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Assessment of long term change in sediment condition after organic enrichment: defining recovery

Catriona K. Macleod [a,b,*], Christine M. Crawford [a], Natalie A. Moltschaniwskyj [b]

[a] *Marine Research Laboratories, Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Taroona, Tasmania 7053, Australia*
[b] *School of Aquaculture, Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Taroona, Tasmania 7053, Australia*

Abstract

Sediment condition at an Atlantic salmon (*Salmo salar*) culture site in S.E. Tasmania, Australia was evaluated to determine the rate and extent of recovery after removal of farmed fish. By local standards the cage sediment at the start of this survey was markedly degraded but comparison with results from impact studies in Scotland, Canada and Norway suggests that the sediments were considerably less impacted than in northern temperate areas. The impact at the cages diminished rapidly with both time and distance; after only 2 months conditions were markedly improved. The macrobenthos indicated a slower recovery than chemical measures, after 36 months the benthic faunal community structure under the cages still differed from reference conditions even though other sediment measures had recovered. This study highlighted two other key issues in relation to monitoring and management of sediment recovery. First, techniques used to determine impact may not be appropriate for evaluation of recovery. Second, establishment of local baseline standards is extremely important to ensure appropriate evaluation of both impact and recovery.

Keywords: Sediment recovery; Environmental assessment; Monitoring; Organic enrichment; Aquaculture; Video

1. Introduction

Deposition of aquaculture waste from finfish cages can result in organic enrichment. To overcome this it is usual for farmers to leave areas of seabed free from farming activities for a period of time to allow recovery. However, it is currently not clear to what extent sediment recovery occurs or to what degree natural environmental conditions can influence recovery. From the perspective of both farm management and ecosystem protection it is important to have a clear understanding of the processes involved in recovery. The degree and extent of organic enrichment of sediments under cages and the magnitude and scale of impact is dependent on both husbandry parameters and physical, chemical and biological characteristics of the environment (Iwama, 1991; Gowen and Rosenthal, 1993; Wu, 1995; Black, 2001). However, few studies have attempted to evaluate sediment recovery rates and 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). Consequently, the primary objective of this study was to assess the rate of sediment recovery associated with long term fallowing of intensively farmed marine Atlantic salmon cage sites in the temperate waters of south-east Tasmania, Australia.

Many factors influence sediment recovery rate and hence several different techniques have been used as surrogate measures of sediment condition (Hargrave et al., 1997; Morrisey et al., 1998). Some measures (e.g. redox and sulphide) indicate specific chemical aspects of sediment condition (Holmer and Kristensen, 1992; Hargrave et al., 1993) whilst others (e.g. infaunal community structure) reflect a combination of physical, chemical and biological influences (Findlay et al., 1995; Karakassis et al., 1999; Macleod et al., 2002). Characterisation of benthic infaunal communities is one of the most reliable indicators of environmental disturbance, but it can be expensive and time consuming (Wildish et al., 1999; Crawford et al., 2002),

* Corresponding author. Address: Marine Research Laboratories, Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Taroona, Tasmania 7053, Australia.
E-mail address: catriona.macleod@utas.edu.au (C.K. Macleod).

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consequently simpler and more cost effective techniques are frequently used. These include measurement of redox and sulphide, total organic matter and/or total organic carbon levels and more commonly these days, visual assessment of sediment characteristics and epibiota by diver or video.

Visual assessment of sediment and epibiotic status is a simple approach, which has the potential to provide clear, easily interpretable results. It is often used by regulatory authorities to identify areas of major impact. However, the information obtained is generally subjective. A semi-quantitative approach for video evaluation recently developed by Crawford et al. (2001, 2002) has increased the value of video assessment.

The second objective of this study was to determine the suitability of different sediment evaluation techniques for assessment of recovery. A number of countries have produced specific protocols for monitoring the impact of fish farms, for example in Norway and Scotland national standards have been developed, whilst regional recommendations exist in Australia and the Canadian provinces. However, although there will be broad similarities in organic enrichment effects marked geographic differences in the range and scale of measurements, both globally and regionally can be expected. These differences must be taken into consideration when interpreting results among different areas. Furthermore, these guidelines relate specifically to detection of impact, not evaluation of recovery. Although degradation and recovery processes may be similar (Ritz et al., 1989; Karakassis et al., 1999; Black, 2001), it is unlikely that the rates would be equivalent. Sediment recovery is passive whilst degradation results from the active input of waste products. Consequently, the currently acknowledged approaches for evaluating impact may not be appropriate for assessing recovery.

2. Methods

2.1. Study site

The study lease (3.12 ha) was located on the eastern shore of North-West Bay, Tasmania, Australia (Fig. 1). Prior to cessation of farming in August 1999 the farm had been involved in the commercial production of Atlantic salmon (*Salmo salar* L.) for 14 years. Over the preceding four years this site had essentially been stocked continually with little or no fallowing. In the year prior to closure the farm stocked approximately 200–300 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. Current velocity throughout the water column was generally slow (3.4–4.3 cm^{-1}), with rates near the seabed further reduced (Macleod et al., 2002).

