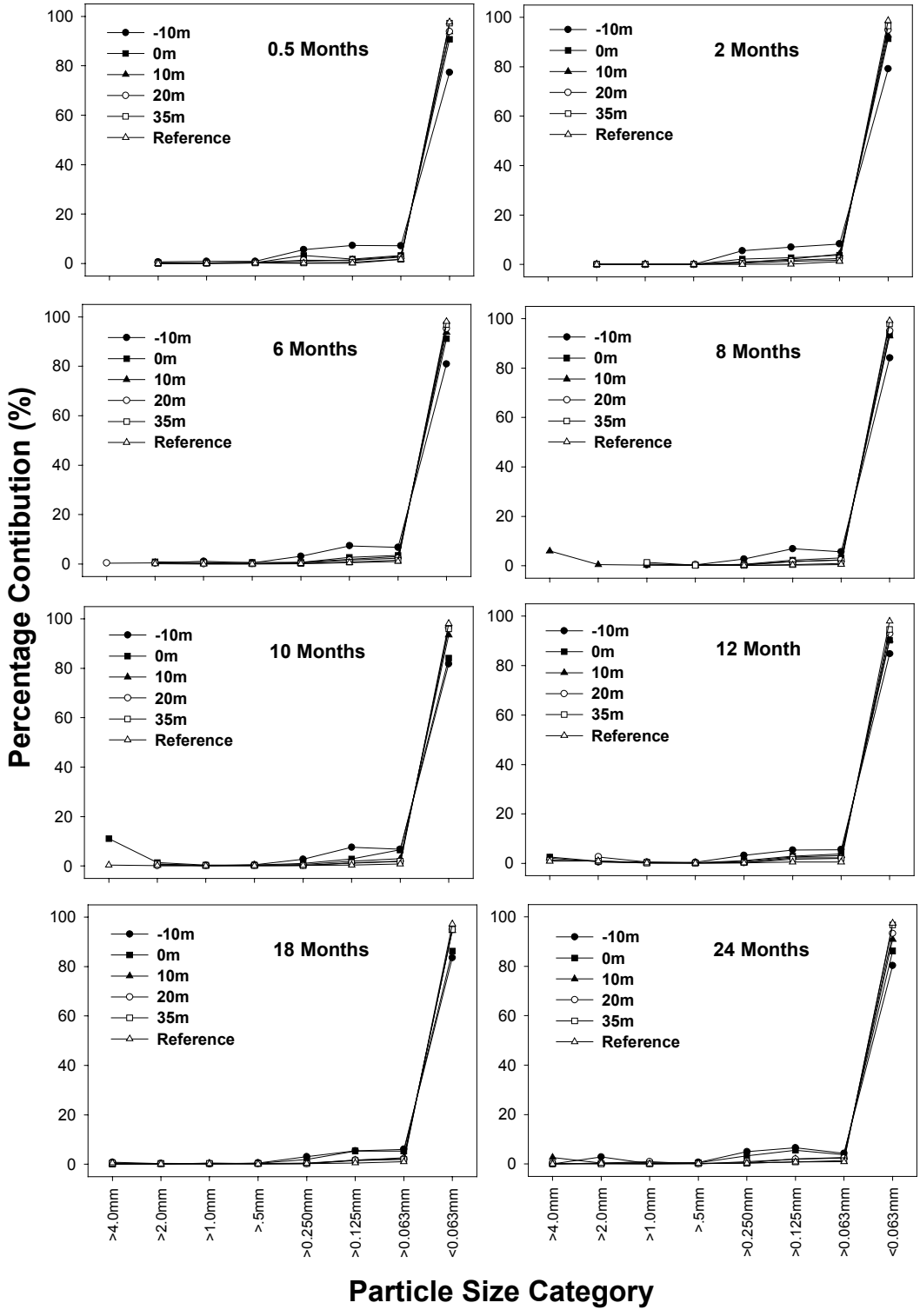


Fig. 6. Mean percentage contribution of each particle size category for each station at 0.5, 2, 6, 12, 18 and 24 months.

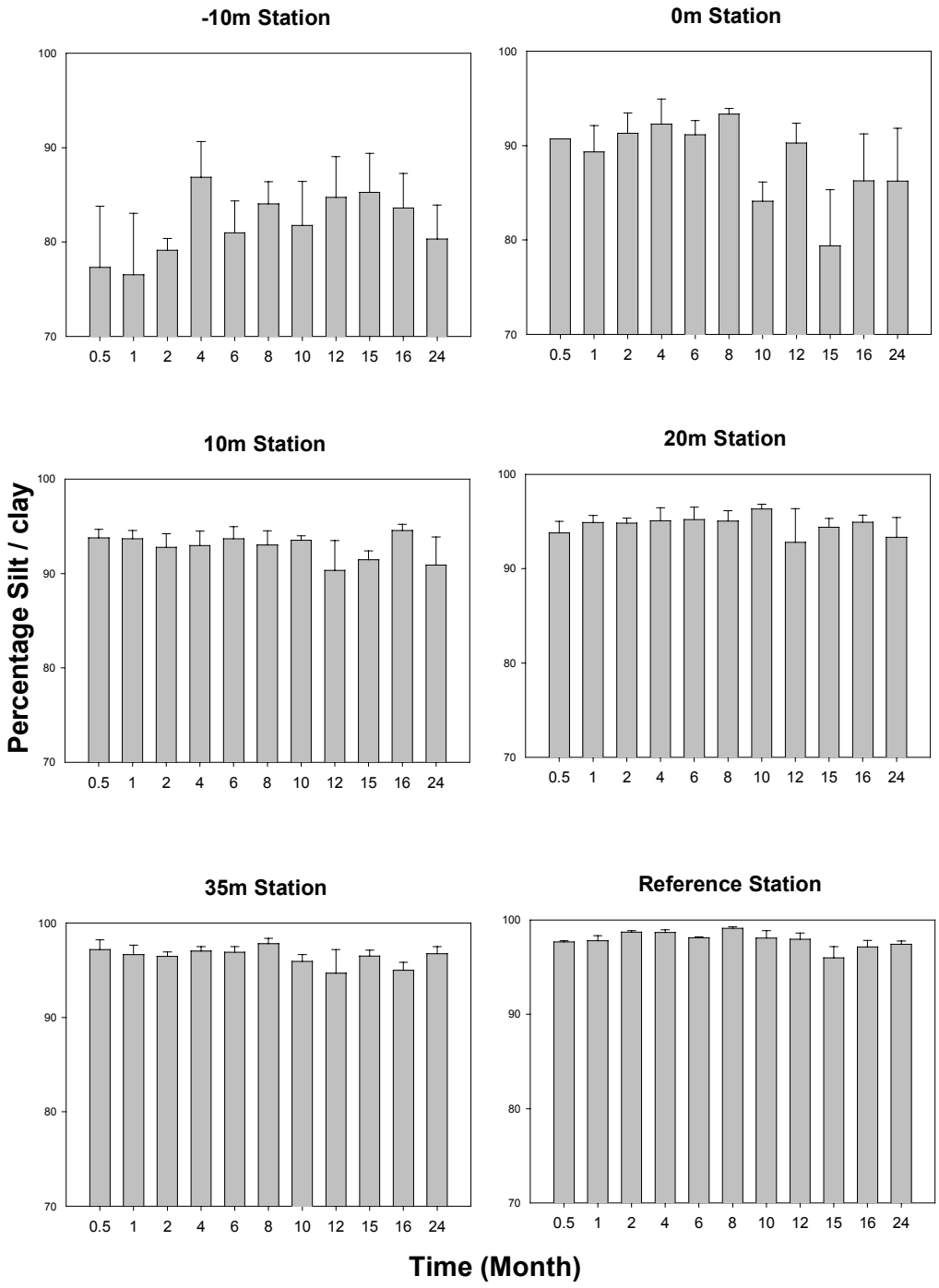


Fig. 7. Percentage silt/clay (<0.063mm) at each station at each sample visit.

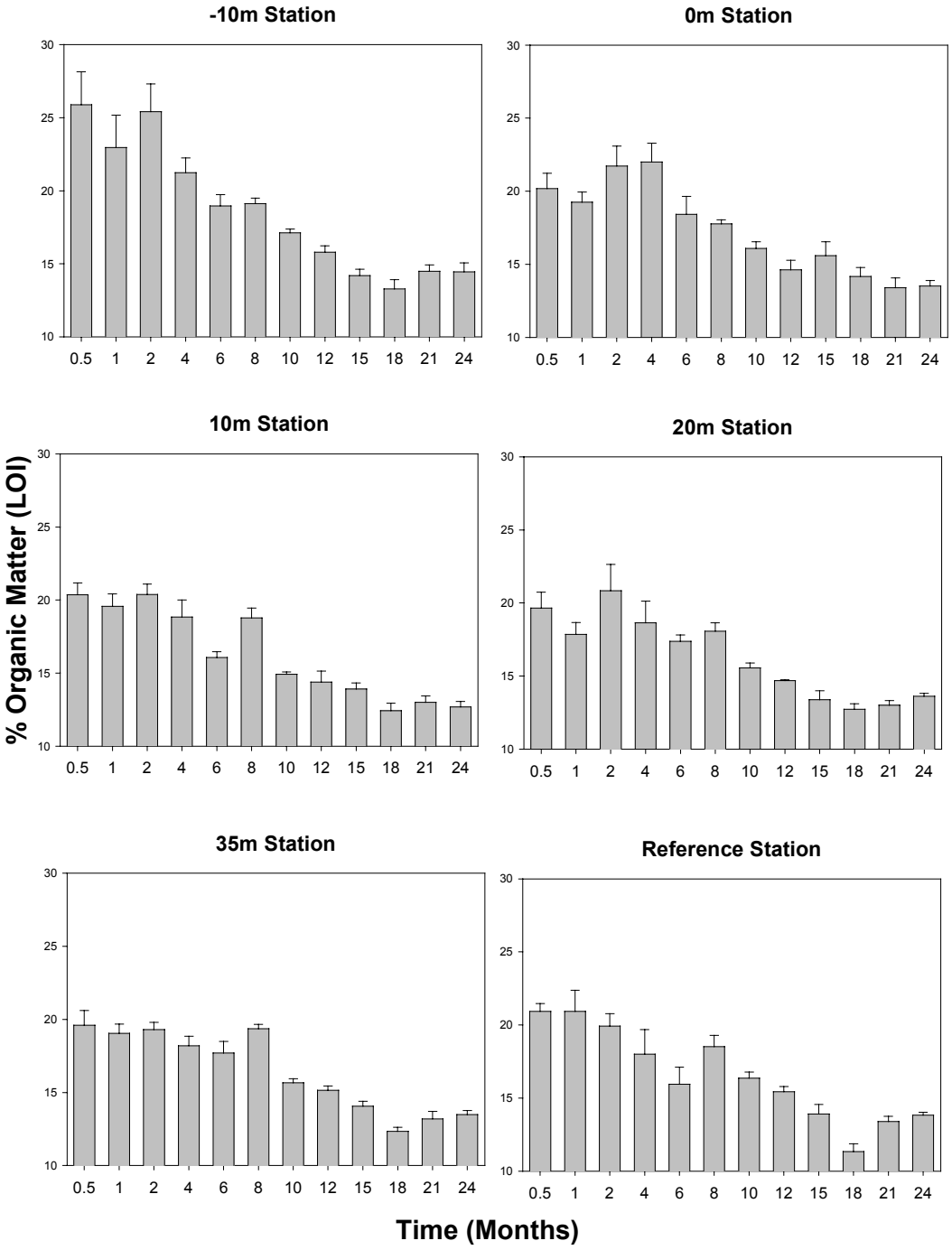


Fig. 8. Percentage organic matter content (+s.e.) at each station and at each sample site.

3.5 Sulphide

Sulphide levels exhibited a clear spatial and temporal gradation of effect (Fig. 9). The highest levels were encountered in the first 4 months at the stations nearest to the cages, with levels decreasing with distance from the cage and over time. The levels were generally similar at both of the measurement depths, although where high sulphide concentrations were encountered the difference between the two depths tended to be more variable.

