# The impacts of an eco-elevator harvester on *Cerastoderma edule* stocks, sediment composition and associated macrofauna within the River Exe estuary

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## DECLARATION

This dissertation is a product of my own work and is not the work of any collaboration.

I agree that this dissertation may be available for reference and photocopying at the discretion of the College.

Vanessa Lee

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## ABSTRACT

There has been much conflict between shellfish fisheries and nature conservation due to the increased use of mechanical harvesting methods and the effects these have on shellfish stocks, non-target macrofauna and sediment characteristics. The cockle, *Cerastoderma edule*, is harvested on the Exe estuary by an eco-elevator harvester. The aims of this study are to determine whether the eco-elevator harvester has caused significant effects on macrofauna species and individual abundance and species diversity, cockle size and abundance and sediment grain size, organic content and permeability. Three surveys were carried out between November and April. Macrofauna composition has shown significant differences between areas; individual abundance ($P=0.000$) and species diversity (H') ($P<0.05$). Three key macrofauna species were analysed; a significant effect ($P<0.05$) occurred between areas for *Hydrobia ulvae* and *Pygospio elegans* abundance in all three surveys. Whereas the abundance of *Corophium arenarium* between areas was significantly different ($P<0.05$) in survey three only. However, densities in the fished areas are higher than the control areas, so it would seem macrofauna are not affected by the disturbance and redistribution does not occur. Cockle abundance varied significantly ($P<0.05$) in all three surveys and cockle widths were significantly different ($P<0.05$) between areas in survey one and two only. Fished areas were however significantly higher than control areas. Although visual disturbances of the sediment were recorded, there were no significant effects ($P>0.05$) between areas in organic content and permeability of the sediment. There was a significant increase in sediment particle size; however this was not a negative impact of fishing activity. The present study did not find any significant impacts resulting from fishing by the eco-elevator harvester on macrofauna composition, *C. edule* abundance or size. No significant impacts from fishing on sediment composition were found either. However, further statistical analysis is required between the three surveys within this study and with the baseline survey to come to a clear conclusion. Further monitoring of cockle stocks and of non-target macrofauna is recommended which will also aid the protection of migratory birds which overwinter on the Exe.

# Chapter 1
# INTRODUCTION

Mechanical methods of harvesting shellfish, such as the tractor or hydraulic dredge, have outmoded traditional manual methods as they are able to collect commercial quantities more quickly and efficiently (Leitao & Gaspar, 2007). There has been much concern and debate as to the effects of these mechanical harvesters on shellfish stocks, non-target macrofauna and their dependent bird populations.

Commercial quantities of the edible cockle, *Cerastoderma edule,* inhabit the large intertidal sediments of the Exe estuary, South West England and are harvested by the Exmouth Mussel Company. Despite the importance of the cockle fishery, the Exe estuary is also a Site of Special Scientific Interest (SSSI), a Special Protection Area under the EU Habitats Directive and a RAMSAR site of wetland importance. Conservation priority includes the protection of migratory birds that overwinter on the estuary. Some species have been identified as declining more than would be expected on the basis of the regional and national trends in numbers over the same period. Brent geese, *Branta bernicla*, and oystercatcher, *Haematopus ostralegus*, were two such species, with unexpectedly high levels of decline over both 10-year (winter 1990/91 to winter 1999/2000) and 5-year periods (winter 1995/96 to winter 1999/2000) (Goss-custard, 2007). Over-fishing has been suggested as one of several possible causes of decline; therefore the environmental impacts of fishing activities need to be assessed.

The cockles within the Exe estuary are fished by an eco-elevator harvester. This harvester was developed by John Bayes (Seasalter Shellfish) and Gary Wordsworth (Othniel Shellfish) to allow for cockles to be depurated and sold live as well as in response to increasing environmental concern (Howard, 1999). The eco-elevator harvester operates by lifting cockles from the sediment bed with jets of water onto an elevator chain (Figure 1 (a)) which gently delivers them into fish boxes on deck (Figure 1 (b). Small cockles and non-target species (<20mm in diameter) can pass through the chain mesh

(Figure 1 (c)), and drop straight back onto the seabed. In contrast, hydraulic dredges operate by fluidising the sediment with jets of water, sieving that sediment through a grid and retrieving cockles via a suction pipe into a revolving drum (Hall & Harding, 1997). This involves the cockles being transported in very damaging highly turbid water flows for long periods of time, which can lead to shell damage. The eco-elevator harvester, however, only uses 10% of the power used by hydraulic dredges (Howard, 1999) and should consequently cause less harm to the shells. Devon Sea Fisheries Committee manages the shellfisheries on the Exe. They have implemented byelaws to cover mussel harvesting, a minimum size for winkles and the protection from over fishing of shellfish beds (). However, a byelaw for minimum landing size (MLS) of cockles has not been set; the size collected is dictated by commercial viability and market demands which is generally larger than 24mm (Robbins, 2009).

(a) (b) (c)

Figure 1: (a) Raised elevator chain of harvester (Photo V.Lee). (b) Cockles collected in fish box (Photo V.Lee). (c) Mesh of elevator chain (Photo V.Lee).

Research into the effects on macrofauna composition, cockle stocks and sediment parameters by mechanical harvesters has been widely reported and there is a need for this type of investigation for the eco-elevator harvester. This study has therefore been commissioned by Natural England in conjunction with Devon Sea Fisheries Committee as the final investigation of a 12 month trial fishery and its impact on the Exe estuary.

## 1.1 Aims

The aims of this study are to determine whether the eco-elevator harvester has caused significant effects on:

1. Macrofauna species and individual abundance and species diversity
2. Cockle size and abundance
3. Sediment grain size, organic content and permeability

## 1.2 Hypothesis

$H_0$: The eco-elevator harvester will not cause significant effects on macrofauna composition, cockle stocks and sediment parameters as seen with other types of mechanical cockle harvesters.

$H_1$: The eco-elevator harvester will cause significant effects on macrofauna composition, cockle stocks and sediment parameters as seen with other types of mechanical cockle harvesters.

# CHAPTER 2
# LITERATURE REVIEW

Harvesting causes physical disturbance to intertidal sediments and consequently disturbs the variety of macrofauna species which inhabit them. This may disrupt population, community or ecosystem structure. For example, changes in species abundance may affect the abundance of other species that prey on, compete with or are eaten by the dominant species (Hiddink, 2003), and/or changes to sediment topography may affect deposition of organic matter or macrofauna larvae (Dernie *et al.,* 2003). The intermediate disturbance hypothesis, proposed by Connell (1978; cited by Dernie *et al.,* 2003), states the importance of preventing the competitive exclusion by dominant species to maintain species diversity. Competitively dominant species eliminate subordinate species at low disturbance, however too much disturbance leads to local extinctions (Dial & Roughgarden, 1998), thus disturbance should be at an intermediate level. Disturbance effects depend on a variety of factors:

## 2.1 Type of fishing gear

Tractor dredges use an inclined horizontal blade to skim cockles onto a conveyor belt and into a revolving coarse sieve, which traps large individuals and allows sediment and smaller cockles to be released back onto the sea bed (Cotter *et al.,* 1997). Hydraulic or suction dredges, on the other hand, operate by fluidising the sediment with jets of water, dislodging the cockles from the substratum and transporting them to the surface by a suction pipe (Coffen-Smout, 1998). The immediate effect of hydraulic dredging has shown up to 30% reductions in the number of species and 50% reduction in number of individuals, with recovery times varying from 14-56 days (Hall & Harding, 1997). In particular, Hiddink (2003) found that suction dredging for cockles in the Wadden Sea caused significant negative effects on densities of *Macoma balthica*; and this effect persisted one year after dredging. There were, however, no significant effects on the mudsnail *Hydrobia ulvae* or *C. edule*. In comparison, Ferns *et al.* (2000) observed reductions in densities of the tube dwelling species *Pygospio elegans* and *H. ulvae* 100 days after harvesting by a tractor dredge. Also numbers of *Bathyporeia pilosa* did not fully recover 111 days after dredging. However, despite results showing a major impact on macrofauna communities, sampling was not continued for long enough to determine how long they took to recover.

Cockles that are transported through suction pipes and sorted through sieves are subjected to repeated physical shocks. Several studies have reported that this has resulted in damage to cockle shells and slowed their burrowing behaviour, even when shells are not damaged (Cotter *et al*., 1997, Coffen-Smout & Rees, 1999, Mendonca *et al.,* 2008). This may result in long term reductions in cockle stocks. Coffen-Smout (1998) described damage rates of 11-14% from several hydraulic suction dredgers. Cotter *et al.* (1997) also observed damaged discards of cockles, in this case from a tractor dredge, which were therefore unable to rebury. However, observations a year after dredging showed no further mortalities and spatfall was not significantly different to control areas, concluding that the effects of dredging on cockle stocks was negligible. A reburrowing experiment, using tagged and marked cockles and simulated physical shocks, found cockles had been transported by the tide, 200m away from where they were released (Coffen-Smout & Rees, 1999). Also they found more small cockles were able to reburrow before the tide flooded compared to medium or large cockles, although younger smaller cockles are thought to be most vulnerable to mechanical damage due to their lighter shells (Cotter *et al.,* 1997). Coffen-Smout & Rees (1999) also observed cockle discards from hydraulic dredges will reburrow more effectively when harvesting takes place at high tide than at low tide, as they will not be disturbed or carried away as the tide comes in. Cockles broken during harvesting and left to decompose at the site may temporarily contribute to anoxic conditions in the sediment (Mendonca *et al.,* 2008), making conditions unfavourable to other macrofauna.

## 2.2 Sediment type and macrofauna community structure

Muddy sands have a more structured community, consisting of sedentary and tube-dwelling species, such as *P. elegans*, whereas less stable habitats are comprised of more opportunistic, mobile species, for example *Eurydice pulchra* and *B. pilosa*. Bolam & Fernandes (2003) investigated the ecological significance of dense aggregations of *P. elegans* and the tube-beds they create. An increase in abundance of other macrofauna was found within the *P. elegans* patches attributable to the polychaete tubes stabilising the sediment, and creating refuges against physiological stress and predators. However, several studies have observed that macrofauna in these stable habitats recover less quickly than those in unconsolidated coarse sediments (Gubbay & Knapman, 1999, Collie *et al.,* 2000, Ferns *et al.,* 2000). For the latter, Collie *et al.* (2000) suggest adaptations to periodic sediment resuspension and smothering result in higher recovery rates of macrofauna. It is important not to classify habitats by the particular nature of the sediment; intertidal sandflats inhabited by high densities of spionids will be more stable than sandlfats with relatively little macrofauna, and therefore will have a higher impact from fishing (Kaiser *et al.,* 2001).

Macrofauna also possess adaptations to other potential stresses encountered on estuaries, such as dessication, extreme temperature and reduced time for feeding, which allow for rapid colonisation (Bolam *et al*., 2004). Estuaries are also exposed to high degrees of anthropogenic stress. Elliott & Quintino (2007) developed the term Estuarine Quality Paradox which describes the difficulty of distinguishing natural stress from human-induced stress due to the similarity between the features of organisms and assemblages in both estuaries and anthropogenically-stressed areas.

Reductions in macrofauna densities suggest the habitat is less suitable as a consequence of dredging and therefore, results in either mortality, redistribution of motile species from dredged to undredged areas, or both (Hiddink, 2003). An experiment by Coffen-Smout & Rees (1999) showed an active movement of cockles away from disturbed areas. Other mobile species are capable of undertaking mass movement during a single tide, such as *B. pilosa*; however Ferns *et al*. (2000) observed a failure of this species to recolonise after 39 days of dredging. Defensive secretions released by injured macrofauna may explain the long recovery time. As demonstrated by Dernie *et al*. (2003), juvenile lugworms, *Arenicola marina,* and spionid worms may deter settlement of other species onto disturbed habitats by releasing defensive chemicals. Lugworms are also known to have negative effects on the abundance of juveniles of other polychaetes such as *Nereis diversicolor*, *P. elegans*, and *C. edule*, because their burrowing behaviour causes high sediment resuspension which affects larval settlement (Mendonca *et al.,* 2008). Gaspar *et al.* (2003) also suggest that species reductions may persist if the disturbed area is large comparative to the remainder of the habitat and a dilution effect cannot occur. If there is a larger area of the habitat that remains undisturbed, the impact of disturbance is lessened.

