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The Environmental Effects of
Blue Mussel (*Mytilus edulis*)
Aquaculture in Port Phillip Bay

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**L.J. McKinnon, G.D. Parry, S.C. Leporati, S. Heislars, G.F. Werner,
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Contents

Abstract.....	1
Introduction	2
Impacts on the pelagic environment.....	3
Impacts on the benthic environment	4
Present study.....	5
Methods	6
Study Site.....	6
Sampling methods.....	6
Redox potential.....	6
Sediment Particle Size Analysis	6
Total Organic Carbon	6
Benthic infauna	7
Video Survey of epifauna.....	7
Results.....	8
Redox Potential.....	8
TOC and Sediment Particle Size.....	8
Benthic Infauna.....	8
Benthic Epifauna.....	9
Discussion.....	10
Acknowledgements	12
References.....	13

List of tables and figures

Table 1. ANOVA table for Redox with distance from farm and depth within the substrate.	15
Table 2. Summary of analyses of linear regressions of species diversity and species richness against distance from farm and of multiple regressions of species diversity and species richness against distance from farm, particle size (<63µm), and sediment organic content (TOC).	15
Table 3. Linear regression coefficients, significance levels and R2 values of abundance of indicator taxa with distance from the farm for species observed on video transects.	16
Table 4. Summary of results of environmental monitoring of shellfish aquaculture from selected relevant studies.	17
Table 5. Summary of analyses of linear regressions of abundance (LogN+1) against distance from farm and of multiple regressions of abundance (LogN+1) against distance from farm, particle size (<63µm), and sediment organic content (TOC) for the 10 most common benthic infaunal species.	18
Table 6. Mean and standard deviation of density (per m2) of selected species from Grassy Point video sled tows.	20
Figure 1. Locality map of Grassy Point Aquaculture Zone, indicating approximate location of sampling site.	21
Figure 2. Schematic of study site indicating relative location of transects and indicative randomly selected sampling points.	22
Figure 3. Redox potential (mV) ± SE with distance from the farm (m).	23
Figure 4. East and west plot showing a regression of % total organic carbon against distance from the farm.	23
Figure 5. East and west plot showing a regression of % sediment fines (<63µm) against distance from the farm.	24
Figure 6. % composition of particle size fraction with distance from the farm	24
Figure 7. Multi-dimensional scaling plot of benthic taxa communities from the study site.	25
Figure 8. East and west plot showing a regression of species diversity against distance from the farm.	25
Figure 9. East and west plot showing a regression of species richness against distance from the farm.	26
Figure 10. Percent mussel cover and density of 11-Arm stars with distance from the farm	27
Figure 11. (a), Distribution pattern of <i>Phoronopsis albomaculata</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance (log(n+1)) of <i>Phoronopsis albomaculata</i> against distance from the farm.	28
Figure 12. (a), Distribution pattern of <i>Lumbrineris Cf. latreilli</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance (log(n+1)) of <i>Lumbrineris Cf. latreilli</i> against distance from the farm.	29
Figure 13. (a), Distribution pattern of <i>Capitellid sp.1</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance (log(n+1)) of <i>Capitellid sp.1</i> against distance from the farm.	30
Figure 14. (a), Distribution pattern of <i>Edwardsia vivipara</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance (log(n+1)) of <i>Edwardsia vivipara</i> against distance from the farm.	31
Figure 15. (a), Distribution pattern of <i>Notomastus chrysosetus</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance (log(n+1)) of <i>Notomastus chrysosetus</i> against distance from the farm.	32

Figure 16. (a), Distribution pattern of <i>Nemertea</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance ($\log(n+1)$) of <i>Nemertea</i> against distance from the farm.	33
Figure 17. (a), Distribution pattern of <i>Ampharete sp.1</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance ($\log(n+1)$) of <i>Ampharete sp.1</i> against distance from the farm.	34
Figure 18. (a), Distribution pattern of <i>Nephtys inornata</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance ($\log(n+1)$) of <i>Nephtys inornata</i> against distance from the farm.	35
Figure 19. (a), Distribution pattern of <i>Marphysa sp.1</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance ($\log(n+1)$) of <i>Marphysa sp.1</i> against distance from the farm.	36
Figure 20. (a), Distribution pattern of <i>Artacamella dibranchiata</i> on and near the study site, the boundaries of which are shown; (b), east and west plot showing a regression of abundance ($\log(n+1)$) of <i>Artacamella dibranchiata</i> against distance from the farm.	37

Abstract

The environmental impacts on the benthos of a 3 ha mussel farm in Port Phillip Bay were examined by comparing the sediments and fauna beneath the farm with those between 5, 25 and 50 m from the farm. Sediments contained up to 60% fines (<63 μ m) and the percentage of fines decreased significantly with distance from the farm. There were no significant trends in % total organic carbon or redox potential with distance from the farm, although the lowest redox potential values were recorded beneath the farm.

Spatial differences in infaunal communities determined from MDS plots suggested that community structure varied from west to east across the study area, but did not indicate that communities beneath the farm were distinct from those further from the farm. Infaunal species diversity and richness were greatest beneath the farm, and both parameters decreased significantly with distance from the farm, but not with either sediment particle size (%<63 μ m) or sediment total organic carbon. Two infaunal species were more abundant beneath the farm than further from the farm, and one infaunal species was more abundant further from the farm. Mussels dislodged from the longlines, and eleven-armed seastars, which prey upon the mussels, were more abundant beneath the farm than beyond the farmed area.

Environmental impacts from long term mussel farming in Port Phillip Bay were small and restricted to the immediate vicinity of the farmed area. The low impact of this mussel farm compared to impacts of farms in other regions appears to result from the relatively low production of farms in Port Phillip Bay (up to 15-30t/ha/year).

Introduction

The Environment Conservation Council (ECC 2000) recently recommended the introduction of several new and expanded aquaculture zones in Victorian coastal waters. These include a 440 ha aquaculture zone near Flinders in Western Port Bay, a 200 ha zone near Portland, and ten zones within Port Phillip Bay with a total area of 2044 ha. The zones within Port Phillip Bay are located near Avalon (Land based, 17 ha), Bates Point (25 ha), Beaumaris (25 ha), Clifton Springs (315 ha), Dromana (20 ha), Grassy Point (252 ha), Kirk Point-Werribee (200 ha), Mount Martha (150 ha), Pinnacle Channel (1000 ha) and Pt Lillias (Land based, 40 ha). Management plans are being developed for each aquaculture zone, as recommended by ECC.

This report was commissioned by Fisheries Victoria to assess the impact of current mussel farming activities on the marine environment and to assist with the preparation of the management plans for aquaculture fisheries reserves in Victorian coastal waters.

Bivalve aquaculture in Victoria is presently based on blue mussels (*Mytilus edulis*). The total aquaculture production of blue mussels in Victoria was almost 1600 tonnes in 2001/02, worth approximately \$3.7M (Anon. 2002a). It is anticipated that blue mussels will continue to be the main species farmed within the new and expanded aquaculture fisheries reserves in Victoria.

Bivalves can be cultured on suspended offshore rafts or longlines, intertidal racks and in-sediment culture, however the only method presently used in Victoria is subsurface longline culture. A typical Victorian mussel longline configuration consists of a 250m long rope (longline), anchored at either end with a two-tonne concrete block. The central section (100-120m) is suspended within approximately 2m of the surface by attached floats. Mussel growing ropes (4-5m in length) are hung vertically from the central section of the longline at intervals of 40-80cm. Typically there are 300 growing ropes per longline, and each growing ropes produces 15 - 30 kg of mussels over a 12-18 month production cycle. Farms generally have a total of 6-10 longlines per 3 ha licensed area, resulting in an annual production of <15-30 tonnes/ha of licence area. The mussel industry in Victoria has a year-round culture season,

with maximum product availability from July to January.

Studies have been conducted on the environmental impacts of mussel, oyster and clam aquaculture in various parts of the world (Haven & Morales-Alamo 1966; Dahlback & Gunnarsson 1981; Asmus & Asmus 1991; Mallet & Carver 1991; Hatcher *et al.* 1994; Grant *et al.* 1995; Gilbert *et al.* 1997; Kaiser *et al.* 1998; Chamberlain *et al.* 2001), however few studies on the environmental impacts of shellfish aquaculture have been undertaken in Australia.

In this report the benthic impacts of subsurface longline mussel culture in Port Phillip Bay, Victoria, including physico-chemical and biological impacts, are detailed and discussed.

The process of extensive subsurface on-growing of bivalve molluscs is the stage with the greatest potential for environmental impacts (Smaal 1991). Bivalves consume food naturally present in their environment (Kasper *et al.* 1985) and, in contrast to finfish farming, bivalve aquaculture does not require any addition of food or nutrients.

Commercial bivalve culture can lead to environmental impacts in both pelagic and benthic environments. Pelagic impacts can include the depletion of phytoplankton and/or the alteration of nitrogen cycles in the water column. In bivalve culture operations sited in areas of adequate water exchange, pelagic impacts are of less concern than benthic impacts (Hatcher *et al.* 1994; Gavine & McKinnon 2002).

Benthic impacts can include a build-up of live mussels and shells, and faeces and pseudofaeces (biodeposition). Pseudofaeces includes sediment, diatoms and other matter filtered from the water by the gills and entrained in mucus, but expelled from the mussel before it enters its mouth. Mussel shells can provide sites of attachment for large epibiota such as tunicates, sponges and tube worms, as well as providing an increased food supply for epibenthic and pelagic predators. The accumulation of faeces and pseudofaeces may affect benthic diversity and increase the representation of species that consume detritus. Furthermore, changes in the benthic fauna and changes in nutrient inputs may disrupt nitrogen

cycles in the sediment. Where biodeposition is excessive, sediments may become anoxic (Kaiser *et al.* 1998). Surveys of the benthic habitat of potential bivalve culture sites are now common practice to avoid impacts on seagrass, macroalgae and other significant habitats.

Impacts on the pelagic environment

Mussels filter and ingest particulate organic matter (POM) such as phytoplankton, detritus, microzooplankton and bacteria from the water column (Gosling 1992), while faeces, pseudofaeces, ammonium, orthophosphate and silicate are released (Dame *et al.* 1991). Filtration rates and growth of mussels depend on stocking density and the availability of POM (Côté *et al.* 1993). Mussel beds can reduce phytoplankton biomass in shallow environments through filtration. Conversely, mussel beds may also significantly promote primary production through the release of nutrients by the mussels (Asmus & Asmus 1991). Ammonium released by mussels is immediately available for localised phytoplankton production in surface waters (Kasper *et al.* 1985). However, large-scale commercially intensive bivalve aquaculture may lead to depletion of nutrients, particularly nitrogen which plays a crucial role in the productivity of a coastal ecosystem and may lead to localised food-limitation for bivalve production (Kasper *et al.* 1985).

The balance between net nutrients extracted and net nutrients returned to the water column is difficult to quantify, but mass balance equations and nutrient flow models have been used to determine the overall effects of bivalve aquaculture on nutrient loads (Rodhouse *et al.* 1985; Gavine & McKinnon 2002). A large percentage (42 % N and 58% C) of nutrients filtered from the water column by mussels are excreted directly into the water column through the protonephridium and the gills (Rodhouse *et al.* 1985). The remaining nutrients are either harvested (removed completely from the system), consumed by scavengers/decomposers, or released as faeces. A decline in phytoplankton biomass due to bivalve consumption can reduce nutrient demand, especially for ammonia, which can in turn promote higher nitrate concentrations within bivalve culture zones through increased bacterial nitrification. Therefore, at high density, bivalve culture can alter the dominant

biogeochemical process in the water column by stimulating nitrification (Souchu *et al.* 2001).