The levels at the reference station were very low (less than 20uM) and often below the detection level of the probe. In contrast the sulphide levels at the cage stations (-10m and 0m) were much higher. One month after vacation of the lease levels were around 800uM and 550uM at the –10m and 0m stations respectively (Fig. 9). In comparison with the reference conditions these differences were highly significant (ANOVA 1cm: N=35, df= 5, F=15.103, p=<0.001; ANOVA 4cm: N=35, df= 5, F=23.683, p=<0.001). The sulphide concentration at the cage stations decreased markedly over time, and after 24 months levels at both the –10m and 0m stations were below 20uM.

The early samples from the 10m station (1-4 months) also had high mean sulphide levels but the concentration varied greatly between replicates and they were not significantly different from the reference stations.

3.6 Macrofauna

Two-way ANOSIM analysis of selected times (0.5, 2, 6, 12, 18 and 24 months) and the *a priori* defined groupings of cage (-10m and 0m), farm (10m and 20m), boundary (35m) and reference indicated that overall there were highly significant differences both between and within times and groups (Table 3).

Table 3. Two-way ANOSIM of groups and selected times (0.5, 2, 6, 12, 18 and 24 months) based on *a priori* group classification (Cage, Farm, Boundary and Reference)

Hypothesis 1. Test for differences between Times (averaged across all *a priori* groups)

Sample statistic (Global R): 0.298

Significance level of sample statistic: 0.1%

All pairwise test time combinations were highly significant (<0.005)

Hypothesis 2. Test for differences between *a priori* groups (averaged across all times)

Sample statistic (Global R): 0.423

Significance level of sample statistic: 0.1%

All pairwise test time combinations were highly significant (0.001)

Cluster analysis of the benthic infaunal data showed that the cage stations could be separated (-10m and 0m) from all other stations (Groups 1 and 2, Fig. 10). The 0.5, 1 and 2 month samples at the –10m station were differentiated at an overall similarity level of approximately 23% (group 1); whilst almost all the remaining cage stations were separated at a similarity level of around 35% (group 2). The remaining samples form a large group (group 3) with an overall similarity of around 40%. Within this main group the reference samples largely clustered together at a similarity level in the order of 47%.

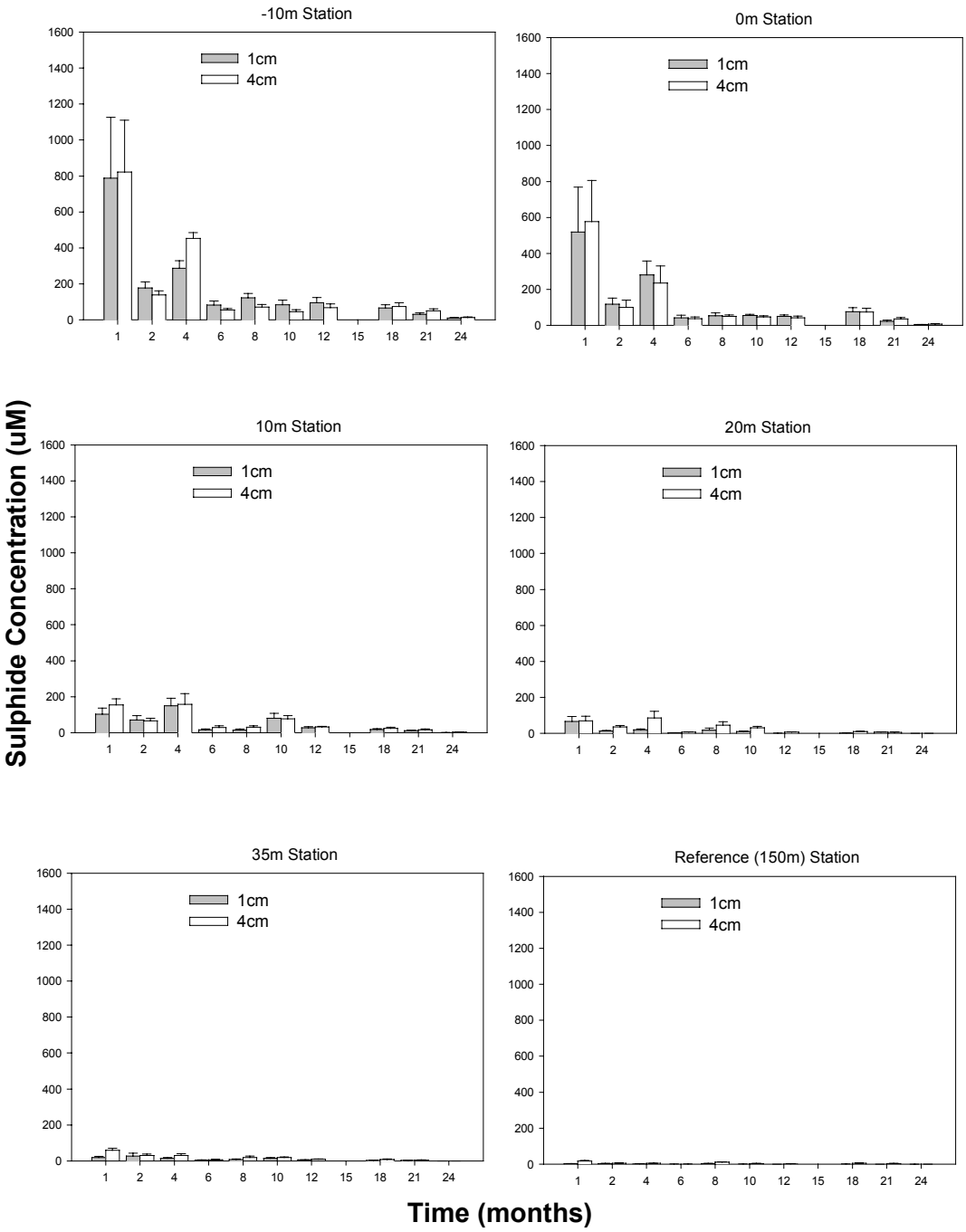


Fig. 9. Sediment sulphide concentration in µM at 1cm and 4cm depth (+ s.e.). (Data not available for 0, 0.5 or 15 months).

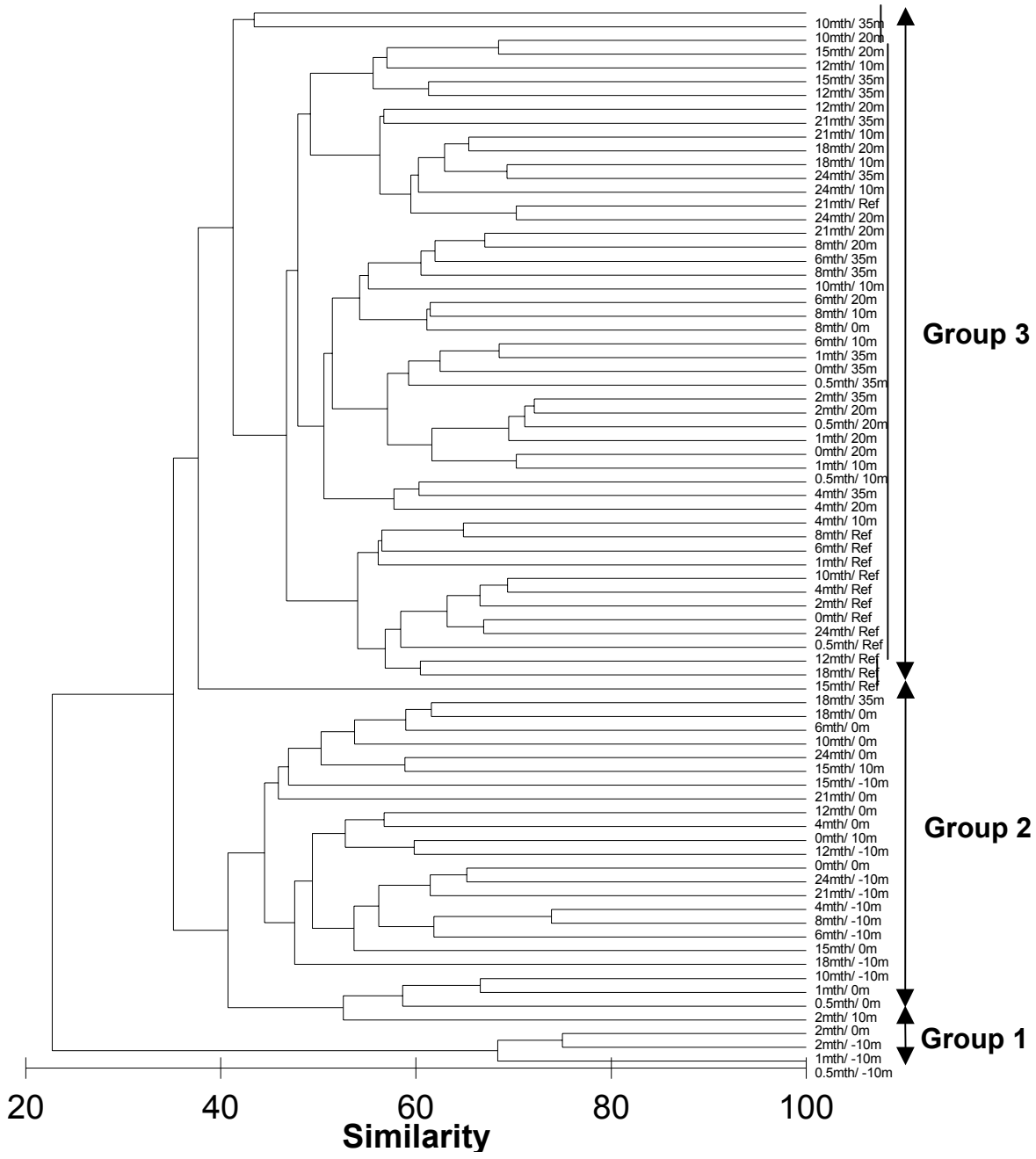


Fig. 10. Cluster analysis – Dendrogram for species abundance data.

The 2 dimensional ordination plot representing the cluster groups (Fig. 11) illustrates more clearly the changing community structure and consequent progression of stations both spatially and temporally. There is a pattern of spatial progression (left to right) represented by the community differences between the –10m stations, the 0m stations, the 10m stations and the 20m, 35m and reference stations. At the cage stations (-10m and 0m) a temporal gradation was also evident within the spatial distribution; the earliest cage samples tending towards the left and the later samples tending towards the right. The demarcation between the cluster groups is not well defined indicating that both the temporal and spatial changes in community structure were gradual rather than sudden.

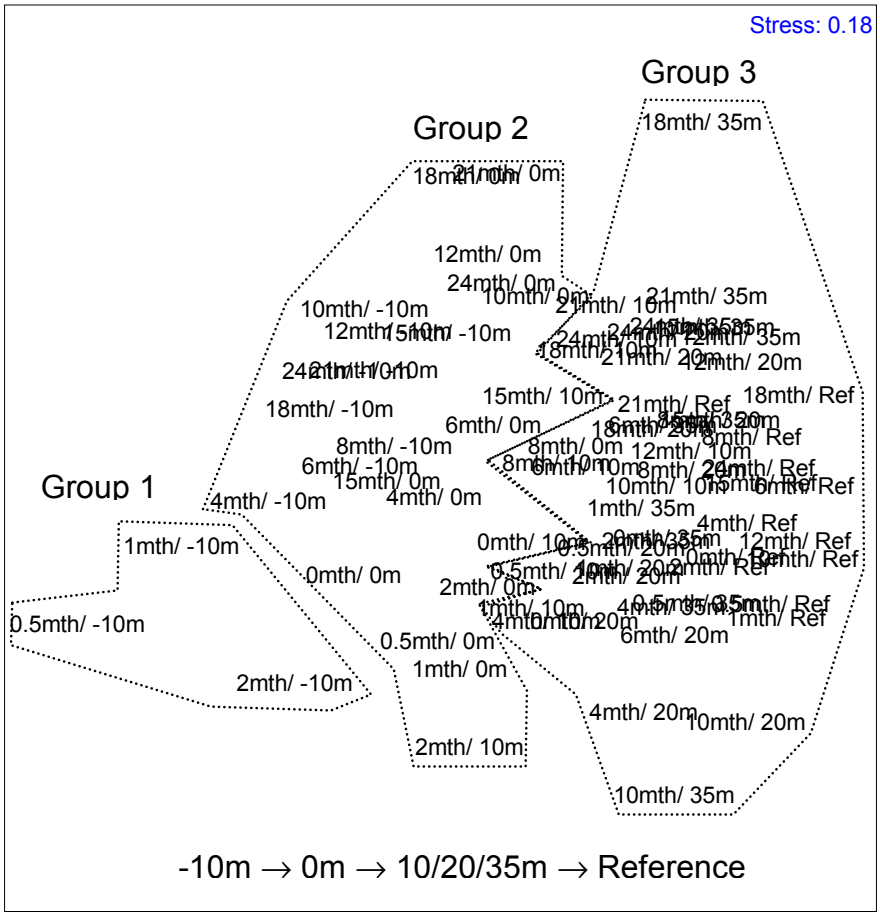


Fig. 11. Ordination analysis – 2-dimensional MDS plot of species abundance data. Stress=0.18.

