THE EFFECTS OF AN ECO-ELEVATOR COCKLE HARVESTER ON MACROFAUNA ASSEMBLAGE, COCKLE POPULATIONS AND SEDIMENT PARAMETERS WITHIN AN INTERTIDAL SAND FLAT.

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Executive Summary

Previous methods of mechanical harvesting for the bivalve *Cerastoderma edule* have caused disturbance effects to macrofauna assemblage composition, abundance and diversity, as well as negative sediment changes. This study investigated the effects of a new type of mechanical fishery method for *C.edule*, an eco-elevator harvester, to assess the effects to target and non-target macrofauna and sediment changes in the Exe Estuary, UK. Two fished and two control plots were defined on the intertidal sand bank with macrofauna and sediment samples collected prior to fishing activity, to assess the baseline. Samples were taken in an identical manner 34 days, 139, 227 and 286 after fishing activity had commenced. Significant differences were observed for, Phi size and percentage organics during the survey, however these were not attributable to fishing activity. No significant differences were seen for permeability of sediment. Macrofauna assemblage, abundance and diversity were not adversely affected by fishing activity throughout the survey although significant differences were seen. *C.edule* size and abundance were not significantly different as an effect of fishing. The activity of the eco-elevator harvester did not cause a significant negative impact to macrofauna, target species or environmental parameters in the study time period. However due to the importance of the Exe Estuary, it is recommended that intensity and size of area fished should remain the same.

Introduction

Physical disturbances to sediment both natural and mechanical and the effects thereafter on benthic community structures have been widely investigated (Hall 1994, Hall & Harding 1997). Natural physical disturbances such as storms have been shown to be vital for the regulation of an ecosystem and a controlling factor in spatial and temporal composition for intertidal and soft sediment habitats (Connell 1978, Probert 1984, Hall 1994). There has however, been increasing conflict between human activity and ecosystem sustainability of estuaries in the UK within the last 40 years, resulting from fishing pressure exerted by mechanical equipment (Hall & Harding 1997 & Eleftheriou 2000).

Conflicts of resource have previously led to exploitation of estuarine and intertidal habitats with severe ecological consequences as exhibited by the Dutch Wadden Sea fishery collapse in the early 1990's (Swart & Andel 2008). Intensive suction dredge fishing caused the collapse of the cockle *Cerastoderma edule* and mussel *Mytilus edulis* stocks resulting in high mortality of migratory bird populations which relied on the shellfish as an over wintering food source (Camphuysen et al. 1996). The collapse in Holland increased the pressure and subsequent intensity on UK cockle fisheries in order to supply the European market (Hall et al. 1997). Increased pressure on the cockle fishing industry resulted in mechanical methods replacing traditional hand collection to meet demand. However, many of the mechanical methods used resulted in the reduction in target fauna abundance, which caused the mortality of birds, reduction in associated macrofauna abundance and diversity, as well as changes in the sediment such as grain size and topography. As a result of increased fishing effort, and ecological concerns, widespread research into the effects of mechanical harvesting has been carried out. (Spencer et al 1998).

Previous research into mechanical harvesting has focused on two main areas: the impact on bird populations and the reduced abundance and diversity in target and non target species. Atkinson et al. (2003) showed through population models that changes in Oystercatcher populations directly correlated with a low abundance of available food sources of cockles and mussels as a result of suction dredging. Interference competition also intensifies in Oystercatchers when cockle beds are reduced which is attributable to shellfishing activity (Goss-Custard et al. 2004).

The impact of mechanical harvesting upon the benthic assemblage and target species varies with gear used, but there is also wide variation between studies. Impact surveys widely use marine macrofauna as an ecological indicator due to the high diversity of phyla, the ease of sampling and the potential to indicate stress or disturbance, as well as the intrinsic relationship with sediment (Salas et al. 2004, Patrício et al. 2009). Suction dredging can cause significant negative impact on cockle abundance, non-target fauna abundance, diversity, and assemblage and sediment parameters such as "scarring" of the sediment (Hall & Harding 1997, Kraan et al. 2007, Piersma et al. 2001). Post-suction dredging recovery time for the macrofauna abundance and diversity varies with area. Hall & Harding (1997) observed recovery of macrofauna assemblage after 56 days whilst Hinndink (2003) did not find macrofauna assemblage recovery until 1 year after dredging activity. The time taken for recovery of the target species was considerably longer. Piersma et al., (2001) stated that it took 8 years for cockle stocks to recover in the Dutch Wadden Sea. Tractor dredging in some cases has been shown to result in >100 days recovery time for *Pygospio elegans* and *Hydrobia ulvae* (Ferns et al. 2000). Halls & Harding (1997), however, noted that tractor dredging activity had little effect on macrofauna assemblage and abundance, although seasonal variation and recruitment were influencing factors. Detailed knowledge of the variability and influence of factors (e.g. seasonality) needs to be understood in order to successfully manage estuaries.