Although the impacts of shellfish farming on cycling of nitrogen may be considerable, a range of factors exist which may influence nitrogen cycling in any one body of water. These include farm-specific factors (scale of operation, production density etc.), natural processes (natural flushing, tidal exchange etc.) and external sources of N (natural, domestic and agri-industrial sources of N). In Port Phillip Bay nitrogen inputs vary widely within and between years (CSIRO 1996). Such high degree of variability in nitrogen renders it intrinsically difficult to measure in the context of environmental effects of mussel farming.

Pelagic environmental impacts of bivalve culture are greatly influenced by the intensity of the aquaculture operation and physical characteristics of the waterway, particularly tides, depth, and currents. (De Casabianca *et al.* 1997) suggested that bivalve aquaculture was the major cause of eutrophication in Thau Lagoon, France, a 75 km² lagoon that produces 35,000 tonnes of mussels and oysters annually. Twenty percent of Thau Lagoon is occupied by aquaculture, the lagoon has a mean depth of 4.5 m and is characterised by a lack of tides (Souchu *et al.* 2001). In Thau Lagoon, the high-density of cultivation and the limited exchange of water between farmed and non-farmed areas (Mazouni *et al.* 1996) predisposes this region to problems of eutrophication. It is noted that only 1.05% of a much larger area (1950 km²) is available for bivalve culture in Port Phillip Bay and water exchange between farmed and unfarmed areas is facilitated by a greater tidal influence and the bay's greater depth (mean 13 m).

High-density bivalve aquaculture production may also cause an increase in competition between bivalves and zooplankton for food resources (Rodhouse & Roden 1987). Tenore *et al.* (1982) found that high-density mussel culture area in Spain caused cultured mussels to out-compete zooplankton and other filter feeders.

In summary, bivalve aquaculture may impact the pelagic environment through the alteration of nutrient cycling, but such impacts may only be expected to occur where high density bivalve aquaculture occurs.

Impacts on the benthic environment

The primary benthic environmental impact of suspended bivalve aquaculture is the build up of faeces and pseudofaeces on the benthic environment directly below the culture area (Anon. 2002b). The impact of biodeposits from suspended mussel culture on surrounding sediments depends on stocking density, farm productivity, amount of food available to mussels, and the age of the farm. In addition, local current patterns, height of growing lines above the seabed, orientation of farm lines, water depth and settling velocities of biodeposits all influence the area impacted by biodeposits (Jaramillo *et al.* 1992; Chamberlain *et al.* 2001). A hazard assessment undertaken by Gavine & McKinnon (2002) identified that the most important environmental issues related to bivalve culture in Victorian marine waters were impacts on sediment quality.

Bivalve aquaculture redirects suspended organic matter to the benthos and may convert nutrients into larger particles in the form of faeces and pseudofaeces (biodeposition), which can lead to localised sediment enrichment (Grant *et al.* 1995). Faeces and pseudofaeces are generally composed of chloropigments, organic carbon, organic nitrogen and biogenic silica. During blooms, diatoms may become entrained in pseudofaeces and even where the production of pseudofaeces is lower than that of faeces, undigested algae in pseudofaeces remain an important source of energy and chlorophyll *a* for other trophic levels (Navarro & Thompson 1997). High sedimentation rates caused reduced community diversity and density beneath mussel farms in Spain and Ireland (Tenore *et al.* 1982; Chamberlain *et al.* 2001), an effect which is further compounded if significant habitats such as seagrass or kelp beds lie beneath suspended culture sites (Kasper *et al.* 1985).

Grant *et al.* (1995), studied the impacts of mussel culture on benthic community composition and found that the community was dominated by molluscs attracted to the enriched organic matter (a product of biodeposition) and the mussels that had fallen from lines. Overall, the impact of the culture operation on the benthic community appeared to be minor, with little impact on species diversity evident as a result of mussel culture (Grant *et al.* 1995). Tenore *et al.* (1982) contrasted the benthic regimes of two Spanish coastal embayments: an intensive

mussel culture area (approximately 2000 mussel culture rafts), and an extensive mussel culture area (<100 mussel culture rafts). It was found that forty years of intensive mussel culture (2000 rafts) resulted in a generally low biomass and low diversity polychaete-nematode dominated assemblage in the benthic macrofauna, with decreased biomass within farmed areas. This is in comparison to less intensive mussel culture (<100 rafts) maintaining a highly diverse benthic community in the culture area (Tenore *et al.* 1982).

Microbial and meiofaunal responses to mussel culture biodeposition were studied by Mirto *et al.* (2000). Chloroplastic pigments, proteins and lipids were found to accumulate in the sediments directly under the culture areas. Microbial assemblages increased in density and meiofaunal densities also changed beneath mussel farms. Densities of turbellarians, ostracods and kinorhynchs decreased, while the abundance of copepods remained constant or increased (Mirto *et al.* 2000).

Hatcher *et al.* (1994) found that the greatest response of the benthic community to suspended mussel culture was increased ammonium release throughout the year due to increased sedimentation. Suspended bivalve culture may promote anoxic sediments when the organic enrichment of the sediment surface is excessive and leads to the build up of mats of the sulphate reducing bacteria, *Beggiatoa* spp. Under these circumstances the biomass and diversity of benthic infauna is reduced (Dahlback & Gunnarsson 1981; Rosenberg & Loo 1983). Crawford *et al.* (2001) studied the effects of oyster and mussel culture on sediment condition, assessing redox, sulphide and carbon levels, but no significant differences were found between sites outside, at the boundary and inside the farmed areas. However, sediment condition was found to differ between farms due to differences in sediment composition and seagrass coverage.

In summary, benthic impacts of bivalve aquaculture are predominantly due to increased sedimentation, resulting in changes to benthic community structure, organic content of sediments and decreased oxidative capacity of sediments. The greatest benthic impacts occur in areas of high-density bivalve aquaculture.

Present study

The existing Victorian mussel aquaculture industry is based around longline production at three major areas: Clifton Springs and Grassy Point aquaculture zones in Port Phillip Bay and Flinders aquaculture zone in Western Port (total area 954 ha). Three smaller areas have also been farmed in Port Phillip Bay: Beaumaris, Mount Martha and Dromana aquaculture zones (total area 12 ha). Each zone is generally comprised of 3ha areas. Farmed areas are separated by at least one vacant 3ha area.

The objectives of this study were to determine the benthic environmental impacts of long-term mussel farming in Port Phillip Bay, and to contribute to the development of appropriate environmental monitoring and management strategies for aquaculture areas.

Impacts of the farm were measured by comparing the physical characteristics of sediments and the fauna beneath the farm with those on transects 5 m, 25 m, and 50 m from the farm, in the direction of the prevailing currents, using benthic grab samples and video footage.

Method

Study Site

The mussel farm selected as the study site was within the Grassy Point aquaculture zone (Figure 1). The Grassy Point aquaculture zone was established in 1989 and contains 36 three hectare blocks arranged in three rows of 12 blocks each. The blocks in each row are separated by 40m, and each row of blocks is separated by about 100m. Active farms are separated by at least one vacant 3ha block. Only eighteen of the 36 areas are designated for bivalve shellfish aquaculture, and 16 of these are currently farmed. The long axis of the zone extends east-west which is the approximate direction of the water currents on ebb and flood tides. Water depth in the zone varies between 10-15m.

The mussel farm used in the study was chosen due to its high stocking density (by Victorian standards), long history of farming (at least 10 years) and its location on a corner of the aquaculture zone. The corner location ensured that any impacts that extended outside the aquaculture zone were likely to be from the farm being studied, rather than an adjacent farm.

Sampling methods

Parallel transects were located 5, 25, and 50 m from longlines at the eastern and western boundaries of the farm, and two transects were located immediately below the longlines within the licence area (Figure 2). Transects were approximately the same length as the longlines.

Five 0.1m² Smith-McIntyre sediment grab samples were taken at random along each of 8 transects which were approximately parallel to the orientation of the mussel longlines (Figure 2). Sites sampled within the farm along transects immediately adjacent to the longlines were given a distance of 0 m (distance from the farm) for the purposes of the analyses.

A 500g subsample of sediment was collected from each grab sample, stored in a "Whirlpak" and frozen for later analysis of particle size and total organic carbon (TOC).

Redox potential

Redox potential (also referred to as ORP – oxidation-reduction potential) was measured immediately after sample collection by removing a cover on the Smith-McIntyre grab and inserting

a probe into undisturbed surface sediments. Redox potential was measured using a Cyberscan 100 pH/Redox meter, at the surface and at depths of 1 cm and 4 cm within the sediment sample.

Two-way ANOVA was used to test for differences in redox potential between measurements taken at different depths (0, 1, 4 cm) and on transects within and at various distances (0, 5, 25, 50 m) from the mussel longlines.

Sediment Particle Size Analysis

Each subsample of sediment was dry-sieved through a sieve stack of 3275, 1000, 850, 600, 500, 250, 125 µm sieves, either by hand or using a sieve shaker. The size fraction that passed through the 125 µm was then wet sieved through a 63µm sieve. The material remaining on each sieve, and material that passed through the 63µm sieve were carefully removed and dried at <65°C then weighed to 0.001g.

One-way ANOVA was used to test for differences in the proportion of each sediment size fraction between samples taken at different distances (0, 5, 25, 50 m) from the mussel longlines. Spatial variation in the <63µm fraction was analysed in more detail as this was the largest sediment fraction at the study site. A linear regression of the proportion of <63µm fraction in each sample was calculated against distance (0, 5, 25, 50 m) from the mussel longlines for each transect.

Total Organic Carbon

Total organic carbon (TOC) was estimated for each sediment sample using the following methods:

Dried, homogenised sediment (typically about 10 mg) was weighed onto 25 mm fibre glass filters (eg Gelman type AE or Whatman GF/C), acidified with 16% hydrochloric acid to convert carbonates to CO₂ and combusted at 500°C for 4 hours. Following the acidification step, filters were rinsed 2 - 3 times with about 2 ml of carbon-free water. Filters were placed in the solids analysis unit of the Astro 2001 Carbon Analyser, where the organic carbon material was combusted at 850°C to carbon dioxide in a stream of oxygen with a copper monoxide catalyst. Detection was by infra-red analyser.

Linear regressions of the percentage total organic carbon were calculated against each sediment grain size fraction (3275, 1000, 500, 250, 63, <63 μm) and distance (0, 5, 25, 50 m) of the sampled transects from the mussel longlines.

Benthic infauna

Each grab sample was washed through a 1.0 mm sieve using seawater to separate infauna. The retained fraction was preserved in buffered formalin. Preserved samples were sorted at least to Family level, and the most abundant ten taxa identified to species.

Differences in community structure at different distances from the mussel longlines were examined using multidimensional scaling (MDS) and measures of species diversity, including species richness and Shannon-Weiner species diversity.

Differences between sites sampled on transects within and adjacent to the mussel farm were examined using Bray-Curtis (B-C) dissimilarity measures (Bray & Curtis 1957). This measure was chosen because it is not affected by joint absences, it gives more weighting to abundant than rare species, and it has consistently performed well in preserving 'ecological distance' in a variety of simulations on different types of data (Faith *et al.* 1987). Double square root ($=N^{1/4}$) transformations were applied to all data before calculating B-C dissimilarity measures. These transformations were made to prevent abundant species from influencing the B-C dissimilarity measures excessively (Clarke & Green 1988; Clarke 1993).