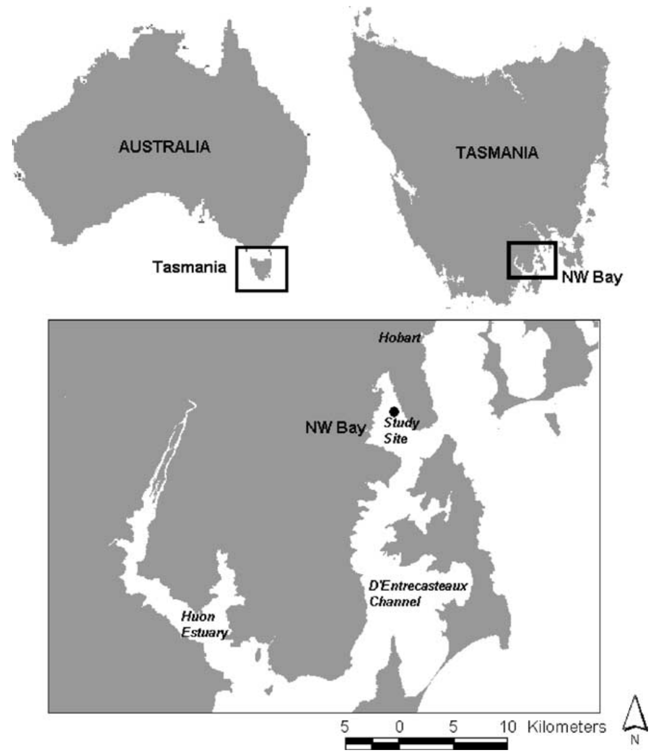

Fig. 1. Location of study site.

Two cages were selected at random for this study (Fig. 2). These cages had comparable farming histories, i.e. fish biomass and feeding levels were equivalent. At each cage fixed transects were positioned on the seabed running from directly beneath the cage (−10 m) to 35 m from the cage edge (Fig. 2). Stations were established at −10 m (centre cage), 0 m (cage edge), 10 m, 20 m and 35 m. Reference stations for each transect were located

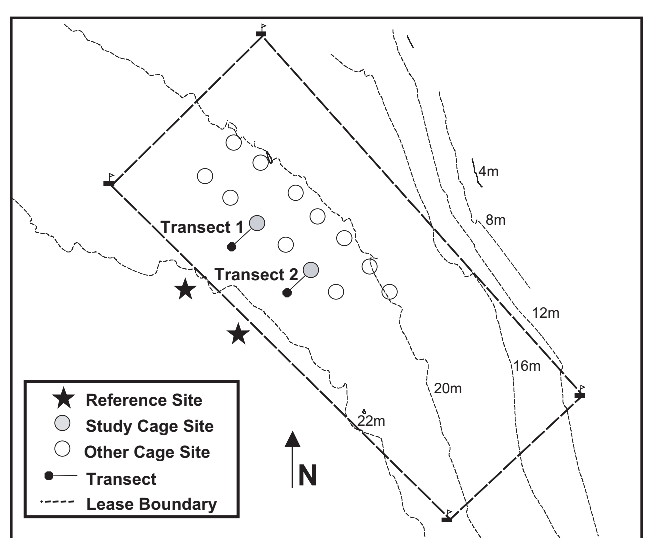

Fig. 2. Location of cage study sites and reference positions within study lease.

150 m from the cages, directly in line with the fixed transects (Fig. 2) and at similar depths. The positions of 0 m, 35 m and reference (150 m) stations for each transect were determined using a differential global positioning system (DGPS).

Initial sampling at each station was undertaken two weeks after removal of fish and then 1, 2, 6, 12, 24 and 36 months thereafter. Benthic samples were collected by diver for macrofaunal and physical/chemical analyses. Video footage was collected along the transect line and from an area within a 2 m radius of the reference station.

2.2. Physical/chemical analyses

At each station three replicate core samples were collected using perspex tubes (250 mm long and 45 mm internal diameter). A single sub-sample (4 ml) was taken from each replicate core at a depth of 4 cm for measurement of sulphide using a Cole-Parmer 27502-40 silver/sulfide electrode as per Wildish et al. (1999). Sulphide standards were prepared before each sampling event and electrode calibration curves were determined.

