

# Long-term Recovery – Review of sediment condition at Marine Farm lease No.76 (Gunpowder Jetty), North West Bay

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**Title**

Long-term Recovery –Review of sediment condition at Marine Farm lease No.76 (Gunpowder Jetty), North West Bay

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## **Non-technical summary**

This study was undertaken to assess the Gunpowder Jetty site (Marine Farm lease No.76) 10 years after it was finally vacated to determine whether the benthic, visual and physical-chemical conditions were consistent with background conditions and to what extent the system has recovered. The results suggest that, based on the selected environmental condition variables assessed, the sediments around the Gunpowder lease have largely recovered. Whilst some parameters still showed small differences between locations these do not appear to be having a major effect on the ecology or nutrient mineralisation processes and consequently there is no evidence that the farming activity has had any permanent impact on the ecology in this area.

**KEYWORDS:** aquaculture impacts, environmental assessment, benthic infaunal assessment, sediment biogeochemistry, nutrients dynamics

## **Acknowledgements**

We would like to thank the staff at DPIPWE for their help in compiling the data, in analysing the video data and in providing feedback and suggestions along the way (particularly Graham Woods and Eric Brain) and Marc Santos and John Preston for assistance with field sampling.

## 1. Introduction

Management of the environmental impact of finfish farms requires a good understanding of the local physical, chemical and biological processes, and a reliable understanding of enrichment and recovery status. Localised aquaculture impacts have been well studied (e.g. 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 . Buschmann et al., 2006; Hargrave et al., 1997; Janowicz and Ross, 2001; Karakassis et al., 2002). Organic enrichment is the main effect of cage fish farming and areas of localised organic enrichment can arise when fish feed and faeces settle on the seabed (Brown et al., 1987; Lumb, 1989; Holmer and Kristensen, 1992; Hargrave et al., 1993; Karakassis et al., 2000). 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 and Bradbury, 1987, Gowen et al., 1988; Rosenthal and Rangeley, 1988; Holmer and Kristensen, 1992, Hargrave et al., 2008). As this organic matter degrades the sediment oxygen demand increases and the underlying sediments become deoxygenated (Hargrave et al., 2008). This results in a lowered redox (oxidation-reduction) potential and increased sediment sulphide concentrations (Hargrave et al., 2008). Sediments under cages can also be enriched with zinc and copper sourced from feed or antifouling agents. These conditions will affect the prevailing benthic communities, changing the species mix and in the worst case scenarios where the sediments are anoxic, azoic conditions can result (Gowen and Bradbury, 1987).

Whilst it is important to have a good understanding of the extent to which the sediments are impacted, it is also important to understand the potential for recovery, as this determines how long an impact will persist and the potential for reversibility of any effects. To date estimates of recovery vary greatly, from less than six months (Brooks et al., 2003; Ritz et al., 1989) to more than 5 years (Brooks et al., 2004). This variability in the estimates of recovery probably reflects differences in both underlying ecosystem conditions and farm management but may also be related to the specific recovery end-point employed, as the benchmark against which recovery is being judged can have a significant impact on recovery time estimations (Macleod et al., 2004b, Keeley et al., 2013). That said, most studies tend to target short-term recovery, with only a very few addressing longer-term remediation. There are still relatively few studies which have been conducted over long enough timeframes to

identify “complete” recovery i.e. to pre-impact condition or to conditions consistent with those of selected control sites (e.g. Karakassis et al., 1999; Macleod et al., 2004b).

Against this background, a study was initiated in 1999 jointly by the aquaculture industry (Aquatas Tasmania Australia Pty Ltd) and the Department of Primary Industries, Water and Environment (DPIWE) 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 sediment 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 following of an intensively farmed marine finfish cage site in the cool temperate waters of Tasmania. The results of this research were published in Macleod et al (2004) and the key findings suggesting that sediments under the cage were highly impacted at the time that the cages were removed but that the extent of impact diminished with both time and distance from the cages. However, the results suggested that although most of the biogeochemical parameters assessed had returned to conditions equivalent to those at the reference after 2 years, the benthic community structure still differed significantly.

This project follows on from that original study to evaluate the benthic, visual and physical-chemical condition of the sediments at the Gunpowder lease almost 13 years after it was first assessed, and 10 years after it was finally vacated. These results provide a valuable insight into the rate and extent to which the sediment has recovered and inform our understanding of ecosystem resilience, providing valuable information for long-term management of aquaculture lease areas to ensure the sustainability of ongoing operations.

### **Study Aims:**

- Evaluate whether sediments within and around the Gunpowder Jetty lease have recovered such that they are consistent with reference conditions.
- Determine whether there is any significant difference in community composition between any of the sampled sites
- Assess whether there is any significant difference in functional performance between the farm/cage and reference sites

## **2. Methods**

Sampling was undertaken at the same sites and in a manner consistent with the protocols described in the original study (Macleod et al., 2002). In the initial study all samples were collected by diver however, more recent safety precautions now prohibit diving under these conditions and so the sediment samples in the current study were all collected by core or grab, with the video footage collected by ROV.

Included in this temporal comparison are the data collected one week after removal of cages, the data from 24 months after removal and the most recent data collected almost 13 years after initial cage removal (which is approximately 10 years after complete vacation of the farm in July 2003).

## 2.1 Sample Site Location

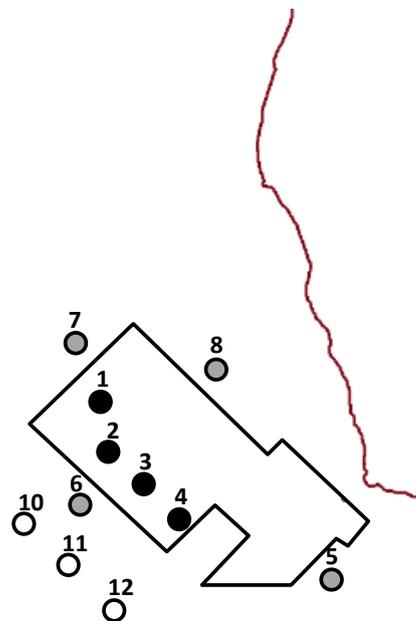
The Gunpowder lease (lease 76), in North-West Bay, Tasmania, was first granted to Aquatas Pty Ltd in 1985 and operated as a commercial salmon farming operation until August 1999. When in operation, the farm was a smolt ongrowing site, occupying an area of 3.12ha, with up to 16 production pens including 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 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 (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". The site was restocked briefly after the initial study, from mid-October 2002 to early-April 2003, and was permanently vacated shortly thereafter.

Three cage positions were randomly selected prior to the removal of the cages, in late August 1999 for inclusion in the initial study, and fixed transects were positioned on the seabed running from directly beneath the cage (-10m) to a distance 35m from the cage boundary (Fig. 1, Table 1). Reference stations were established 150m from the cage position. The positions of all stations were fixed using a differential global positioning system (DGPS). In this follow-up study only the -10m and 150m stations were re-assessed.

In the final recovery assessment samples were also collected from reference and compliance positions originally sampled as part of the DPIPWWE original baseline survey at this farm site. Consequently, the full suite of samples surveyed includes; 6 sites consistent with the original University of Tasmania study (i.e. the -10m and 150m for each of the original transect positions) and 6 sites collected from DPIPWWE's original baseline survey (Fig. 1). In combination these sites reflect 4 previously farmed sites (1-4), 4 boundary/compliance sites (5-8) and 4 unfarmed reference sites (9-12); the location of the sites is shown in Fig. 1 and their position information and identification in each of the previous studies is shown in Table 1.

**Table 1.** Site identification details – showing current and previous labels, eastings and northings and relevant impact category.