Slow recovery of adult populations may also be due to the consequences of tractor dredges exposing and dispersing anoxic layers to the surface, resulting in the release of sulphides into the upper layers of the sediment. This may disrupt the redox potential discontinuity (RPD) layer (Ferns *et al.,* 2000), which could be an obstacle for reburrowing by macrofauna displaced by the fishing gear. This is supported by observations by Bolam *et al.* (2004) of reduced redox potential profiles produced by increased organic content of the sediment. This resulted in a species-specific decline in macrofaunal recolonisation; *Tubificoides benedii* and *Streblospio shrubsolii* (a spionid) were negatively affected, whereas *H. ulvae* and *Hediste diversicolor* showed no response to increased organic matter. Piersma *et al*. (2001), on the other hand, suggest that the sediment becomes unattractive for settlement of bivalves because silt and organic material are resuspended in the water column by dredges and transported away by the tide. Effects of sediment resuspension include; nutrient release from the sediment, increased biological oxygen demand and smothering of feeding and respiratory organs (Kaiser *et al.,* 2001). However the effects are more pronounced in deep-water environments that are relatively undisturbed. The quantity of sediment resuspended depends on sediment grain size; higher on mud and fine sand compared to coarse sand (Kaiser *et al.,* 2001). Deposition of organic matter and macrofauna larvae may be reduced as a result of tracks being left in the sediment by harvesters. Tractor dredges have been reported to leave tracks in the sediment which in stable sediments have been visible for more than six months (Gubbay & Knapman, 1999). Hydraulic suction dredges have also been known to create furrows between 0.5-3.5 m wide and 5-60 cm deep (Dernie *et al.*, 2003).

Stillman *et al*. (2001) discovered that harvesting of larger cockles may enhance spat settlement of cockles on the Exe estuary, based on evidence that spat recruitment is inversely dependent on adult density. Harvesting of larger cockles by tractor dredge has also been demonstrated by Mendonca *et al.* (2008) to enhance the survival of smaller macrofauna. This is explained by the fact larger cockles have high filtration rates which may inhibit their own and other species larval settlement and growth. In particular, the removal of cockles led to the proliferation of the spionid polychaete *P. elegans*. This is supported by Bolam & Fernandes (2003) who propose competition, predation and sediment disturbance from burrowing by *C. edule* are responsible for the negative effect on *P. elegans*. Specifically, *C. edule* has been shown to inhale settling *P. elegans* larvae. In contrast, Ferns *et al.* (2000) found a significant reduction in *P. elegans* of 83% after tractor dredging. Populations of opportunistic species such as *P.elegans* are unstable and dense beds have been found to be replaced by subsequent colonisers (Bolam & Fernandes, 2003) which could account for such reductions in number.

### 2.3 Seasonality

Temperature and therefore time of year can also influence abundances of macrofauna species. Spawning takes place mainly in spring for species such as *Nephyts hombergii* and *M. balthica*. Phytoplankton blooms occur in autumn which makes this the best time of year for *H. ulvae*, *C. edule* and other bivalve recruitment. Only in *P. elegans* and some populations of *B. pilosa* does breeding occur throughout the year (Ferns *et al.,* 2000).

Recruitment of young cockles is very variable from year to year (Franklin, 1972, Beukema & Dekker, 2005). Weather, tidal conditions and availability of suitable food are important factors for spawning and settlement. Rapid recovery of macrofauna observed after disturbances can be partly attributed to the reproductive characteristics of dominant species. Certain polychaetes ensure there is an adequate supply of larvae to recolonise disturbed areas by either producing larvae that remain in the plankton for a short time before settling or those that go through complete benthic development (Bolam *et al.,* 2004). Similarly, tubificid oligochaetes directly replenish populations by spending their whole life cycles below the sediment. Avoiding dredging during key spawning periods could reduce time required for the recovery of macrofauna communities (Gubbay & Knapman, 1999). Cockles are sensitive to low temperatures resulting in an increase in their overall biomass during spring and summer, followed by a decrease from September-March (Smit *et al.*, 1998). Studies of mechanical harvesting on the Thames estuary presented results on effects on undersized cockles. Dredging for a short period over a bed of spat during their first winter had little effect on the numbers surviving to fishable size. However, prolonged dredging especially during the cockle's second growing season from May-October resulted in significant reductions in numbers.

### 2.4 Scale of the habitat

Within the Wadden Sea 1500 ha of a total of 5000 ha of intertidal flats were harvested for cockles and resulted in considerable effects on target and non-target species (Piersma *et al.,* 2001). However a study by Hall & Harding (1997) showed that when dredging took place within an area of only 7 ha, although there were high levels of mortality, recovery of macrofauna was rapid.

### 2.5 Over-exploitation of shellfish stocks

One case of severe ecological damage was the collapse of the Dutch Wadden Sea fishery in the early 1990's. Suction dredging for cockles caused an 8 year long decline in bivalve stocks (Piersma *et al.,* 2001). This was explained by reduced spatfall due to the loss of fine silts within the dredged areas. The fine grained sediments which attract settling bivalve larvae were unable to build up due to the filter feeding bivalves no longer producing faecal pellets. This re-processing of deposited material primarily increases the sediments water content, resulting in less stable and more thrixotropic conditions which results in easier burrowing (Little, 2000).

Over-exploitation of shellfish also has consequences on bird populations which over-winter on estuaries. In particular, oystercatcher numbers declined on the Wadden Sea following a number of winters with low shellfish stocks. Reducing the biomass of shellfish available per oystercatcher forces birds to feed at higher densities, thus interference competition for food is intensified (Stillman *et al*., 2001, Goss-Custard *et al.,* 2004). The consequent high mortality of oystercatchers has led to the development of several simulation behaviour-based models to investigate the effects of mussel and cockle fisheries on the survival and numbers of over-wintering birds. Stillman *et al*. (2001) carried out a simulation model of increasing fishing intensities by suction dredging on the Exe estuary and Burry inlet, South Wales. Comparisons in mortality rates of oystercatchers between sites with equal fishing efforts were discovered. Bird populations on the Exe have more difficulty in meeting their energy demands in the absence of fishing, contributing to the high mortality after small increases in fishing effort compared to the equivalent increase on the Burry inlet not affecting mortality. In a similar experiment, Atkinson *et al*. (2003) investigated the relationship between the state of the shellfish stocks and changes in oystercatcher and knot survival in the Wash, eastern England. Like the Wadden Sea, during the 1990s shellfish stocks on the Wash collapsed due to a long run of poor recruitment following high fishing rates. Atkinson *et al*., (2003) concluded that a reduction in juvenile settlement, rather than survival resulted in oystercatcher populations declining from ca. 40,000 to 11,000 birds. One of the key findings of this study was that during the years prior to the first mortality event mussel stocks were much higher which suggests the mussels were sustaining the oystercatcher population. This is supported by Smit *et al.*

(1998) who suggest that mussel beds provide a more reliable food source when cockle stocks are low, and therefore fishing on newly developing as well as existing intertidal mussel beds should be highly restricted. Furthermore, Stillman *et al*. (2001) suggest fishing should be reduced during periods or winters in which birds are already having difficulty surviving.

## CHAPTER 3
## MATERIALS AND METHODS

### 3.1 Study Site

The study was undertaken at Cockle Sands on the Exe estuary, located North West of Exmouth, Devon, UK (Figure 2). Molluscan shellfish farming is the largest single commercial fishery on the Exe, with its extensive sand flats supporting Mussel, *Mytilus edulis,* beds and Pacific Oyster, *Crassostrea gigas,* beds, as well as dense *Cerastoderma edule* beds. In addition there is small scale recreational and commercial hand raking for shellfish and the site is dug for bait by anglers and also used by crab tilers. Fishing for cockles by the eco-elevator harvester has been carried out on the study site 20 minutes twice a week during spring tides. This ceased twelve months before the survey began.

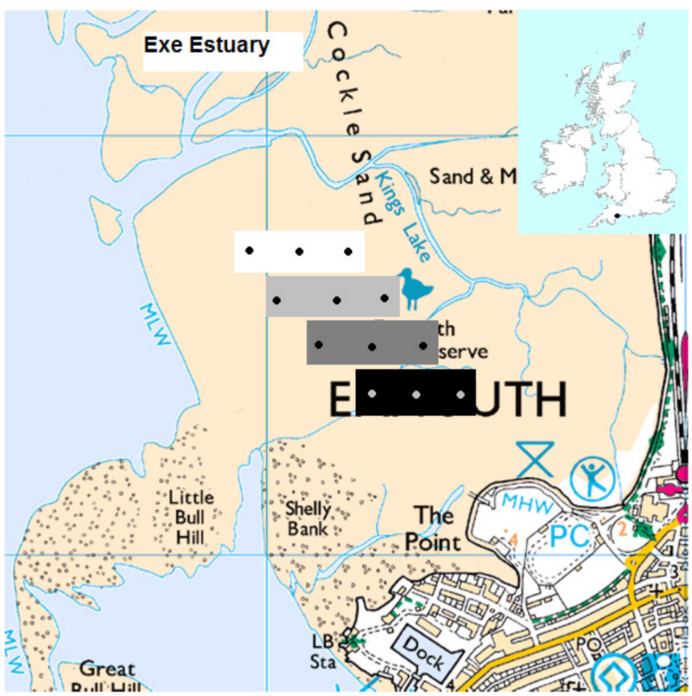



Figure 2: Map of Exe Estuary showing areas where samples were taken. (Not to scale) ■ = control area 1 ■ = fished area 1 ■ = control area 2 □ = fished area 2. • = Sampling sites

Two 100 $m^2$ treatment (fished) areas were selected along with two 100 $m^2$ control plots; 175 m were taken between each area. The areas were located by using a portable global positioning system (GPS) along a bearing of 310$^o$ from a yellow marker post to the flag pole of Powderham Castle. The coordinates in Table 1 mark the start of each area. Three sites within each area were selected haphazardly by walking 100 m along a line transect following a bearing of 270$^o$. Sampling took place within each of these sites.

| Area | Coordinates |
|---|---|
| Fished area two | 50 37.588 003 25.708 |
| Control area two | 50 37.533 003 25.605 |
| Fished area one | 50 37.475 003 25.499 |
| Control area one | 50 37.404 003 25.369 |

Table 1: GPS coordinates (latitude and longitude) for the treatment and control areas

A baseline survey was carried out by Hulme (2009) in May and 23 days later fishing commenced, followed by the first sampling survey being carried out on 23rd July. This study includes three more sampling surveys continued from this, carried out every two-three months. The first survey in this study was completed on 5$^{th}$ November, the second survey on 1$^{st}$ February and the third on 1$^{st}$ April.

### 3.2 Sampling of the Macrofauna

Four cores (10cm diameter x 100$cm^3$ volume) of sediment were taken haphazardly from the three sites in each area, sieved through a 1mm mesh and any macrofauna retained in the sieve were preserved in 70% ethanol for analysis in the laboratory. Macrofauna were subsequently identified to the lowest possible taxonomic level, using the appropriate dichotomous key (Hayward & Ryland 1995), under a low power dissection microscope.

### 3.3 Sampling of the cockles

A 0.3 $m^2$ quadrat was placed haphazardly at each site within each area. Sediment was removed within the quadrat from a depth of 6 cm and sieved through a 1 cm sieve. Widths of the cockles retained on the sieve were measured to the nearest millimetre and abundance was recorded.

In addition, 1 kg samples were taken from the eco-elevator harvester during fishing activity on the same day as sampling and width measurements recorded to the nearest millimetre. The numbers of broken and dead cockles were also recorded.

### 3.4 Sampling of the sediment

A sediment core (2 cm diameter x 20 $cm^3$ volume) was taken at each site within each area, placed in labelled polythene bags and kept frozen until required for analysis. Permeability of the sediment was measured at each site within each area by dropping an 80 cm steel rod from a height of 30 cm and measuring the depth to which it penetrated the sediment.

Sediment samples were dried for 24 hours in a 35$^{o}$C oven to remove moisture from the samples and then passed through a 1mm sieve, to remove any large coarse sediment. Grain size analysis was then carried out using the Malvern Long-bed Mastersizer 2000 particle sizer. Five sub-samples from each site sample were taken and each sub-sample was tested five times. Sediments were classified using the Wentworth Scale. The major size classes determined are gravel (-2 phi to –5 phi), sand (+4 phi to –1 phi), silt (+5phi to +7 phi) and clay (+8 phi and smaller). Mean particle size (phi) was used in the analysis.

To measure sediment organic content, the loss on ignition method (LOI) was employed. Samples were placed in crucibles and their weight measured before oven-drying to constant weight for 24 h at 105$^{o}$C. The samples were then re-weighed, burnt in a muffle furnace at 550$^{o}$C for 4 h and re-weighed again. The LOI was then calculated using the following equation (Heiri *et al.,* 2001):

$$LOI_{550} = ((DW_{105}-DW_{550})/DW_{105})*100$$

DW represents the dry weight of the sample in grams before combustion and after heating to 550$^{o}$C. $LOI_{550}$ is represented as a percentage and is proportional to the amount of organic carbon contained in the sediment.