SIMPER analysis identified that the main species contributing to the group separations (Table 4) were *Capitella capitata* complex and *Malacoceros tripartitus.* At the group 1 stations these two species account for approximately 93% of the overall group similarity. To achieve an equivalent level of group similarity for group 2 requires the contribution of 13 species. In group 2 the two most significant species, *Euphilomedes* sp. (MoV 18) and *Theora fragilis* only account for 36% of the group similarity. Both *Capitella capitata* complex and *Malacoceros tripartitus* were present at the group 2 stations, but at relatively low abundance levels and generally in association with either the -10m or 0m stations. Although not listed within the six most important species, the common brittle star *Amphiura elandiformis* was also an important species characterising unimpacted conditions, particularly in the early samples (Appendix 1).

To determine whether there were seasonal variations in community structure the reference stations were examined independently. Cluster analysis and ordination of the data (Fig. 12) indicated 3 groups, but there was no clear temporal pattern in the group distinctions. These groups were formed at very high overall sample similarity levels (>55%), suggesting a high degree of commonality. The species composition of these groups (Appendix 1) indicated that they were very similar and that the differences

between the groups resulted from subtle changes in abundance levels rather than species replacement.

Table 4. SIMPER output for the full community assessment indicating a) and b) average abundance per sample, ratio (average similarity / st.dev. similarity), % similarity and cumulative % similarity of the six most important species in each of the main groups and c) average abundance per sample, ratio (average similarity / st.dev. similarity) and cumulative % similarity of the three species which most clearly distinguish the main groups identified by cluster analysis

Species Name	Av.Abund.	Ratio	Percentage Similarity	Cumulative % Similarity
a) GROUP 1				
Capitella capitata complex	121.34	3.26	70.72	70.72
Malacoceros tripartitus	23.43	2.72	21.87	92.59
Nebalia longicornis	2.69	2.55	1.86	94.45
Nassarius nigellus	1.39	2.96	1.16	95.61
Prionospio kulin	1.00	2.60	1.12	96.73
Simplisetia amphidonta	1.03	3.19	0.91	97.64
b) GROUP 2				
Euphilomedes (MoV 18)	5.70	1.67	18.20	18.20
Theora fragilis	6.25	1.19	17.57	35.76
Capitella capitata complex	16.09	0.69	17.43	53.20
Corbula gibba	3.81	1.80	11.08	64.28
Simplisetia amphidonta	2.24	2.28	7.70	71.98
Nassarius nigellus	1.65	1.30	5.04	77.03
b) GROUP 3				
Euphilomedes (MoV 18)	4.32	1.71	15.40	15.40
Theora fragilis	3.02	1.87	11.86	27.26
Lysilla jennacubinae	1.61	1.87	7.34	34.60
Corbula gibba	1.70	2.04	7.06	41.66
Euchone limnicola	1.81	1.20	5.95	47.62
Nassarius nigellus	1.42	1.52	5.86	53.48

Species Name	Group 2 Av.Abund.	Group 1 Av.Abund.	Ratio	Cumulative % Similarity
c) BETWEEN GROUPS				
Capitella capitata complex	16.09	121.34	2.32	60.35
Malacoceros tripartitus	2.97	23.43	2.92	74.20
Theora fragilis	6.25	0.17	1.07	78.31

Species Name	Group 3 Av.Abund.	Group 1 Av.Abund.	Ratio	Cumulative % Similarity
c) BETWEEN GROUPS				
Capitella capitata complex	0.46	121.34	4.32	60.82
Malacoceros tripartitus	0.32	23.43	4.51	74.44
Euphilomedes sp.(MoV 18)	4.32	0.58	1.01	76.72

Species Name	Group 2 Av.Abund.	Group 3 Av.Abund.	Ratio	Cumulative % Similarity
c) BETWEEN GROUPS				
Capitella capitata complex	16.09	0.46	0.81	17.95
Theora fragilis	6.25	3.02	1.02	25.21
Euphilomedes sp.(MoV 18)	5.70	4.32	1.18	31.54

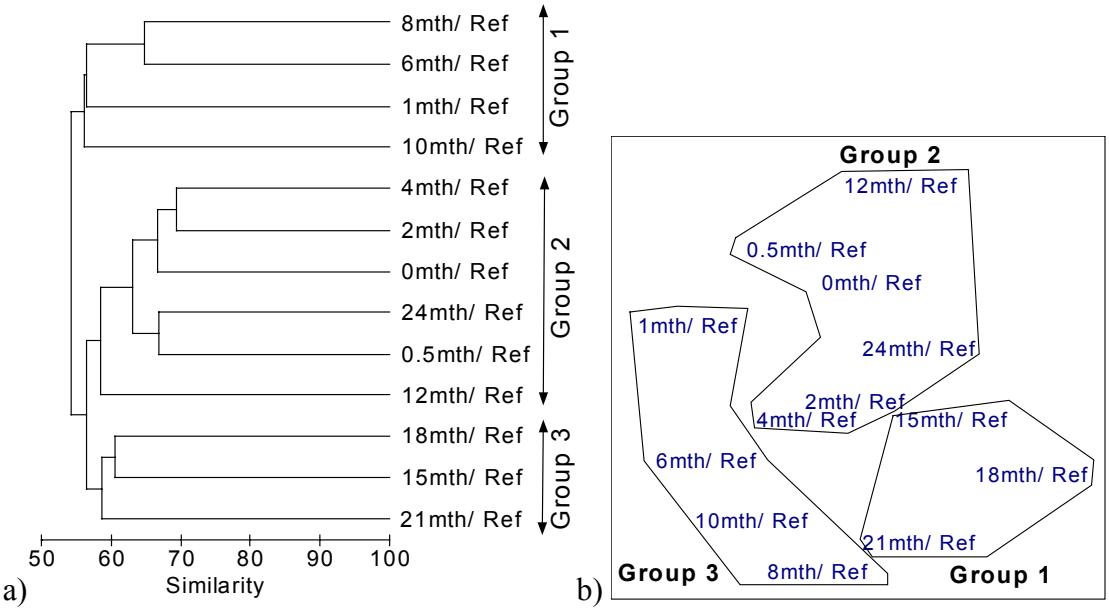


Fig. 12. Multivariate output for species abundance data for the reference stations. a) Cluster analysis - Dendrogram and b) MDS ordination plot (Stress=0.17).

The differentiation of the cage associated stations (-10m and 0m) from the reference stations was very clear (Fig. 13 a and b). The reference stations (group 1) separated from the other stations at a similarity level of 27%, forming a group with an overall similarity level of approximately 55% (Fig. 13 a and b). The initial samples from the stations directly under the cages (-10m, sampled in the first two months) separated from the remaining cage stations at a similarity level of 34% (group 2). The stations from directly under the cages for the first two months are positioned on the far right of the plot and the reference stations are on the far left of the plot, with the remaining cage stations forming a central group.

The relationship between the farm (10m and 20m stations) and reference stations can be seen in Figure 14. The 20m station at 10 months stands alone, primarily as a result of an increased abundance of *Amphiura elandiformis* (Species List –Appendix 1) . The remaining stations form three groups (Fig. 14 a and b); but at a high similarity level (approximately 44%). Group 3 contains the majority of the reference samples whilst groups 1 and 2 contain all the farm station samples. The earlier samples (first 6-8 months) are in group 1 whilst the later samples are in group 2 (Fig. 14 a and b). The MDS plot still shows a progression across the plot from right to left (group 1 to 3) and from the early farm samples through later farm samples to reference stations (Fig. 14b). However, the groups were not as clearly separated as the cage stations and references and the stress level for the plot is relatively high, 0.21. Differences in the species composition within each group (Appendix 1) indicate that the abundances of *Amphiura elandiformis*, *Lysilla jennacubinae* and *Nucula pusilla* were generally higher at the reference stations, although increased at the farm stations over time, whilst numbers of *Euphilomedes* sp. (MoV 18), *Theora fragilis* and *Corbula gibba* were on the whole lower at the reference stations but increased at the farm stations over time.

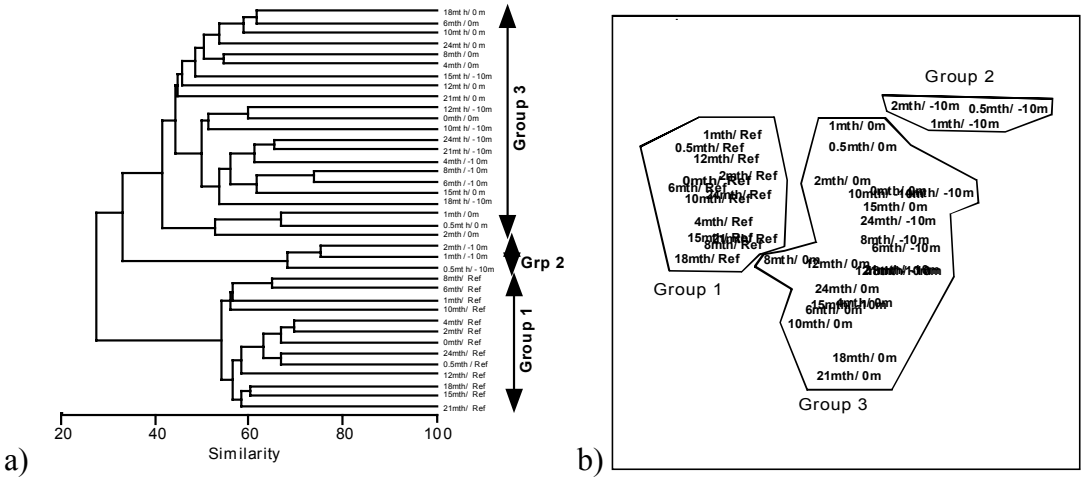


Fig. 13. Multivariate output for species abundance data for the reference and cage (–10m and 0m) stations. a) Cluster analysis - Dendrogram and b) MDS ordination plot (Stress=0.15).

Figure 15 a and b shows the association between the boundary and reference stations. Two stations were identified as distinct from the main groups; the 35m station at both 10 and 18 months. Reference to the species information (Appendix 1) indicates that the fauna at 18 months differed as a result of greatly increased numbers of three species of amphipod, *Jassa marmorata*, *Ericthonius* sp. and *Corophium ascherusicum* (all commonly associated with fouling communities), and which occurred in only one sample. The differences in the samples from 35m at 10 months could be attributed to a slight increase in abundance of Phoronid sp. and a reduction in the numbers of *Theora fragilis*. The communities at the remaining stations were very similar. It is difficult to discern any other groups in the data as all further dichotomies occur at sample similarity levels greater than 47%.

The numbers of species and abundance are presented as both total (the maximum number recovered from all replicates at each station) and mean (the average number found between the replicates at each station). Two-factor analysis of variance (station and time) for the univariate indicators of community structure indicated that there were significant interactions between station and time for all measures (number of species ($N=676$, $df=55$, $F=1.929$, $p<0.001$), abundance ($N=676$, $df=55$, $F=5.442$, $p<0.001$) and Shannon diversity ($N=676$, $df=55$, $F=2.956$, $p<0.001$)).