Site ID	Original ID	Easting/ Northing	Category
1	MFB 1	524242 / 5234269	Farm
2	TAFI T1 (-10m)	524250 / 5234204	Farm
3	TAFI T2 (-10m)	524299 / 5234159	Farm
4	TAFI T3 (-10m)	524348 / 5234111	Farm
5	MFB 2	524208 / 5234349	Compliance
6	MFB 6	524214 / 5234129	Compliance
7	MFB 3	524396 / 5234312	Compliance
8	MFB 5	524549 / 5234032	Compliance
9	MFB 7	523918 / 5234687	Reference
10	TAFI T1 (150m)	524139 / 5234105	Reference
11	TAFI T2 (150m)	524199 / 5234051	Reference
12	TAFI T3 (150m)	524260 / 5233990	Reference



**Figure 1.** Location of sampling sites in relation to the original Gunpowder Jetty lease. (Reference - ○ , Compliance - ● , Farm - ● ).

## 2.2 Sediment Characterisation(Physical-Chemistry Properties)

At each sample station three replicate core samples were collected using perspex tubes 250mm long and 45mm internal diameter. Samples were collected for analysis of particle size composition, organic content, loss on ignition, (LOI), redox potential, sulphide concentration. Redox potential and sulphide concentration were measured at two points (1cm and 4cm) in a single core from each triplicate.

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,

500 $\mu$ m, 250 $\mu$ m, 125 $\mu$ m and 63 $\mu$ m). The sediment retained on each sieve was dried and weighed and the percentage of the total sample weight calculated. The fraction smaller than 63 $\mu$ m was determined by calculation of the difference between the initial sample weight and the combined weight of the retained fractions.

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 1M HCl. The samples were oven dried for 24 hours at 60°C 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.

In the laboratory sub-samples (4ml) were taken from each replicate core for sulphide analysis 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 to establish electrode calibration curves.

Redox potential data was not analysed in this study as the original data from 1999 and 2002 was considered doubtful due to contamination of the redox probe during the study, and it would appear that a similar issue may have occurred with the 2012 data.

Sediment samples for metal (copper and zinc) analysis were collected using a multicore sampling device. Upon retrieval the cores were sealed using latex bungs. Overlying water was carefully removed, and the top 30 mm of sediment extruded using a plunger; the sediment samples from all 3 mini cores collected at each position were combined and homogenised in a 120 ml jar. Samples were refrigerated prior to analysis for total recoverable copper by CSIRO.

Samples of porewaters for evaluation of copper and zinc loading were collected from the bulk sediment samples. Duplicate tubes were centrifuged at 3500 rpm for 15 minutes, at 4°C. The supernatant from each of the duplicate sub-samples was combined to give sufficient volume for analysis, and filtered through 0.45  $\mu$ m filters into 10 mL acid washed vials. This process was undertaken in a nitrogen filled glove box to avoid introducing additional oxygen to the samples. Samples were acidified with Merck Suprapure Nitric Acid to 0.2% v/v. A procedural blank (deionised water) was prepared with every 10 samples. Samples were sent to CSIRO for analysis. Where results are reported as below detection, the limit of reporting has been halved for the purpose of plotting and statistical analysis.

### **2.3 Faunal Characterisation**

In the initial study five replicate samples were collected by diver for assessment of the benthic macrofaunal community structure at each sample sites using hand held 150 mm diameter PVC pipe corers to a depth of 100mm (sampling area of 0.0177m<sup>2</sup>). Samples were placed immediately into mesh bags (0.875mm), and then rinsed on the boat prior to being 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. In 2012 the same processing and identification protocols were followed but only 3 replicate samples were collected at each sample position using a Van Veen grab (sampling area of 0.0675m<sup>2</sup>).

### **2.4 Video Footage**

Camera drops/ ROV footage was collected at each location in 1999 and 2012 and analysed by DPIPWE according to the criteria outlined in Marine Farm Environmental Monitoring protocols and described by Crawford et al. (2001).

### **2.5 Mesocosm Incubations**

Three replicate large mesococm core samples were collected for incubations from each of the original TAFI sites at the -10m and 150m positions (sites 2, 3, 4, 10, 11 and 12), a total of 18 cores. Incubations were profiled for oxygen and nutrient flux rates assessed (start/ end).

Sediments for mesocosms were collected by box corer. A mesocosm core (300 mm x 150 mm diameter) was pushed into the sediment to a depth of 100 mm, and capped at the bottom. The mesocosm core was gently filled with bottom water collected from the site, and then the top cap put in place. Cores were gently transferred to a Nally bin filled with site water, ice-blocks were added to keep the samples cool during transport and cores were kept upright and as stable as possible to allow sediments to settle. Cores were transported to the laboratory as soon as practicalable. At the laboratory, cores were carefully transferred to the incubation system, large round tubs filled with Tarooma seawater, and allowed to equilibrate; chiller units kept the incubation temperatures equivalent to that of the site bottom water.

In addition sufficient site water to incubate all cores (approx. 400 L) was collected from just above the sediments. Site water was collected from about 1 m off the bottom (to avoid re-suspending sediments) at one sample location at each farm using a length of tubing and a bilge pump. The tubing was flushed with at least 60 L to ensure all water was from the bottom. Carboys were triple rinsed then filled with site bottom water. Site water was kept at a temperature equivalent to that of the relevant site bottom water (determined from CTD profile data) until used in incubation of mesocosms.

Sediments were allowed to equilibrate (24 hours) in the laboratory before incubations commenced. Mesocosm cores were flushed with site water for 20-30 minutes before incubations commenced. Incubations were undertaken in the dark, as the depths from which the samples were collected would preclude phytoplankton or microphytobenthic activity. Water samples were collected for dissolved nutrients (ammonia, nitrate + nitrite, phosphate) and pH analysis from each individual core at the start and end (~ 4 hours) of the incubation to estimate sediment nutrient fluxes. Dissolved oxygen was monitored at the start and end of each incubation using a Hach HQ40d meter with LDO101-15 probe. Data loggers continuously monitored O<sub>2</sub> levels (3 minute intervals) in one or two cores from each treatment. Temperature was maintained at a level approximately equivalent to field conditions using programmable heater/chiller units. Nutrient samples were analysed by Analytical Services Tasmania, with a minimum reporting limit of 2 µg/L as N or P. Nutrient flux rates in µmol/m<sup>2</sup>/hr were calculated from the incubation time, the measured height of water in each core, the surface area of the core and the concentration changes during incubation (Eriksen et al., 2012).

## 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 location and time was used to assess variations in particle size, organic content, sulphide concentration, macroinvertebrate abundance, number of species and Shannon diversity. One-way ANOVA to test for differences in sediment fluxes and metal loadings between locations.

Multivariate analyses were conducted on the community assessment results to identify patterns within the data, with the results displayed as ordination plots using multi-dimensional scaling (MDS). Other than where indicated, 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 farm, compliance and reference at each time to test within and between these *a priori* groups. SIMPER analysis was used to determine if any particular species were indicative of these patterns (Clarke and Warwick, 2001). All the multivariate analysis techniques were included in the Plymouth Routines in Multivariate Ecological Research (PRIMER) software package.