Ordinations of triangular matrices of dissimilarities were obtained using non-metric MDS options in the PRIMER program package (Clarke & Gorley 2001). The configuration presented was the best solution (ie. exhibited the lowest 'stress' values, or least distortion) from 100 random starts.

Linear regressions of species richness and Shannon-Weiner species diversity were calculated against distance from the nearest mussel longline (0, 5, 25, 50 m). Multiple regression of species richness and Shannon-Weiner species diversity were also calculated against sediment particle size (<63 μm), percentage organic content and distance from the nearest mussel longline (0, 5, 25, 50 m).

For each of the ten most abundant species, linear regressions of $\log(\text{abundance}+1)$ were calculated against distance from the nearest mussel longline (0, 5, 25, 50 m). Multiple linear regressions were

also calculated of $\log(\text{abundance}+1)$ against sediment particle size (<63 μm), % organic content and distance from the nearest mussel longline (0, 5, 25, 50 m).

Video Survey of epifauna

A sled-mounted video camera was towed at a velocity of 1-2 kts along each of the 8 transects (Figure 2). Transects were located 5, 25, and 50 m from longlines at the eastern and western boundaries of the farm, and two transects were located within the licence area (Figure 2). The exact location of the camera in relation to the towing vessel was determined using the position of the vessel, obtained using GPS, and known depth, length of tow rope and direction of tow. Video footage of each transect was divided into three sections of approximately equal tape duration. Within each section, the density of each of eight indicator species was estimated by counting individuals over a known area, determined by the width of field on the video tape and the distance of each section. The width of field was calibrated using a wire mesh grid of known dimensions, and attached to the sled within the field of view of the camera. The indicator species chosen were: mussels (*Mytilus edulis*), eleven-arm seastars (*Coscinasterias calamari*), scallop (*Pecten fumata*), flathead (*Platycephalus bassensis*), little rock whiting (*Neoodax balteatus*), mollusc eggcase (unidentified species), spider crab (*Leptomithrax gaimardii*) and biscuit star (*Tosia magnifica*). Percentage mussel cover was estimated for each transect by analysing the percentage cover of mussels in each of three randomly selected areas within each section of tape.

Linear regressions of the % cover of mussels and the number of each indicator species/tape section were calculated against distance (0, 5, 25, 50 m) from the nearest mussel longline.

Results

Redox Potential

Redox potential varied significantly between transects (Table 1) but no clear trend was apparent (Figure 3). Conversely, within any one transect, oxidative conditions decreased with depth (Table 1, Figure 3). At all depths the lowest redox values were observed beneath the mussel farm. But the next lowest redox values were obtained 50 m from the farm, where sediments were the most similar to those on the farm. The higher redox values obtained 5 and 25 m from the farm may be artefacts of the coarser sediments on these transects. During field measurements redox values took some time to stabilise, and in sandier sediments values increased progressively for 1-3 minutes before readings stabilised. In these coarser sediments water drained readily from the jaws of the Smith McIntyre grab, and readings may have been elevated by more oxygenated water being drawn lower in the sediment as water drained out the bottom.

TOC and Sediment Particle Size

TOC varied between 0.5-1.2% throughout the sampling area, but showed no significant change with distance from the farm (Figure 4). TOC was negatively correlated with the coarser size fractions, 3275, 1000, 500 μm ($p < 0.05$, $p < 0.0001$ and $p < 0.005$ respectively, see Appendices). TOC was positively correlated with the $< 63 \mu\text{m}$ size fraction, but this correlation was not significant ($p = 0.09$).

Overall, the percent composition of fines ($< 63 \mu\text{m}$) decreased with distance from the farm (Figure 5), however the percentage of fines from some samples was greater 5 and 25 m from the farm than on the farm itself. The largest size fraction in most sediments was fines ($< 63 \mu\text{m}$) and sediments contained up to 60% fines (mean 25-35%, Figure 6). The proportion of most coarse fractions ($> 125 \mu\text{m}$) was higher on transects further from the farm, and was highest 50 m from the farm (Figure 6). In particular, the 1000, 850, 600, 500 and 250 μm size fractions were highest 50 m from the farm and most of these sediment fractions showed a progressive increase with distance from the farm (Figure 6).

Benthic Infauna

Spatial differences in infaunal communities determined from MDS plots showed communities beneath the farm were not distinct from transects near the farm (Figure 7). There was no clear change in communities with distance from the farm, but communities on the west and east of the farm appeared to differ, suggesting there may be a trend in community structure from west to east across the study area (Figure 7). The stress value of 0.25 is high and indicates that patterns are not clear and "plots could be dangerous to interpret" (Clarke 1993).

The greatest species diversity was found on the farm (Figure 8) and linear regressions indicate infaunal species diversity and species richness both decrease significantly with distance from the nearest mussel longline (Figure 8, Figure 9, Table 2). Multiple regressions of infaunal species richness and species diversity with distance from the farm, particle size ($< 63 \mu\text{m}$) and total organic carbon, indicate that species richness and species diversity were significantly correlated with distance from the farm, but not with either sediment particle size ($< 63 \mu\text{m}$) or sediment organic content (Table 2).

A total of 104 taxa were identified from benthic infaunal samples taken from the study site. Trends in the abundance of the 10 most common taxa with distance from the study site are summarised in the Appendices. Linear regressions of the abundance ($\log N + 1$) of the ten most common taxa with distance from the nearest longline, and multiple regressions of the abundance ($\log N + 1$) of the ten most common taxa with distance from the farm, particle size ($< 63 \mu\text{m}$) and total organic carbon, are summarised in the appendices. No correlation with percentage of sediment fines was observed for any of the 10 most common taxa recorded, and only the abundance of the polychaete *Lumbrineris cf. latreilli* showed a significant decrease in abundance with %TOC ($p = 0.05$). *Lumbrineris cf. latreilli* was significantly more abundant further from the farm ($p = 0.02$), whereas *Notomastus chrysosetus* and *Nemerteanus* were more abundant on and near the farm ($p = 0.04$ and $p = 0.02$ respectively).

Benthic Epifauna

Mussels and eleven armed seastars showed a significant increase in density near the farm (Table 3, Figure 10). Mussels were observed up to 25m west of the study site, and up to 5m east of the study site. No significant regression was observed between abundance of epifauna and distance of the video transect from the farm (Table 3) for the remaining six of the eight taxa recorded.

Scallops, little rock whiting and mollusc eggcases were observed on each video transect and were common throughout the sampling area. Sand flathead and spider crabs were observed on five, and biscuit stars observed on three of the nine video transects (see Appendices).

No *Beggiatoa* mats were observed, nor was any outgassing observed from the sediment.

Discussion

Differences between physical characteristics and biological communities beneath the mussel farm and nearby were small. Sediment particle size changed with distance from the farm, with a higher percentage of fine sediments on and near the farm, possibly due to the accumulation of pseudofaeces beneath the suspended mussel longlines. However, if faeces and pseudofaeces accumulated beneath the farm they did not significantly affect the organic content of sediment, which was consistently low (0.5-1.2%), and was not significantly elevated near the farm.

Redox potential is a standard method for categorising the intensity of reducing conditions in marine sediments, and low values may indicate more anaerobic conditions caused by farming practices (Gavine & McKinnon 2002). Redox potential decreased similarly with sediment depth on and near the farm, but differences between sites on and more distant from the farm are probably associated with differences in grain size, which change sediment porosity and the diffusion rate of oxygenated water, rather than to their proximity to the mussel farm. Mats of the sulphur reducing bacteria *Beggiatoa*, indicative of high organic loads and strongly anaerobic sediments, were never observed on the video transects.

Some measurements of redox potential recorded in this study may be unreliable due to drainage of water from the grab during measurements, and small differences in methods between studies mean that direct comparisons of redox values must be made cautiously. However, in this study redox potential in the sediment varied from -105.6 to 143.9mV at the sediment surface/water interface, -141.0 to 83.5mV at 1cm depth, and -169.0 to -52.0mV at 4cm depth, from sites within the farm to sites 50m from the farm. Redox potential near mussel farms in Ireland decreased from around 0mV at the sediment/water interface, to -100mV at 1cm depth and greater, but at sites 40m from farmed areas redox potential remained positive at all substrate depths measured (Chamberlain *et al.* 2001). The results of Chamberlain *et al.* (2001) indicate an impact of mussel farming on sediment organic loads occurring in the vicinity of the mussel farms. Positive redox measurements (200mV-400mV) were recorded from the substrate/water interface to 4.0cm substrate depth on Tasmanian farms (Crawford *et al.* 2001). In Port Phillip Bay, oxygen is generally consumed within the top 3mm of the

sediment (Burke 1995). The usefulness of redox potential in determining the effects of aquaculture on the oxidative capacity of the sediment below the surface may be limited in Port Phillip Bay due to the naturally reducing conditions present in the sediment.

The MDS plot indicates that differences between infaunal communities over the area sampled were small, with an east-west trend in communities being more likely than differences between communities beneath the farm and those nearby. The high stress value for the MDS (0.25) indicates that communities sampled on and near the farm are not particularly heterogeneous, and differences need to be interpreted cautiously (Clarke 1993).

The highest infaunal species diversity was found beneath the farm, and both species diversity and species richness decreased significantly with distance from the farm.

Of the ten most common taxa recorded in this study, the abundance of three taxa (*Lumbrineris Cf. latreilli*, *Notomastus chrysosetus* and *Nemertea*) decreased significantly with distance from the farm, and the abundance of most of the remainder also decreased with distance from the farm, although not significantly. The higher diversity found beneath the mussel farm in this study contrasts markedly with results of many similar studies, where species diversity and species abundance decreased beneath shellfish farms in many parts of the world (Tenore *et al.* 1982; Mirto *et al.* 2000; Chamberlain *et al.* 2001).

The only conspicuous change in epibenthic fauna associated with the mussel farm were aggregations of mussels and eleven armed seastars directly beneath the mussel longlines on the farm. Both species were only found at low densities between longlines and mussel/seastar aggregations extended only approximately 10m from the edge of the farm. It appears that mussels detached from growing ropes attracted seastars to the farm. No self-sustaining mussel beds were observed beneath the longlines, apparently due to predation by the seastars. Similar observations have been made in shellfish farms elsewhere where detached mussels beneath longlines attract predators, scavengers or other molluscs (Tenore *et al.* 1982; Kasper *et al.* 1985; Grant *et al.* 1995; Crawford *et al.* 2001).

The changes observed in this study included a small increase in fine sediments, a small increase in infaunal richness and diversity, and an increase in the abundance of 11 armed seastars consuming mussels beneath the farm. All these changes are localised and relatively minor, and are probably readily reversible by natural (or other acceptable) processes following cessation of farming in an area, but this can only be confirmed by additional experimental studies.

Changes measured beneath the Victorian mussel farm were much smaller than those observed in most other studies (Table 4). The relatively small impacts found in this study may be the result of the relatively low production of Victorian and Tasmanian mussel farms. For example, in Ireland where infaunal diversity is markedly reduced beneath mussel farms, production is approximately 200-300 tonnes/ha/year (calculated from data in (Chamberlain *et al.* 2001), many times higher than 15-30 tonnes/ha/year found at the present study site. Shellfish production in Tasmania is similar to that in Victoria and may

reflect lower nutrient levels in Australian waters. Biodeposition of faeces and pseudofaeces beneath farms elsewhere has been found to increase with shellfish productivity, and may result in increased sediment organic content, more strongly anaerobic sediments, *Beggiatoa* mats, and a reduced infaunal diversity. None of these were observed in this study.