Commercial quantities of the edible cockle *Cerastoderma edule*, inhabit the large intertidal sediments of the Exe Estuary, South West England. However the area is also a SSSI, Special Protection Area and a RAMSAR site of wetland importance. Conservation priority includes the protection of the overwintering migratory birds which comprise of Brent Geese, *Branta bernicula*, and oystercatchers, *Haematopus ostralegus*. Both species have suffered unexpected high mortality rates over a 10 year period from winter 1990 until winter 2000 (Goss-custard, 2007). Over-fishing has been suggested as a possible contributing factor; therefore the environmental impacts of cockle fishing activity need to be assessed. Despite widespread investigation into the effects of mechanical cockle harvesters, little has been conducted into the effect of the eco-elevator harvester.

The eco-elevator harvester was developed by John Bayes (Seasalter Shellfish) and Gary Wordsworth (Othniel Shellfish) to allow for cockles to be exported and sold alive as well as in response to increasing environmental concern (Howard 1999). The elevator harvester operates during high tide attached to an adapted dredge boat. The system differs from other mechanical harvesting techniques in that it lifts the cockles up from the sediment bed with jets of water onto an elevator chain. The elevator chain has an average diameter of 20 mm (see Figure 1 a and c) which allows for any undersized cockles to drop straight back down onto the seabed. Hydraulic dredges on the other hand operate by fluidising the sediment with jets of water in front of a cutting blade, sediment is sifted through a grid and cockles are retrieved from a suction pipe into a revolving drum. Sediment and small cockles pass through screens and return to the seabed (Hall & Harding 1997). Research by Coffen-Smout and Rees (1999) into hydraulic dredges showed that reburial was more effective when harvesting took place at high tides. Tractor dredges, unlike hydraulic dredges, operate during low water; the dredge is pulled along by the tractor with a blade which “skims” the sediment. Cockles and sediment are transported onto a conveyor into a rotating drum where sediment and small cockles pass through creating ridges of sediment in the vehicle tracks (Cotter et al. 1997, Ferns et al. 2000). The shells of small cockles can be damaged when removing larger cockles by the rotation of the drum used in both hydraulic and tractor dredges (Coffen-Smout 1998). Damaged cockles left to decompose may temporarily cause unfavourable macrofauna conditions by contributing to anoxic conditions in the sediment (Mendonca et al., 2008).

(a) (b) (c)

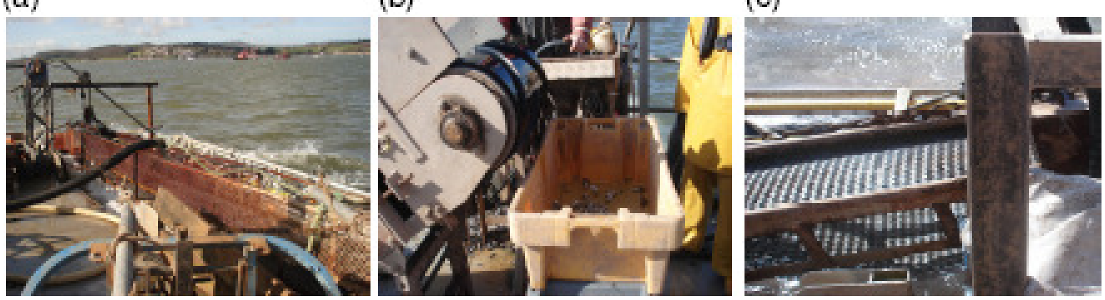


Figure 1: (a) raised eco-elevator harvester; (b) cockles collected in fish box; (c) mesh of elevator chain. (Photos by V.Lee)

The eco-elevator harvester system uses only 10% of the power used by conventional hydraulic dredge, small cockles and non target species can pass through the chain directly back into the same path (Howard 1999). There is currently no UK based minimum landing size (MLS) for cockles, although individual Sea Fishery Committees can place a MLS via bylaws. Devon Sea Fisheries Committee does not have a set MLS:-, the size collected is dictated by commercial viability and market demands generally >24mm (Robbins 2009, Blood-Smyth 2009 Personal communication).

Further investigation into this fishing method is needed to identify any ecological implications related to its application upon estuarine sand flats. This gap in knowledge and the importance of the Exe estuary in terms of conservation has led to Natural England and Devon Sea Fisheries Committee commissioning a 12 month study.