## 3.5 Statistical analysis

### 3.5.1 Univariate analysis

The software package SPSS was used to carry out univariate statistical analysis. Univariate analysis of variance was used for all sample data including post-hoc analysis using Fisher's least significant difference (LSD). In addition, the Levene's test of homogeneity of variance was applied. Alternatively, whenever normality assumptions failed by Kolmogrov-Smirnov testing, the non-parametric Kruskal-Wallis test was used (Kinnear & Gray, 2008). If significant differences were found this was followed by the Mann-Whitney U test to compare two independent groups of sampled data.

PRIMER (Plymouth Routines in Multivariate Ecological Research) v5 software package was used to calculate Shannon-Weiner diversity index (H') for the macrofauna data.

### 3.5.2 Multivariate analysis

PRIMER v5 software package was used for multivariate statistical analysis. Multidimensional scaling (MDS) was used to graphically coordinate differences in macrofauna species composition onto two-dimensional charts Bray-Curtis similarity matrixes were used to construct the plots. The similarities percentage procedure (SIMPER) was also adopted to examine species contribution between areas.

## CHAPTER 4
## RESULTS

### 4.1 Macrofauna composition

In total 25 different taxa were identified with *Cerastoderma edule* juveniles recorded separately; most of the taxa were successfully identified to species level.

Analysis of species number showed no significant differences ($P>0.05$) between fished and control areas in all three surveys (Table 2). Also there was no significant difference between surveys ($P=0.547$). Whereas, all three surveys showed significant differences ($P<0.05$) in individual abundance ($P=0.000$) between areas. Mann-Whitney U tests established a significant difference ($P<0.05$) between fished area one and fished area two, and between fished area two and both control areas for all surveys (Appendix A). As can be seen in Figure 3 the mean individual abundances are considerably higher in the fished two areas compared to other areas. Individual abundance between surveys did not differ significantly ($P=0.840$).

| | Species number | Species diversity (H') |
|---|---|---|
| Survey | P value | P value |
| 1 | 0.187 | **0.003** |
| 2 | 0.190 | **0.005** |
| 3 | 0.290 | 0.111 |

Table 2. P values from univariate analysis of variance of macrofauna composition between areas

Species diversity (H') varied significantly ($P<0.05$) between areas in survey one and survey two (Table 2). Post hoc tests for survey one established a significant difference between fished area one and fished area two (Appendix B); with fished area one having a higher mean diversity (Figure 3). Fished area one and two varied significantly ($P<0.05$) to control area one (Appendix B); control area one had a higher mean diversity than both fished areas (Figure 3).

Analysis of survey two species diversity (H') showed a significant difference ($P<0.05$) between control area one and fished area two and between control area one and two (Appendix B). Survey three analysis showed no significant differences between areas (P>0.05) (Table 2). The between surveys analysis revealed a significant difference in fished areas two between survey one and survey three only (P=0.313). The mean species diversity increased in fished area two between survey one and three (Figure 3).

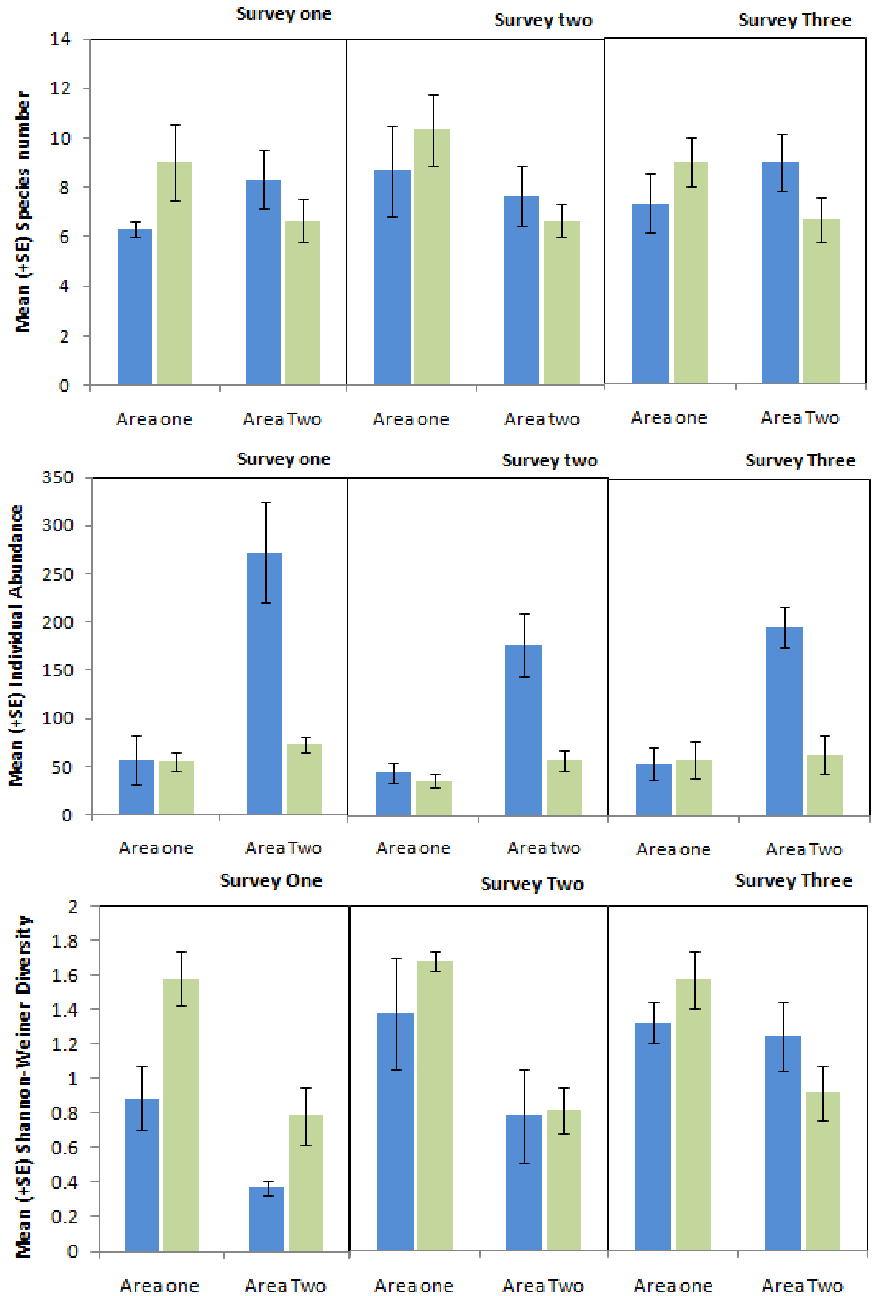



Figure 3. Species number, individual abundance and Shannon-Weiner diversity for fished ( ) and control ( ) areas

Three key species were analysed to assess disturbance on a species level due to their high abundance within the samples. *Hydrobia ulvae* were the most dominant species, accounting for 49.42% of all the samples, *Pygospio elegans* was the second dominant species, comprising 14.84% of all the samples and is also an opportunistic species. *Corophium arenarium*, a short term coloniser, was the third species to be assessed. The rest of the samples were composed of bivalves, other polychaetes, oligochaetes, amphipods and nematodes.

A significant effect ($P<0.05$) occurred between areas for *H. ulvae* abundance in all three surveys (Table 3). In survey one abundance in fished area two was significantly different ($P<0.05$) to fished area one and both control areas (Appendix C). This was the same in survey two except fished area one was also significantly different to control area two and the two control areas also varied significantly (Appendix C). Very low abundances were recorded in fished area one and control area one (Figure 4).
Survey three compared the same to survey two with the exception of fished area one differing significantly to control area one instead of control area two (Appendix C). Abundances in fished area one increased slightly to numbers nearer those in control area two, whereas they remained low in control area one (Figure 4). Figure 4 also shows the mean *H. ulvae* abundance for fished area two is higher than in the other areas but decreases from survey one to survey three. The Kruskall-Wallis test confirms this is a significant decrease ($P=0.026$). There were no other significant differences in areas between surveys ($P>0.05$).

| | *H. ulvae* abundance | *P. elegans* abundance | *C. arenarium* abundance |
|---|---|---|---|
| Survey | P value | P value | P value |
| 1 | **0.000** | **0.011** | 0.692 |
| 2 | **0.000** | **0.05** | 0.294 |
| 3 | **0.000** | **0.000** | **0.006** |

Table 3. P values from Kruskall-Wallis analysis of key species abundance between areas

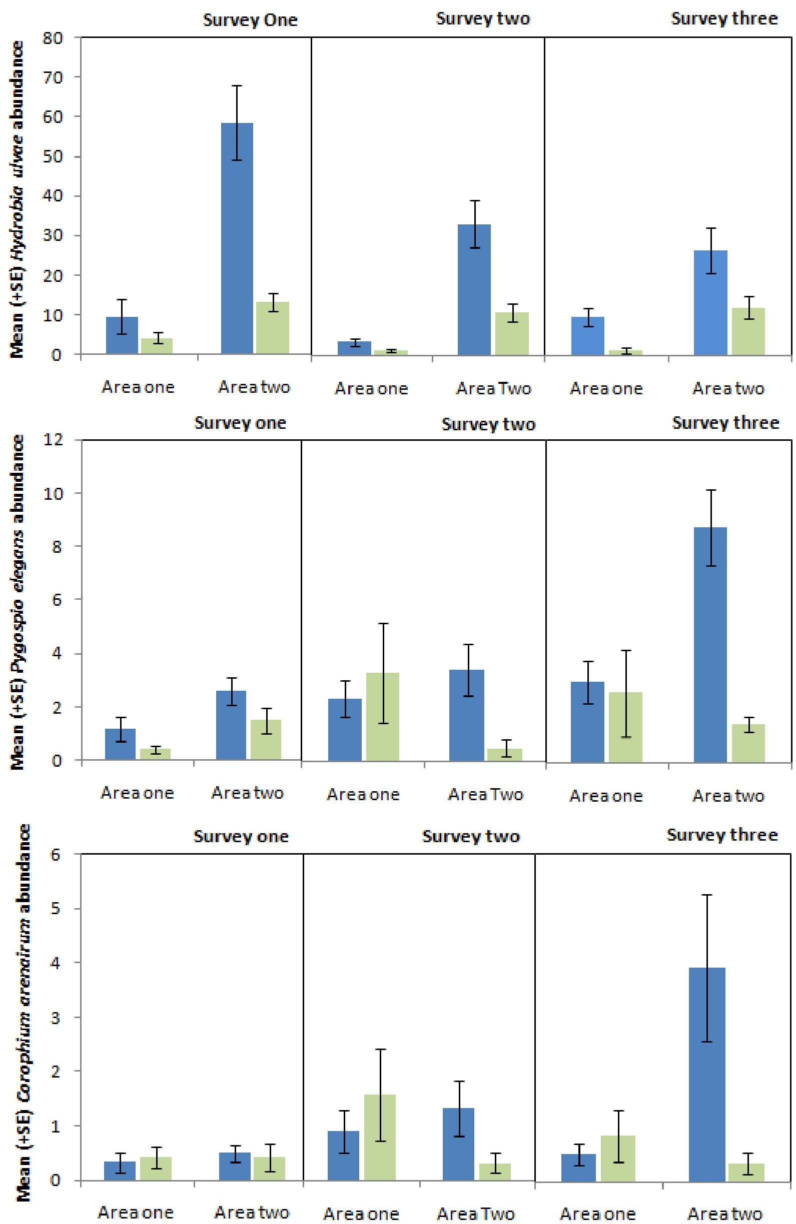



Figure 4. Mean abundance of key species for fished ( ) and control ( ) areas

The abundance of *P. elegans* was significantly different (P<0.05) between areas in all three surveys (Table 3). In survey one Mann-Whitney U tests established that the abundance in fished area two was significantly different (P<0.05) to fished area one and control area one (Appendix D). In survey two fished area one and fished area two were significantly different to control area two (Appendix D). Figure 4 shows this is due to an increase in abundance in fished area one and control area one, and a decrease in abundance in control area two. This is followed by an increase in abundance in fished area two samples from survey three, whereas the other area abundances have remained relatively the same (Figure 4). Mann-Whitney tests confirm this increased abundance in fished area two is significantly different to the other three areas (Appendix D). Between survey analysis also reveals fished area two is significantly different to the same area in survey one (P=0.001) and two (P=.008). Control area two is significantly different between survey one and two and between survey two and three. Figure 4 shows a decrease in abundance in this area from survey two and then an increase again in survey three.

The abundance of *C. arenarium* between areas was significantly different in survey three only (Table 3). Fished area two had a higher mean abundance compared to the other areas in survey three (Figure 4). Mann-Whitney U tests confirm that fished area two differs significantly to the other areas (Appendix E). Between survey analysis reveals that fished area two differs significantly between survey one and survey three (P=0.010).