The observed environmental impacts caused by the mussel farm in this study need to be considered when designing monitoring programs for future farms. Some monitoring of conditions beneath farms (eg. video monitoring for *Beggiatoa* mats or outgassing from sediments, or direct measurements of sediment redox potential) may be adequate. But if changes in benthic infauna were to be used for monitoring they would be best undertaken on the highest production farms as the small changes anticipated will not be detected with low levels of benthic sampling across a large and variable aquaculture zone.

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Table 1. ANOVA table for Redox with distance from farm and depth within the substrate.

Source	DF	SS	MS	F	p-value
Distance	3	29806.17	9935.39	3.42	0.02
Depth	2	388995.09	194497.54	67.06	<0.0001
Interaction	6	19696.72	3282.79	1.15	0.35
Error	66	191438.51	2900.58		
Total	71	610239.76			

Table 2. Summary of analyses of linear regressions of species diversity and species richness against distance from farm and of multiple regressions of species diversity and species richness against distance from farm, particle size (<63µm), and sediment organic content (TOC). Significant (P<0.05) regression coefficients or partial regression coefficients are shown in bold.

Dependent Variable	Independent variable (s)	Reg coeff	Std Err	p
Species diversity (h)	Distance	-0.0078	0.0032	0.02
	Distance	-0.0069	0.0033	0.05
	Particle size	0.0079	0.0071	0.27
	TOC	0.5941	0.3948	0.14
Species richness (n)	Distance	-0.1942	0.0583	0.002
	Distance	-0.1634	0.0632	0.01
	Particle size	0.0632	0.1138	0.17
	TOC	0.4437	7.4678	0.95

Table 3. Linear regression coefficients, significance levels and R² values of abundance of indicator taxa with distance from the farm for species observed on video transects.

Species	b	P	R ²
Mussels (<i>Mytilus edulis</i>)	-1.26	0.002	0.77
11 Arm seastar (<i>Coscinasterias calamari</i>)	-0.08	0.007	0.72
Scallop (<i>Pecten fumata</i>)	-0.03	ns	0.55
Sand flathead (<i>Platycephalus bassensis</i>)	-0.002	ns	0.22
Little rock whiting (<i>Neoodax balteatus</i>)	-0.008	ns	0.07
Mollusc eggcase	-0.002	ns	0.15
Spider crab (<i>Leptomithrax gaimardi</i>)	-0.003	ns	0.33
Biscuit star (<i>Tosia magnifica</i>)	-0.001	ns	0.26

Table 4. Summary of results of environmental monitoring of shellfish aquaculture from selected relevant studies.

Source	Estimated Production Density (Country)	Reported Environmental Impacts		
		Benthic infauna / epifauna	Organic Loading	Redox
(Dahlback & Gunnarsson 1981)	>200 t/ha/yr (Sweden)		Increase in organic material beneath culture	
(Tenore <i>et al.</i> 1982)	~1500 t/ha/yr (Spain)	Lower macroinfaunal biomass and species diversity. Increased biomass beneath mussel rafts	High organic content of bottom sediments. Much of organic load utilised by mussel raft epifauna	
(Grant <i>et al.</i> 1995)	4x10 ⁶ mussels/ha (Canada)	Increased biomass beneath mussel culture due to fallen mussels		
(Mirto <i>et al.</i> 2000)	400t/yr (Italy)	Significant changes in meiofaunal density	Some organic carbon enrichment of sediments beneath mussel farm	Oxygen penetration into sediment reduced beneath mussel farm
(Chamberlain <i>et al.</i> 2001)	200-350 t/ha/yr (Ireland)	Reduced macrobenthic infauna density at higher production density	Elevated organic carbon at higher production density	Negative redox values near high production farm. Positive redox values away from high production farm
(Crawford <i>et al.</i> 2001)	7-24.5 t/ha (Tasmania, Australia)	No major changes to benthic infauna	No significant trends in organic carbon along farm transects	No negative redox measurements found beneath farms

Table 5. Summary of analyses of linear regressions of abundance (LogN+1) against distance from farm and of multiple regressions of abundance (LogN+1) against distance from farm, particle size (%<63µm), and sediment organic content (TOC) for the 10 most common benthic infaunal species. Significant (P<0.05) regression coefficients or partial regression coefficients are shown in bold.

Species	Rank	Independent variable(s)	Reg coeff	Std Err	p
<i>Phoronopsis albomaculata</i>	1	Distance	-0.0055	0.0056	0.34
		Distance	-0.0024	0.0045	0.59
		Particle size	-0.0177	0.0098	0.08
		TOC	0.0915	0.5493	0.87
<i>Lumbrineris Cf. latreilli</i>	2	Distance	0.0049	0.0033	0.15
		Distance	0.0069	0.0034	0.05
		Particle size	0.0081	0.0072	0.27
		TOC	-0.9824	0.4014	0.02
<i>Capitellid sp.1</i>	3	Distance	-0.0028	0.0044	0.53
		Distance	0.0007	0.0037	0.85
		Particle size	0.0125	0.0081	0.13
		TOC	-0.4679	0.4344	0.29
<i>Edwardsia vivipara</i>	4	Distance	0.0005	0.0049	0.93
		Distance	0.0044	0.0045	0.33
		Particle size	-0.0056	0.0092	0.55
		TOC	0.6453	0.5830	0.28
<i>Notomastus chrysosetus</i>	5	Distance	-0.0053	0.0027	0.05
		Distance	-0.0045	0.0022	0.04
		Particle size	0.0021	0.0045	0.65
		TOC	0.2085	0.2557	0.42
<i>Nemertea</i>	6	Distance	-0.0059	0.0023	0.01
		Distance	-0.0055	0.0023	0.02
		Particle size	-0.0054	0.0048	0.27
		TOC	0.3209	0.2674	0.24

Species	Rank	Independent variable(s)	Distance	Particle size	TOC
<i>Ampharete sp.1</i>	7	Distance	-0.0030	0.0033	0.38
		Distance	-0.0018	0.0026	0.49
		Particle size	0.0058	0.0053	0.28
		TOC	-0.0880	0.3164	0.78
<i>Nephtys inornata</i>	8	Distance	0.0003	0.0032	0.92
		Distance	-0.0015	0.0026	0.57
		Particle size	0.000008	0.0052	0.99
		TOC	0.4980	0.3614	0.18
<i>Marphysa sp.1</i>	9	Distance	-0.0037	0.0021	0.09
		Distance	-0.0037	0.0020	0.07
		Particle size	0.0049	0.0042	0.25
		TOC	-0.0727	0.2369	0.76
<i>Artacamella dibranchiata</i>	10	Distance	0.0010	0.0038	0.79
		Distance	-0.0006	0.0036	0.87
		Particle size	-0.0059	0.0070	0.41
		TOC	0.2230	0.4602	0.63

Table 6. Mean and standard deviation of density (per m²) of selected species from Grassy Point video sled tows.

		W50	W25	W10	FW	FE	E10	E25	E50
11 Arm Stars	Mean	0.19	0.21	0.17	0.52	0.45	0.22	0.09	0.12
	SD	0.04	0.14	0.17	0.13	0.40	0.22	0.08	0.01
Scallops	Mean	0.18	0.31	0.38	0.35	0.43	0.18	0.29	0.49
	SD	0.07	0.17	0.13	0.11	0.04	0.04	0.10	0.11
Flathead	Mean	0.01	0.00	0.01	0.00	0.03	0.01	0.00	0.02
	SD	0.02	0.00	0.01	0.00	0.04	0.01	0.00	0.03
Rock Whiting	Mean	0.18	0.07	0.19	0.08	0.18	0.07	0.05	0.18
	SD	0.02	0.06	0.10	0.03	0.19	0.06	0.06	0.04
Mollusc eggcase	Mean	0.01	0.04	0.01	0.01	0.01	0.04	0.02	0.01
	SD	0.02	0.05	0.01	0.01	0.03	0.04	0.03	0.01
Spider crab	Mean	0.02	0.04	0.02	0.03	0.00	0.00	0.01	0.00
	SD	0.02	0.05	0.00	0.04	0.00	0.00	0.01	0.00
Biscuit Star	Mean	0.00	0.00	0.01	0.00	0.00	0.00	0.02	0.00
	SD	0.00	0.00	0.01	0.00	0.01	0.00	0.01	0.00
Mean Mussel %	Mean	5.26	5.36	3.47	0.89	0.40	0.00	0.00	0.40
	SD	6.38	8.13	7.25	2.68	0.79	0.00	0.00	0.79

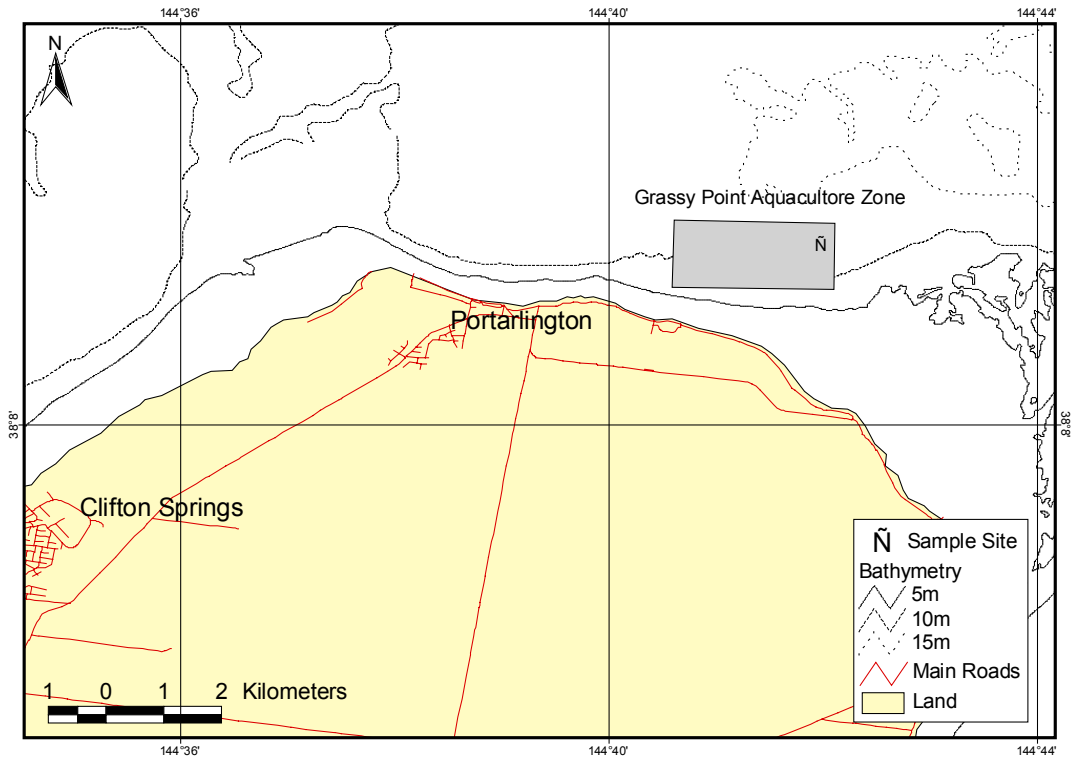


Figure 1. Locality map of Grassy Point Aquaculture Zone, indicating approximate location of sampling site.