The mean number of species was lowest at the cage (–10m and 0m) stations, particularly over the first 2-4 months (Fig. 16). The total number of species at the –10m station was initially clearly lower than the reference stations but increased steadily over the first six months. The data also suggest that the total number of species was greatest around the autumn sample times (8 and 21 months). In contrast total and mean abundance levels (Fig. 17) were extremely high at the –10m station 2 weeks after the cages were removed (0.5 months) and declined rapidly although remaining considerably higher than reference conditions for the first 6-8 months. The Shannon index (Fig. 18) was generally low at all sites (below 2.0). However, although index values were above 1.0 at the reference, boundary (35m) and farm (10m and 20m) stations, they fell well below this level for the first 4 months at the –10m station.

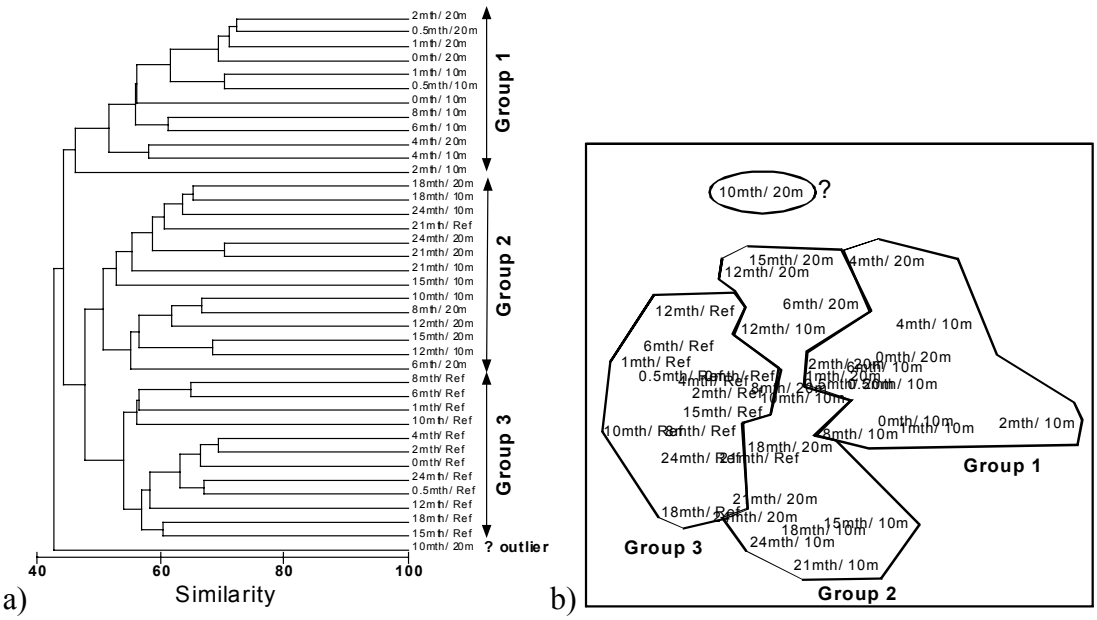


Fig. 14. Multivariate output for species abundance data for the reference and farm (10m and 20m) stations. a) Cluster analysis - Dendrogram and b) MDS ordination plot (Stress=0.21).

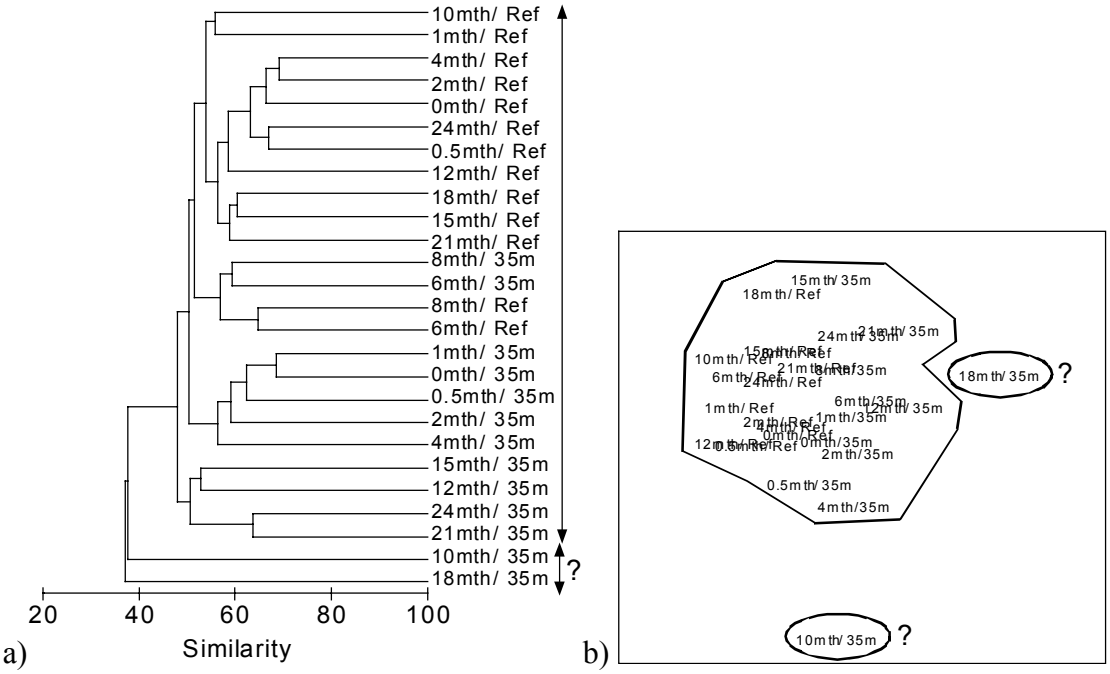


Fig. 15. Multivariate output for species abundance data for the reference and boundary (35m) stations. a) Cluster analysis - Dendrogram and b) MDS ordination plot (Stress=0.20).

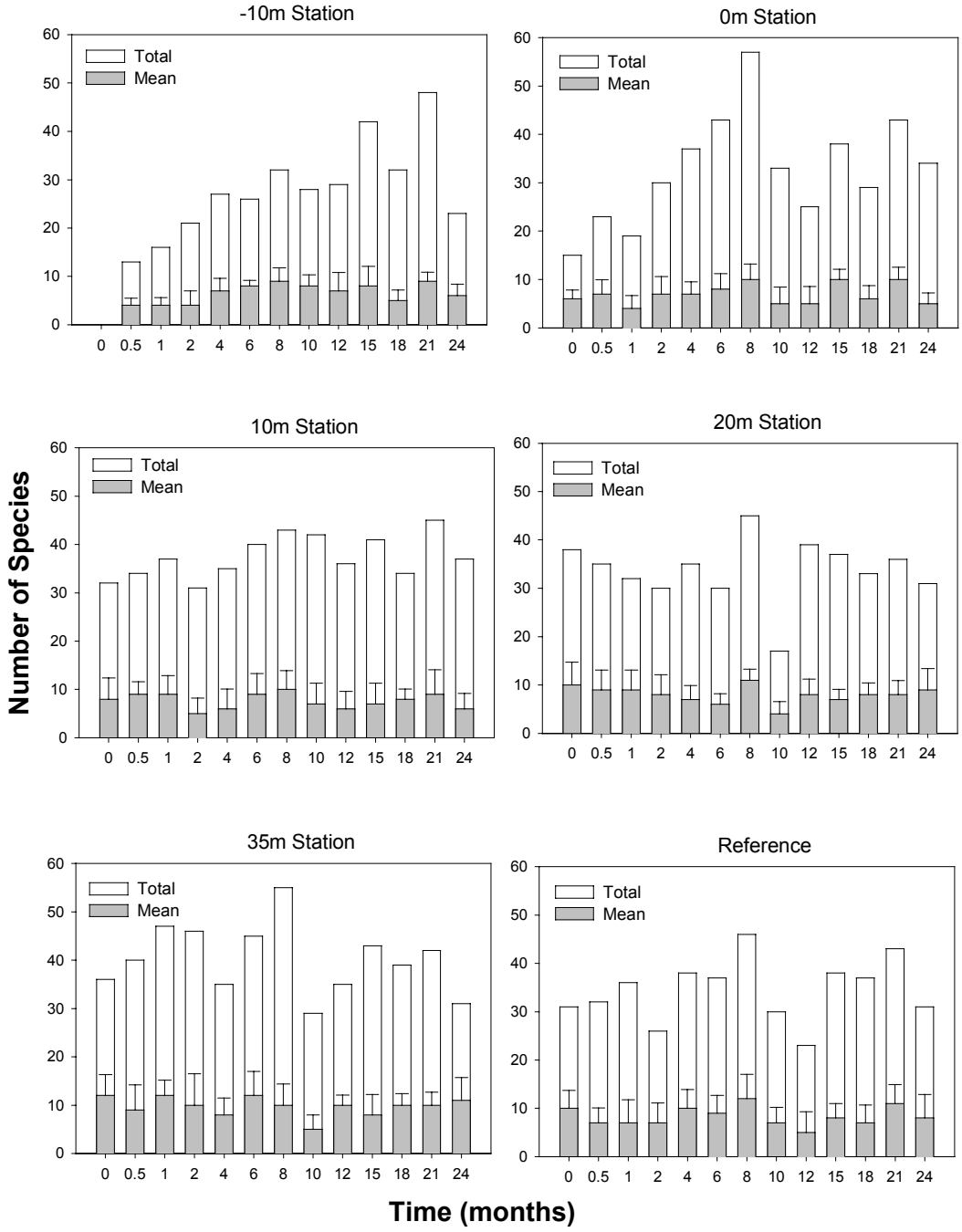
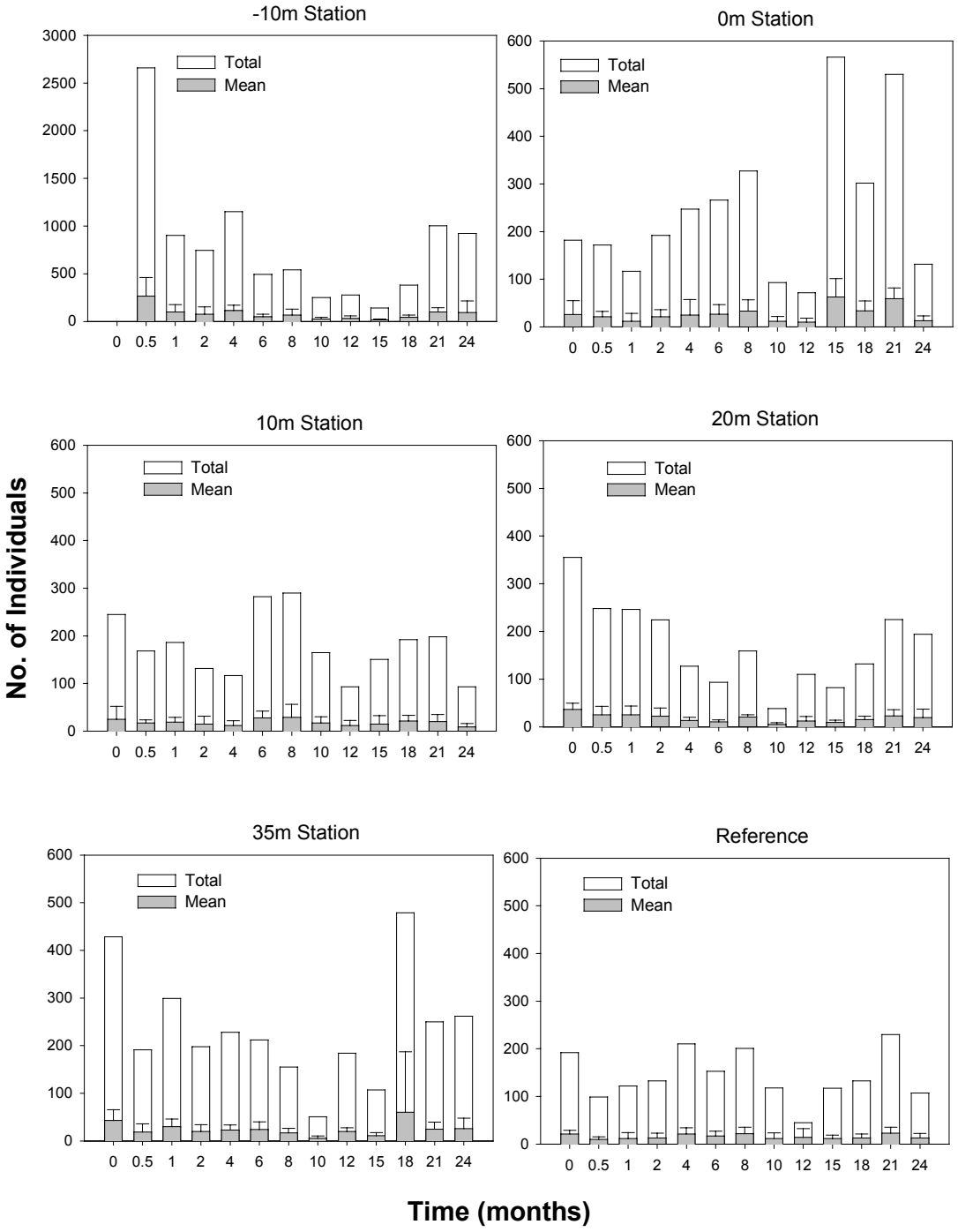


Fig. 16. Total and mean number of species (+ s.d.) at each sample station and at each sample time.