SIMPER analysis showed that dominant species in the fished areas varied between surveys (Table 4). The control areas however were dominated by similar species. Differences in species contribution can be seen between fished and control areas.

| | | | |
|---|---|---|---|
| **Survey one** | | | |
| Fished | | Control | |
| Species | % contribution | Species | % contribution |
| *Hydrobia ulvae* | 70.31 | *Hydrobia ulvae* | 65.49 |
| *Cerastoderma edule* | 12.34 | *Cerastoderma edule* | 9.27 |
| *Cerastoderma edule juvenile* | 7.76 | *Pygospio elegans* | 6.24 |
| | | *Eteone sp.* | 4.96 |
| | | *Corophium arenarium* | 3.27 |
| | | *Bathyporeia pilosa* | 3.12 |
| **Survey two** | | | |
| Fished | | Control | |
| Species | % contribution | Species | % contribution |
| *Hydrobia ulvae* | 55.06 | *Hydrobia ulvae* | 69.17 |
| *Pygospio elegans* | 13.82 | *Bathyporeia pilosa* | 5.77 |
| *Cerastoderma edule* | 10.37 | *Pygospio elegans* | 5.59 |
| *Corophium arenarium* | 6.99 | *Eteone sp.* | 3.99 |
| *Cerastoderma edule juvenile* | 6.89 | *Corophium arenarium* | 3.19 |
| **Survey three** | | | |
| Fished | | Control | |
| Species | % contribution | Species | % contribution |
| *Hydrobia ulvae* | 46.54 | *Hydrobia ulvae* | 49.19 |
| *Pygospio elegans* | 24.94 | *Pygospio elegans* | 15.13 |
| *Cerastoderma edule* | 9.92 | *Bathyporeia pilosa* | 11.71 |
| *Eteone sp.* | 8.77 | *Eteone sp.* | 8.08 |

Table 4. Average similarity (SIMPER) between fished and control areas

MDS ordination (Figure 5) of macrofauna for the three surveys showed that macrofauna communities in control area 1 were clustered away from those found in the other areas. The fished areas and control area two became more dispersed from survey one to survey three.

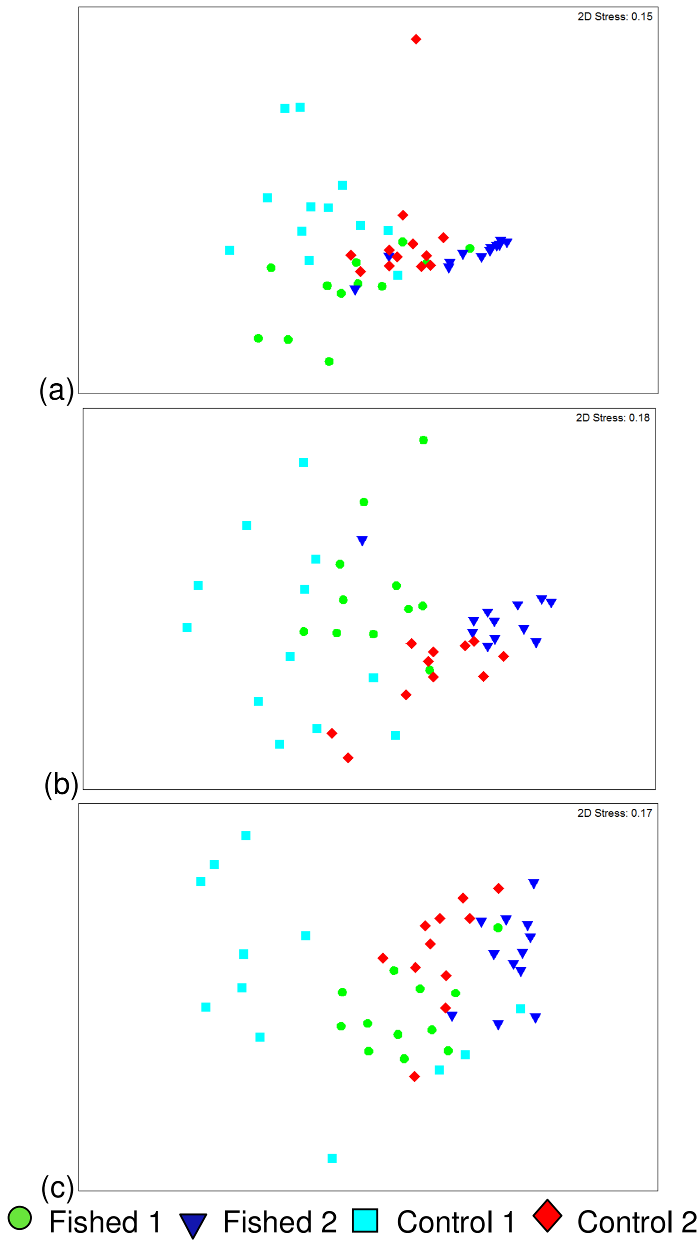



Figure 5. Multi-dimensional scaling configuration of macrofauna data from fished and control areas survey one (a) survey two (b) and survey three (c).

### 4.2 Cockle stocks

The cockle abundance varied significantly ($P<0.05$) in all three surveys (Table 5). In survey one cockle abundance was significantly higher in fished area two compared to the other three areas (Appendix F). Fished area two cockle abundance was also significantly higher ($P<0.05$) in survey two. In addition a significant difference between fished area one and control area one was shown (Appendix F). Abundances in fished area one increased slightly in survey two (Figure 6). Survey three compared the same to survey two, with observed increases in abundance for fished area one (Figure 6). There were no significant differences in areas between surveys ($P>0.05$).

| | Cockle abundance | Cockle width |
|---|---|---|
| Survey | P value | P value |
| 1 | **0.001** | **0.02** |
| 2 | **0.002** | **0.02** |
| 3 | **0.004** | 0.229 |

Table 5. P values from univariate analysis of variance of cockle abundance and Kruskall-Wallis analysis of cockle widths between areas

The cockle widths were significantly different between areas in survey one and two but not survey three (Table 5). Mann-Whitney U tests revealed that in both survey one and two, fished area one varied significantly to fished area two and control area two (Appendix G). Control area one was excluded from statistical tests due to very low numbers of cockles measured. Observed differences of the means in Figure 6 present a slightly higher abundance in fished area one compared to the other areas (excluding control area one). The significant differences may be due to the size of the samples; a larger quantity of cockles were measured in fished area two compared to fished area one. Fished area one and fished area two varied significantly ($P<0.05$) between survey one and survey three (Appendix H). This was the same for survey two. Control area two was not significantly different between surveys ($P=0.583$). There is an observed decrease in cockle width means in both fished areas between survey one and survey three (Figure 6).

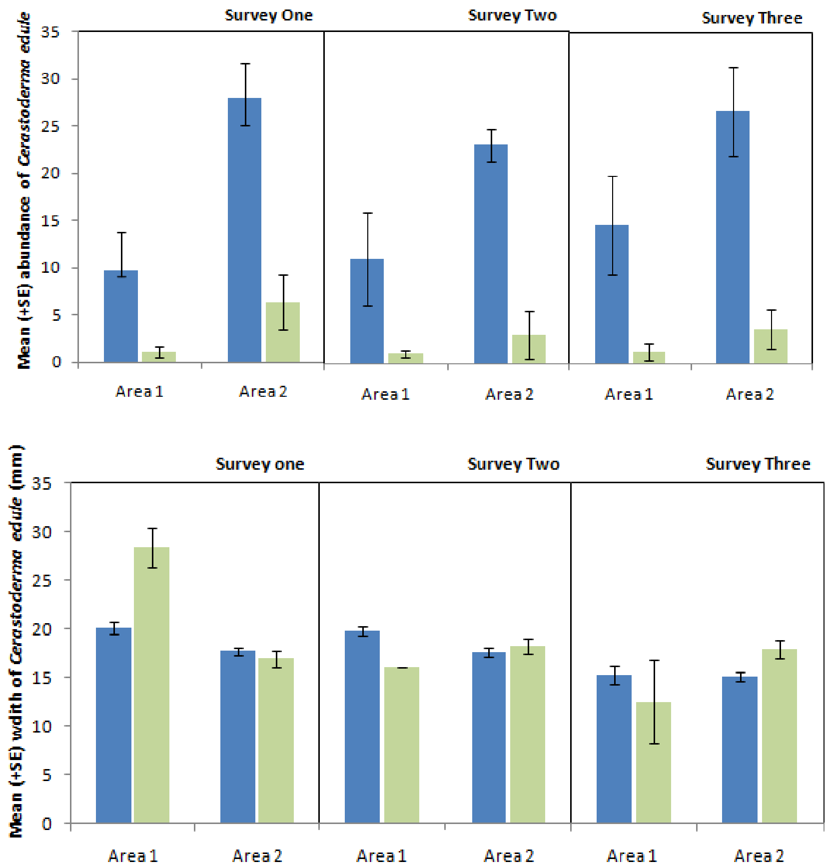
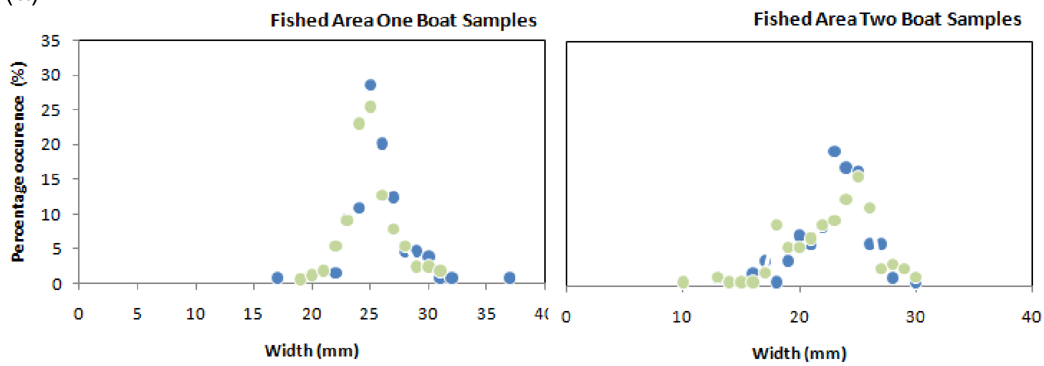



Figure 6. Mean width size (mm) and mean abundance of *C. edule* for fished ( ) and control areas ( )

Comparisons of cockle widths between intertidal samples and boat samples from survey one and survey three presented a significant difference ($P<0.05$) between fished areas (Appendix I). Boat sampling was not carried out during survey two due to time constraints. Cockle widths measured from the boat samples varied significantly between surveys in fished area one ($P=0.000$). The size range for intertidal samples were between 11mm-28mm and 5mm-29mm, from survey one and survey three respectively (Figure 7). The greatest percentage was for 18mm for survey one and 10mm for survey three. In comparison, the largest cockles measured from the boat samples peak at 26mm, ranging from 13mm-37mm, and 25mm, ranging from 10mm-31mm for survey one and survey three respectively.

(a)




(b)

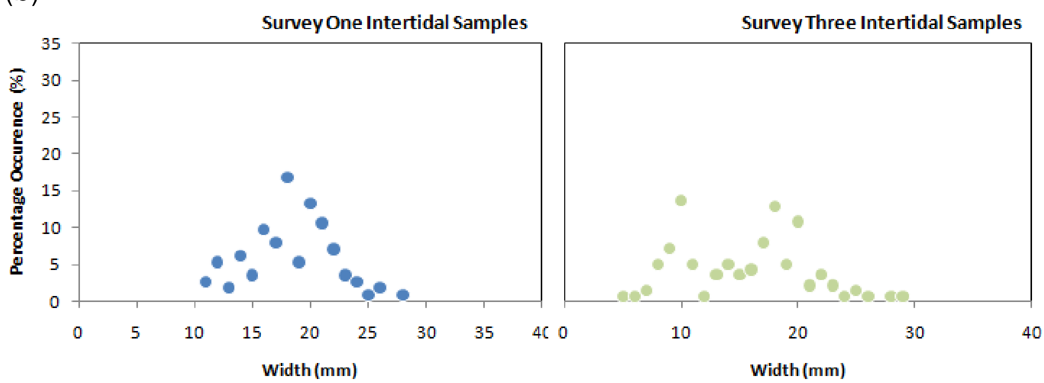



Figure 7. Size distribution graphs for cockle widths (mm) sampled from the eco-elevator harvester (a) and intertidally (both fished areas recorded together) (b). Survey one ( ) and survey three ( )

Figure 7(a) shows that the trend for samples from survey three shifted slightly towards smaller width sizes in fished area one compared to survey one. Figure 7 (b) demonstrates that the intertidal samples had a wider distribution of sizes and the majority were smaller than 20mm.

### 4.3 Sediment parameters

Particle size (phi) varied significantly between areas in all three surveys (P=0.000). In all three surveys fished area one was significantly different to fished area two and control area one (Appendix J). Also fished area two differed significantly to control area two. The mean particle sizes for both fished areas are higher than the control areas (Figure 8). In survey one the mean particle size is slightly higher in fished area one compared to fished area two. However this is reversed in survey two and three (Figure 8). All four areas in survey one are significantly different ($P<0.05$) to their equivalents in survey two and three. Survey two and three are not significantly different (P>0.05).