Aims

The aim of this study was to determine a baseline level of macrofauna, sediment, cockle size and abundance for the intertidal sand flats on the Exe estuary, and to examine if the eco-elevator harvester caused significant impact upon:

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

Hypothesis

H_0: The eco-elevator harvester will not cause any significant effect on macrofauna composition, cockle stocks or sediment parameters as seen by other types of mechanical cockle harvesters.

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

Methodology

Study Site: The study was conducted at Cockle Sands on the Exe Estuary which lies North West of Exmouth (50^0 37.2' N and 50^0 38' N.) (Figure 3). It is a Site of Specific Scientific interest (SSSI), Special Protection Area (under the EU bird directive) and RAMSAR site of Wetland importance. The Exe supports large *Mytilus edulis* beds and *Zostera nolti* beds, as well as dense *Cerastoderma edule* beds. The study site had previously undergone limited fishing activity for cockles by the eco-elevator harvester (20 minutes twice a week during spring tide); however this stopped 12 months before sampling was carried out. Sporadic hand raking for cockles was still carried out during this time, however; the site is also dug by anglers for bait and used by crab tilers (Sheehan et al 2008).

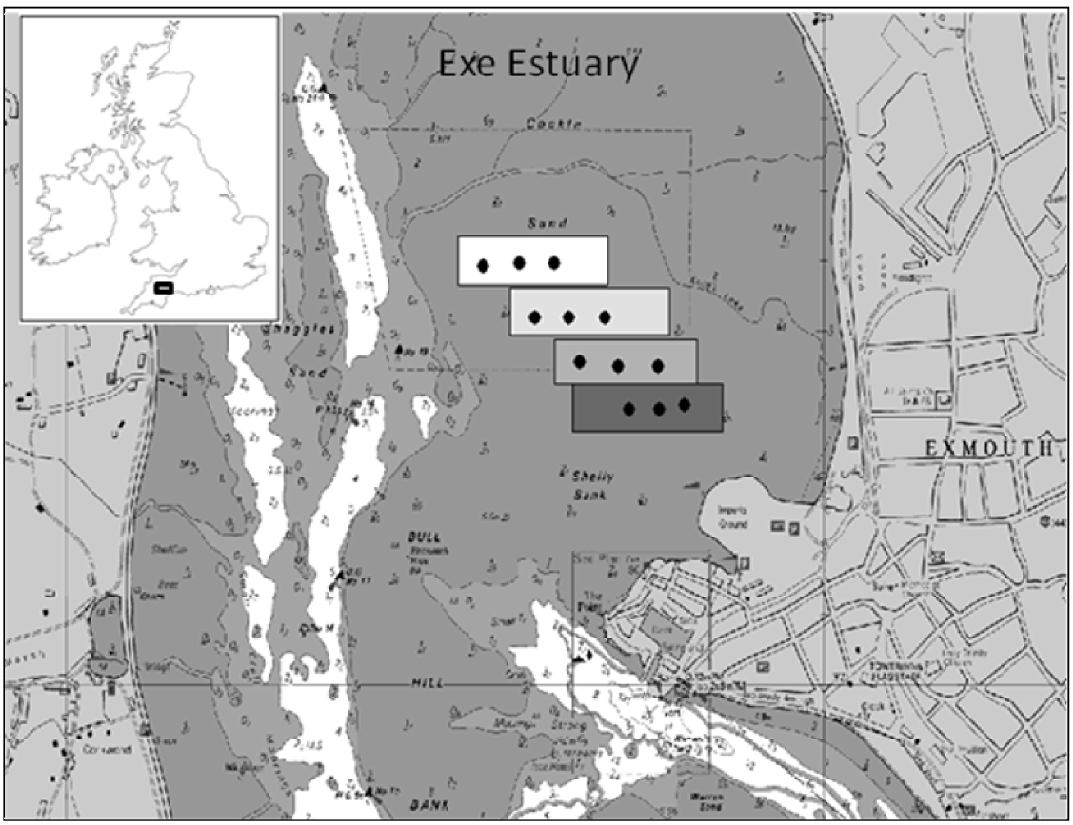


Figure 2. Map of Exe Estuary (Devon, UK image courtesy of Hydrographical office) showing plots where baseline samples were taken. (Not to scale) =control plot 1 = fished plot 1 =control plot 2 = fished plot 2. = Sampling sites for baseline survey

Experiment 1: effect of fishing upon macrofauna assemblage composition.