After taking samples for sulphide measurements, the remaining sediment was extruded and sectioned. Half of the top 4 cm from two cores was collected for sediment particle size analysis. A sub-sample of each was passed wet through a graded series of sieves (4 mm, 2 mm, 1 mm, 500 μm, 250 μm, 125 μm and 63 μm). The sediment retained on each sieve was dried and weighed and the percentage of the total sample weight calculated. The fraction <63 μm was determined as the difference between the initial sample weight and the combined weight of the retained fractions. Total organic matter was determined by the loss on ignition technique (Greiser and Faubel, 1988) modified as follows; samples collected from the top 4 cm of each core were homogenised and a sub-sample of approximately 2–5 g taken, excess carbonate was removed from the samples by (1) sieving to remove large shell fragments and (2) neutralising any remaining carbonate by acidification with 1 N HCl. The samples were then oven dried for 24 h at 60 °C before being transferred to a muffle furnace for 4 h at 500 °C. The weight of organic material was calculated as the difference between oven dried and final furnace ashed weights.

2.3. Macrofaunal assessment

Macrofaunal data were collected from stations representative of cage impacts (−10 m/0 m), farm effects (10 m) and unimpacted reference conditions (150 m). At all stations 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 100 mm (sampling area of 0.0177 m^2). Samples were collected by diver and transferred immediately to mesh bags (0.875 mm^2 mesh); on the boat the bags were rinsed and transferred to containers with 10% formalin in seawater. In the laboratory each sample was sieved to 1 mm, sorted and the animals retained were identified to the lowest possible taxonomic level and enumerated.

2.4. Video

Video footage was obtained using a Hi-8 underwater colour video camera. Video recordings were assessed at each station and environmental variables were scored as an average value for all frames observed 2 m either side of the stations. Videos were scored according to the criteria described by Crawford et al. (2001). The variables measured included a numeric categorisation of sediment colour, *Beggiatoa* density, presence of gas bubbles, feed pellets or farm debris, prevalence of burrows, casts and tracks, abundance of molluscs, ophiuroids, annelids and small fish, and the occurrence of locally common seastar species (e.g., *Coscinasterias muricata* and *Asterias amurensis*).

2.5. Statistical analysis

Univariate data were analysed by Analysis of Variance (ANOVA) with homogeneity of variances checked using residual plots. Data were untransformed. A two-way fixed effects model ANOVA, with factors station and time, was used to assess variation in particle size, organic matter, sulphide concentration and macroinvertebrate diversity (Shannon index, Shannon and Weaver, 1963). Tukey's Honestly Significant Difference post-hoc test was used following a significant ANOVA result.

Multivariate analyses were conducted on the community data and video results using the ecological research software package PRIMER© (PRIMER, 2001). Benthic replicates were combined and square root transformed to adjust the importance of species dominants. Macrofaunal and video data were analysed from 3 positions representative of cage effect (−10 m and 0 m combined), more general farm effects (10 m) and unimpacted conditions (150 m).

The data are displayed as ordination plots using nonmetric multi dimensional scaling (MDS). SIMPER analysis was used to determine if any particular species or factors were indicative of these patterns (Clarke and Warwick, 2001). The interaction among groups and time in the macrofaunal data was evaluated using one-way ANOSIM for the group*time combinations. Where this was significant, pairwise comparisons were made. Video data were assessed using two-way crossed analysis of similarities (ANOSIM) to test for differences in community composition among groups within each time and for differences over time (allowing for the fact that there may be differences between groups). As only a single video assessment was undertaken for each transect there was insufficient power to calculate significance levels for a one-way ANOSIM of all group and time combinations.

3. Results

3.1. Particle size distribution

The predominant sediment type at all stations was silt/clay (<0.063 mm). There was no significant change in the silt/clay fraction at each station over time ($F_{\text{time}*\text{station}} = 0.859$, $df = 18, 40$, $P = 0.625$) or through time ($F_{\text{time}} = 0.549$, $df = 6, 40$, $P = 0.766$), but there were significant differences between stations ($F_{\text{station}} = 24.64$, $df = 5, 40$, $P < 0.001$). Post-hoc comparisons showed that the proportion of the silt/clay component was significantly lower at the −10 m station than at any of the other stations and was significantly reduced at the 0 m station compared with the reference (Fig. 3).

3.2. Organic matter measurement

Organic content was significantly different among stations ($F = 5.67$, $df = 5, 123$, $P < 0.001$) and times

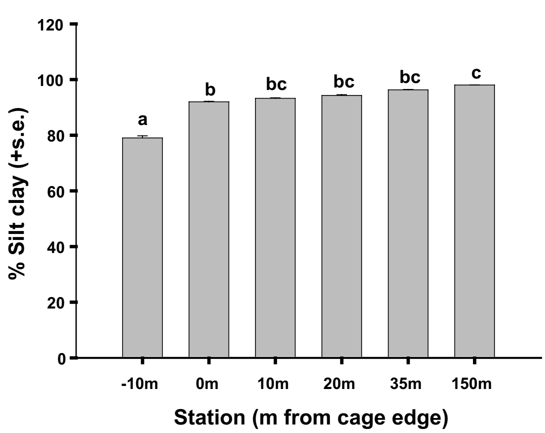

Fig. 3. Mean percentage silt/clay (<0.063 mm) (+s.e.) at each sample station averaged across all times ($n = 54$). Where the letters above each bar differ results were significantly different.