Sample analyses were specifically designed to address the specified study aims:

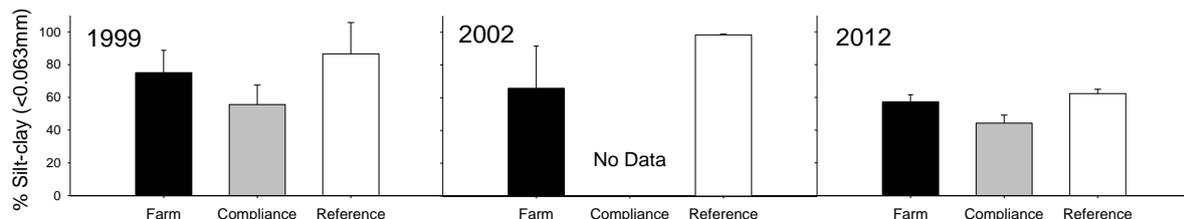
1. Determine whether sediments within and around the Gunpowder Jetty lease have recovered such that they are consistent with reference conditions.  
*- comparison of data from the reference, farm and compliance positions, with a single mean for the combined replicates at each position acting as a single replicate for the respective condition.*

2. Determine whether there is any significant difference in community composition between any of the sampled sites
  - *comparison of faunal data for replicate cores taken from each of the sampling positions in 2012*
3. Determine whether there is any significant difference in functional performance between the reference and previously farmed cage sites
  - *comparison of replicate mesocosm samples (x3) from farm and reference positions in 2012*

### 3. Results and Discussion

#### 3.1 Particle Size Distribution

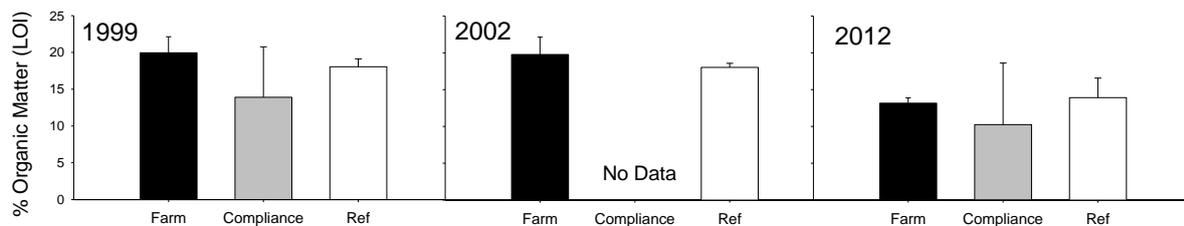
The sediment particle size was generally fine at all sites, with the silt-clay fraction (<0.063mm) comprising more than 50% of the sediment at all sites and times. At the end of farming (1999) there was no significant difference in the percentage silt-clay in the sediments from any sites, even though levels at the compliance sites were lower than in the reference sediments ( $p = 0.07$ ,  $df_{2,7}$ ,  $F = 3.95$ ) (Fig. 2). The percentage silt clay in sediments has not changed markedly at the farm sites since over time, but would appear to have declined slightly at the reference sites over this period, with the average dropping from 98% ( $\pm 0.5$ ) to 62% ( $\pm 2.7$ ) ( $p < 0.01$ ,  $df_{2,6}$ ,  $F = 8.09$ )



**Figure 2.** Average percentage silt/clay (<0.063mm) (s.d.) for farm, compliance and reference sites in 1999, 2002 and 2012.

### 3.2 Organic Matter Measurement

There was no significant difference between organic matter levels at farm, compliance or reference within each year, nor was there any difference in farm levels between 1999 and 2002 (Fig. 3). However, over time there has been a marked decline in organic matter content at the farm sites since 1999 ( $p < 0.01$ ,  $df_{2,6}$ ,  $F = 18.02$ ).

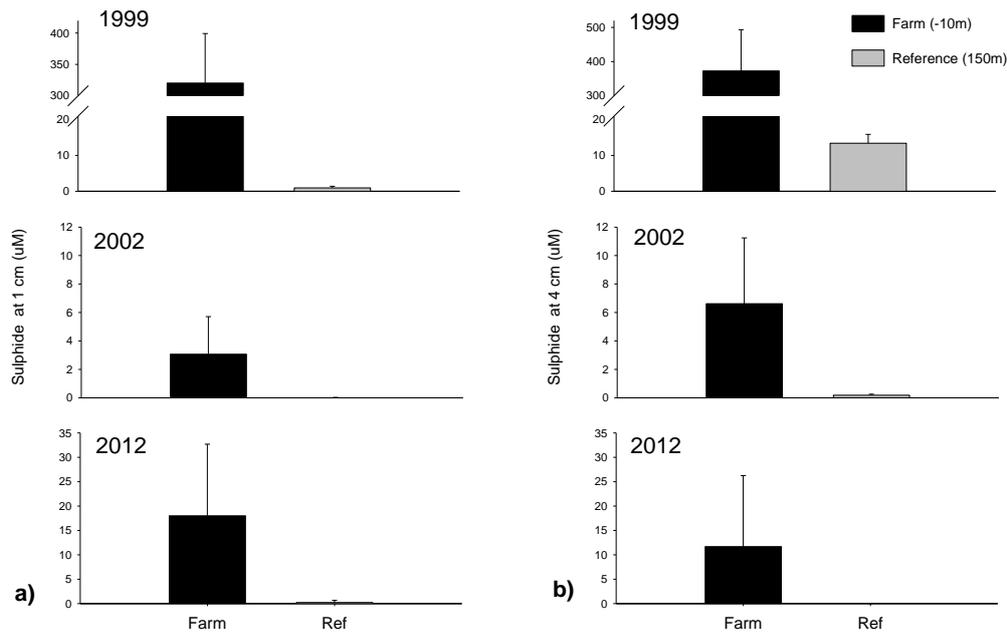


**Figure 3.** Percentage organic matter content (s.d) for farm, compliance and reference sites in 1999, 2002 and 2012.

### 3.3 Sulphide

Sulphide levels were very high at the farm stations (both at 1 and 4 cm depth) at the point at which farming ceased (1999) but dropped at the end of the initial recovery period (2002) and have remained low ever since (Fig.4). Sulphide levels at the reference stations have consistently been much lower, levels being negligible in 2002 and 2012. Consequently the interaction between location and time was highly significant ( $p < 0.01$ ,  $df_{2,6}$   $F = 147.55$ ).

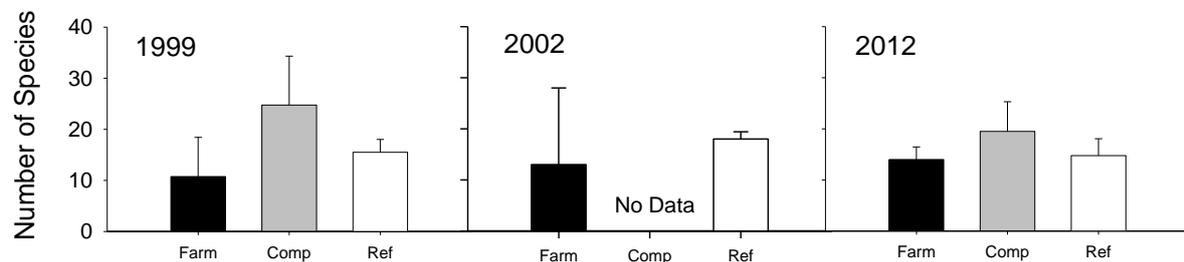
Whilst the sulphide concentrations at the farm stations remained higher than that recorded at the reference stations at all times, it is important to note that levels have decreased markedly since farming ceased and are consistent with levels reported from un-farmed areas elsewhere (Macleod et al., 2004a,b).



**Figure 4.** Sulphide levels (s.d) at a) 1cm and b) 4 cm for farm and reference sites in 1999, 2002 and 2012 (Note that sulphide was not sampled at compliance stations).