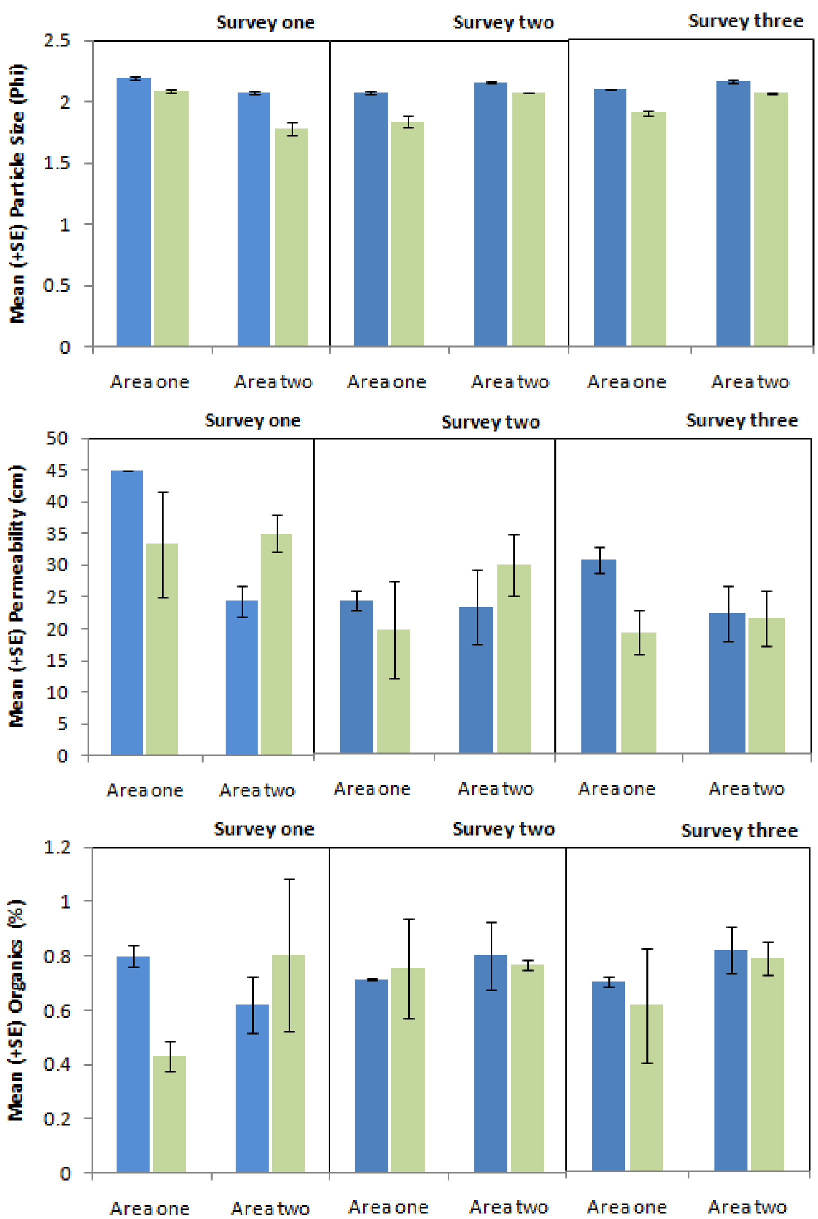



Figure 8. Sediment parameters for fished ( ) and control areas ( )

Sediment permeability did not differ significantly (P>0.05) between areas in all three surveys (Table 6). Between survey analysis however did reveal a significant difference in fished area one permeability between survey one and two, survey one and three and survey two and three (Appendix K).

Permeability is much higher in fished area one from survey one compared to the other two surveys (Figure 8).

| | Permeability | Organic content |
|---|---|---|
| Survey | P value | P value |
| 1 | 0.072 | 0.338 |
| 2 | 0.633 | 0.637 |
| 3 | 0.206 | 0.501 |

Table 6. P values from univariate analysis of variance of permeability (cm) and organic content (%) between areas

Sediment organic content did not differ significantly (P>0.05) between areas in all three surveys (Table 6). Also, between survey analysis did not reveal any significant differences (P>0.05).

The tracks left by the cockle harvester were also observed during survey three and ten days later. As can be seen in Figure 9 track marks were reduced after 10 days.

(a) (b)

Figure 9. Track marks left by the eco-elevator harvester a few hours after fishing (a) and 10 days later (b) (Photos V.Lee)

# CHAPTER 5
# DISCUSSION

The non-parametric Kruskal-Wallis test was used for macrofauna individual abundance, and abundance of the key species. It was also used for the analysis of cockle widths and particle (phi) size. These sets of data all had a significant P value from Kolmogrov-Smirnov tests indicating they were not normally distributed and therefore the non-parametric test was best to use. Although the organic content and permeability data were normally distributed, the homogeneity of variance assumption failed. Monte Carlo studies have shown that the one-way ANOVA is, to some extent, robust to small to moderate violations of the assumptions of the model, such as homogeneity of variance and normality of distribution (Kinnear & Gray, 2008). Therefore, univariate analysis of variance was used for organic content and permeability data, in addition to species number, species diversity and cockle abundance

Due to haphazard sampling adopted in the methods, there is the possibility that core samples were not all taken from areas directly fished by the eco-elevator harvester, especially as the boat does not harvest in parallel lines. The minimum sample spacing depends on the spatial heterogeneity of the intertidal area (Durell *et al.*, 2005); results obtained would be reflective of interactions occurring in the area. Using GPS coordinates meant that sample sites were easy to locate and will be easy to relocate in the future.

## 5.1 Macrofauna composition

Marine macrofauna are a widely used ecological indicator for impact surveys due to their high diversity of species with different tolerance to stress and the relative ease of sampling (Patricio *et al.*, 2009).

The analysis of macrofauna species number in this study indicated that the eco-elevator harvester did not have a significant effect. There was a significant difference in individual abundance between fished area two and the other areas in all three surveys; however the individual abundance was higher in this area. There were also no significant differences between surveys; therefore the eco-elevator harvester did not have a negative effect. The analysis of species diversity (H’) illustrated a significant difference between fished area one and control area one; however, between survey one and three the species diversity increased in both fished areas indicating the eco-elevator harvester was not having a negative impact. The lower diversity in the first two surveys may have been due to other factors such as bad weather conditions experienced over the winter. From observations of the means, fished area one is presented as having a lower species number and diversity to its control area throughout the three surveys. Although these differences are not significant this presents a caveat to possible future declines within the fished area compared to its control. This is particularly important as fished area one is favoured more by the fisherman due to overall higher abundances compared to fished area two, and therefore more time is spent harvesting in fished area one. Fished area two was only harvested ¾ time over the survey year (Clarke, 2010).

Comparisons of macrofauna composition between this study (Figure 3) and the baseline survey and initial survey after fishing commenced investigated by Hulme (2009) show similar if not higher observed means (Appendix L). This indicates the eco-elevator harvester is not causing any negative widespread changes. The fishing trial did not commence until late summer, allowing macrofauna to spawn and achieve recruitment. If fishing is allowed to continue during peak times of recruitment, i.e. summer of this year, there may be a negative effect on both target and non-target macrofauna composition.

Studies into other mechanical methods, such as hydraulic and tractor dredging, have resulted in a reduction in species number, diversity and individual abundance (Hall & Harding, 1997, Ferns *et al.*, 2000). The Exe estuary is exposed to frequent disturbance from naturally occurring currents and tidal forces, and from activities such as digging for bait and trampling. It could be possible therefore, that the macrofauna community in the Exe has adapted to regular disturbances, as suggested by Bolam *et al.* (2004) or is maintained in a permanently altered state prior to fishing due to these disturbances (Collie *et al.,* 2000). In addition, less stable intertidal sand habitats have been demonstrated to consist of macrofauna communities which are able to recover quickly from disturbance compared to stable muddy habitats (Gubbay & Knapman, 1999 & Ferns *et al.,* 2000). Medium and fine sands usually have an abundant macrofauna, but because muds contain more organic matter faunal densities are usually highest in these environments (Gray, 1981). This is supported by Durell et al. (2005), who found the highest abundances within the Exe were found in well-consolidated muddy sites with high organic content. The size of the fishing area may also explain the lack of impact by the eco-elevator harvester. Hall & Harding (1997) studied dredging in an area of 7 ha, which produced high levels of macrofauna mortality, however recovery was rapid. The fished areas on the Exe comprise of just 200m$^2$ in total, therefore species recovery should also be rapid and a dilution effect might be occurring as the disturbed area is small compared to the surrounding area (Gaspar *et al.,* 2003).

SIMPER analysis was used for identifying which species primarily account for observed differences in macrofauna composition between areas. It is also useful for identifying species typical of a specific environmental type. The findings of a higher number of dominant species in the control areas are consistent with species diversity; control areas had a higher species diversity compared to the fished areas. In particular the control areas had higher contributions of *B. pilosa* and *Eteone sp.*. An abundance of *B. pilosa* reflects unstable sediment characteristics and, being a mobile species is capable of undertaking mass movement during a single tide; therefore *B. pilosa* may have migrated from disturbed fished areas (Ferns *et al*. 2000). Further univariate analysis of the abundance of B. Pilosa would have been useful. However, analysis of macrofauna composition concluded the eco-elevator harvester did not have a significant impact. Seasonal migration into the water column by this species may lead to underestimated abundances when core sampling is used (Eleftheriou, 2000).

MDS ordination of macrofauna for the three surveys did not show a clear separation of sites which may reflect continuity of sediment characteristics. Control area one was particularly clustered away from the other areas. This correlates with the univariate analysis of low individual abundances in control area one, however species diversity was higher than all the other areas. The other areas became more dispersed from survey one to survey three suggesting the species composition was changing between surveys. This correlates with an increase in species diversity between surveys revealed by univariate analysis of variance.

Key species were examined to assess any impact at a species level. The decrease in *Hydrobia ulvae* abundance in fished area two from survey one to survey three was significant, however the abundance was still higher than in control area two ruling out disturbance by the eco-elevator harvester. In addition, fished area one abundance remained lower than in fished area two, but was still higher than the abundances in control area one. The high abundance in survey one is comparable to the initial survey carried out by Hulme (2009) after fishing commenced (Appendix M). This may be a result of the phytoplankton blooms occurring in autumn providing optimum conditions for *H. ulvae* recruitment (Ferns *et al.,* 2000). Reductions in abundance over the three surveys could be attributable to predation or such variables as tidal height and salinity. However, the distribution of *H. ulvae* has been difficult to explain in terms of any one variable (Walters, 1980). Single-site sampling has been questioned with regards to migrating macrofauna species as high-turnover rates have been recorded for mobile juvenile species (Armonies & Hartke, 1995).

*Pygospio elegans* abundance in fished area two was significantly different to the other three areas. However, as with the abundance of *H. ulvae,* the fished area abundance was higher than the control area, therefore the eco-elevator harvester did not have any impact on *P. elegans* abundance. *P. elegans* is an opportunistic species with the ability to quickly recolonise disturbed areas. They are important in stabilising the sediment by building tubes (Bolam & Fernandes, 2003) and those inhabiting intertidal sandflats will be more stable than sandlfats with relatively little macrofauna, and therefore the impact from fishing will be greater. However, this is not observed in this study. Increases in *P. elegans* correlates with an observed decrease in permeability. This does not correlate with studies demonstrating that dense arrays of tube-builders enhance sediment permeability (Bolam & Fernandes, 2003). If this does indeed occur and harvesting was to cause mortalities or relocations of such species the sediment characteristics would change. The increase in *P.elegans* also correlates with a decrease in cockle widths in survey three. Harvesting of larger cockles has been demonstrated by Mendonca *et al.* (2008) to enhance the survival of *P. elegans*. Competition, predation and sediment disturbance from burrowing by *C. edule* is reduced when cockles are removed by harvesting, resulting in increases in *P. elegans*. On the other hand, populations of opportunistic species are unstable and dense beds have been found to be replaced by subsequent colonisers (Bolam & Fernandes, 2003) which could account for the observed reductions in mean abundance in survey one and two compared to the surveys carried out by Hulme (2009) (Appendix M).

The high abundance of *C.arenarium* in fished area two from survey three was not significantly different to control area two and therefore it was not likely the eco-elevator had any impact. An increase in abundance of other macrofauna was found within *P. elegans* patches within a study by Bolam & Fernandes (2003). This is attributable to the polychaete tubes stabilising the sediment, and creating refuges against physiological stress and predators, which could explain the increased *C. arenarium* abundance correlating with the increased abundance of *P.elegans* in survey three.

As densities in control areas are not higher than those in fished areas, it would seem macrofauna are not affected by the disturbance and a redistribution from harvested to control areas is therefore not necessary.