($F = 43.90$, $df = 6, 123$, $P < 0.001$) but the interaction between station and time was not significant. At the 10 m station organic content was consistently higher than all the other stations, and no other differences were seen (Fig. 4a). Initial organic matter levels were high at all stations (c. 20%) (Fig. 4b). Levels generally declined by between 30–40% at all stations during the first two years (Fig. 4b). However, in the last twelve months organic matter increased slightly and the overall reduction from the start to end of study was only 10–25%.

3.3. Sulphide

Sediment sulphide levels exhibited a clear spatial and temporal gradation of effect (Fig. 5) and a significant interaction between station and time of sampling was identified ($F_{\text{Station}*\text{Time}}$ 7.244, $df = 25, 17$, $P < 0.001$). Sulphide concentration at the cage stations decreased markedly over time. Only in the first 2 months were there significant differences amongst stations (Fig. 5). At 1 month the sulphide levels were highest at the −10 m station followed by 0 m and then 10 m stations. Levels were considerably lower at the 20 m, 35 m and reference (150 m) stations and these stations were not significantly different. At 2 months only the −10 m station levels were significantly higher (greater than × 100) than at the reference stations. Sediment sulphide levels diminished both over time and with distance from the cage site. Levels at the 20 and 35 m stations levels remained equivalent to reference throughout the study. Sulphide concentrations at the 0 and 10 m stations were similar to the reference within 2 months and by 6 months the −10 m stations were comparable to the reference (Fig. 5). After 36 months there were no significant differences between any of the stations.

3.4. Macrofauna

Changes in diversity, using the Shannon diversity index, indicated an interaction between station and time ($F_{\text{Station}*\text{Time}} = 2.72$, $df = 30, 210$, $P < 0.001$). Diversity was consistently >1.0 at the reference and at stations

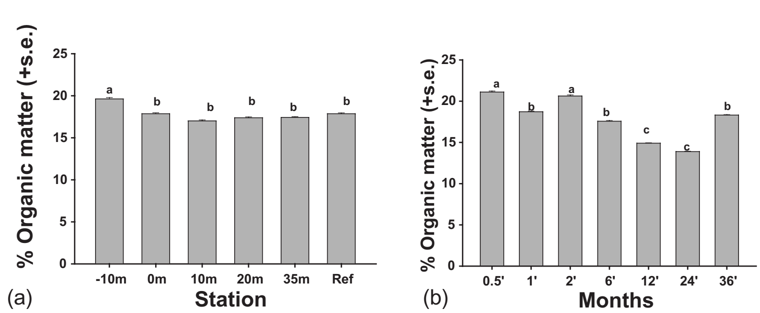

Fig. 4. Average percentage organic matter content (+s.e.) at (a) each sample station and (b) at each sample time. The letters above the bars indicate the results of Tukeys post hoc test, means with different letter are significantly different.

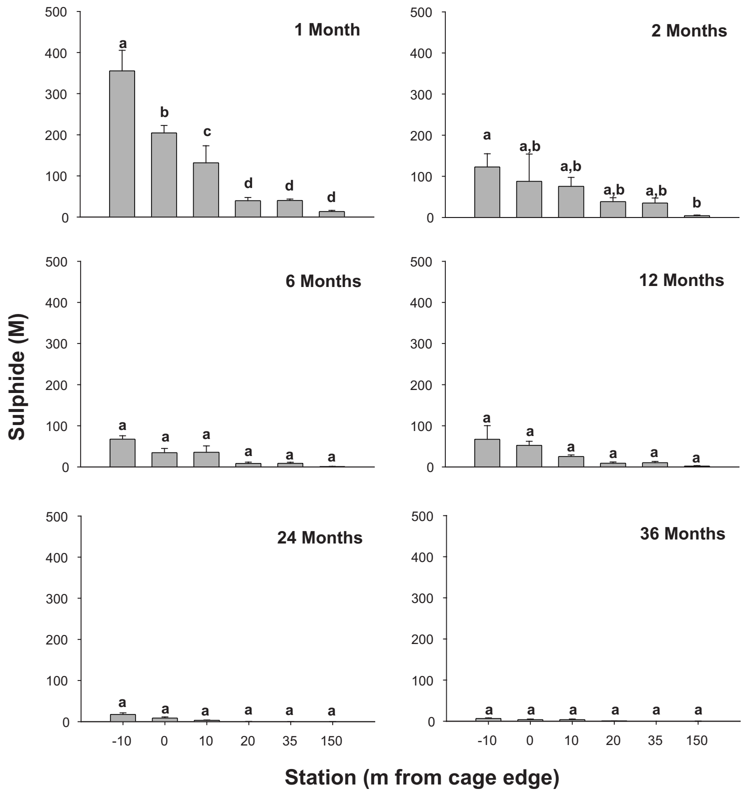

Fig. 5. Average sediment sulphide concentration in μM at 4cm depth (+s.e.) at each sample station and time. (Data not available for 0.5 months.) The letters above the bars indicate the results of Tukeys post hoc test, means with different letter are significantly different.