### 3.4 Macrofaunal Diversity

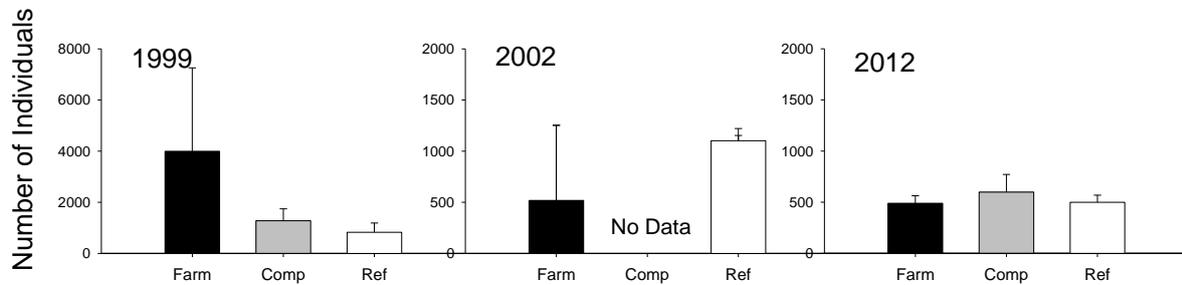
There were differences in the average number of species recorded from the farm, compliance and reference sites over time ( $p < 0.01$ ,  $df_{2,66}$   $F = 11.68$ ) (Fig. 5). This appears to be associated with the compliance site; in 1999 more species were recovered from the compliance locations than in 2012 ( $p < 0.05$ ,  $df_{1,22}$   $F = 6.57$ ) and at the end of the study in 2012 there were significantly more species at the compliance site than either the farm or reference sites ( $p < 0.01$ ,  $df_{2,66}$   $F = 11.68$ ). However, it is important to note that these differences were still relatively minor with the lowest average number of species reported being 10 and the highest 25.



**Figure 5.** Average number of species (s.d.) at farm, compliance and reference sites in 1999, 2002 and 2012.

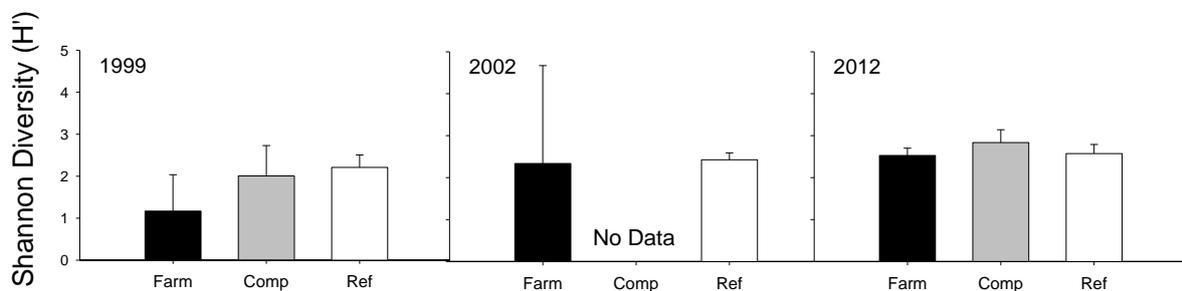
Total abundance has decreased markedly at farm sites since the cessation of farming ( $p < 0.01$ ,  $df_{2,33}$   $F = 6.93$ ) (Fig. 6). Similarly abundances have also declined significantly at the compliance sites ( $p < 0.01$ ,  $df_{1,22}$   $F = 11.68$ ). Although abundances were very much lower at the

reference sites than at either compliance or farm sites in 1999 and 2002 average abundances at these sites were still slightly elevated relative to the results for 2012 ( $p < 0.05$ ,  $df_{2,27}$   $F = 3.48$ ). However, in 2012 there was no significant difference between any of the sampling positions ( $p = 0.13$ ,  $df_{2,33}$   $F = 2.15$ ).



**Figure 6.** Average abundance per m<sup>2</sup> (s.d) at farm, compliance and reference sites in 1999, 2002 and 2012.

Total diversity improved markedly at the farm sites between 1999 and 2002 ( $p < 0.01$ ,  $df_{2,27}$   $F = 15.41$ ) and has remained at a similar level ever since (Fig. 7), although the variability observed at the farm site in 2002 appears to have stabilised. Once again, in 2012 there was no significant difference between any of the sampling positions ( $p = 0.21$ ,  $df_{2,33}$   $F = 1.60$ ).



**Figure 7.** Shannon diversity (s.d) at farm, compliance and reference sites in 1999, 2002 and 2012.

### 3.5 Nutrient Flux Measures

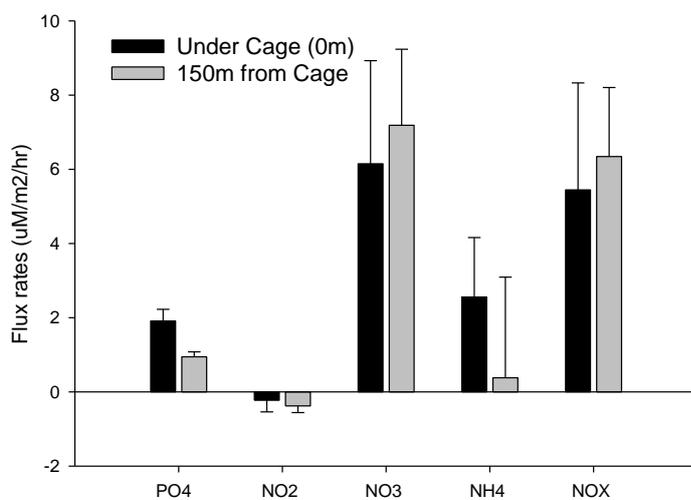
Where waste food and faeces are present in the benthos, they will be mineralised in the sediments and released directly back into the water column as soluble forms that can fuel algal growth (eutrophication) in the water column. The transformation and cycling of nutrients (oxygen, nitrogen, phosphorous) between sediments and the water column is therefore intimately linked to the rate of organic enrichment and degradation.

Consequently the rate at which these processes occur can provide some insight into the condition and performance of benthic habitats.

Generally, high ammonia and phosphorus fluxes from the sediment reflect organically enriched and oxygen depleted sediments. Similarly, fluxes of nitrate in the opposite direction, i.e. from the water column into the sediments, may reflect organically enriched and oxygen depleted sediments. This is because the process of nitrification that creates and supplies nitrate for denitrifying microbes in the sediments is limited in oxygen depleted sediments and the "denitrifiers" must therefore rely on nitrate sourced from the water column.

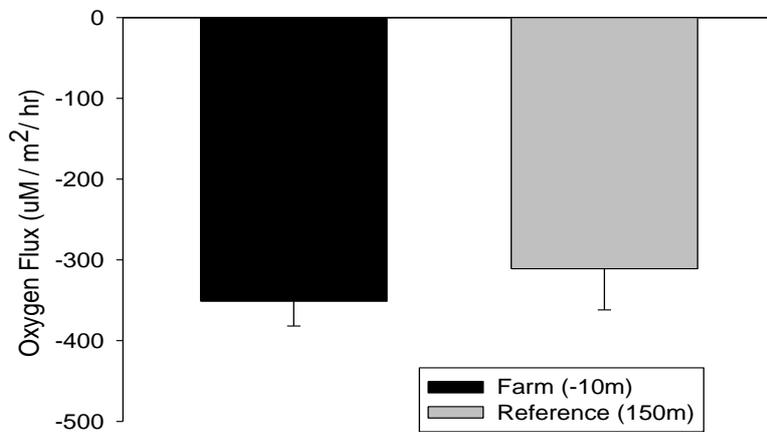
Although the average  $\text{NH}_4$  flux at the site 150m from the cage was less than half that under the cage the replicate results were highly variable and so not significantly different (Fig. 8). Furthermore, these fluxes are very low (by several orders of magnitude) in comparison to fluxes previously recorded under or near operational sea-cages (e.g. Holmer and Kristensen, 1994; Christensen et al., 2000; Holmer et al., 2002, 2003; Bissett et al., 2009; Erickson et al., 2012), and within the range of fluxes reported from reference conditions (e.g. Christensen et al., 2000; Morrisey et al., 2000; Bissett et al., 2009; Erickson et al., 2012). Similarly, the fluxes of  $\text{NO}_x$ ,  $\text{NO}_2$ ,  $\text{NO}_3$  and  $\text{PO}_4$  measured at the farm and reference sites are consistent with fluxes reported from reference conditions elsewhere (e.g. Christensen et al., 2000; Morrisey et al., 2000; Bissett et al., 2009; Erickson et al., 2012), including previously reported fluxes from other areas remote from fish farming within NW Bay (Macleod et al., 2008). There was no significant difference between the flux rates for any of the nitrogen species at the previously sampled cage and reference positions. Nitrate, ammonia and  $\text{NO}_x$  all indicate positive effluxes of nutrients from the sediments whilst nitrite was generally absorbed.