### 5.2 Cockle stocks

The abundance of cockles was significantly higher in the fished area two compared to the other areas. This reflects on the preference of harvesting within the fished area one. Along with the observed increases in abundance in fished area one, this reveals the eco-elevator harvester had no impact on cockle abundance. This is also supported by observations of an increase in abundances in fished area two from the initial survey after fishing commenced by Hulme (2009) (Appendix N). In addition, abundances in fished area one remained constant. One explanation for this could be the finding that harvesting of larger cockles may enhance spat settlement of cockles as spat recruitment is inversely dependent on adult density (Stillman *et al*., 2001). Also Coffen-Smout and Rees (1999) indicate that suspension and reburial are part of the normal dynamics of cockle beds, therefore *C. edule* may be able to recover from harvesting disturbances. The cockle widths were significantly greater in fished area one compared to fished area two and control area two. This would not be expected as fished area one is fished more intensively and has lower abundances. The significant differences are due to spatial and temporal variation, rather than a result of fishing activity, which are high in estuarine environments (Ysebaert & Herman, 2002). There was also a significant decrease between surveys; however, the observed means between the two surveys carried out by Hulme (2009) are similar to this study (Appendix N). The eco-elevator harvester has not had an effect on cockle stocks.

Research into other mechanical harvesters has reported damages to cockle shells resulting in reduced burrowing behaviour and having a negative effect on cockle stocks (Cotter *et al*., 1997, Coffen-Smout & Rees, 1999, Mendonca *et al.,* 2008). Although a significant effect of harvesting did not occur, the eco-elevator harvester still caused damage to a few cockles as observed when measuring widths of boat samples. However the numbers were not large enough for it to be significant. The over-exploitation of cockle stocks by suction dredging within the Wadden Sea has been widely reported and highlights the importance of management of shellfish stocks. An 8-year long decline in bivalve stocks in this area led to the high mortality of migratory birds (Piersma *et al.,* 2001). Although there were no signifcant effects of the eco-elevator harvester on cockle stocks in this study, caution needs to be taken due to the importance of the Exe for migratory birds, especially as bird populations on the Exe already have difficulty in meeting their energy demands even in the absence of fishing (Stillman *et al.*, 2001).

Size distributions of boat samples revealed that the mesh on the elevator was not allowing some cockles smaller than 20mm to pass through onto the seabed. The smallest cockle was measured at 10mm from the boat samples. This is due to a fault with the mesh, which also has effects on cockle shell damage (Clarke, 2010). Even though the mesh will be expensive to refit, not allowing smaller cockles to rebury and re-establish themselves in the sediment could result in declines in cockle stocks.

There was a significant difference between cockle widths from boat samples and those from intertidal samples. The majority of intertidal samples were less than 25mm, indicating that smaller, younger cockles are present within the intertidal flats. There was a significant decrease in cockle widths from boat samples within fished area one between surveys suggesting the abundance of larger cockles is decreasing within this area. However, the intertidal width sizes were the highest in fished area one, although the mean size peaked at 20mm. Caution therefore is needed, especially if fished area one is to be harvested more intensively than fished area two in the future and annual recruitment of bivalves is characterised by year-to-year variability (Beukema & Dekker, 2005).

## 5. 3 Sediment composition

Sediment compostion is often considered to be the most important variable affecting macrofauna composition and is therefore critical in assessing disturbances to target and non-target macrofauna (Durell *et al.,* 2005).

The particle sizes (phi) in both fished areas were significantly higher than their control areas. However, from Figure 8 these significant differences are not very pronounced. Further statistical analysis is required between this study and the two surveys carried out by Hulme (2009) to conclude whether the eco-elevator harvester is impacting on sediment particle size. Certainly, increases in grain size can be seen from the baseline survey (Appendix O). Increased particle size means finer sediment is present within the fished areas and this fine sediment is favoured for settlement by small bivalves. Resuspension of this fine sediment and organic matter during dredging has been found to have a negative effect on settling bivalve larvae and macrofauna abundance (Piersma *et al*., 2001). Therefore the increase in sediment particle size would be a positive impact by the eco-elevator harvester as long as the mean particle size does not increase any further. A decrease in particle size, i.e. towards coarser sediment, may be an indication of reduction in cockle abundance as cockles add muddy sediment and increase organic matter by production of biodeposits (Beukema & Dekker, 2005).

The sediment found within the experimental area can be defined as fine-medium well-sorted sand. Well-sorted sediments are typical of high-energy areas, such as estuaries with high wave and current activity (Gray, 1981). Macrofauna in less consolidated sand habitats have been demonstrated to recover quicker from disturbance (Gubbay & Knapman, 1999, Collie *et al.,* 2000, Ferns *et al.,* 2000). Beukema & Dekker (2005) found that relationships of environmental conditions such as mud content with macrofauna abundance take a bell shaped (Gaussian) form. Moderate changes in sediment composition at extreme values, i.e. areas with very low mud content such as the experimental area, would cause significant effects on bivalve abundance. This is further evidence for the lack of impact of the eco-elevator harvester on sediment composition as there are no adverse effects on *C. edule* abundance.

Permeability of the sediment did not differ significantly between areas, however fished area one had a higher permeability compared to survey two, which then increased again in survey three. The high permeability in survey one was more likely to be due to weather or tidal conditions, with high rainfall and high winds at this time. Organic content also did not differ significantly between areas and surveys. Therefore fishing activity did not cause a change in organic content or permeability of the sediment.

Permeable sand facilitates bioirrigation of the sediment by tube burrowing macrofauna and reduces anoxic conditions to provide a good supply of oxygen needed to regulate the distribution of microbial and macrofuna communities (Volkenborn *et al.,* 2007). Tractor dredges have been shown to expose and disperse anoxic layers which have a negative impact on macrofauna burrowing (Ferns *et al.,* 2000). Fishing activity could reduce the organic content of sediment as seen by tractor dredges resuspending organic matter into the water column (Gubbay & Knapman, 1999). Also deposition of organic matter may be reduced as a result of tracks being left in the sediment by harvesters which could have a negative effect on macrofauna recruitment (Piersma *et al*., 2001). The physical disturbance of the sediment by the eco-elevator harvester was apparent from the tracks left after fishing activity (Figure 9a). Tractor dredges have also been reported to leave tracks for more than six months in stable sediments (Gubbay & Knapman, 1999). However, in the less consolidated sediment of the intertidal sandflat tracks were reduced after just 10 days (Figure 9b).

Loss of ignition (LOI) was used as a method to estimate organic content within the sediment samples. Although a commonly used and simple method, Heiri *et al*. (2001) indicate several factors that may influence results. LOI can be dependent on exposure time and for mixed sediments 4h was a reasonable time at 550$^{o}$C. LOI is also dependent on sample size and this should be kept the same throughout the experiment. Limitations of sediment analysis in impact surveys have been highlighted by Dernie *et al*. (2003). Core samples mask interactions which occur in the top surface layer of the sediment. Also macrofauna can cause stratification of sediments into different grain size distributions and sediment treated and analysed do not correlate well with this. In this study no chemicals were used for sediment analysis therefore natural aggregates such as faecal pellets were not broken down and so may be more representative of the natural environment the macrofauna encounters. Also diversity indices, such as the Shannon-Weiner diversity index used in this study, have been suggested as not being sensitive enough to handle some impacts on benthic diversity (Eleftheriou, 2000). Future studies would also include sediment chemistry and water quality measures to rule out factors such as eutrophication which could impact on macrofaunal communities.

Although other mechanical harvesters have resulted in significant effects on macrofauna composition, cockle stocks and sediment parameters, results of any single study are highly specific with respect to fishing gear, disturbance regime, habitat and environment (Collie *et al.* 2000).

# CHAPTER 6
# CONCLUSION

The present study did not find any significant impacts resulting from fishing by the eco-elevator harvester on macrofauna composition, *C. edule* abundance or size. No significant impacts from fishing on sediment composition were found either. Therefore the null hypothesis is accepted. Nevertheless, further statistical analysis is required between the three surveys within this study and with the baseline survey and intitial survey carried out by Hulme (2009) after fishing commenced. This will conclude whether the eco-elevator harvester has had any significant impacts throughout the twelve month trial fishery. The considerably smaller size of the fished areas compared to other studies on mechanical harvesters is a strong contributing factor to explain the lack of impact by the eco-elevator harvester. The intertidal habitat continues to provide favourable conditions after disturbance as indicated by the lack of significant reductions in macrofauna composition, cockle stocks and sediment parameters.

Further monitoring of cockle stocks and of non-target macrofauna is recommended, especially if the fished area one is to continue to be harvested more intensively. Monitoring will also aid the protection of migratory birds which overwinter on the Exe.

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# APPENDICES

## Appendix A

Mann-Whitney U tests for macrofauna individual abundances

Fished area one = 1, Fished area two = 2, Control area one = 3, Control area two = 4

Survey One

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 7.38 | 88.50 |
| | 2 | 12 | 17.63 | 211.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 10.500 |
| Wilcoxon W | 88.500 |
| Z | -3.553 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.63 | 211.50 |
| | 3 | 12 | 7.38 | 88.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 10.500 |
| Wilcoxon W | 88.500 |
| Z | -3.556 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

Ranks

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.17 | 206.00 |
| | 4 | 12 | 7.83 | 94.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 16.000 |
| Wilcoxon W | 94.000 |
| Z | -3.236 |
| Asymp. Sig. (2-tailed) | **.001** |
| Exact Sig. [2*(1-tailed Sig.)] | .001[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 6.96 | 83.50 |
| | 2 | 12 | 18.04 | 216.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 5.500 |
| Wilcoxon W | 83.500 |
| Z | -3.849 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

## Survey two

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.75 | 213.00 |
| | 3 | 12 | 7.25 | 87.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 9.000 |
| Wilcoxon W | 87.000 |
| Z | -3.638 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.67 | 212.00 |
| | 4 | 12 | 7.33 | 88.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 10.000 |
| Wilcoxon W | 88.000 |
| Z | -3.581 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

Survey three

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 6.71 | 80.50 |
| | 2 | 12 | 18.29 | 219.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 2.500 |
| Wilcoxon W | 80.500 |
| Z | -4.016 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.71 | 212.50 |
| | 3 | 12 | 7.29 | 87.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 9.500 |
| Wilcoxon W | 87.500 |
| Z | -3.616 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.92 | 215.00 |
| | 4 | 12 | 7.08 | 85.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 7.000 |
| Wilcoxon W | 85.000 |
| Z | -3.755 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

## Appendix B

Post hoc analysis for macrofauna species diversity between areas

Survey One

| (I) Area | (J) Area | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| **1** | **2** | .520300* | .2100259 | **.038** | .035979 | 1.004621 |
| | **3** | -.692200* | .2100259 | **.011** | -1.176521 | -.207879 |
| | 4 | .102033 | .2100259 | .640 | -.382287 | .586354 |
| **2** | 1 | -.520300* | .2100259 | .038 | -1.004621 | -.035979 |
| | **3** | -1.212500* | .2100259 | **.000** | -1.696821 | -.728179 |
| | 4 | -.418267 | .2100259 | .082 | -.902587 | .066054 |
| 3 | 1 | .692200* | .2100259 | .011 | .207879 | 1.176521 |
| | 2 | 1.212500* | .2100259 | .000 | .728179 | 1.696821 |
| | 4 | .794233* | .2100259 | **.005** | .309913 | 1.278554 |
| 4 | 1 | -.102033 | .2100259 | .640 | -.586354 | .382287 |
| | 2 | .418267 | .2100259 | .082 | -.066054 | .902587 |
| | 3 | -.794233* | .2100259 | .005 | -1.278554 | -.309913 |

Survey two

| (I) Area | (J) Area | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| 1 | 2 | .594300 | .3158406 | .097 | -.134030 | 1.322630 |
| | 3 | -.308667 | .3158406 | .357 | -1.036996 | .419663 |
| | 4 | .560067 | .3158406 | .114 | -.168263 | 1.288396 |
| **2** | 1 | -.594300 | .3158406 | .097 | -1.322630 | .134030 |
| | **3** | -.902967* | .3158406 | **.021** | -1.631296 | -.174637 |
| | 4 | -.034233 | .3158406 | .916 | -.762563 | .694096 |
| **3** | 1 | .308667 | .3158406 | .357 | -.419663 | 1.036996 |
| | 2 | .902967* | .3158406 | .021 | .174637 | 1.631296 |
| | **4** | .868733* | .3158406 | **.025** | .140404 | 1.597063 |
| 4 | 1 | -.560067 | .3158406 | .114 | -1.288396 | .168263 |
| | 2 | .034233 | .3158406 | .916 | -.694096 | .762563 |
| | 3 | -.868733* | .3158406 | .025 | -1.597063 | -.140404 |