10 m or more from the cage edge (Fig. 6). It dropped to approximately 0.5 at the −10 m station for the first 2 months and at the 0 m station at 1 month, and only at these positions and times was the diversity significantly lower than the reference (Fig. 6). Diversity at the 10, 20, 35 m and reference stations did not differ significantly over time (Fig. 6a) and the diversity at these stations was comparable at each sampling time (Fig. 6b).

One-way ANOSIM of all group and time combinations for the full community dataset indicated a significant interaction between group and time (Global $R = 0.454$, $P < 0.001$) which suggests that the spatial groups responded differently over time. There were no differences between the farm and reference stations through time but the cage communities changed progressively over time (Table 1a and b). Except at 24 months the cage and reference communities were significantly different at all equivalent times (Table 1a). The cage community at 0.5 months was significantly different to all other times after 1 month (Table 1b). A marked change in the community structure was evident after 6 months, after which the cage positions differed significantly from the initial cage community. By 36 months the cage stations were different from all times earlier in the study, and although not significantly different from the farm station, they were still significantly different from the reference station (Table 1b). The only difference between the farm and reference communities was at 0.5 months (Table 1b).

The two dimensional ordination plot (Fig. 7) shows a spatial progression with the cage stations on the far left of the plot, the reference stations on the far right of the plot and the farm stations forming a central group. At the cage stations a temporal gradation was also evident within the spatial distribution; the earliest impacted samples tending towards the left and the later samples tending towards the right. The demarcations between the cage and farm groups and between the farm and reference groups are not well defined indicating that both the temporal and spatial changes in community structure were gradual rather than sudden. However, the differentiation of the cage stations from the reference stations was clear.

3.5. Video assessment

Two-way ANOSIM of the a priori groups (cage, farm and reference) and time indicated significant differences between both groups (Global $R = 0.326$, $P < 0.001$) and times (Global $R = 0.288$, $P < 0.001$). Pairwise comparisons showed that within groups there were significant differences between the cage and all other groups (Cage/Farm $R = 0.250$, $P = 0.002$; Cage/Reference $R = 0.511$, $P = 0.001$) and also between the farm and reference