There were significant differences in phosphate ( $\text{PO}_4$ ) concentrations ( $df_{1,5}$ ,  $F=23.80$ ,  $p=0.008$ ), but the overall rate was once again very low; the observed difference well within the flux range that might be expected from normal, oxygenated fine sediments. Fluxes from actively farmed sediments easily being an order of magnitude higher (Erickson et al., 2012).



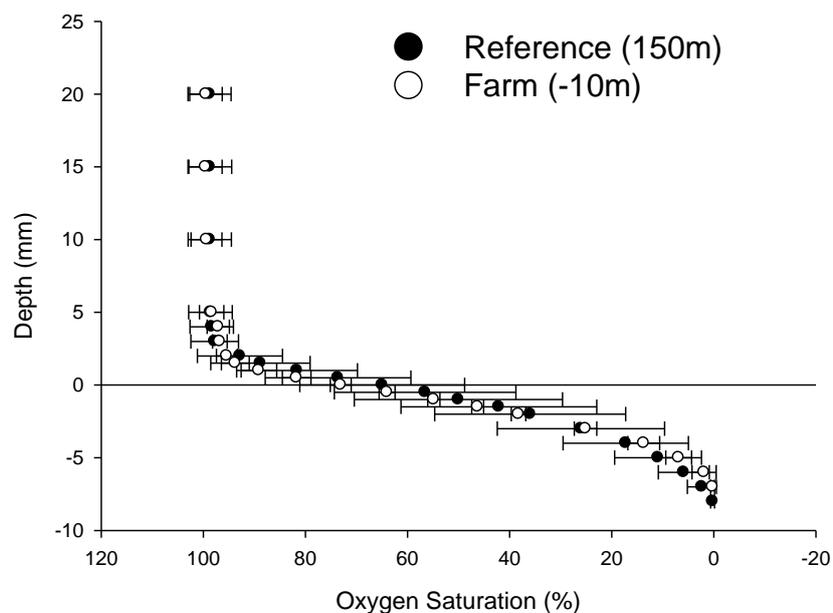
**Figure 8.** Nutrient flux rates (s.d) in 2012 associated with sediments collected from the original cage transect sampling positions (-10m and 150m).

Oxygen flux rates provide a result that is consistent with the interpretation of the other nutrients (Fig. 9). Oxygen is being drawn into sediments but not at an excessive rate, which suggests a healthy respiring benthic ecology, and there is no significant difference between the levels at the farm and at the reference locations.



**Figure 9.** Oxygen flux rates (s.d) in 2012 associated with sediments collected from the original cage transect sampling positions (-10m and 150m).

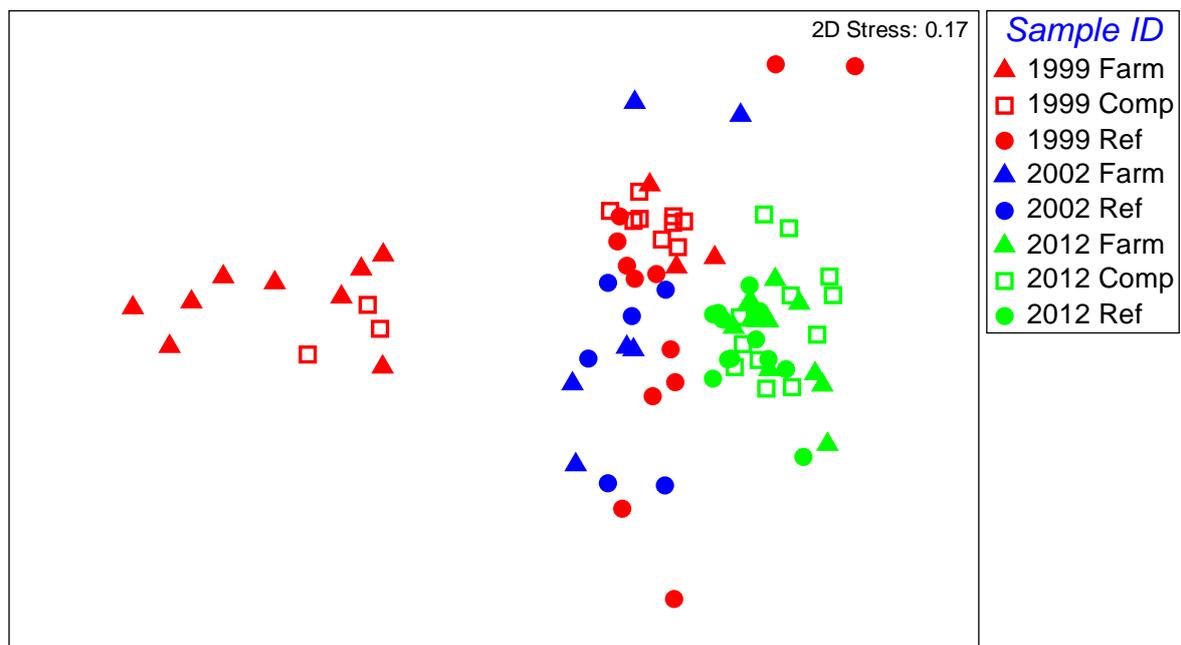
The water overlying the sediments was well oxygenated and although there was a fairly rapid drop off in oxygen levels within the first few millimetres of the sediment, this is as might be expected in such fine (silt-clay) sediments (Fig. 10). There is no evidence that the farm sediments differed significantly from those of the reference locations.



**Figure 10.** Oxygen penetration depths (s.d) for farm (-10m) and reference (150m) sites in 2012. Sediment interface is shown at 0mm.

### 3.6 Macrofauna

The ordination of the sites shows that there were significant differences in the community structure of the farm sites in 1999 compared to all other sites and times (Fig. 11). The ordination plot and associated ANOSIM analysis shows that in 2002 (after 3 years recovery) there had been considerable recovery at the farm positions such that for the most part the farm sites were indistinguishable from the reference, and that 10 years on the community structure and variability at the farm compliance and reference sites was indistinguishable but that these communities were as a whole different to those sampled on previous years, such that there appears to have been a general temporal shift.



**Figure 11.** MDS ordination plot (Stress=0.17) showing the relative positions of farm, compliance and reference sites based on the benthic community composition of all replicate samples collected in 1999, 2002 and 2012.

Examining the nature of these changes overall shows that there were significant differences between years (Global R = 0.478,  $p < 0.05$ ) as well as between the different categories of site (Global R = 0.234,  $P < 0.05$ ), but the differences between years were greater (Global R values). The communities at the farm sites at the end of production (1999) were dominated by opportunistic species indicative of organic enrichment (i.e. *Capitella capitata*, *Malacoceros tripartitus* and *Nebalia longicornis*), which together accounted for more than 86% of the overall similarity of communities at these sites and were quite different to those at reference locations, where the community was characterised by the brittle star *Amphiura elandiformis*, previously established as a useful indicator of clean sediment conditions and a number of other filter and surface deposit feeding polychaetes, bivalves and crustaceans more commonly associated with uncontaminated environmental conditions such as (Table 2). In contrast there was a lot of overlap in the communities at the farm, compliance and

reference locations in 2012, with differences for the most part being associated with changes in abundances of these key species rather than in species replacements. The species mix also suggests relatively good ecological conditions, with no obvious dominants and a broad range of functional types represented.

**Table 2.** SIMPER output for the full community assessment indicating the average within group similarity and the average abundance per group, variability (st.dev. ratio), individual % contribution to the group similarity and cumulative % similarity of the key species in each of the main groups over time.