Note that for the –10m station the range for the number of individuals is different.

Fig. 17. Total and mean number of individuals (+ s.d.) at each sample station and at each sample time.

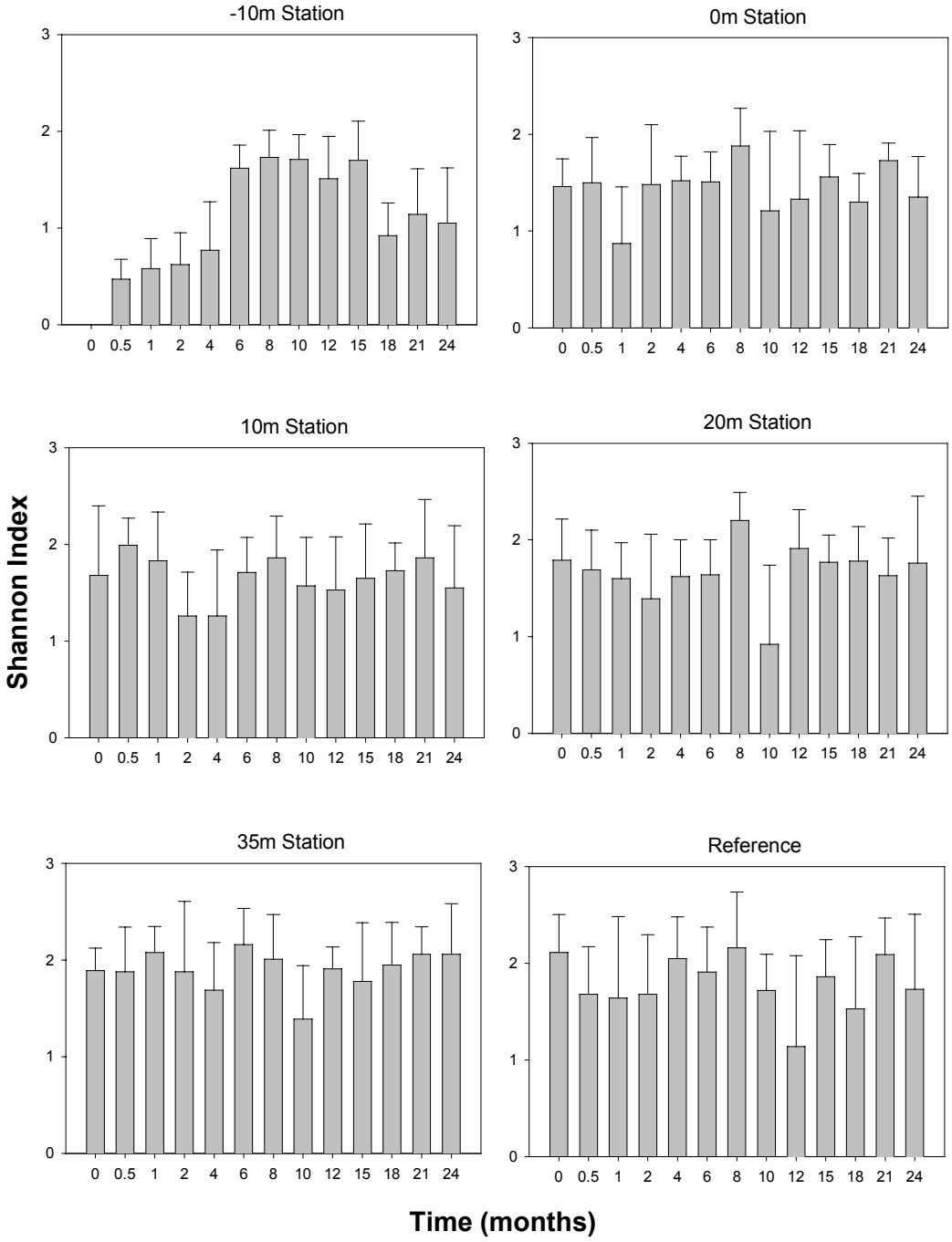


Fig. 18. Shannon index (+ s.d.) at each sample station and at each sample time.

3.7 Video Assessment

Two-way ANOSIM of time and the *a priori* groups (cage (-10m and 0m), farm (10m and 20m), boundary (35m) and reference indicated significant differences between both times and groups (Table 5). Within groups there were significant differences between the reference and all other groups and between the cage and farm groups.

Table 5. Two-way ANOSIM of sample station groups and times based on *a priori* group classification (Cage, Farm, Boundary and Reference)

Hypothesis 1. Test for differences between Times (averaged across all *a priori* groups)

Sample statistic (Global R): 0.211

Significance level of sample statistic: 0.1%

Hypothesis 2. Test for differences between *a priori* groups (averaged across all times)

Sample statistic (Global R): 0.209

Significance level of sample statistic: 2.8%

Pairwise Tests:

Groups Used	Statistical Value (R)	Significance Level
(Cage, Farm)	0.335	0.001
(Cage, Boundary)	0.112	0.122
(Cage, Reference)	0.310	0.001
(Farm, Boundary)	-0.069	0.781
(Farm, Reference)	0.272	0.003
(Boundary, Reference)	0.355	0.014

Cluster analysis of the video assessment data identified two clear groups. The –10m stations over the first two months (0.5, 1 and 2 months) and at 6 and 8 months were distinguished at the first dichotomy and form group 1, with an overall similarity level of around 17% (Fig. 19). The remaining stations, group 2 were further separated into three smaller cluster groups. Group 2a included the –10m stations at 4, 12, 15 and 21 months and the 0m stations at 0 and 12 months. The similarity level at which groups 2b and 2c separated was high (50%). Group 2b was composed of the 0m stations at 0.5, 1 and 4 months. The MDS plot suggests that the community structures of the group 2c stations were very similar. The ordination (Fig. 20) reveals a spatial gradation in the stations across the plot which largely separates the –10m and 0m stations from the remaining stations. These remaining stations were indistinguishable from one another. SIMPER analysis (Table 6) of the two main groups identified the presence of *Beggiatoa*, the profusion of burrows and faunal tracks and sediment colour as the primary factors responsible for the group determination.

Although the video assessment could not clearly distinguish between the stations further than 10m from the cages, the station distribution matrix that resulted from the video analysis still corresponded well with that produced by the macrofaunal assessment as shown by RELATE analysis (Global sample statistic = 0.482, $p<0.001$).

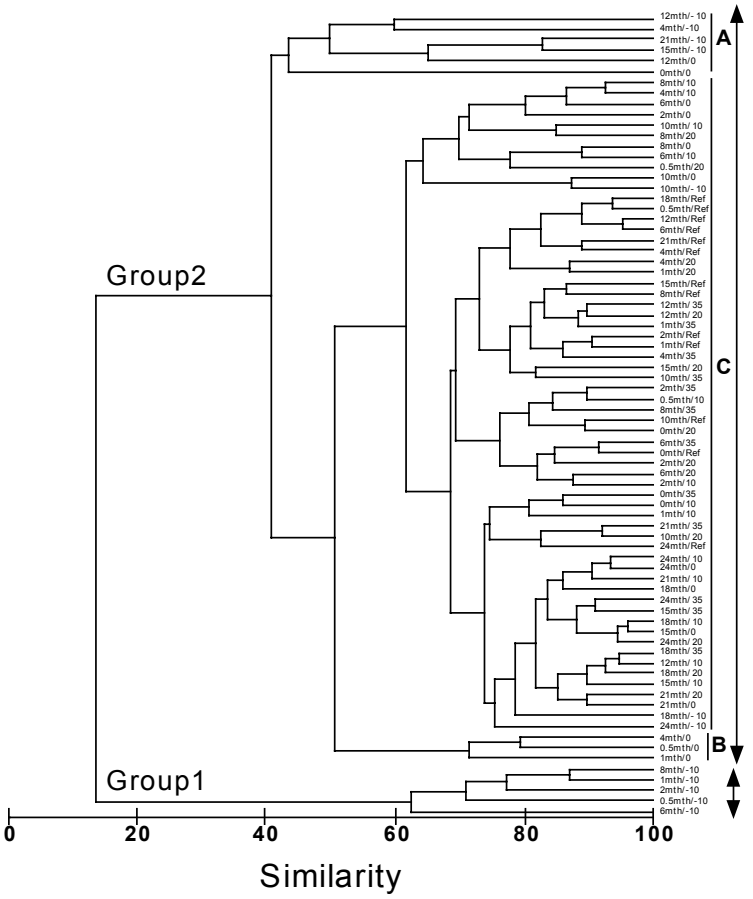


Fig. 19. Cluster analysis – Dendrogram for video assessment data from all sample stations where the video footage was usable.

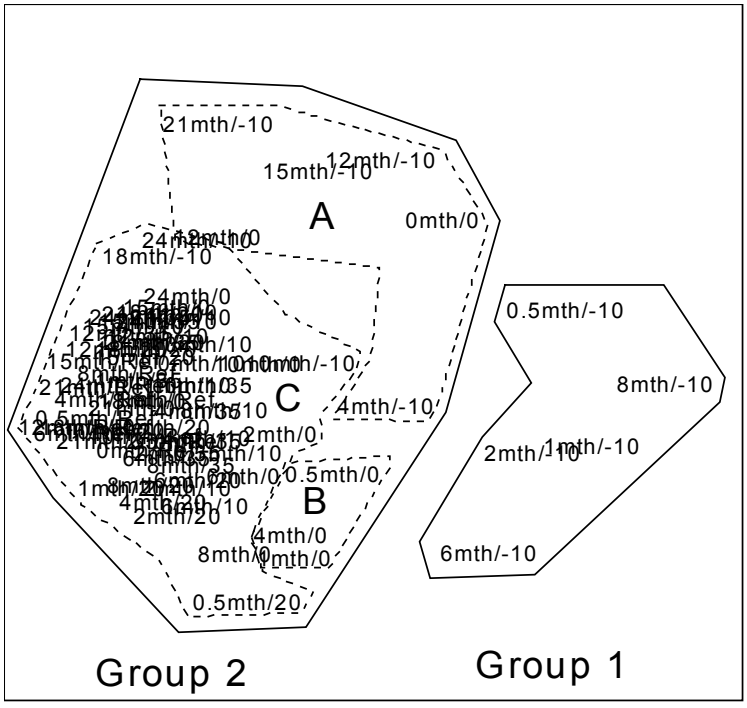


Fig. 20. Ordination analysis – 2-dimensional MDS plot of video assessment data from all sample stations where the video footage was usable. Stress=0.15.