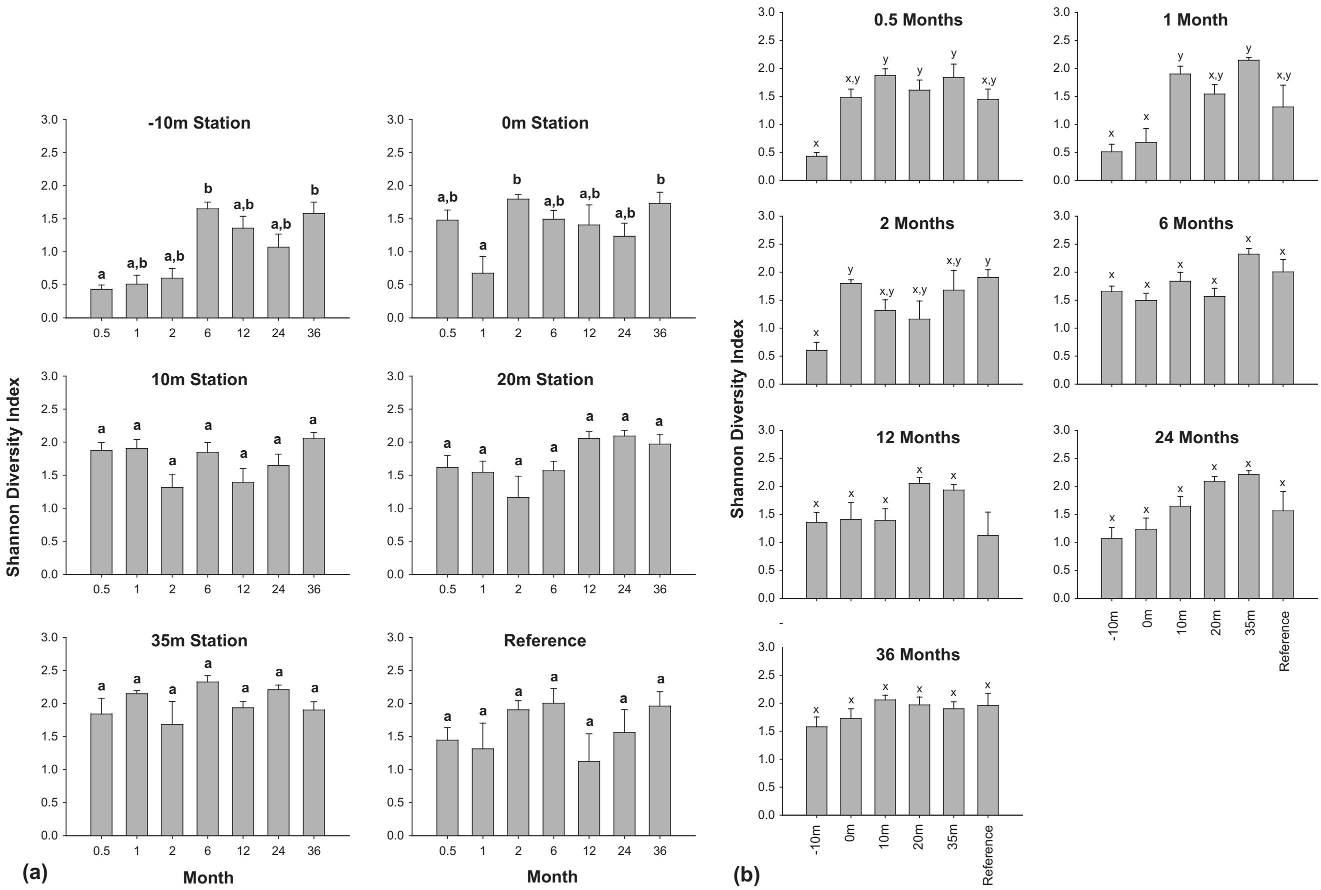

Fig. 6. Average Shannon index value (+s.e.) at each (a) sample station and (b) time. The letters above the bars indicate the results of Tukeys post hoc test for comparisons within each station over time (a,b,c) and for all stations within each time (*x*, *y*, *z*), means with different letter are significantly different.

Table 1
Probability values from the one-Way ANOSIM comparison of benthic data for selected sample station groups (C-cage and R-reference) and times (Bonferroni corrected $n = 84$, $P < 0.0006$)

(a) Cage vs reference at each time	Probability
0.5 month	**0.0001**
1 month	**0.0006**
2 months	**0.0002**
6 months	**0.0003**
12 months	**0.0002**
24 months	0.003
36 months	**0.0001**

(b) Group/time	0.5C	1C	2C	6C	12C	24C
1C	0.2219					
2C	**0.0023**	0.1845				
6C	**0.0001**	**0.0006**	**0.0005**			
12C	**0.0001**	0.0037	0.0062	0.0043		
24C	**0.0001**	0.0015	**0.0006**	0.0028	0.0098	
36C	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**

(a) Comparison of cage and reference communities within times, (b) comparison of cage communities over time. Categories with significant differences are shown in bold. The only significant difference between the farm and cage communities was at 0.5 months ($P = 0.0002$) and there were no significant differences between the farm and reference communities, so farm results not shown.

groups (Farm/Reference $R = 0.418$, $P = 0.001$). Pair-wise comparisons of all time combinations, adjusted for multiple comparisons ($n = 21$, $P = 0.002$), indicated that the visual condition of the sediments at 0.5 months was

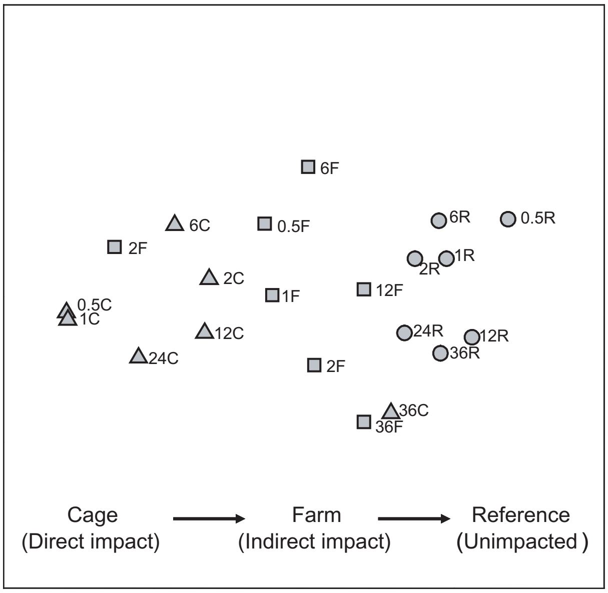

Fig. 7. Ordination analysis—2-dimensional MDS plot of species abundance data. Stress = 0.12. The prefix indicates the time of sampling in months whilst the symbol and suffix indicate the impact group, cage (▲C), farm (■F) and reference (●R) respectively.

significantly different from that at 12 months. Video footage at 2 months differed from that at 12 and 24 months and footage from 6 months was significantly different to that from 24 months. However, differences between communities at the later sample times (12, 24, 36 months) were less obvious. Ordination of the video assessment data reveals a spatial gradient in the stations across the plot which largely separates the cage stations in the first 12 months from the remaining stations (Fig. 8). These remaining stations were statistically indistinguishable from one another. SIMPER analysis (Table 2) of the two main groups identified profusion of burrows and faunal tracks, sediment colour and the presence of *Beggiatoa* as the primary factors in the group determination.