*Group 1999 Farm*

Average similarity: 31.73

Species	Av.Abund	Sim/SD	Contrib%	Cum.%
Capitella sp.(MoV 2558)	38.14	0.92	51.21	51.21
Malacoceros tripartitus	21.73	0.70	25.82	77.03
Nebalia longicornis	7.90	0.51	9.64	86.67
Corbula gibba	4.23	0.27	3.28	89.95
Nassarius spp	3.43	0.39	2.70	92.65

*Group 1999 Comp*

Average similarity: 37.37

Species	Av.Abund	Sim/SD	Contrib%	Cum.%
Corbula gibba	12.00	1.34	19.06	19.06
Lysilla jennacubinae	9.22	1.03	13.84	32.90
Euphilomedes spp.	8.59	1.19	13.42	46.33
Ostracod sp.	8.69	1.03	10.62	56.95
Theora lubrica	7.71	0.99	10.00	66.95
Capitella sp.(MoV 2558)	9.17	0.22	3.81	70.76
Nephtys spp.	2.61	0.66	2.83	73.59
Nassarius spp	2.71	0.66	2.82	76.41

*Group 1999 Ref*

Average similarity: 28.11

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Lysilla jennacubinae	7.50	5.28	0.80	18.79	18.79
Amphiura elandiformis	7.14	4.57	0.96	16.26	35.05
Theora lubrica	7.96	4.25	0.56	15.13	50.18
Corbula gibba	7.78	3.81	0.82	13.54	63.73
Nucula pusilla	7.43	3.64	0.60	12.95	76.67
Lumbrinereidae spp.	4.49	1.72	0.64	6.11	82.78
Euphilomedes spp.	6.06	1.27	0.41	4.53	87.31
Nassarius spp	3.39	0.97	0.42	3.47	90.77

*Group 2012 Farm*

Average similarity: 55.81

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Callianassa limosa	7.52	11.80	7.47	21.14	21.14
Theora lubrica	7.52	11.80	7.47	21.14	42.27
Phoronid sp.	5.64	6.50	1.06	11.65	53.93
Euchone limnicola	5.64	6.35	1.07	11.38	65.31
Lysilla jennacubinae	5.01	4.77	0.84	8.55	73.86
Corbula gibba	5.01	4.64	0.84	8.32	82.17
Nassarius spp	4.38	3.37	0.68	6.04	88.21
Melitid sp.	3.39	1.76	0.42	3.15	91.36

*Group 2012 Comp*

Average similarity: 44.09

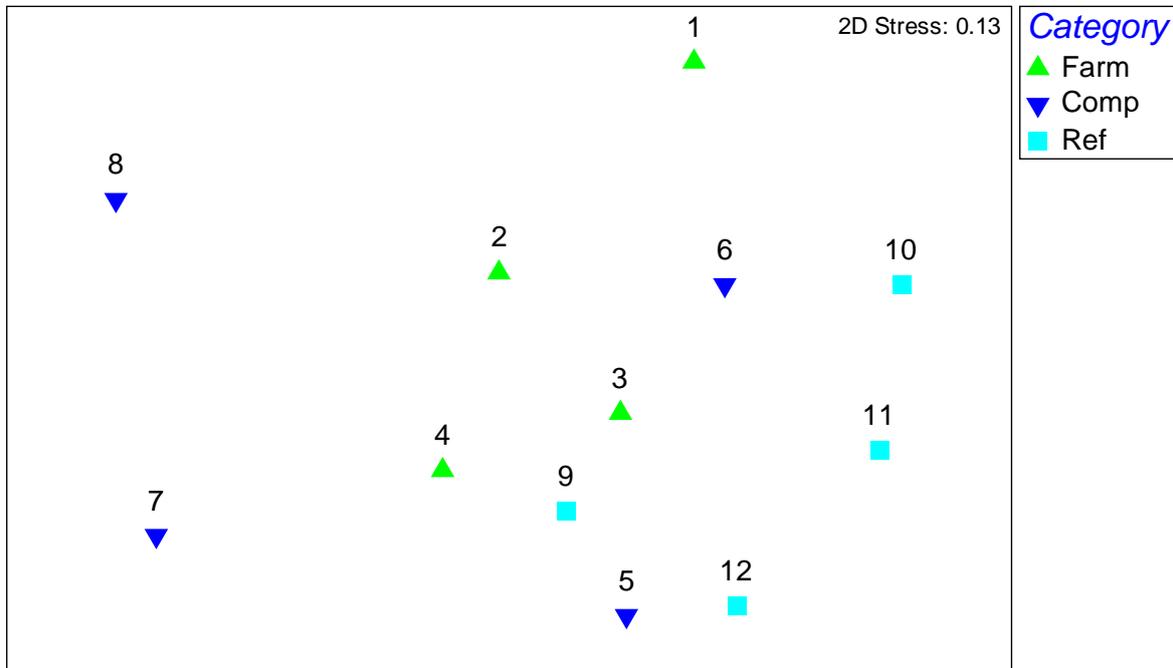
Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Theora lubrica	6.89	7.90	1.98	17.91	17.91
Euchone limnicola	6.26	6.12	1.39	13.87	31.78
Callianassa limosa	5.64	5.70	1.06	12.92	44.69
Nassarius spp	5.01	4.02	0.81	9.11	53.81
Corbula gibba	4.38	3.71	0.66	8.41	62.21
Phoronid sp.	4.38	3.43	0.64	7.77	69.98
Nemertean sp.	4.38	3.09	0.64	7.01	77.00
Lysilla jennacubinae	4.38	2.84	0.66	6.44	83.43
Kalliapseudes sp.	3.13	1.15	0.41	2.61	86.04
Mediomastus australiensis	3.13	1.13	0.42	2.57	88.61
Oedicerotid spp.	2.51	0.77	0.31	1.76	90.37

*Group 2012 Ref*

Average similarity: 57.52

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Callianassa limosa	7.52	11.50	7.56	19.99	19.99
Theora lubrica	7.52	11.50	7.56	19.99	39.99
Corbula gibba	6.26	7.98	1.41	13.87	53.86
Nassarius spp	6.26	7.67	1.42	13.33	67.19
Amphiura elandiformis	5.01	4.91	0.84	8.53	75.71
Euchone limnicola	4.38	3.47	0.67	6.03	81.74
Lysilla jennacubinae	3.76	2.66	0.53	4.63	86.37
Phoronid sp.	3.76	2.49	0.54	4.34	90.71

Comparison of the community composition data for each of the locations sampled in 2012 shows a high degree of overlap between the sites, such that there was no significant difference between the farm, compliance and reference communities (Global R = 0.113, p=0.17) (Fig.12).



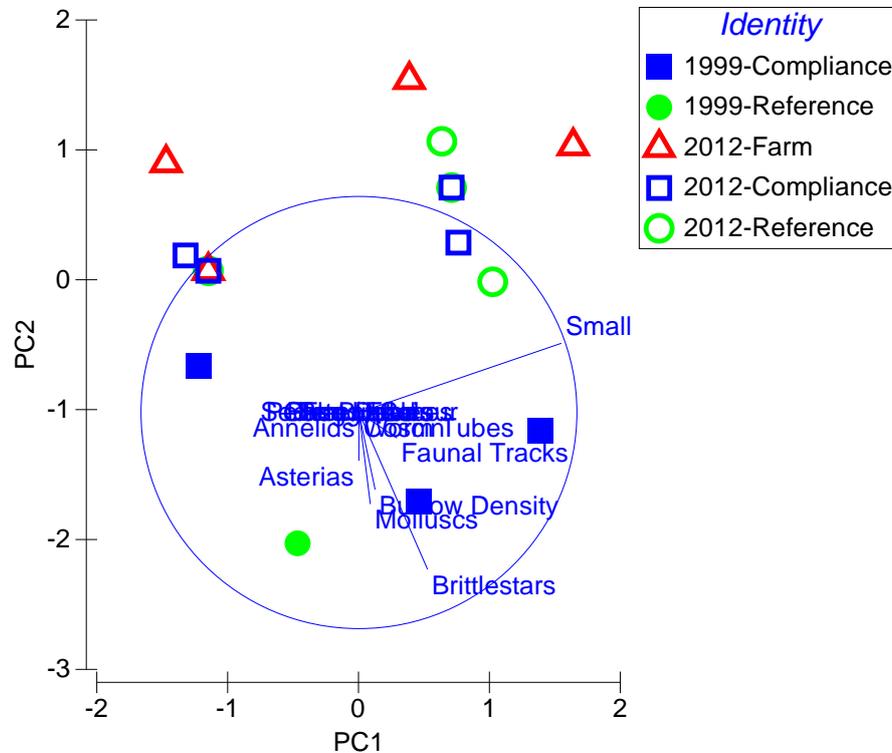
**Figure 12.** MDS ordination plot (Stress=0.13) showing the relative positions of farm, compliance and reference sites based on the benthic community composition at each of the locations sampled in 2012.