Table 6. SIMPER output for the video assessment indicating a) and b) average abundance, ratio (average similarity / st.dev. similarity), % similarity and cumulative % similarity of the most important factors in each of the main groups and c) average abundance, ratio (average similarity / st.dev. similarity) and cumulative % similarity of the five factors which most clearly distinguish the main groups identified by cluster analysis

Species Name	Av.Abund.	Ratio	Percentage Similarity	Cumulative % Similarity
a) GROUP 1				
Beggiatoa density	1.80	3.39	40.89	40.89
Sediment colour	1.70	5.67	40.53	81.42
Worm cast density	0.70	3.52	14.31	95.72
b) GROUP 2				
Burrow density	2.08	2.94	41.71	41.71
Density of faunal tracks	1.44	1.55	24.39	66.10
Mollusc abundance	0.92	1.56	14.89	80.99
Level of debris	0.58	0.56	5.97	86.95
Small fish abundance	0.37	0.63	4.49	91.44

Species Name	Group 2 Av.Abund.	Group 1 Av.Abund.	Ratio	Cumulative % Similarity
c) BETWEEN GROUPS				
Burrow density	2.08	0.10	2.83	18.01
Beggiatoa density	0.00	1.80	3.35	35.14
Sediment colour	0.13	1.70	2.95	50.06
Density of faunal tracks	1.44	0.00	2.04	62.80
Mollusc abundance	0.92	0.10	1.58	70.35
Worm cast density	0.04	0.70	2.32	76.77

4. Discussion

Aquaculture like any other intensive farming activity has the potential to generate and release pollutants into the environment. Intensive cultivation of fish generates large amounts of particulate organic waste (Gowen and Bradbury, 1987; Gowen *et al*., 1988; Hall *et al*., 1990; Hansen *et al*., 1990; Ye *et al*., 1991). However, the type and scale of environmental change will depend on many influences. Farm management practices, background environmental conditions and climatic conditions can all significantly effect the extent of change. Where water flows are limited it is likely that the organic enrichment effects associated with cage aquaculture will be confined to a relatively small area (Brown *et al*., 1987; Lumb, 1989; Holmer and Kristensen, 1992; Hargrave *et al*., 1993). Results from this study suggest that any benthic impacts associated with the deposition of organic matter from the Gunpowder Jetty lease site would be highly localised. The overall flow rates were low, particularly in the lower 5m, which suggests that the dispersion of waste material (feed and faeces) around the cages would be limited and therefore that benthic impacts would be confined to a relatively small area around and to the north-west and south-east of the cages. The high levels of silt and clay in the sediment also suggests that both within the lease area and at the reference stations the sediments were "depositional" as defined by Rosenthal *et al*. (1988) and that the extent of waste dispersion from the cages would be limited.

Although the main current flow reflected the tidal movements in the area, it was clear from the data that prevailing weather conditions can have a significant influence both on the speed and direction of flow, particularly in relation to the surface water. Severe weather conditions in late December 1999 resulted in an increase in the main current flow throughout the entire water column and a swing in the predominant current direction. Storm events such as this could have a significant effect on the pattern of waste dispersion and, depending on the frequency and timing of these events, may to some extent mitigate environmental impacts.

Fish farm sediments have been found to have a very high organic matter content which can be strongly anoxic and rich in sulphides (Brown *et al*., 1987; Frogh, 1991). The present results agree with these previous studies; the sediments associated with the cages (-10m and 0m) were highly impacted at the time that the cages were removed. The highest sulphide levels observed in the present study were recorded at the cage stations (-10m and 0m), with maximum values of 2700 μM at the –10m station. Wildish *et al*. (1999), in Atlantic Canada, classified sediments associated with salmon culture into four categories by sulphide levels; oxic A, oxic B, hypoxic, anoxic. These categories can be equated to those described by Pearson and Rosenberg (1978) using the macrofaunal community structure (normal, transitory, polluted and grossly polluted). Using these characterisations the highest sulphide levels observed in the present study indicated hypoxic and polluted conditions.

The sulphide levels at all stations diminished over time, suggesting that these sediments were indeed in the process of recovering. However, after an initial decrease at the cage stations after 1 month, sediment sulphide levels increased again at 4 months (January 2000). As sulphide production in the sediments is largely a function of the activity of sulphate reducing bacteria and these bacteria are strongly influenced by temperature, the

warmer temperatures encountered in January 2000 may have facilitated sulphate reduction. After 4 months the levels decreased again and remained low.

In a similar study in the Huon Estuary Redox, bulk parameters of organic matter (%C, %N and stable isotope ratios C:N) and lipid concentrations were evaluated over one year at two cages with very different background environmental conditions and farming history in an attempt to determine the extent of sediment recovery at fish (McGhie *et al.*, 2000). In both instances the bulk organic matter parameters and the lipid concentrations indicated that there was still a significant portion of organic waste in the sediments after twelve months, whilst the redox results, like the sulphide measures in the present study, suggested a more rapid return to oxic conditions. This suggests that some measures of the sediment chemistry (i.e. redox/sulphide), although a useful guide to the sediment oxidant status in degradation (Crawford *et al.*, 2002), do not always provide such a clear indication of recovery.

Environmental impacts of aquaculture may also be reflected in increases in sediment organic matter levels. Yet, evaluation of organic matter content has not always been shown to be a useful measure of farm impact (eg. Johannessen *et al*., 1994; Hargrave *et al*., 1997; CSIRO Huon Estuary Study Team, 2000; Macleod, 2000; Crawford *et al*., 2002). In the present study organic matter levels recorded from all samples were very high, (~20% in association with cages and ~18% at the reference stations). Levels directly under the cages were generally higher than those reported from farming operations under similar environmental conditions either overseas, 9.5% (Brown *et al*., 1987), or locally 16-17% (Macleod, 2000). However, in the recent Huon estuary study comparably high levels (18-24%) were observed at stations in the upper reaches of the estuary where the input of terrestrial organic material was significant (CSIRO Huon Estuary Study Team, 2000). In order to determine whether the levels obtained from the reference stations in the current study were representative of North West Bay in general and that these sites were not being affected by cages, the results were compared with organic matter (LOI) from several other sites within the bay with similar depth and grain size composition (Fig. 21). These samples had been collected as part of another study to assess and monitor nutrients and habitat in North West Bay (Jordan *et al*., 2002) and were processed in an identical manner to the samples already collected in the present study. The results from these stations (Table 7) indicate that organic matter level (LOI) was generally highest at depths greater than 20m and was on the whole higher in the rest of the bay than at the Gunpowder reference locations. It is therefore unlikely that the organic matter levels at the Gunpowder references were influenced by the cages.

Regardless of origin the present data indicate that the amount of organic matter declined at all stations both spatially, with distance from the cage, and temporally, over the course of the study, but that the greatest change occurred in the under cage sediments (-10m). At the end of the present study overall organic matter levels had fallen 30%. One possible explanation may be that these changes resulted from longer term changes or cycles in the natural depositional character of the bay. However, differences between cage and reference sites were not significant and no seasonal patterns were evident. The proportion of the organic material at the study sites which could be attributed to sources other than fish farming is unknown. At times of high rainfall the North West Bay River (which represents ~79% of the freshwater input to the bay) could influence the level of

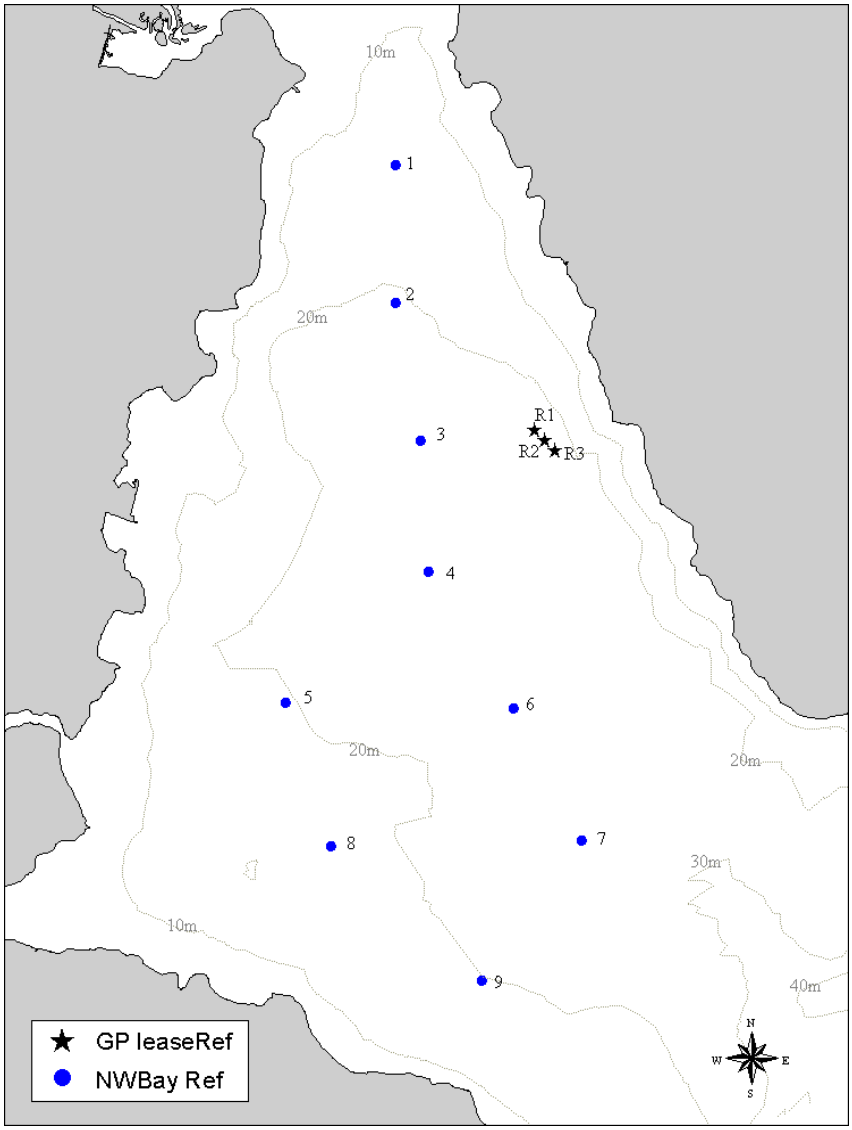


Fig. 21. Location of Gunpowder Jetty lease reference positions and additional sites from which samples were collected for analysis of organic matter and grain size composition in North West Bay.

organic material in the system (Jordan *et al*., 2002). However, the monthly rainfall for the period of this study was not particularly variable (50-100mm) and the most recent flood event occurred eight months before sampling commenced (ie. January 1999, Jordan *et al*., 2002). Aside from aquaculture, the main anthropogenic input of organic material into the bay would be treated sewage (Jordan *et al*., 2002). Although no data is available it is unlikely that sewage is a major contributor of particulate organic matter and improvements to the Dru Point sewage treatment plant in September 1999 have resulted in an appreciable improvement in the quality of the effluent water (J. Doole, Kingborough Council, pers.comm.).

Another indicator of farm impact at the cage stations was the presence of larger particles (125 – 250 µm) in the samples from the –10m stations. Visual examination of these samples identified that these particles included fragments of bone and fish vertebrae. The particles were smaller than that which would arise from the cultured fish

themselves and were present at levels suggesting that their occurrence was not natural. These bone fragments were only identified from the –10m station samples which suggests that they were in someway connected to the cage culture operations and it is possible that they may have originated from the fishmeal used in the manufacture of the commercial aquaculture diets.

Table 7. Site depth, percentage silt/clay (<0.063mm) and organic matter levels (LOI) for selected sites in North West Bay (May, 2001) and at the Gunpowder Jetty reference stations (June, 2001)

Location	Depth (m)	% Silt/clay	Mean % Organic Matter (LOI)	Standard deviation
1	17	95.4	15.7	1.41
2	20	91.5	18.1	0.76
3	22	96.0	18.7	0.26
4	24	98.1	18.4	0.91
5	19	85.2	17.4	0.72
6	25	78.4	18.9	0.51
7	26	97.0	18.7	0.38
8	18	22.3	9.2	0.13
9	19	72.3	15.0	0.41
Gunpowder Reference				
June 2001 (T21)	23	97.4	13.4	0.88

Changes in sediment chemistry associated with fish farm sediments will also result in changes in the macrobenthic community (Brown *et al*., 1987; Weston, 1990; Hargrave *et al*., 1997; Karakassis *et al*., 1998). At the time of cage removal the macrobenthic community structure at the cage associated stations (-10m and 0m) was very clearly impacted. Species diversity (Shannon index) was very low and abundance levels greatly increased, dominated by the opportunistic polychaete *Capitella capitata* complex. This species (previously recorded in Australia as simply *Capitella capitata*) is thought to be ubiquitous and has long been associated with areas of high organic enrichment (Pearson and Rosenberg, 1978; Gowen *et al*., 1991). It is a burrowing deposit feeder and is an opportunist. This species was described in Pearson and Rosenberg's (1978) review as indicative of the "polluted" zone, which is defined as an area which has partially recovered or is some distance from the "dead"/azoic zone and is characterised by an impoverished fauna. Cuomo (1985) showed that sulphide concentrations between 100-1000 µM were optimal for *Capitella* spp. settlement, nmetamorphosis and survival, which suggests that for the first four months the cage stations (-10m and 0m) were extremely suitable for *Capitella* spp colonisation.