## Appendix C
Mann-Whitney U tests for *Hydrobia ulvae* abundance

Survey one

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 7.46 | 89.50 |
| | 2 | 12 | 17.54 | 210.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 11.500 |
| Wilcoxon W | 89.500 |
| Z | -3.495 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 18.08 | 217.00 |
| | 3 | 12 | 6.92 | 83.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 5.000 |
| Wilcoxon W | 83.000 |
| Z | -3.875 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.08 | 205.00 |
| | 4 | 12 | 7.92 | 95.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 17.000 |
| Wilcoxon W | 95.000 |
| Z | -3.179 |
| Asymp. Sig. (2-tailed) | **.001** |
| Exact Sig. [2*(1-tailed Sig.)] | .001[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 3 | 12 | 8.46 | 101.50 |
| | 4 | 12 | 16.54 | 198.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 23.500 |
| Wilcoxon W | 101.500 |
| Z | -2.811 |
| Asymp. Sig. (2-tailed) | **.005** |
| Exact Sig. [2*(1-tailed Sig.)] | .004[a] |

Survey two

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 7.21 | 86.50 |
| | 2 | 12 | 17.79 | 213.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 8.500 |
| Wilcoxon W | 86.500 |
| Z | -3.677 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 8.67 | 104.00 |
| | 4 | 12 | 16.33 | 196.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 26.000 |
| Wilcoxon W | 104.000 |
| Z | -2.692 |
| Asymp. Sig. (2-tailed) | **.007** |
| Exact Sig. [2*(1-tailed Sig.)] | .007[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 18.00 | 216.00 |
| | 3 | 12 | 7.00 | 84.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 6.000 |
| Wilcoxon W | 84.000 |
| Z | -3.839 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 17.04 | 204.50 |
| | 4 | 12 | 7.96 | 95.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 17.500 |
| Wilcoxon W | 95.500 |
| Z | -3.155 |
| Asymp. Sig. (2-tailed) | **.002** |
| Exact Sig. [2*(1-tailed Sig.)] | .001[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 3 | 12 | 7.29 | 87.50 |
| | 4 | 12 | 17.71 | 212.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 9.500 |
| Wilcoxon W | 87.500 |
| Z | -3.645 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

Survey three

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 8.92 | 107.00 |
| | 2 | 12 | 16.08 | 193.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 29.000 |
| Wilcoxon W | 107.000 |
| Z | -2.489 |
| Asymp. Sig. (2-tailed) | **.013** |
| Exact Sig. [2*(1-tailed Sig.)] | .012[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 17.50 | 210.00 |
| | 3 | 12 | 7.50 | 90.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 12.000 |
| Wilcoxon W | 90.000 |
| Z | -3.515 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 18.25 | 219.00 |
| | 3 | 12 | 6.75 | 81.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 3.000 |
| Wilcoxon W | 81.000 |
| Z | -4.035 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 15.42 | 185.00 |
| | 4 | 12 | 9.58 | 115.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 37.000 |
| Wilcoxon W | 115.000 |
| Z | -2.023 |
| Asymp. Sig. (2-tailed) | **.043** |
| Exact Sig. [2*(1-tailed Sig.)] | .045[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 3 | 12 | 7.46 | 89.50 |
| | 4 | 12 | 17.54 | 210.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 11.500 |
| Wilcoxon W | 89.500 |
| Z | -3.544 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

## Appendix D

Mann-Whitney U tests for *P. elegans* abundance

Survey one

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 9.58 | 115.00 |
| | 2 | 12 | 15.42 | 185.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 37.000 |
| Wilcoxon W | 115.000 |
| Z | -2.069 |
| Asymp. Sig. (2-tailed) | **.039** |
| Exact Sig. [2*(1-tailed Sig.)] | .045[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 16.96 | 203.50 |
| | 3 | 12 | 8.04 | 96.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 18.500 |
| Wilcoxon W | 96.500 |
| Z | -3.240 |
| Asymp. Sig. (2-tailed) | **.001** |
| Exact Sig. [2*(1-tailed Sig.)] | .001[a] |

## Survey two

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 15.42 | 185.00 |
| | 4 | 12 | 9.58 | 115.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 37.000 |
| Wilcoxon W | 115.000 |
| Z | -2.209 |
| Asymp. Sig. (2-tailed) | **.027** |
| Exact Sig. [2*(1-tailed Sig.)] | .045[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 16.04 | 192.50 |
| | 4 | 12 | 8.96 | 107.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 29.500 |
| Wilcoxon W | 107.500 |
| Z | -2.627 |
| Asymp. Sig. (2-tailed) | **.009** |
| Exact Sig. [2*(1-tailed Sig.)] | .012[a] |

<u>Survey three</u>

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 8.38 | 100.50 |
| | 2 | 12 | 16.63 | 199.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 22.500 |
| Wilcoxon W | 100.500 |
| Z | -2.891 |
| Asymp. Sig. (2-tailed) | **.004** |
| Exact Sig. [2*(1-tailed Sig.)] | .003[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 16.92 | 203.00 |
| | 3 | 12 | 8.08 | 97.00 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 19.000 |
| Wilcoxon W | 97.000 |
| Z | -3.106 |
| Asymp. Sig. (2-tailed) | **.002** |
| Exact Sig. [2*(1-tailed Sig.)] | .001[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 18.13 | 217.50 |
| | 4 | 12 | 6.88 | 82.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 4.500 |
| Wilcoxon W | 82.500 |
| Z | -3.943 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Appendix E**

Mann-Whitney U tests for *C. arenarium* abundance

## Survey three

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 1 | 12 | 8.96 | 107.50 |
| | 2 | 12 | 16.04 | 192.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 29.500 |
| Wilcoxon W | 107.500 |
| Z | -2.571 |
| Asymp. Sig. (2-tailed) | **.010** |
| Exact Sig. [2*(1-tailed Sig.)] | .012[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 15.88 | 190.50 |
| | 3 | 12 | 9.13 | 109.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 31.500 |
| Wilcoxon W | 109.500 |
| Z | -2.467 |
| Asymp. Sig. (2-tailed) | **.014** |
| Exact Sig. [2*(1-tailed Sig.)] | .017[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Abundance | 2 | 12 | 16.54 | 198.50 |
| | 4 | 12 | 8.46 | 101.50 |
| | Total | 24 | | |

| | Abundance |
|---|---|
| Mann-Whitney U | 23.500 |
| Wilcoxon W | 101.500 |
| Z | -2.970 |
| Asymp. Sig. (2-tailed) | **.003** |
| Exact Sig. [2*(1-tailed Sig.)] | .004[a] |

## Appendix F

Post hoc analysis for cockle abundance between areas

Survey one

| (I) Area | (J) Area | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| **1** | **2** | -18.33* | 4.391 | **.003** | -28.46 | -8.21 |
| | 3 | 8.67 | 4.391 | .084 | -1.46 | 18.79 |
| | 4 | 3.33 | 4.391 | .470 | -6.79 | 13.46 |
| **2** | **1** | 18.33* | 4.391 | **.003** | 8.21 | 28.46 |
| | **3** | 27.00* | 4.391 | **.000** | 16.88 | 37.12 |
| | 4 | 21.67* | 4.391 | .001 | 11.54 | 31.79 |
| **3** | 1 | -8.67 | 4.391 | .084 | -18.79 | 1.46 |
| | **2** | -27.00* | 4.391 | **.000** | -37.12 | -16.88 |
| | 4 | -5.33 | 4.391 | .259 | -15.46 | 4.79 |
| **4** | 1 | -3.33 | 4.391 | .470 | -13.46 | 6.79 |
| | **2** | -21.67* | 4.391 | **.001** | -31.79 | -11.54 |
| | 3 | 5.33 | 4.391 | .259 | -4.79 | 15.46 |

Survey two

| (I) Area | (J) Area | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| **1** | **2** | -12.00* | 4.110 | **.019** | -21.48 | -2.52 |
| | **3** | 10.67* | 4.110 | **.032** | 1.19 | 20.14 |
| | 4 | 8.00 | 4.110 | .087 | -1.48 | 17.48 |
| **2** | 1 | 12.00* | 4.110 | .019 | 2.52 | 21.48 |
| | **3** | 22.67* | 4.110 | **.001** | 13.19 | 32.14 |
| | **4** | 20.00* | 4.110 | **.001** | 10.52 | 29.48 |
| 3 | 1 | -10.67* | 4.110 | .032 | -20.14 | -1.19 |
| | 2 | -22.67* | 4.110 | .001 | -32.14 | -13.19 |
| | 4 | -2.67 | 4.110 | .535 | -12.14 | 6.81 |
| 4 | 1 | -8.00 | 4.110 | .087 | -17.48 | 1.48 |
| | 2 | -20.00* | 4.110 | .001 | -29.48 | -10.52 |
| | 3 | 2.67 | 4.110 | .535 | -6.81 | 12.14 |

Survey three

| (I) Area | (J) Area | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| **1** | **2** | -12.00* | 5.185 | **.049** | -23.96 | -.04 |
| | **3** | 13.33* | 5.185 | **.033** | 1.38 | 25.29 |
| | 4 | 11.00 | 5.185 | .067 | -.96 | 22.96 |
| **2** | **1** | 12.00* | 5.185 | **.049** | .04 | 23.96 |
| | **3** | 25.33* | 5.185 | **.001** | 13.38 | 37.29 |
| | **4** | 23.00* | 5.185 | **.002** | 11.04 | 34.96 |
| **3** | **1** | -13.33* | 5.185 | **.033** | -25.29 | -1.38 |
| | **2** | -25.33* | 5.185 | **.001** | -37.29 | -13.38 |
| | 4 | -2.33 | 5.185 | .665 | -14.29 | 9.62 |
| 4 | 1 | -11.00 | 5.185 | .067 | -22.96 | .96 |
| | 2 | -23.00* | 5.185 | .002 | -34.96 | -11.04 |
| | 3 | 2.33 | 5.185 | .665 | -9.62 | 14.29 |

## Appendix G

Mann-Whitney U tests for cockle widths between areas

Survey one

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Width | 1 | 29 | 73.57 | 2133.50 |
| | 2 | 84 | 51.28 | 4307.50 |
| | Total | 113 | | |

| | Width |
|---|---|
| Mann-Whitney U | 737.500 |
| Wilcoxon W | 4307.500 |
| Z | -3.176 |
| Asymp. Sig. (2-tailed) | .001 |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Width | 1 | 29 | 29.29 | 849.50 |
| | 4 | 19 | 17.18 | 326.50 |
| | Total | 48 | | |

| | Width |
|---|---|
| Mann-Whitney U | 136.500 |
| Wilcoxon W | 326.500 |
| Z | -2.950 |
| Asymp. Sig. (2-tailed) | .003 |

Survey two

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Width | 1 | 33 | 65.86 | 2173.50 |
| | 2 | 69 | 44.63 | 3079.50 |
| | Total | 102 | | |

| | Width |
|---|---|
| Mann-Whitney U | 664.500 |
| Wilcoxon W | 3079.500 |
| Z | -3.416 |
| Asymp. Sig. (2-tailed) | .001 |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Width | 1 | 33 | 23.42 | 773.00 |
| | 4 | 9 | 14.44 | 130.00 |
| | Total | 42 | | |

| | Width |
|---|---|
| Mann-Whitney U | 85.000 |
| Wilcoxon W | 130.000 |
| Z | -1.972 |
| Asymp. Sig. (2-tailed) | .049 |
| Exact Sig. [2*(1-tailed Sig.)] | .052[a] |

## Appendix H
Mann-Whitney U tests for cockle widths between surveys

### Fished area one

**Ranks**

| | Survey | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Widths | 1 | 84 | 95.51 | 8023.00 |
| | 3 | 80 | 68.84 | 5507.00 |
| | Total | 164 | | |

| | Widths |
|---|---|
| Mann-Whitney U | 2267.000 |
| Wilcoxon W | 5507.000 |
| Z | -3.615 |
| Asymp. Sig. (2-tailed) | **.000** |

**Ranks**

| | Survey | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Widths | 2 | 69 | 87.66 | 6048.50 |
| | 3 | 80 | 64.08 | 5126.50 |
| | Total | 149 | | |

| | Widths |
|---|---|
| Mann-Whitney U | 1886.500 |
| Wilcoxon W | 5126.500 |
| Z | -3.340 |
| Asymp. Sig. (2-tailed) | **.001** |

Fished area two

**Ranks**

| | Survey | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Widths | 1 | 84 | 95.51 | 8023.00 |
| | 3 | 80 | 68.84 | 5507.00 |
| | Total | 164 | | |

| | Widths |
|---|---|
| Mann-Whitney U | 2267.000 |
| Wilcoxon W | 5507.000 |
| Z | -3.615 |
| Asymp. Sig. (2-tailed) | **.000** |