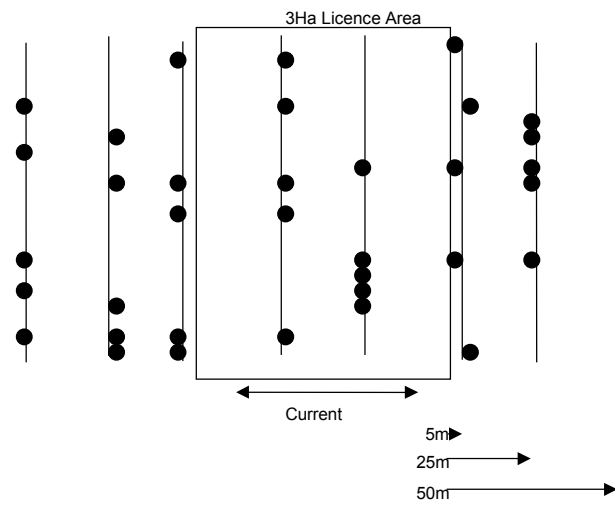


Figure 2. Schematic of study site indicating relative location of transects and indicative randomly selected sampling points.

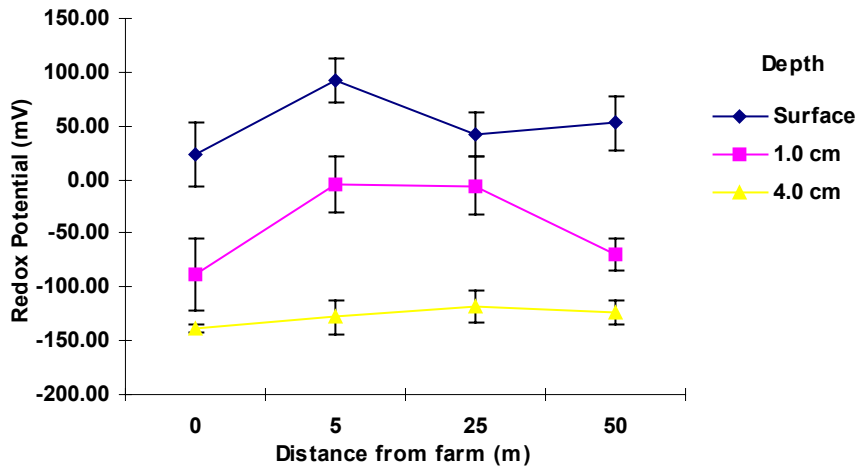


Figure 3. Redox potential (mV) \pm SE with distance from the farm (m).

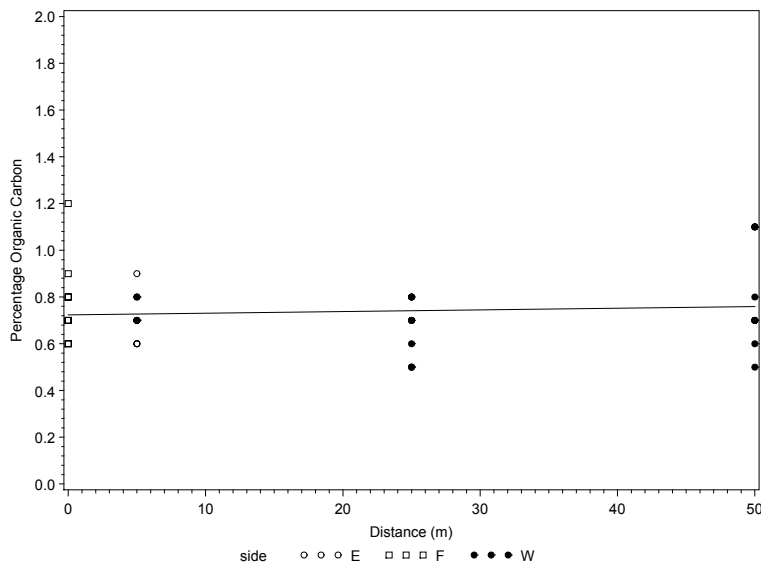


Figure 4. East (o) and west (●) plot showing a regression of % total organic carbon against distance from the farm (□). Probability of a regression coefficient of zero is 0.60.

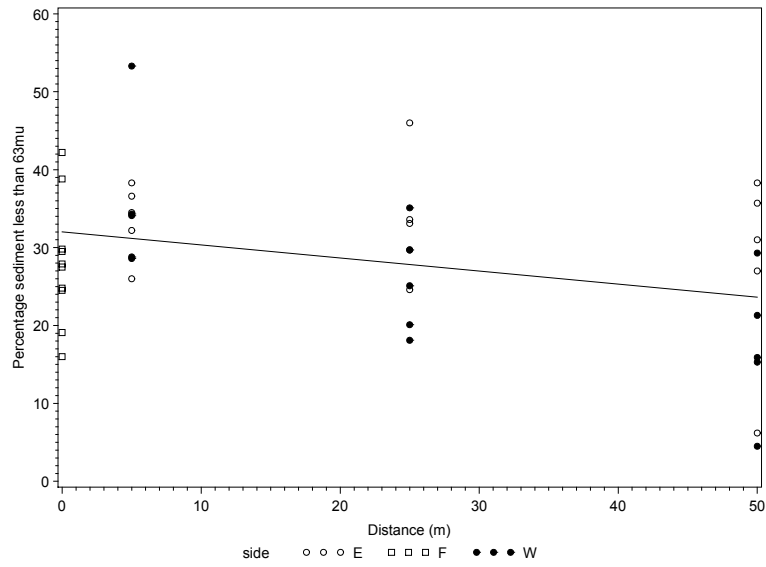


Figure 5. East (o) and west (●) plot showing a regression of % sediment fines (<63µm) against distance from the farm (□). Probability of a regression coefficient of zero is 0.03.

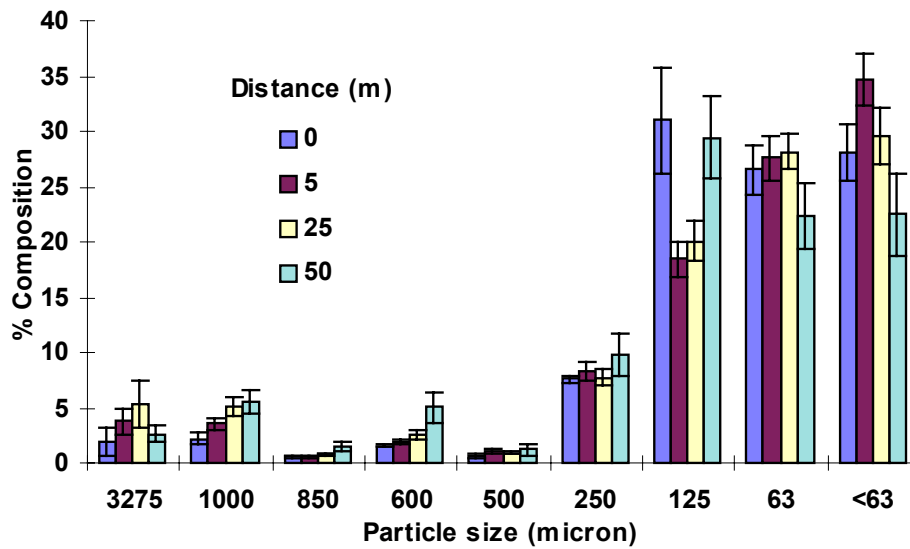


Figure 6. % composition of particle size fraction with distance from the farm

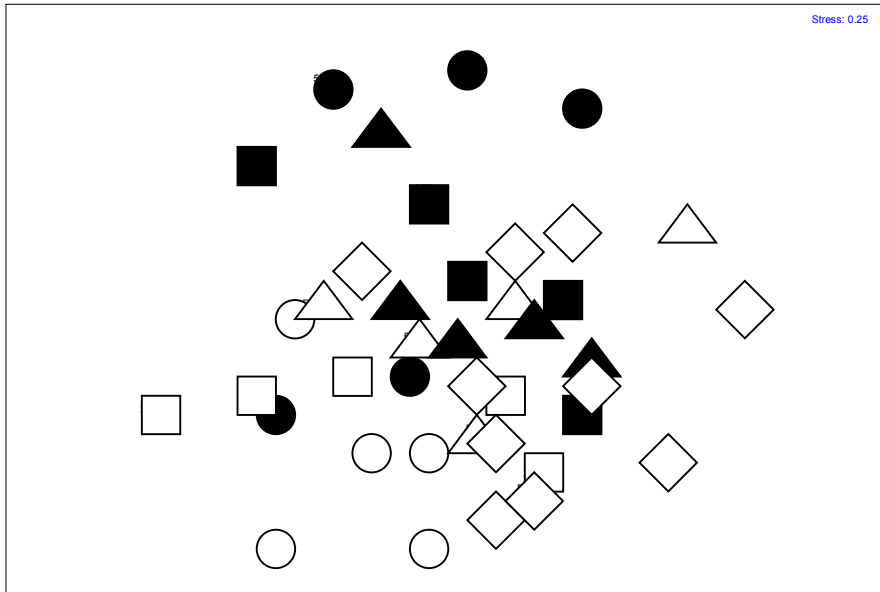


Figure 7. Multi-dimensional scaling (MDS) plot of benthic taxa communities from the study site. ●, east 50m transect; ○, west 50m transect; ■, east 25m transect; □, west 25m transect; ▲, east 5m transect; △, west 5m transect; ◇, farm. Stress = 0.25.

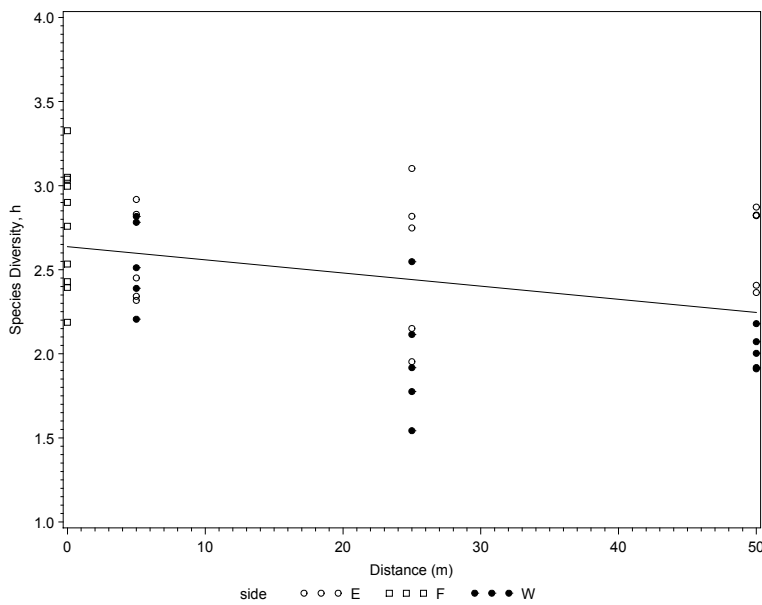


Figure 8. East (○) and west (●) plot showing a regression of species diversity against distance from the farm (□). Probability of a regression coefficient of zero is 0.02.

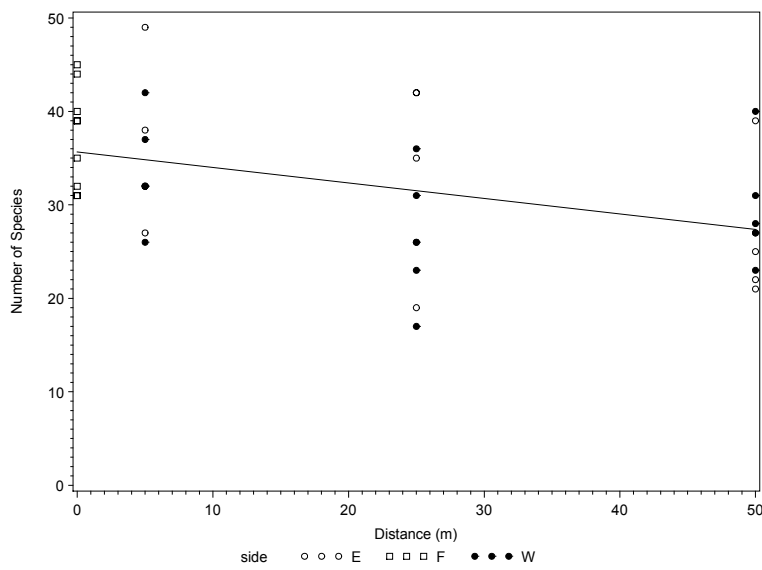


Figure 9. East (o) and west (●) plot showing a regression of species richness against distance from the farm (□). Probability of a regression coefficient of zero is 0.01.