Other species found to be associated with the highly impacted areas in the current study included *Malacoceros tripartitus* and to a lesser extent *Euphilomedes* sp. (MoV 18). Spionid polychaetes, such as *M. tripartitus,* are often associated with areas of high organic enrichment. This species is typically a deposit feeder which collects particles by spreading its long palps over the sediment surface, but in some cases may also use the palps to filter feed (Hutchings, 1984). *M. tripartitus* is a relative of *M. fuliginosa*

which has been associated with areas of high organic enrichment in the northern hemisphere (Thrush, 1986; Johannessen *et al*., 1994; Henderson and Ross, 1995). The ecology of the ostracod *Euphilomedes* sp. (MoV 18) is less well known, but it too is a burrowing deposit feeder and as such may be expected to survive in areas with increased organic material. Ecologically it may be similar to *Euphilomedes carcharodontabe* which has been described in conjunction with a bivalve (*Pavilucino tenuisculpta*) and *Capitella capitata* as one of the "three little pigs", a group of species which were indicative of organic enrichment at an ocean outfall in California (Maurer *et al*., 1993; Zmarzly *et al*., 1994).

The macrofaunal data also clearly indicate that the community structure had changed in a manner suggesting that recovery of the sediments was occurring over time. Generally, diversity was similar at the 0m, 10m, 20m, 35m and reference stations, however, there was a reduction at the –10m station over the first 4 months which indicated a marked community disturbance. The number of species increased at both the –10m and 0m station over the first 8 months suggesting that the community was recovering and indicating an enhancement zone at this time (Pearson and Rosenberg, 1978). The data for the total number of species also showed a recruitment pulse at 8 and 21 months (autumn) at all stations although the differences were not significant. This result is consistent with those of Crawford *et al*. (2002) who identified that in Tasmania, recruitment bias in the data should be considered for autumn sampling.

The community structure at the under cage stations (-10m) could be described as indicative of polluted / hypoxic conditions (Pearson and Rosenberg, 1978; Wildish *et al*., 1999) at the initial sample visit and remained highly impacted for the next 2 months. At four months, the benthic community at the –10m and 0m stations appeared to have undergone a marked recovery, although the fauna was still different to the reference stations. Nonetheless, a moderate impact could still be distinguished at the –10m and 0m stations at the end of the study, 24 months after the cages had been removed. At the farm stations (10 and 20m) it was difficult to discern a clear impact at any time although the community at the 10m station often contained transitional species. It was not generally possible to differentiate the 35m and 20m stations on the basis of the faunal composition and differences between the 35m and reference stations were minor. The faunal differences were due to subtle changes in species abundance and the presence, in low numbers, of particular species. For instance the heart urchin, *Echinocardium cordatum,* was recorded at the 20 and 35m stations but not at the reference stations and similarly the bivalve, *Nucula pusilla,* was recorded from the reference stations but was rarely encountered at the 20 or 35m stations. These differences are more likely to be indicative of natural background variation rather than any direct effects from the aquaculture operations. This in turn suggests that spatially the impact from the cages does not appear to have markedly affected the community structure 35m beyond the cage boundary, even at the time of cage removal.

Although not listed within the six most important species, the common brittle star *Amphiura elandiformis* was also an important species characterising unimpacted conditions, particularly in the early samples. Increases in the abundance and biomass of echinoderms have been found to be associated with a reduction in the degree of pollution (Gray *et al*., 1988). Also, Hylland *et al*. (1997) showed significant growth reduction in brittle stars in response to organic enrichment and hypoxia. More recently,

Macleod (2000) identified that the fauna at unimpacted stations from two aquaculture farm locations in southern Tasmania were characterised by a range of echinoderm species and was generally dominated by *A. elandiformis*. This suggests that in Tasmania *A. elandiformis* is characteristic of a climax community comparable to that defined by *A. filiformis* or *A. chiajei* in north temperate waters (O'Connor *et al*., 1983; Hollertz *et al*., 1998) or *Amphioplus* (*Lymanella*) *laevis* in tropical waters (Warwick and Ruswahyuni, 1987). Two other species were of particular significance at the unimpacted sites, *Lysilla jennacubinae* and *Nucula pusilla*. *L. jennacubinae,* a terebellid polychaete, is a selective surface deposit-feeder (Hutchings, 2000). Terebellids are often extremely abundant in benthic communities and are generally found in non-polluted environments (Hutchings, 2000). The genus *Lysilla* may be compared with *Terebellides* in the Pearson and Rosenberg (1978) organic enrichment model indicating normal/ oxic conditions. The genus *Nucula* is also identified in the Pearson-Rosenberg model as indicative of normal conditions.

At the moderately impacted locations abundances of two introduced bivalves, *Theora fragilis* and *Corbula gibba* and the ostracod *Euphilomedes* sp. (MoV 18), increased in comparison with both the cage and reference conditions. Both *T. fragilis* and *C. gibba* are tolerant of reduced oxygen levels and increased organic enrichment (Tamai, 1996; Furlani, 1996). Tamai (1996) showed that *T. fragilis* was tolerant of severe hypoxia (DO 1.3-1.4 mg/l) and was unaffected under conditions of moderate hypoxia (DO 2.2-2.4mg/l). That numbers decreased at the cage stations suggests that the extent of hypoxia and enrichment at the cages were inhibitory. However, abundances of *T. fragilis* and *C. gibba* also decreased at most stations over time which implies that the return to normal/ oxic environmental conditions may also in some way either directly or indirectly restrict these species.

Increased densities of the amphipods *Jassa marmorata, Ericthonius sp.* and *Corophium ascherusicum* were recorded at the 35m station at 18 months. *J. marmorata* is commonly found in fouling communities (Franz and Mohamed, 1989) and relatives of both *Ericthonius sp.* and *C. ascherusicum* have also been associated with marine fouling populations. Communication with the farm indicated that around 18 months they were removing and cleaning mooring blocks and marker bouys. Consequently it is likely that the presence of these species in this sample was a result of accidental deposition of material on the bottom when mooring lines and surface markers bouys were relocated.

Video assessment also identified the impact under the cages (-10m), for up to ten months after cage removal. Analysis of the video footage distinguished the areas of highest impact but was less successful than either sulphide measurements or macrofaunal assessment at distinguishing the areas of intermediate impact. The video assessment suggested that the size and thickness of the *Beggiatoa* mats had increased at –10m stations between 0.5 and 1 month after removal of the cages. Diver comments recorded at 0 months indicated that *Beggiatoa* mats were not evident under the cages but that the sediment was black and smelt strongly of hydrogen sulphide at the surface. The presence of *Beggiatoa* mats at the –10m stations as late as 8 months after cage removal suggests that the sediment was still anoxic. *Beggiatoa* mats develop at the interface between hypoxic and anoxic conditions, requiring the presence of both sulphide and oxygen. Consequently the diver observations of no *Beggiatoa* under the

cages at initial sampling and subsequent increase in mat density in the first month may indicate that at the time of cage removal the water overlying the sediment was anoxic and therefore *Beggiatoa* development was inhibited. However, if at the initial sampling the sediment and overlying water was anoxic it is unclear why large numbers of *Capitella capitata* complex were present at this time.

The results of this study suggest that at the time that the cages were removed the sediment directly beneath the cages was highly impacted, in all probability hypoxic/anoxic, but that this impact was very localised. Sediment conditions improved rapidly with distance from the cage such that there was no detectable impact 35m from the edge of the cages. Recovery of the sediments was most rapid in the first 2 months, after which the rate of recovery slowed. There were marked differences in the overall evaluation of recovery between the different assessment techniques. The sediment chemistry responded to the changing environmental conditions more quickly than the benthic infaunal community. Sediment sulphide levels beneath the cages recovered to background levels within 6 months of the cages being removed. However, differences were still apparent in the benthic community structure after 24 months.

5. Conclusions

- The water flow in and around the lease area was low and the overall system highly depositional.
- The sediments at the cage locations were highly impacted at the time of cage removal and could be classified as polluted/ hypoxic (Pearson and Rosenberg, 1978; Wildish *et al*., 1999).
- The level of impact diminished rapidly with distance from the cages. The effect was very much reduced at the 10-20m stations and the influence of the cage operations was not generally discernible at the 35m station.
- Initial recovery from polluted to transitional conditions occurred quite quickly (1-2 months) but subsequent recovery appeared to be slower.
- Sulphide concentration in the sediments under the cages returned to oxic or normal levels within 6 months of the cages being removed.
- Organic enrichment effects were apparent in the benthic community for a lot longer than in chemical measures. Recovery to background conditions of the macrofaunal community under and adjacent to the cages will take longer than 24 months.

- The main species indicative of polluted conditions were *Capitella capitata* complex and *Malacoceros tripartitus*. Increased abundances of the introduced bivalves *Theora fragilis* and *Corbula gibba* and the ostracod, *Euphilomedes* sp. (MoV 18), appeared to be associated with transitional conditions whilst in unimpacted conditions species such as *Nucula pusilla, Lysilla jennacubinae* and *Amphiura elandiformis* were common.

- Bacterial mats (*Beggiatoa* spp), blackened sediments and gas bubbles were clear visual indicators of impacted sediments.

- Absence of *Beggiatoa* spp may indicate deterioration (i.e. that the water overlying the sediment surface is anoxic) rather than improvement of environmental conditions. Consequently other factors or temporal information should always be considered.

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References

Beveridge, M.C. (1996) Cage Aquaculture. Fishing News Books, Blackwell Science Ltd, U.K.

Black, K.D. (2001) Environmental impacts of aquaculture. Sheffield Academic Press Ltd, U.K.

Braaten, B. (1991) Impacts of pollution from aquaculture in six Nordic countries. Release of nutrients, effects, and waste water treatment. In *Aquaculture and the Environment*. 1991. N.De Pauw and J. Joyce (Eds) European Aquaculture Society Special Publication No.16, Gent, Belgium.

Brown, J.R., Gowen, R.J. and McLusky, D.S. (1987) The effect of salmon farming on the benthos of a Scottish sea loch. *J. Exp. Mar. Biol. Ecol.* **109**: 39-51.

Chang, B.D. and Thonney, J.P. (1992) Overview and environmental status of the New Brunswick salmon culture industry. *Bulletin of Aquaculture Association of Canada 92-93*: pp.61-63.

Cheshire, A., Westphalen, G. Smart, A. and Clarke, S., (1996) Investigating the environmental effects of sea-cage tuna farming. II. The effect of sea-cages. Department of Botany, University of Adelaide.

Clarke, K.R. and Warwick, R.M. (2001) Change in marine communities: an approach to statistical analysis and interpretation, 2nd edition. PRIMER-E: Plymouth.

Cole-Parmer (no date) Operating instructions for Cole-Parmer Silver/Sulfide Electrodes. Cole-Parmer Instrument Company, Illinois, USA. p.20.

Crawford, C.M., Mitchell, I.M. and Macleod, C.K.A. (2001) Video assessment of environmental impacts of salmon farms. *ICES Journal of Marine Science* 59: 433-463.

Crawford, C.M., Macleod, C.K.A. and Mitchell, I.M. (2002) Evaluation of techniques for environmental monitoring of salmon farms in Tasmania. *TAFI Technical Report No.8. January 2002.* Tasmanian Aquaculture & Fisheries Institute – University of Tasmania, Hobart, Australia.

CSIRO Huon Estuary Study Team. (2000) Huon Estuary Study – Environmental research for integrated catchment management and aquaculture. Final report to Fisheries Research and Development Corporation. Project number 96/284, June 2000. CSIRO Division of Marine Research, Marine Laboratories, Hobart, Australia.