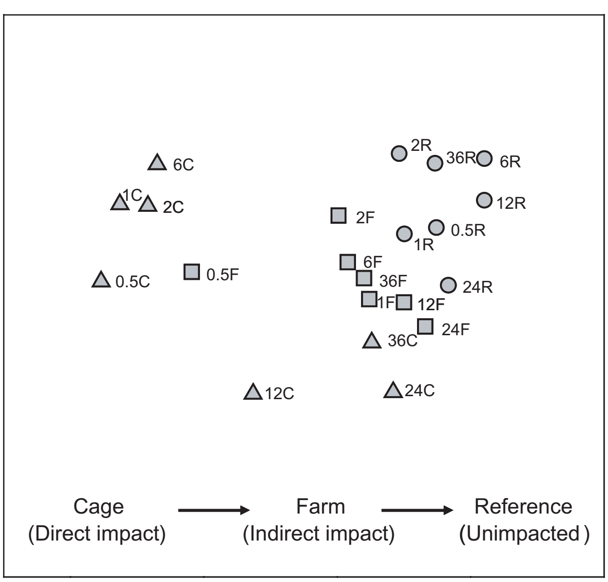

Fig. 8. Ordination analysis—2-dimensional MDS plot of video assessment data from all sample stations where video footage was usable. Stress = 0.09. The prefix indicates the time of sampling and the symbol and suffix indicate the impact group, cage (▲C), farm (■F) and reference (●R) respectively.

4. Discussion

The high levels of silt and clay in the sediment of the study site signify that both within the lease area and at the reference stations the sediments were "depositional" (Rosenthal et al., 1988) and the flow rates indicate that the extent of waste dispersion from the cages would be limited. This suggests that the benthic impacts from aquaculture operations would be highly localised but would also not be readily mitigated by natural hydrographical processes. Fish farm sediments generally have a very high organic matter content which can be strongly anoxic and rich in sulphides (Brown et al., 1987; Frogh and Schaanning, 1991; Brooks et al., 2003). Recent regulations proposed for British Columbia, Canada (Levings et al., 2002) identify the "trigger" standard for sulphide as 1300 μM, whilst Scottish regulations set their minimum limit for action at 3800 mg/kg sediment dry wt (~20,000 μM) (SEPA, 1998). Under these guidelines even the most impacted samples from the current study would fall well within the acceptable range. Using the sulphide level categorisation proposed by Wildish et al. (1999) for New Brunswick, Canada, the highest sulphide levels observed in the present study indicate only hypoxic/moderately polluted conditions. The sulphide levels at all stations also diminished rapidly, suggesting that the sediments were recovering and after 24 months levels were indistinguishable from background conditions.

Measurement of organic matter has been widely used as a surrogate for organic enrichment. However, several recent studies suggest that evaluation of organic matter content is not always a useful measure of farm impact (e.g. Johannessen et al., 1994; Hargrave et al., 1997; CSIRO Huon Estuary Study Team, 2000; Macleod, 2000; Crawford et al., 2002; Brooks et al., 2003). 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 study of the nearby Huon estuary comparably

Table 2
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 variables in each of the a priori groups (Cage, Farm and Reference) and (c) average abundance, ratio (average similarity/standard deviation similarity) and cumulative % similarity of the five variables which most clearly distinguish the main groups identified by cluster analysis

Species name	Average abundance	Ratio	Percent similarity	Cumulative % similarity
(a) Group 1				
Sediment colour	1.43	2.21	42.09	42.09
Beggiatoa density	1.10	1.11	22.80	64.89
Worm cast density	0.57	1.27	13.98	78.87
(b) Group 2				
Burrow density	2.21	3.72	39.50	39.50
Density of faunal tracks	1.48	1.86	23.47	62.97
Mollusc abundance	0.96	2.10	14.51	77.48
	Group 2 Average abundance	Group 1 Average abundance	Ratio	
(c) Between groups				
Burrow density	2.21	0.33	2.95	18.72
Sediment colour	0.06	1.43	2.34	32.87
Density of faunal tracks	1.48	0.10	2.20	46.85
Beggiatoa density	0.00	1.10	1.52	57.82
Mollusc abundance	0.96	0.05	1.96	67.01

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). Results for bulk organic matter parameters in the Huon estuary indicated that a significant portion of organic waste remained in the sediments after twelve months and suggested the residual organic material may be more refractory (McGhie et al., 2000) and therefore not as readily available to the biota. Organic matter levels directly under the cages remained high throughout this study, whereas other aspects of the sediment chemistry and biology indicated improvements in sediment conditions. It may be that a large proportion of the organic matter is refractory and therefore is not assimilated. Nonetheless, the results suggest that measurement of organic matter level is a poor indicator of sediment recovery.