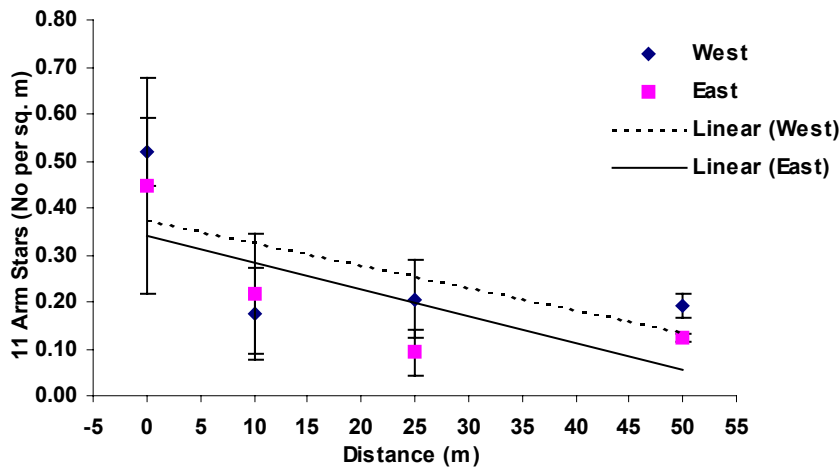
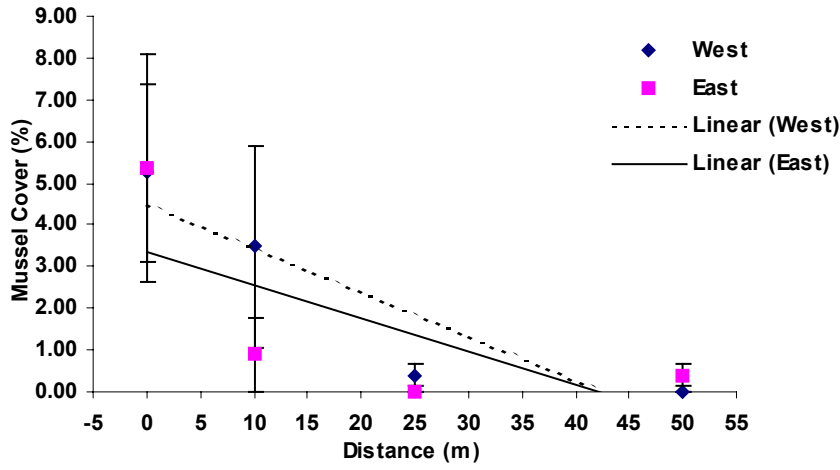


Figure 10. Percent mussel cover and density of 11-Arm stars with distance from the farm

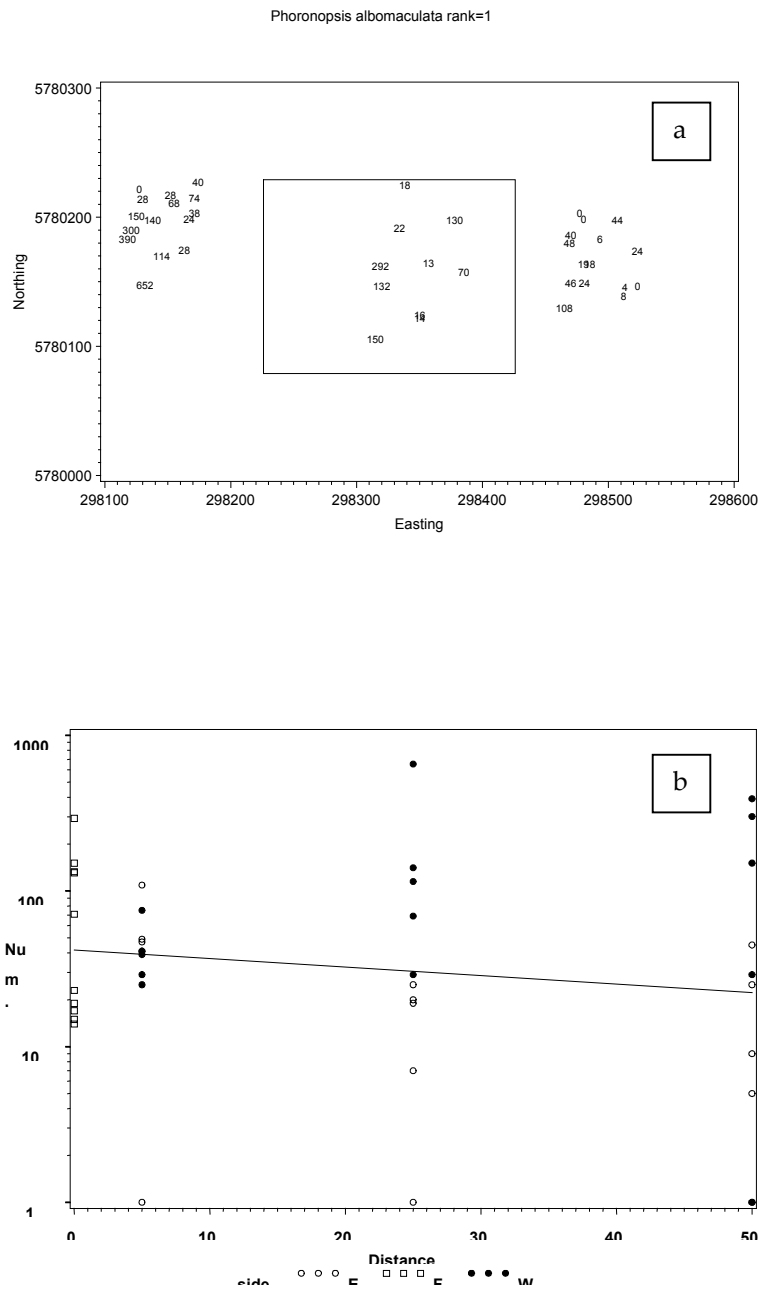


Figure 11. (a), Distribution pattern of *Phoronopsis albomaculata* on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Phoronopsis albomaculata* against distance from the farm (□). Probability of a regression coefficient of zero is 0.34.

Lumbrineris Cf. latreilli rank=2

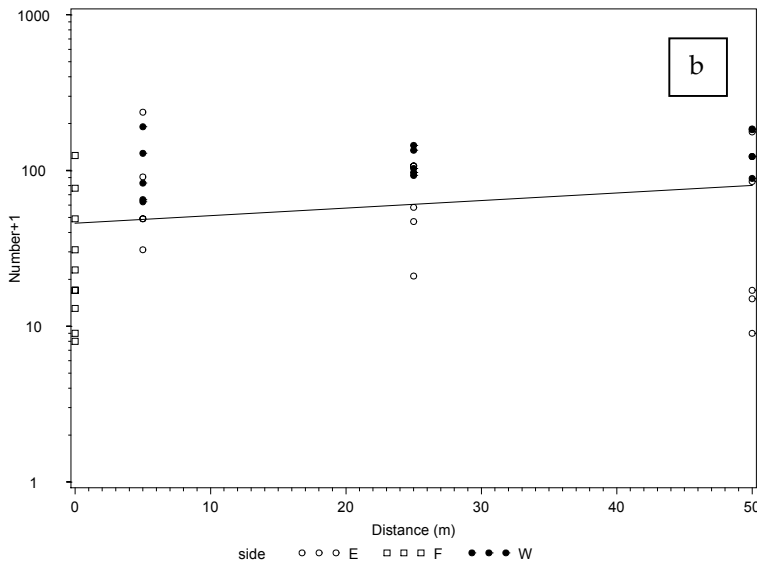
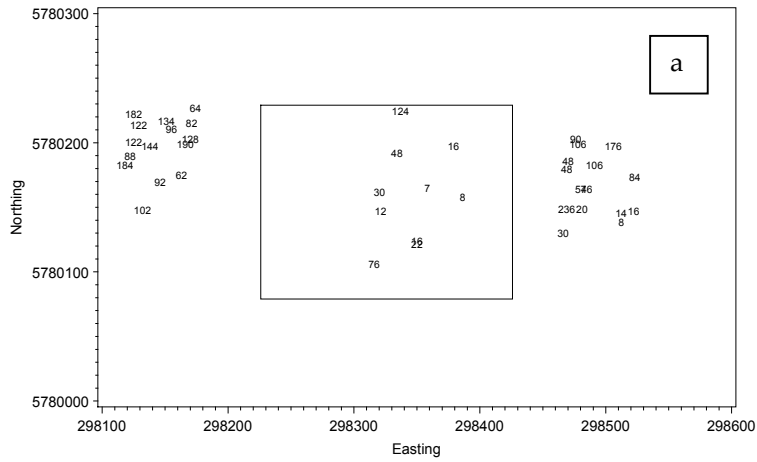


Figure 12. (a), Distribution pattern of *Lumbrineris Cf. latreilli* on and near the study site, the boundaries of which are shown; (b), east (○) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Lumbrineris Cf. latreilli* against distance from the farm (□). Probability of a regression coefficient of zero is 0.15.

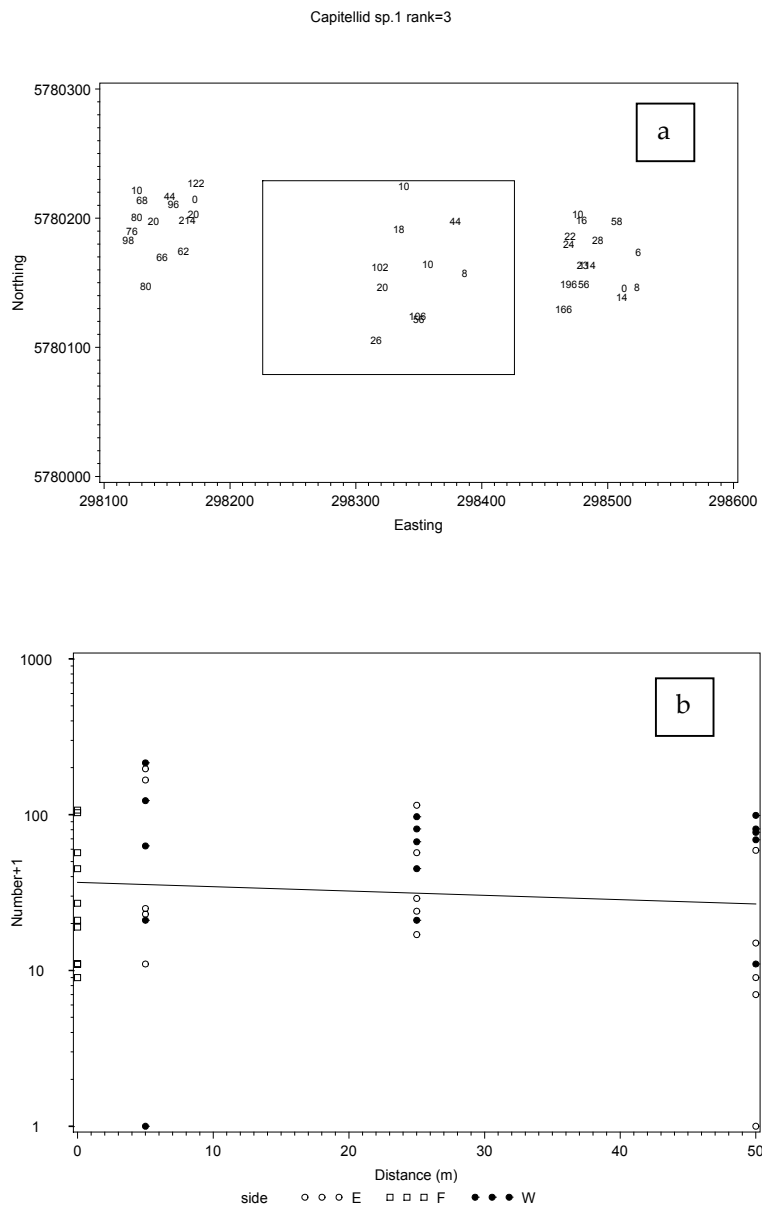


Figure 13. (a), Distribution pattern of *Capitellid sp.1* on and near the study site, the boundaries of which are shown; (b), east (○) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Capitellid sp.1* against distance from the farm (□). Probability of a regression coefficient of zero is 0.53.