**Ranks**

| | Survey | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Widths | 2 | 69 | 87.66 | 6048.50 |
| | 3 | 80 | 64.08 | 5126.50 |
| | Total | 149 | | |

| | Widths |
|---|---|
| Mann-Whitney U | 1886.500 |
| Wilcoxon W | 5126.500 |
| Z | -3.340 |
| Asymp. Sig. (2-tailed) | **.001** |

## Appendix I

Kruskal-Wallis tests for boat and intertidal cockle widths between areas

### Fished area one

**Ranks**

| | Activity | N | Mean Rank |
|---|---|---|---|
| Widths | 1 | 29 | 23.41 |
| | 2 | 129 | 92.11 |
| | Total | 158 | |

| | Widths |
|---|---|
| Chi-Square | 54.507 |
| df | 1 |
| Asymp. Sig. | **.000** |

### Fished area two

**Ranks**

| | Activity | N | Mean Rank |
|---|---|---|---|
| Widths | 1 | 84 | 62.58 |
| | 2 | 165 | 156.78 |
| | Total | 249 | |

| | Widths |
|---|---|
| Chi-Square | 95.973 |
| df | 1 |
| Asymp. Sig. | **.000** |

## Appendix J
Mann-Whitney U tests for particle size between areas

Survey one

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 1 | 15 | 22.37 | 335.50 |
| | 2 | 15 | 8.63 | 129.50 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | 9.500 |
| Wilcoxon W | 129.500 |
| Z | -4.289 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 1 | 15 | 22.00 | 330.00 |
| | 3 | 15 | 9.00 | 135.00 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | 15.000 |
| Wilcoxon W | 135.000 |
| Z | -4.054 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 1 | 15 | 22.93 | 344.00 |
| | 4 | 15 | 8.07 | 121.00 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | 1.000 |
| Wilcoxon W | 121.000 |
| Z | -4.630 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

Survey two

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 1 | 15 | 8.13 | 122.00 |
| | 2 | 15 | 22.87 | 343.00 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | 2.000 |
| Wilcoxon W | 122.000 |
| Z | -4.612 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 1 | 15 | 22.50 | 337.50 |
| | 3 | 15 | 8.50 | 127.50 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | 7.500 |
| Wilcoxon W | 127.500 |
| Z | -4.378 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Ranks**

| | Area | N | Mean Rank | Sum of Ranks |
|---|---|---|---|---|
| Phi | 2 | 15 | 23.00 | 345.00 |
| | 4 | 15 | 8.00 | 120.00 |
| | Total | 30 | | |

| | Phi |
|---|---|
| Mann-Whitney U | .000 |
| Wilcoxon W | 120.000 |
| Z | -4.701 |
| Asymp. Sig. (2-tailed) | **.000** |
| Exact Sig. [2*(1-tailed Sig.)] | .000[a] |

**Appendix K**

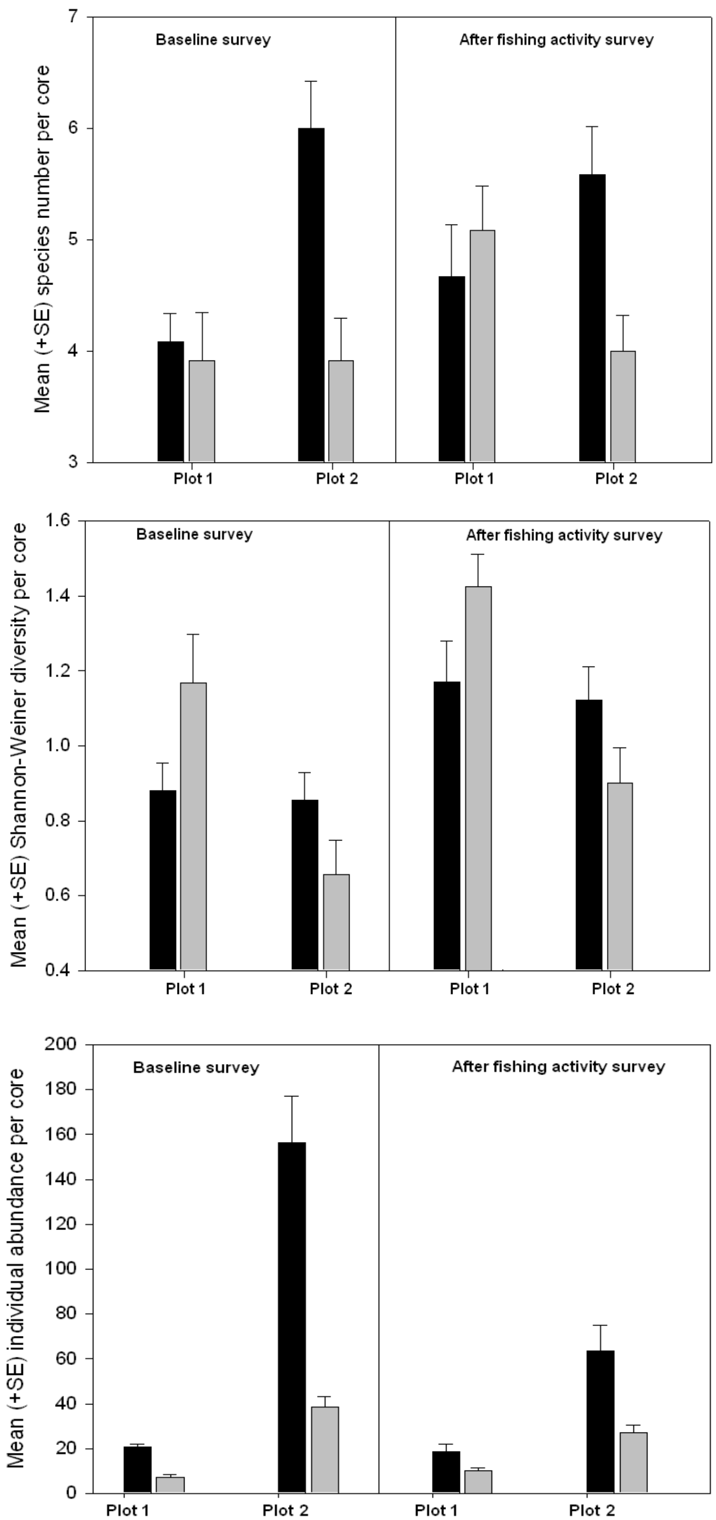
Post hoc analysis of sediment permeability between surveys

Fished area one

| (I) Survey | (J) Survey | Mean Difference (I-J) | Std. Error | Sig. | 95% Confidence Interval Lower Bound | Upper Bound |
|---|---|---|---|---|---|---|
| **1** | **2** | 20.67* | 2.073 | **.000** | 15.59 | 25.74 |
| | **3** | 14.00* | 2.073 | **.001** | 8.93 | 19.07 |
| **2** | 1 | -20.67* | 2.073 | .000 | -25.74 | -15.59 |
| | **3** | -6.67* | 2.073 | **.018** | -11.74 | -1.59 |
| 3 | 1 | -14.00* | 2.073 | .001 | -19.07 | -8.93 |
| | 2 | 6.67* | 2.073 | .018 | 1.59 | 11.74 |

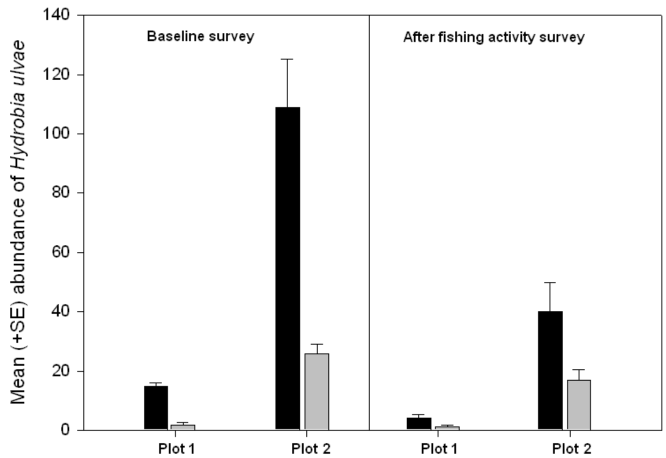
**Appendix L**

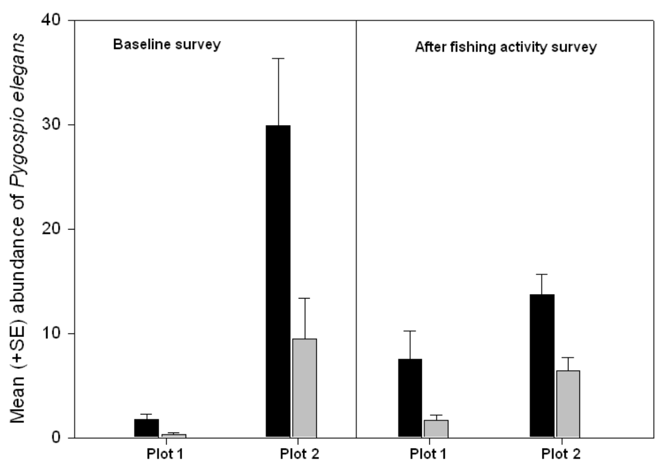
Mean species number, Shannon-Weiner diversity (H’) and individual abundance for ( ) control and ( ) treatment plots (Hulme, 2009)

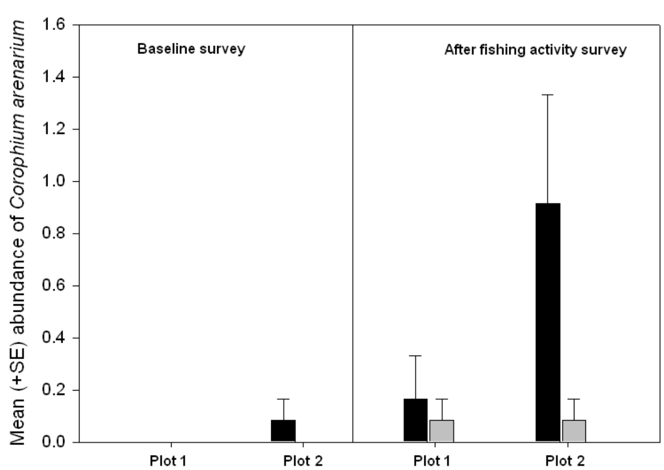


## APPENDIX M

Mean abundance of key species for ( ) control and ( ) treatment plots (Hulme, 2009)

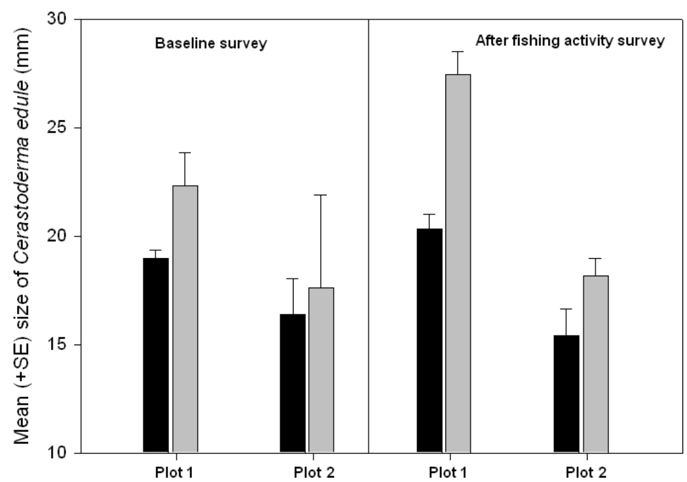


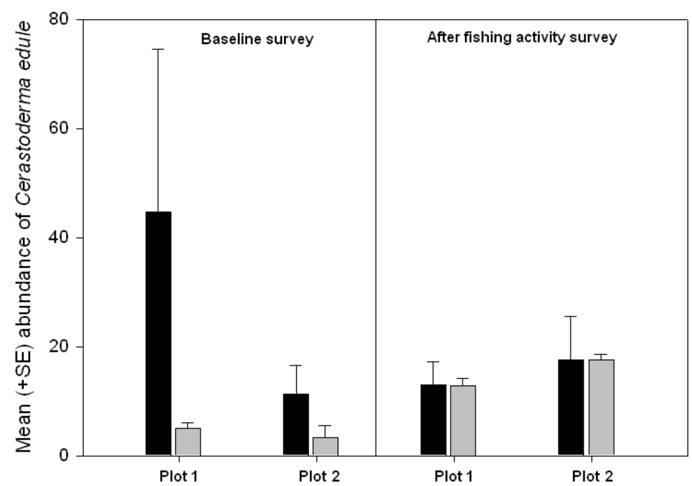



**APPENDIX N**

Figure 7. Mean size (mm) and mean abundance of C.edule for ( ) control and ( ) treatment plots (Hulme, 2009)



## APPENDIX O

Figure 3. Environmental parameters (mean +1SD) for different treatment areas ( ) treament and ( )control (Hulme, 2009)