Cuomo, M.C. (1985) Sulphide as a larval settlement cue for *Capitella sp* I. *Biogeochemistry*. Vol.1 : 169-181.

DPIF, (1997) Marine Farming Development Plans for Tasmania. D'Entrecasteaux Channel, February 1997. Marine Resources Division, Department of Primary Industry and Fisheries, Tasmania.

Findlay, R.H., Watling, L. and Lawrence, M.M. (1995) Environmental impact of salmon net-pen culture on marine benthic communities in Maine: a case study. *Estuaries* 18 (1A): 145-179.

Franz, D.R. and Mohamed, Y. (1989) Short distance dispersal in a fouling community amphipod crustacean, *Jassa marmorata* Holmes. *Journal of Experimental Marine Biology and Ecology* 133 (1-2): 1-13.

Frogh, M. and Schaanning, M. (1991) Benthic degradation in bottom sediments from salmon cage farming at 66 degree N in Norway. In *Aquaculture and the Environment*. *Aquaculture Europe '91*. DePauw,-N.;Joyce,-J.-(comps). 14: 111.

Furlani, D. (1996) A guide to the introduced species in Australian waters. *CRIMP Technical Report No.5. (June 1996).* CSIRO, Hobart, Tasmania, Australia.

Gowen, R.J., Weston, D.P. and Ervik, A. (1991) Aquaculture and the benthic environment: A review. pp187-205. In C.B. Cowey and C.Y. Cho (eds) *Nutritional strategies and aquaculture waste*. Proceedings of the First International Symposium on Nutritional Strategies in Management of Aquaculture Waste. University of Guelph, Guelph, Ontario, Canada, 1990.

Gowen, R.J., Brown, J.R., Bradbury, N.B. and McLusky, D.S. (1988) Investigation into benthic enrichment, hypernutrification and eutrophication associated with mariculture in Scottish coastal waters (1984-1988). Dept. Biological Science, University of Stirling. p.289.

Gowen, R.J. and N.B. Bradbury (1987) The ecological impact of salmonid farming in coastal waters: A review. *Oceanographic and Marine Biology: An Annual Review.* 25: 563-575.

Gowen, R.J. and Rosenthal, H. (1993) The environmental consequences of intensive coastal aquaculture in developed countries: what lessons can be learnt?, in *ICLARM Conference Proceedings*, Vol.31.(eds. R.S.V.Pullin, H.Rosenthal and J.L.Maclean), Phillipines, Manila, pp.102-115.

Gray, J.S., Aschan, M., Carr, M.R., Clarke, K.R., Green, R.H., Pearson, T.H., Rosenberg, R. and Warwick, R.M. (1988) Analysis of community attributes of the benthic macrofauna of Frierfjord/Langesundfjord and in a mesocosm experiment. *Marine Ecology Progress Series*. Vol.46 : 171-180.

Greiser, N. and Faubel, A. (1988) Biotic Factors. pp79-114. In Higgins, R.P. and Theil, H. (eds). *Introduction to the study of meiofauna*. Smithsonian Institution Press, Washington D.C.

Hall, P.O.J., Anderson, L.G., Holby, O., Kollberg, S. and Samuelsson, M.O. (1990) Chemical fluxes and mass balances in a marine fish cage farm. I. Carbon. *Marine Ecology Progress Series.* Vol.61 : 61-73.

Hansen, P.K., Pittman, K. and Ervik, A. (1990) Effects of organic waste from marine fish farms on the seabottom beneath the cages. *International Council for Exploration of the Sea*, C.M.1990/F:34. p9.

Hargrave, B.T., Duplisea, D.E., Pfeiffer, E. and Wildish, D.J. (1993) Seasonal changes in benthic fluxes of dissolved oxygen and ammonium associated with marine cultured Atlantic salmon. *Marine Ecology Progress Series*. Vol.96 : 249-257.

Hargrave, B.T., Phillips, G.A., Doucette, L.I., White, M.J., Milligan, T.G., Wildish, D.J. and Cranston, R.E. (1997) Assessing benthic impacts of organic enrichment from marine aquaculture. *Water, Air, and Soil Pollution*. 99 (1-4): 641-650.

Henderson, AR. and Ross, D.J. (1995) Use of macrobenthic infaunal communities in the monitoring and control of the impact of marine cage fish farming. *Aquaculture Research*. Vol.26 : 659-678.

Hylland, K., Skoeld, M., Gunnarsson, J.S. and Skei, J. (1997) Interactions between eutrophication and contaminants. 4. Effects on sediment-dwelling organisms. *Marine Pollution Bulletin*. 33 (1-6): 90-99.

Hollertz, K., Skoeld, M., Rosenberg, R., Baden, S., Stromberg, J.O., Svane, I., Rosenberg, R., Tiselius, P. (eds) Interactions between two deposit feeding echinoderms: The spatangoid *Brissopsis lyrifera* (Forbes) and the ophiuroid *Amphiura chiajei* Forbes. *Hydrobiologia*. 375-379 (103): 287-295.

Holmer, M. and Kristensen, E. (1992) Impact of marine fish farming on metabolism and sulphate reduction of underlying sediments. *Marine Ecology Progress Series.* 80: 191-201.

Hutchings, P.A. (1984) An illustrated guide to the estuarine polychaete worms of New South Wales. The Australian Museum, Sydney, NSW. p160.

Hutchings, P.A. (2000) Family Terebellidae. Pp 226-232 in Beesley, P.L., Ross, G.J.B. and Glasby, C.J. (eds) *Polychaetes & Allies: The Southern Synthesis. Fauna of Australia. Vol.4A Polychaeta, Myzostomida, Pogonophora, Echiura, Sipuncula*. CSIRO Publishing: Melbourne xii 465pp.

Ingold, W. (1982) Redox measurement – Principles and problems. Ingold Messtechnik Ag, Zurich, Switzerland.

Iwama, G.K. (1991) Interactions between aquaculture and the environment. *Critical Reviews in Environmental Control*. Vol.21 (2) : 177-216.

Johannessen, P.J., Botnen, H.B. and Tvedten, O.F. (1994) Macrobenthos: before, during and after a fish farm. *Aquaculture and Fisheries Management*. 25: 55-66.

Jordan, A., Doole, J., Archer, L., Lawler, M., Halley,V. and Sanderson, C. (2002) *Assessment and Monitoring of Nutrients and Habitats in North West Bay – Supporting Sustainable Management.* Kingborough Council Natural Resource Management Strategy, Hobart, 120p.

Karakassis, I., Tsapakis, M. and Hatziyanni, E. (1998) Seasonal variability in sediment profiles beneath fish farm cages in the Mediterranean. *Marine Ecology Progress Series*. Vol.162 : 243-252.

Karakassis, I., Hatziyanni, E., Tsapakis, M. and Plaiti, W. (1999) Benthic recovery following cessation of fish farming: a series of successes and catastrophes. *Marine Ecology Progress Series*. Vol. 184 : 205-218.

Lim, S. (1991) Environmental impact of salmon farming on the benthic community in the Bay of Fundy. *Bulletin of the Aquaculture Association of Canada*. Vol.91-3 : 126-128.

Lumb, C.M. (1989) Self-pollution by Scottish salmon farms? *Marine Pollution Bulletin*. Vol.20 (8) : 375-379.

Macleod, C.K. (2000) Techniques for farm-based assessment of sediment health associated with the commercial culture of Atlantic salmon (*Salmo salar* L.) in Tasmania. MSc Thesis, University of Tasmania, Australia.

Maurer, D., Robertson, G. and Gerlinger, T. (1993) San Pedro shelf California: Testing the Pearson-Rosenberg Model (PRM). *Marine Environmental Research*. 35 (4): 303-321.

McGhie, T.K., Crawford, C.M., Mitchell, I.M. and O'Brien, D. (2000) The degradation of fish-cage waste in sediments during fallowing. *Aquaculture*. 187 : 351-366.

O'Connor, B., Bowmer, T. and Grehan, A. (1983) Long-term assessment of the population dynamics of *Amphiura filiformis* (Echinodermata: Ophiuroidea) in Galway Bay (west coast of Ireland). *Marine Biology*. 75 (2/3): 279-286.

Parry, G.D., Currie, D.R. and Crookes, D.P. (1997) Exotic pests in Portland Harbour and environs. MAFRI-Report 1997 (1) 40 pp. Queenscliff, Victoria, Australia.

Pearson, T.H. and Rosenberg, R. (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol. Ann. Rev.* Vol.16 : 229-311.

Pearson, T.H. and Stanley, S.O. (1979) Comparative measurements of the redox potentials of marine sediments as a rapid means of assessing the effect of organic pollution. *Marine Biology*. Vol.53 : 371-379.

Raa, J. and Liltved, H. (1991) An assessment of the compatability between fish farming and the Norwegian coastal environment, pp.51-59. In De Pauw, N. and Joyce, J. (eds) *Aquaculture and the Environment*. European Aquaculture Society Special Publication No.16, Gent, Belgium.

Ritz, D.A., Lewis, M.E. and Ma Shen, (1989) Response to organic enrichment of infaunal macrobenthic communities under salmonid seacages. *Marine Biology*. 103: 211-214.

Rosenthal, H. and Rangeley, R.W. (1988) The effect of salmon cage culture on the benthic community in a largely enclosed bay (Dark Harbour, Grand Manan Island, NB, Canada), pp.207-223. In Rosenthal, H. and Lillelund, K. (eds) *Fish health protection strategies.* Bundesministerium für Forschung und Technologie, Hamburg, Germany.

Rosenthal, H., Weston, D.P., Gowen, R.J. and Black, E. (1988) Report of the *ad hoc* study group on environmental impact of mariculture. International Council for exploration of the Seas (ICES) Co-operative Research Report 154. 81p.

Shannon, C.E. and Weaver, W.W. (1963) The mathematical theory of communications. University of Illinois Press, Urbana, Illinois, USA. p.117.

Tamai, K. (1996) Temporal tolerance of larval *Theora fragilis*. *Fisheries Science*. 62 (6): 996-997.

Thrush, S.F. (1986) The sublittoral macrobenthic community structure of an Irish sea-lough: Effect of decomposing accumulations of seaweed. *Journal of Experimental Marine Biology and Ecology.* 96 (3): 199-212.

Warwick, R.M. (1993) Environmental impact studies on marine communities: pragmatical considerations. *Australian Journal of Ecology*. Vol.18 : 63-80.

Warwick, R.M. and Ruswahyuni, (1987) Comparative study of the structure of some tropical and temperate marine soft-bottom communities. *Marine Biology*. 95 (4): 647-649.

Weston, D.P. (1990) Quantitative examination of macrobenthic community changes along an enrichment gradient. *Marine Ecology Progress Series.* 61: 233-244.

Wildish, D.J., Akagi, H.M., Hamilton, N. and Hargrave, B.T. (1999) A recommended method for monitoring sediments to detect organic enrichment from mariculture in the Bay of Fundy. . *Canadian Technical Report of Fisheries and Aquatic Sciences* No.2286. p.31.

Woodward I.O., J.B. Gallagher, M.J. Rushton, P.J. Machin and S. Mihalenko. 1992. Salmon farming and the environment of the Huon Estuary, Tasmania. DPIF (Tasmania) - Technical Report No. 45.

Wu, R.S.S. (1995) The environmental impact of marine fish culture – towards a sustainable future. *Marine Pollution Bulletin*. Vol.31.: 159-166.

Wu, R.S.S., Lam, K.S., MacKay, D.W., Lau, T.C. and Yam, V. (1994) Impact of marine fish-farming on water quality and bottom sediment: a case study in the sub-tropical environment. *Marine Environmental Research*. Vol.38 : 115-145.

Ye, L., Ritz, D.A., Fenton, G.E. and Lewis, M.E. (1991) Tracing the influence on sediments of organic waste from a salmonid farm using stable isotope analysis. *Journal of Experimental Marine Biology and Ecology*. Vol.145 : 161-174.

Zmarzly, D.L., Stebbins, T.D., Pasko, D., Duggan, R.M. and Barwick, K.L. (1994) Spatial patterns and temporal succession in soft-bottom macroinvertebrate assemblages surrouinding an ocean outfall on the San Diego shelf: Relation to anthropogenic and natural events. *Marine Biology*. 118 (2): 293-307.