Edwardsia vivipara/Edwar rank=4

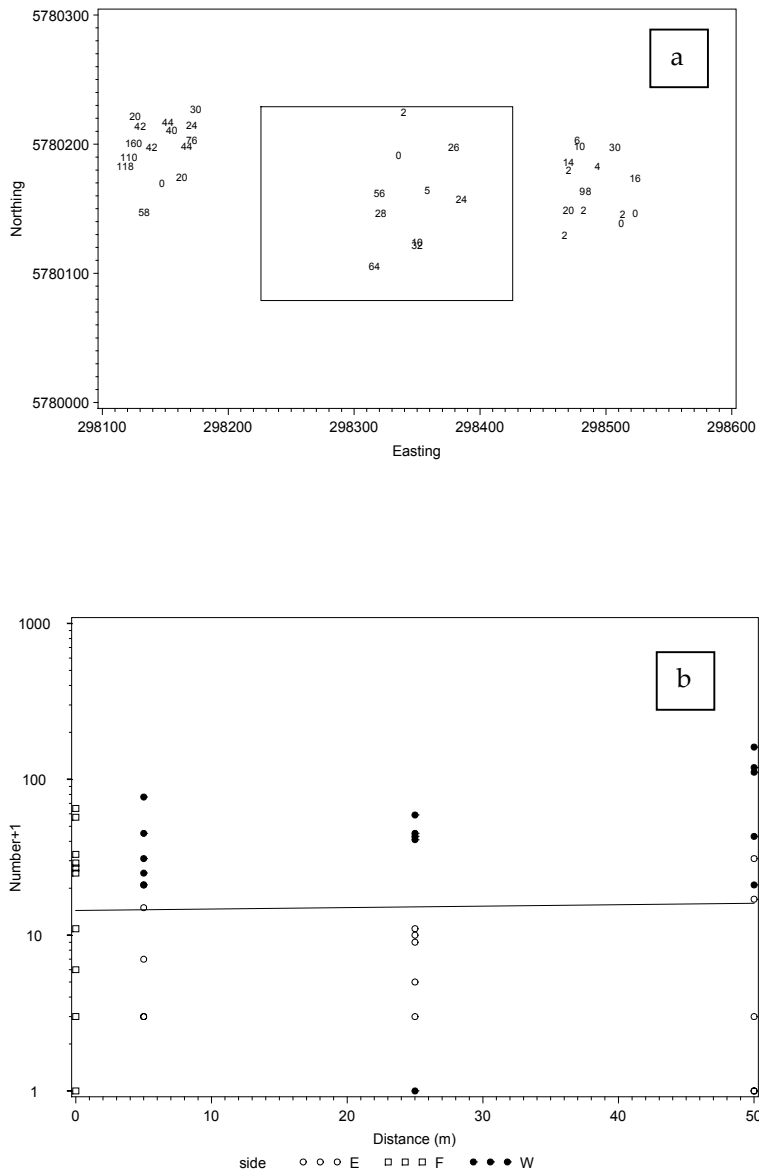


Figure 14. (a), Distribution pattern of *Edwardsia vivipara* on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Edwardsia vivipara* against distance from the farm (□). Probability of a regression coefficient of zero is 0.85.

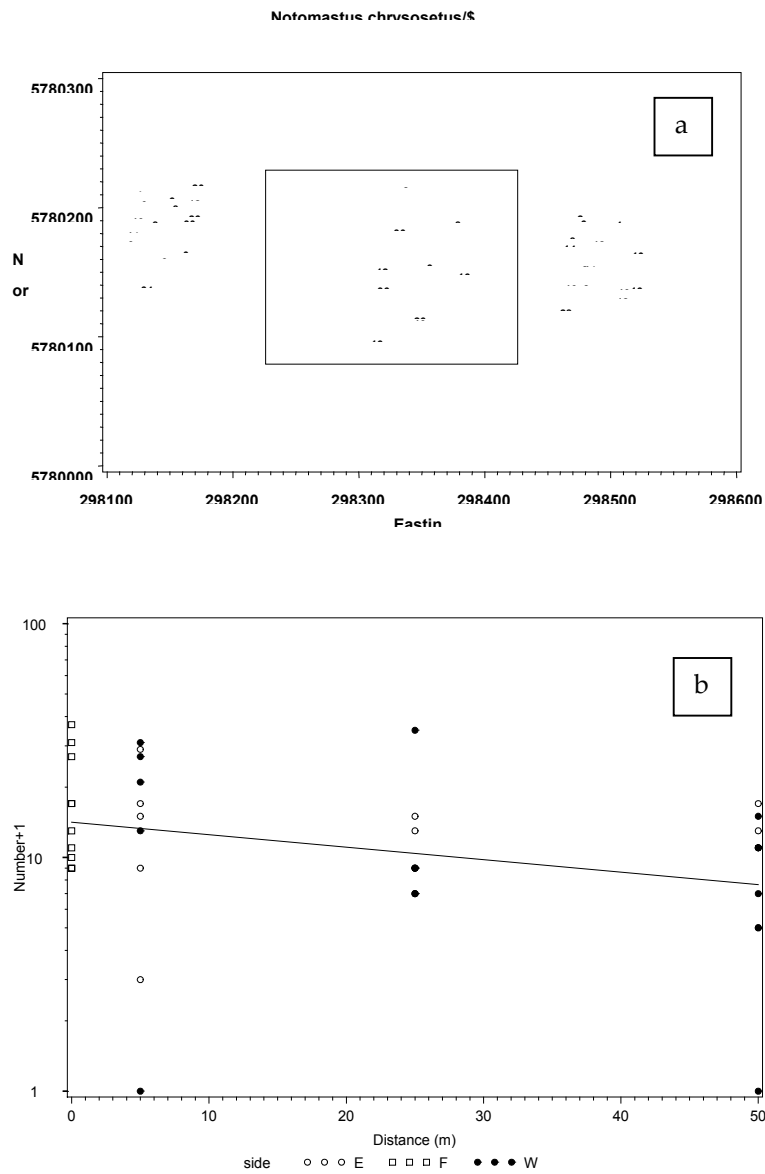


Figure 15. (a), Distribution pattern of *Notomastus chrysosetus* on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Notomastus chrysosetus* against distance from the farm (□). Probability of a regression coefficient of zero is 0.05.

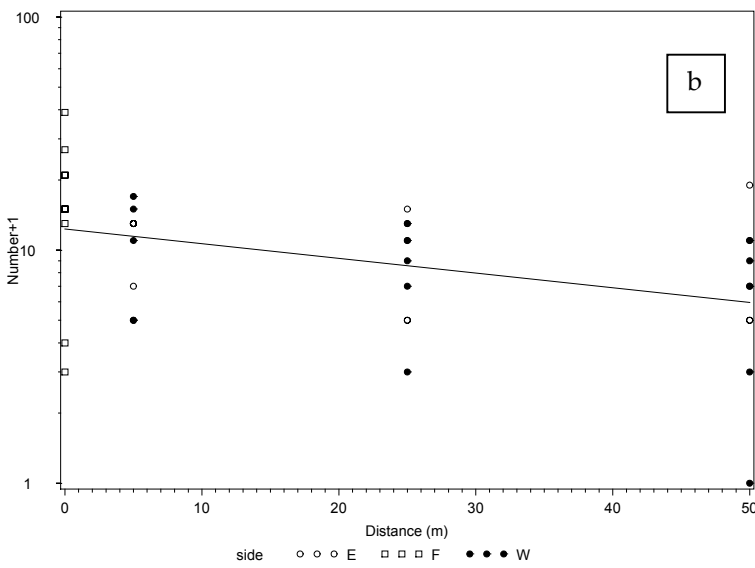
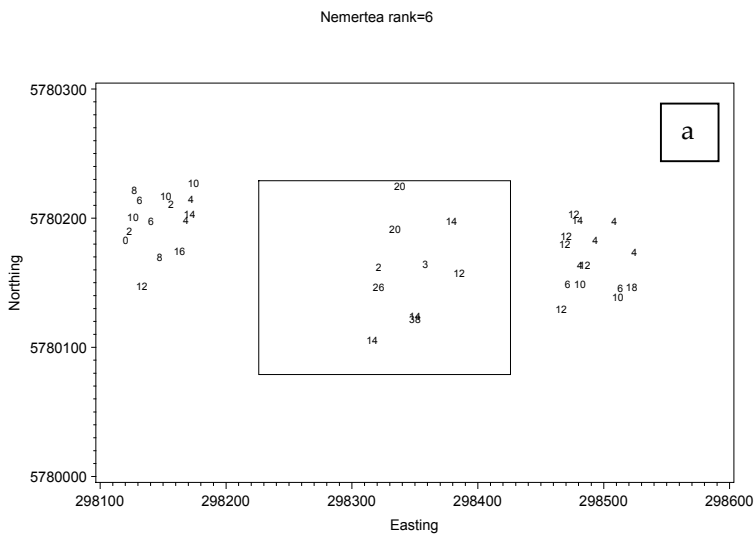


Figure 16. (a), Distribution pattern of Nemertea on and near the study site, the boundaries of which are shown; (b), east (○) and west (●) plot showing a regression of abundance ($\log(n+1)$) of Nemertea against distance from the farm (□). Probability of a regression coefficient of zero is 0.01.