At the time of cage removal the macrobenthic community structures at the cage associated stations (−10 and 0 m) were clearly impacted. The fauna was impoverished, species diversity (Shannon index) was very low, and the community structure was similar to that described by Pearson and Rosenberg (1978) as characterising the "polluted" zone. The community was dominated by the opportunistic polychaete *Capitella capitata* complex, a species indicative of organically enriched conditions. Multivariate analysis of the data clearly indicated that the community structure of these stations changed over time in a manner suggestive of recovery. Nevertheless, the community structure at the under cage stations (−10 m) remained impacted 36 months after the cages had been removed and a moderate impact could still be distinguished at the 0 m stations after 24 months. At the farm stations it was difficult to discern a clear impact at any time, although the community often contained transitional species.

Estimates of benthic infaunal recovery from caged fish farming have ranged from 7 weeks in coastal waters off S.E. Tasmania (Ritz et al., 1989) to 21 months from the west of Scotland (Black, 2001) and greater than 23 months in relation to sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*) culture in Cephalonia Bay, Greece (Karakassis et al., 1999). The variability in these earlier estimates is probably a combination of both ecosystem and farm management differences. The studies by Black (2001) and Karakassis et al. (1999) are more comparable to the current study even though the culture species differ. Black (2001) contrasted his results with those of Karakassis et al. (1999) and noted that the recovery rates seemed to be much higher in the warmer waters. However, local hydrographic conditions influence recovery rates and in quiescent areas recovery may take much longer than in more hydrodynamically energetic areas (Black, 2001). The difference between the rate of recovery observed in the current study and that of the earlier study by Ritz et al. (1989) is probably in part due to differences in background environmental conditions and in part to the expansion and intensification of the industry since that time. There have been significant changes in cage design and stocking densities are now much greater.

Although the benthic community changes observed in the current study were similar to those described elsewhere (Johannessen et al., 1994; Findlay et al., 1995; Karakassis et al., 1999; Wildish et al., 1999; Brooks et al., 2003), there were marked differences between the absolute quantities of the chemical indicators and the level of impact inferred by the changes in community structure. Overall trends were similar, with both sulphide and organic matter levels increasing significantly where major community changes occurred. However, the magnitude of change was very different—sulphide levels were substantially lower under Tasmanian conditions than observed in Canada or Scotland and organic matter levels were considerably higher (Gowen et al., 1988; Holmer and Kristensen, 1992; Wildish et al., 1999; Brooks et al., 2003). This clearly reflects geographical differences and suggests that although trends may be similar absolute levels cannot be directly extrapolated over large spatial scales.

Video is regularly employed by farmers in Tasmania as a means to examine and evaluate seabed condition. Video footage is generally assessed qualitatively and the current study identified several characteristics which represented consistent indicators of severe impact. Bacterial mats (*Beggiatoa* spp.), blackened sediments and gas bubbles are clear visual indicators of impacted sediments (Crawford et al., 2001). The video assessment suggested that the density (size and thickness) of the *Beggiatoa* mats had increased at 10 m stations between 0.5 and 1 month after removal of the cages. The presence of *Beggiatoa* mats at the −10 m stations as late as 6 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 (Frogh and Schaanning, 1991). Consequently diver observations of no *Beggiatoa* under the cages at initial sampling and subsequent increase in mat density in the first month, suggested that initially the sediment was anoxic and *Beggiatoa* development was inhibited. The presence of infaunal species under these conditions is probably a function of the particular species capabilities. These species were generally highly tolerant of hypoxia, and were able to irrigate their burrows by extending tubes into the better oxygenated overlying water.

Assigning values to observed video features allowed direct comparison between locations and over time. The multivariate analysis of the video parameters in the current study suggested that at 10 m from the cage the sediment had recovered sufficiently to be indistinguishable from the reference conditions after only 1 month, but that there was still a significant impact beneath the cages 12 months after their removal. This approach for evaluation of video footage is relatively simple and makes information obtained from video footage more useful. Video data is relatively quick and easy to collect and video results can assist the interpretation and presentation of data produced by other measurement techniques. Video footage is also extremely effective in presenting highly impacted conditions.

This study showed marked differences in the sensitivity of a number of different assessment techniques. Sediment chemistry responded to the changing environmental conditions more quickly than the benthic infaunal community. Video assessment was an effective means of evaluating recovery, although it also indicated a more rapid recovery than the benthic community.

Although the pattern of recovery indicated by the macrofauna in the present study was consistent with that reported from the northern hemisphere (Pearson and Rosenberg, 1978; Johannessen et al., 1994; Findlay et al., 1995; Karakassis et al., 1999; Wildish et al., 1999; Brooks et al., 2003) geographic differences between the levels of impact suggested by associated sediment chemistry measurements were apparent. This highlights the importance of collecting baseline information and determining sediment recovery/degradation rates in relation to local environmental conditions when environmental regulations/guidelines are being established.

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