### 3.7 Video Assessment

The principal components analysis shown in Figure 13 shows the separation of the sites based on the video scores. The first two axes account for approximately 74% of the variation in the dataset with 41% of the variation associated with PC1. The key features separating the stations along PC1 are an increased presence of both Brittlestars and Small Fish. Differentiation of sites along PC2 is not as well defined and generally results from a reduction in Brittlestars along with declines in abundance of Molluscs and Burrow Density. Whilst there does seem to be some evidence that the video footage in 1999 might differ from that in 2012, this is at a very low level and may simply be a data artefact (i.e. a result of improved video capture techniques and clarity of images in 2012), consequently it may be wise not to read too much into this. In addition it is important to note that whilst the presence of brittlestars in a video survey will tend to infer good environmental conditions, the absence of brittlestars does not mean that conditions are in any way degraded.

When the 2012 data is considered in isolation there is little to suggest any major differentiation between the site categories (reference, compliance and farm) (Fig. 13). None

of the usual farm impact assessment criteria (e.g. gas bubbles, sediment colour, *Beggiatoa*, pellets/ faeces) suggested any significant impacts at any of the positions sampled in 2012.



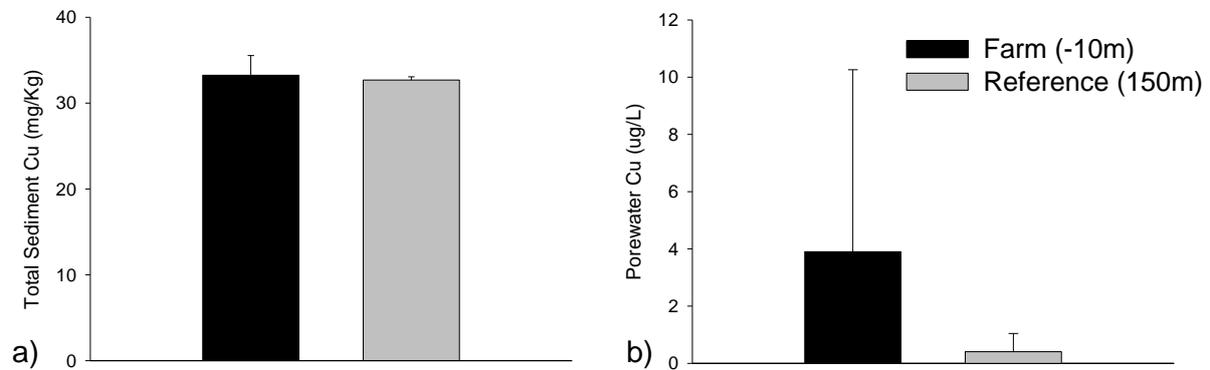
**Figure 13.** Principal components analysis (PCA) showing the relative positions of farm, compliance and reference sites based on the video assessment at each of the locations sampled in 1999 and 2012.

### 3.8 Copper and Zinc loadings

As copper based antifoulants were used in the latter years of production at the Gunpowder site in the follow-up study we assessed total sediment load and porewater concentrations of both copper and zinc.

Total loadings for surface sediments were well below the ANZECC guidelines for sediment copper, with the highest level recorded being 43 mg/kg (i.e. low level guideline of 65mg/kg), and there was no significant difference between levels at the farm and reference locations ( $p=0.11$ ,  $df_{1,5}$   $F=3.52$ ) (Fig.14). Sediment levels were within the range of expectation, being consistent with concentrations reported for reference sites through ongoing salmon farm monitoring (Macleod and Eriksen, 2009) and in a similar range to levels previously reported from SE Tasmania (HEST 2000, Jones et al., 2003, Butler, 2005, Macleod and Helidoniotis, 2005). Porewater copper levels at the reference location were effectively below the detection level of the analysis (<1 $\mu$ g/L) and although the farm levels may appear a little higher the variability was such that these levels were not significantly different from the

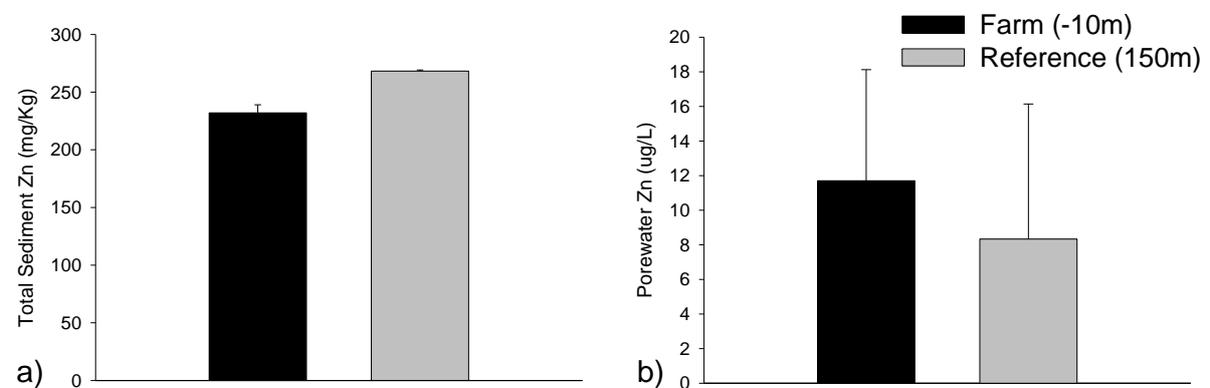
reference sites ( $p=0.10$ ,  $df_{1,19}$   $F=2.91$ ) and were also very low and consistent with previously reported levels.



**Figure 14.** Copper levels (s.d) in a) sediment and b) porewater at the original cage transect sampling positions (-10m and 150m) in 2012.

Sediment zinc loadings were consistently above the ANZECC low level guideline (i.e. 200 mg/kg), with no significant difference between the farm and reference locations ( $p=0.52$ ,  $df_{1,5}$   $F=0.45$ ) (Fig.15). The overall average zinc concentration in the sediments sampled being 263 mg/kg (s.d. 16.68). However, this is consistent with a previous report from Jordan et al. (2002) which noted that both lead and zinc were elevated in NW Bay such that 22% of sites sampled in NW Bay exceeded the low level guideline for zinc. Jordan et al. suggested that the sites with higher zinc levels were generally those with finer sediments from the deeper regions of the Bay and concluded that the source was most likely associated with the local catchment.

Porewater zinc levels were low and at levels unlikely to elicit a biological effect (Stuart Simpson CSIRO-CERC, pers. comm.), and again did not differ significantly between the reference and farm sites ( $p=0.40$ ,  $df_{1,19}$   $F=0.72$ ).



**Figure 15.** Zinc levels (s.d) in a) sediment and b) porewater at the original cage transect sampling positions (-10m and 150m) in 2012.