Ampharete sp.1/\$\$Smythel rank=7

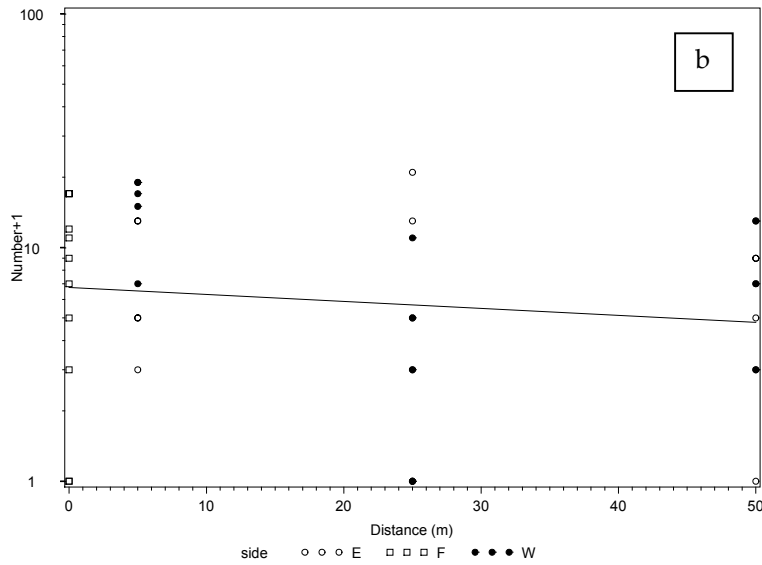
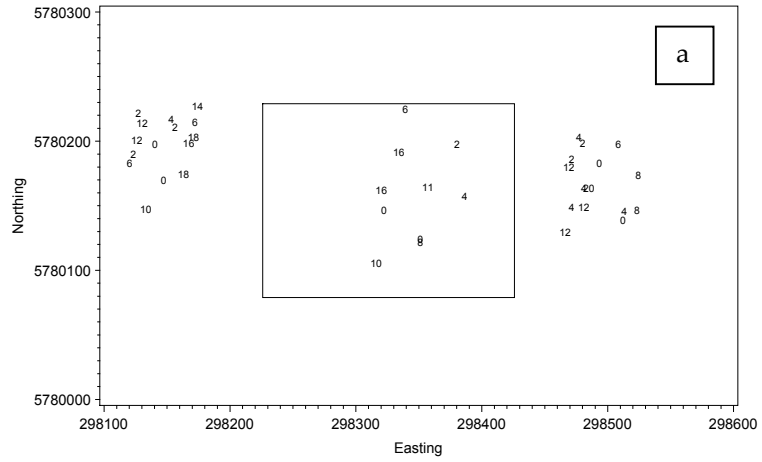


Figure 17. (a), Distribution pattern of *Ampharete sp.1* on and near the study site, the boundaries of which are shown; (b), east (○) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Ampharete sp.1* against distance from the farm (□). Probability of a regression coefficient of zero is 0.38.

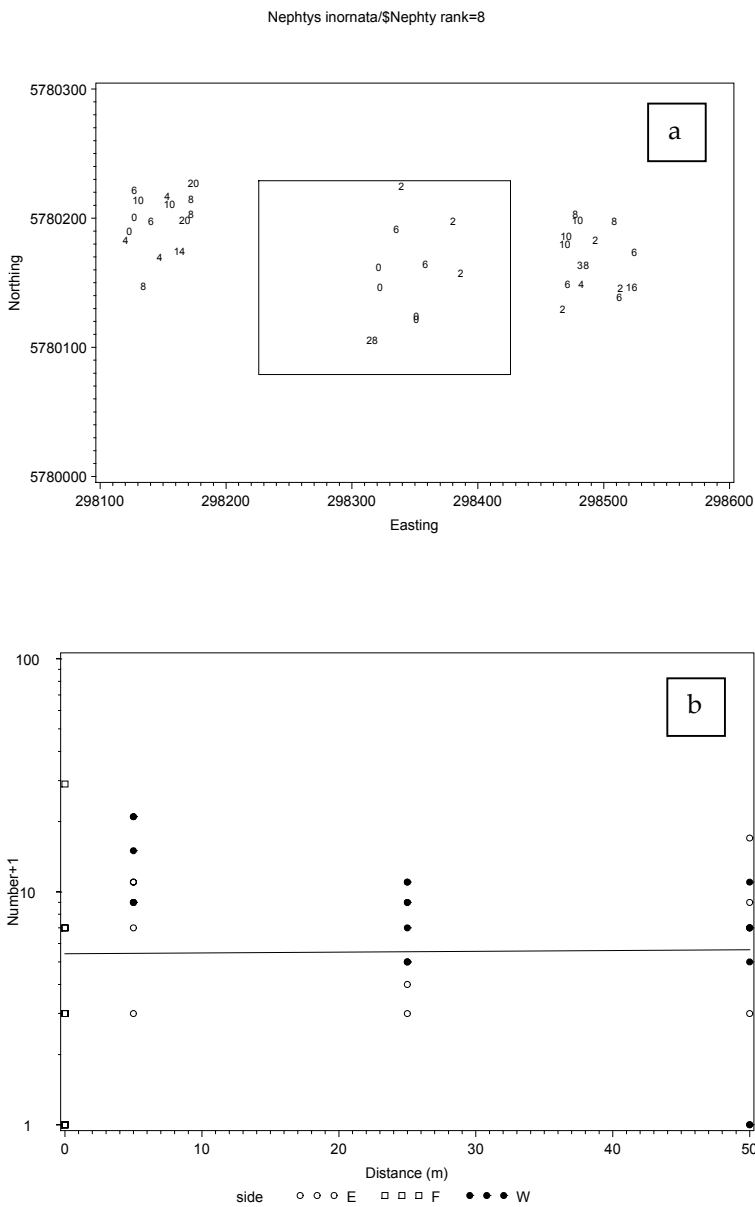


Figure 18. (a), Distribution pattern of *Nephtys inornata* on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Nephtys inornata* against distance from the farm (□). Probability of a regression coefficient of zero is 0.92.

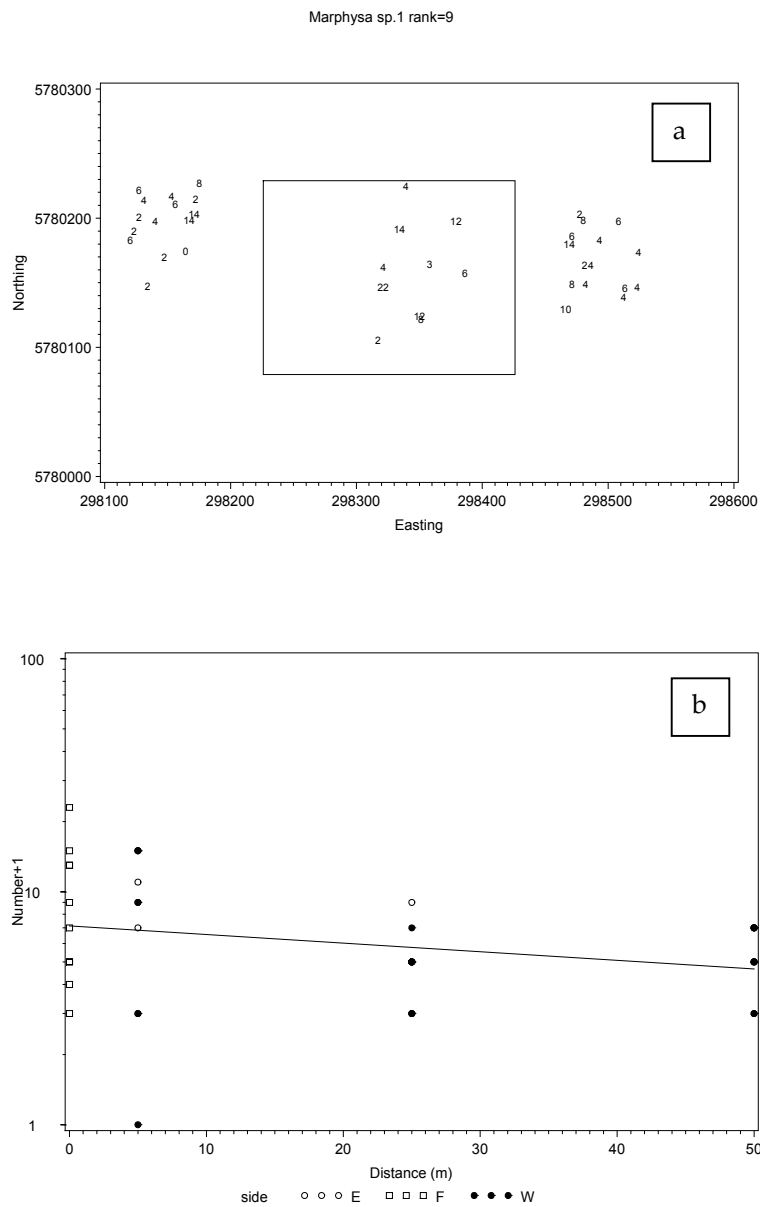


Figure 19. (a), Distribution pattern of *Marphysa* sp.1 on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance (log(n+1)) of *Marphysa* sp.1 against distance from the farm (□). Probability of a regression coefficient of zero is 0.09.

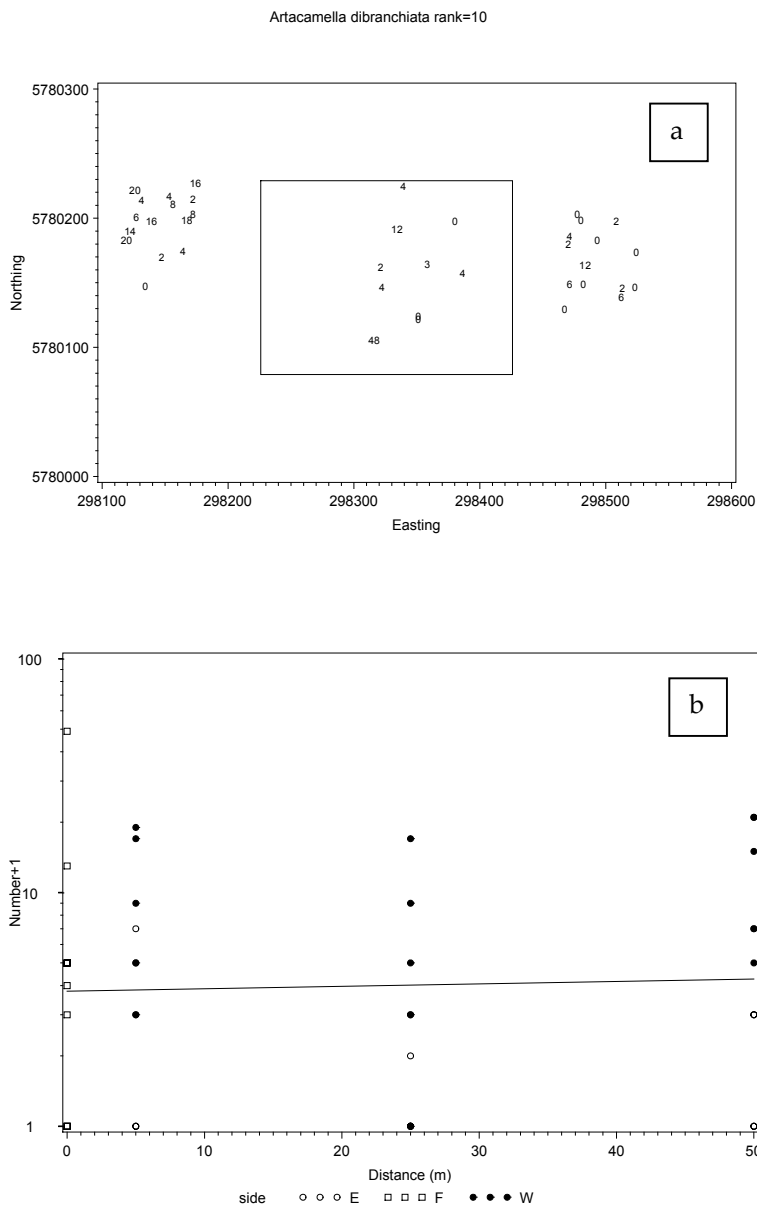


Figure 20. (a), Distribution pattern of *Artacamella dibranchiata* on and near the study site, the boundaries of which are shown; (b), east (o) and west (●) plot showing a regression of abundance ($\log(n+1)$) of *Artacamella dibranchiata* against distance from the farm (□). Probability of a regression coefficient of zero is 0.79.