## 4. Conclusions

- Evaluate whether sediments within and around the Gunpowder Jetty lease have recovered such that they are consistent with reference conditions.
- Determine whether there is any significant difference in community composition between any of the sampled sites
- Assess whether there is any significant difference in functional performance between the farm/ cage and reference sites

In summary it would appear from analysis of the selected environmental condition variables that in 2012 the sediments around the Gunpowder lease have largely recovered.

Sediment geochemical parameters show some small differences but these do not appear to translate to ecological differences or any major changes in sediment biogeochemistry or function. Levels of organic matter are broadly equivalent, with no evidence of any residual organic loadings. Sediments in this region are very fine (predominantly silt-clay), and the farm and reference locations were indistinguishable. The compliance locations had a slightly lower proportion of fines than observed at either the reference or farm locations, but this was quite minimal and consistent within the range of particle sizes that might be expected in this region. Previous sampling in NW Bay would suggest that sediment particle sizes are generally fine with a high proportion of silt-clay but that the composition can vary markedly with depth, with a greater proportion of coarser material in shallower regions (Macleod et al, 2008). Sulphide was still detectable in the sediments collected at the farm locations, but the levels were very low (close to the detection limit for the analysis) and do not appear to have any major impact on the biological community. In terms of abundance, number of species, diversity and overall community structure, the farm sites were largely indistinguishable from the reference or compliance sites.

There is no evidence that any major or substantive changes have occurred at the farm sites following cessation of farming, nor is there evidence that the farming activity has had any permanent impact in this area. Although there does appear to have been a very slight change in community composition over time, this change was evident at all of the sampling stations, was at a very low level and would appear more likely to be a result of natural ecological dynamics. There does not appear to be any significant functional difference in the sediments at the farm sites relative to nearby compliance or reference sites, and the nutrient flux rates are consistent with expectations, suggesting a relatively healthy biological response for this type of sediment. It is also worth noting that over the period the intervening 10 year period since the farm site has been fully vacated there have been a number of other major changes to the local catchment and potential inputs to NW Bay (i.e. significant urban development, and upgrading and relocation of sewage treatment facilities).

## 5. References

- Brooks, K.M., Stierns, A.R., Mahnken, C.V.W., Blackburn, D.B., 2003. Chemical and biological remediation of the benthos near Atlantic salmon farms. *Aquaculture* 219, 355-377
- Brooks, K.M., Stierns, A.R., Backman, C., 2004. Seven year remediation study at the Carrie Bay Atlantic Salmon ( *Salmo salar* ) farm in the Broughton Archipelago, British Columbia, Canada. *Aquaculture* 239, 81-123.)
- 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.
- Buschmann, A. H., Riquelme, V. A., Hernández-González, M. C., Varela, D., Jiménez, J. E., Henríquez, L. A., ... & Filún, L. (2006). A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific. *ICES Journal of Marine Science*, 63(7), 1338-1345.
- Butler, E.C.V. (2005) The Tail of Two Rivers in Tasmania: the Derwent and Huon Estuaries. In: P.J. Wangersky (ed.) *Handbook of Environmental Chemistry*, Vol. 5: Water Pollution, Part H. Estuaries, Springer, New York Berlin Heidelberg Tokyo.
- 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
- Christensen PB, Rysgaard S, Sloth NP, Dalsgaard T, Schwærter S (2000) Sediment mineralisation, nutrient fluxes, denitrification and dissimilatory nitrate reduction to ammonium in an estuarine fjord with sea cage trout farms. *Aqua Microb Ecol* 21: 73-84
- Clarke, K.R. and Warwick, R.M. (2001) *Change in marine communities: an approach to statistical analysis and interpretation*, 2nd edition. PRIMER-E: Plymouth.
- 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.
- 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.
- 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., 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.

- 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 Hargrave et al., 2008
- HEST 2000. Huon Estuary Study – Environmental Research for Integrated Catchment Management and Aquaculture. CSIRO Huon Estuary Study Team. Final report to Fisheries Research and Development Corporation. Project number 96/284, June 2000. CSIRO Division of Marine Research. Marine Laboratories, Hobart. 285 pp. (<http://www.marine.csiro.au/ResProj/CoastEnvMarPol/huonest/index.html>)
- 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. Holmer and Kristensen, 1994;
- Holmer M, Marba N, Terrados J, Duarte CM, Fortes MD (2002) Impacts of milkfish (*Chanos chanos*) aquaculture in carbon and nutrient fluxes in the Balinao area, Philippines. *Mar Pollut Bull* 44: 685-696
- Holmer M, Duarte CM, Heilskov A, Olesen B, Terrados J (2003) Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area, Philippines. *Mar Pollut Bull* 46: 1470-1479
- Janowicz, M., & Ross, J. (2001). Monitoring for benthic impacts in the southwest New Brunswick salmon aquaculture industry. *ICES Journal of Marine Science: Journal du Conseil*, 58(2), 453-459.
- Jones, B.G., Chenhall, B.E., Debretson, F. and Hutton, A.C. (2003) Geochemical comparisons between estuaries with non-industrialised and industrialised catchments: the Huon and Derwent River estuaries, Tasmania. *Australian Journal of Earth Sciences* 50 :653-667.
- 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 Resources Management Strategy, Hobart, 94p.

- 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 : 205-218Karakassis et al. 2000
- Karakassis, I., Tsapakis, M., Smith, C. J., & Rumohr, H. (2002). Fish farming impacts in the Mediterranean studied through sediment profiling imagery. *Marine Ecology Progress Series*, 227, 125-133.
- Keeley, N.B., Macleod, C.K., Hopkins, G., Forrest, B.M. (2013) Spatial and temporal dynamics in macrobenthos during recovery from salmon farm induced organic enrichment: when is recovery complete? *Marine Pollution Bulletin*Marine Pollution Bulletin (In Press).
- Lumb, C.M. (1989) Self-pollution by Scottish salmon farms? *Marine Pollution Bulletin*. Vol.20 (8) : 375-379;
- Macleod, C.K.A., Mitchell, I.M., Crawford, C.M. and Connell, R.D. (2002) Evaluation of sediment recovery after removal of finfish cages from marine farm lease no.76 (Gunpowder Jetty), North West Bay. TAFI Technical Report No.13. January 2002. Tasmanian Aquaculture & Fisheries Institute - University of Tasmania, Hobart, Australia.
- MacLeod, C, Bissett, AP, Burke, C, Forbes, SE, Holdsworth, D\*, Nichols, P\*, Revill, A\*, Volkman, J\* (2004a). Development of novel methods for the assessment of sediment condition and determination of management protocols for sustainable finfish cage aquaculture operations', Aquafin CRC Final Report, Tasmanian Aquaculture & Fisheries Institute, Hobart, Tasmania, Australia (2004)
- Macleod C.K., Crawford, C.M. and Moltschaniwskyj, N.M. (2004b) Assessment of long term change in sediment condition after organic enrichment: defining recovery (2004) *Marine Pollution Bulletin*. 49: 79-88.
- Macleod, C. and Helidoniotis, F. (2005). Ecological status of the Derwent and Huon Estuaries. NHT/NAP Final Report (Project No.46928), Tasmanian Aquaculture & Fisheries Institute, Hobart, Tasmania, Australia
- Macleod C., Revill, A., Volkman, J., Holdsworth, D. (2008) Characterisation of the benthic environment of the D'Entrecasteaux Channel and Huon Estuary. Technical Report Aquafin CRC Project 4.2(2)/ FRDC Project No. 2004/074.
- Macleod, C and Eriksen, R, (2009). A Review Of The Ecological Impacts Of Selected Antibiotics And Antifoulants Currently Used In The Tasmanian Salmonid Farming Industry (Marine Farming Phase), FRDC Final Report (Project No. 2007/246), Tasmanian Aquaculture & Fisheries Institute, Hobart, Tasmania, Australia
- 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.
- Weston, D.P. (1990) Quantitative examination of macrobenthic community changes along an enrichment gradient. Marine Ecology Progress Series. 61: 233-244.
- 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.