The aim of experiment 1 was to carry out a baseline to identify and quantify the intertidal macrofauna assemblage at the experimental site prior to fishing commencing and, from the baseline, examine any effects there-after as a result of the cockle fishing activity. Before the sampling took place two 100 m^2 treatment (fished) plots were designated with two 100 m^2 control plots located before the first treatment plot and between the treatment plots. The plots were located by using a bearing towards landmarks and with a portable global positioning system (GPS) which is accurate to ±10 m. Three sites within each plot were selected haphazardly by walking 100 m along a line transect following the bearing. The first bearing was 310^o from a yellow marker post to the flag pole of Powderdam Castle, with treatment plot 1 along bearing 273^o towards Brunel Tower Starcross and treatment plot 2 along the bearing of 133^o. 175 m were taken between the treatment plots and control plots.

The macrofauna assemblage was sampled on 26th May, 23 days prior to fishing commencing, using a 10 cm diameter corer with a volume of 100 cm^3. Four replicates were taken haphazardly from three sites within each plot. Fishing commenced at the treatment plots on 19th June using the eco-elevator harvester mounted to the side of the Alibi E516, a 10 m French oyster barge, capable of entering the shallower waters of the estuary. The treatment plots were fished for 20 minute intervals, twice a week on spring tides where possible; treatment areas were located by the fisherman using GPS. Samples were taken in an identical manner at each site on 23rd July, 5th November, 1st February and 1st of April (34, 139, 227 and 286 days after fishing had commenced).

Plots	Coordinates
Control Plot 1	50 37.588 003 25.708
Treatment Plot 1	50 37.533 003 25.605
Control Plot 2	50 37.475 003 25.499
Treatment Plot 2	50 37.404 003 25.369

Table 1: GPS coordinates (latitude and longitude) for control and treatment plots.

Macrofauna samples were sieved onsite where possible through a 1 mm mesh before being preserved in 70%. Identification to species or genus levels, where possible, was carried out under a low power dissection microscope using the appropriate dichotomous key (Hayward & Ryland 1995, Crothers 1997);- abundance of each species was also recorded.

Experiment 2: cockle population abundance and size

Cockle abundance was sampled haphazardly at each site within the plots using a 0.3 m^2 quadrat, sediment within the quadrat area to a depth of 6 cm was removed and sieved through a 1 cm sieve to retain adult cockles. Cockle abundance was recorded and width measurements taken to the nearest millimetre.

1 kg samples were also taken from the eco-elevator harvester during fishing activity on the same day of sampling, width was recorded to the nearest millimetre. The number of mascerated cockles was also recorded.

Experiment 3: sediment analysis

Sediment was collected from the centre of each site within each control and treatment plots using a 2 cm diameter corer with a volume of 20 cm^3 for grain size and organic carbon analysis. Physical impacts on the settled structure of the sediment (permeability) was measured at each site within the control and treatment plots by dropping an 80 cm steel rod from a height of 30 cm and measuring the depth to which it penetrated the sediment (Wynberg & Branch 1994). Three permeability measures were taken at each control and treatment plots for the baseline survey and after fishing activity had started on the 23rd July.

Sediment core samples were dried for 48 hours in a 35^{o}C oven to remove moisture from the samples before particle size and organic carbon analysis were carried out. Organic matter was not removed from particle size samples in conjunction with practices carried out by Plymouth Marine Biological Association (Hartley personal communication).

Grain size analysis was carried out using the Malvern Long-bed Mastersizer 2000 particle sizer running the software v.5.4. Samples were sieved through a 1 mm mesh to remove any larger coarse sediment; five sub-samples from each site sample were taken with each sub-sample being tested five times and an average created. Results of the mean grain size (phi) were calculated logarithmically using Folke and Ward graphical models in the GRADSTAT software (Blott &Pye 2001).

Organic carbon analysis was carried out by weighing samples prior to combustion at 450 ^{o}C were reweighed after combustion to establish dry ash weight. Total organic content percentage was then calculated from these weights.

Visual Impact

The visual impact of the eco-elevator harvester was assessed by photographs being taken directly after fishing activity and after 10 days recovery for comparison. Photographs were taken from treatment site 1 on each instance to allow direct comparison.

Statistical Analysis:

Univariate: Analysis was conducted using GMAV5 software package (Underwood et al. 1998). Four way analysis of variance (ANOVA) was carried out for all macrofauna data using survey, treatments, location (plot) and site as factors;- three way ANOVA was conducted on all other data with survey, treatment and location (plot) as factors. Post-hoc analysis was carried out where appropriate using Student Newman-Kuels (SNK) comparisons. Normality and homogeneity of variance was tested using Cochran's C test; appropriate transformations were applied where needed.

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

$H' = -\sum p_i(\log_e p_i)$

(p_i is the proportion of the total sample occurring from the *i*th species)

A two-sample Kolmogorov-Smirnov test was carried out on cockle size populations to test for differences between distributions using the software package SPSS.

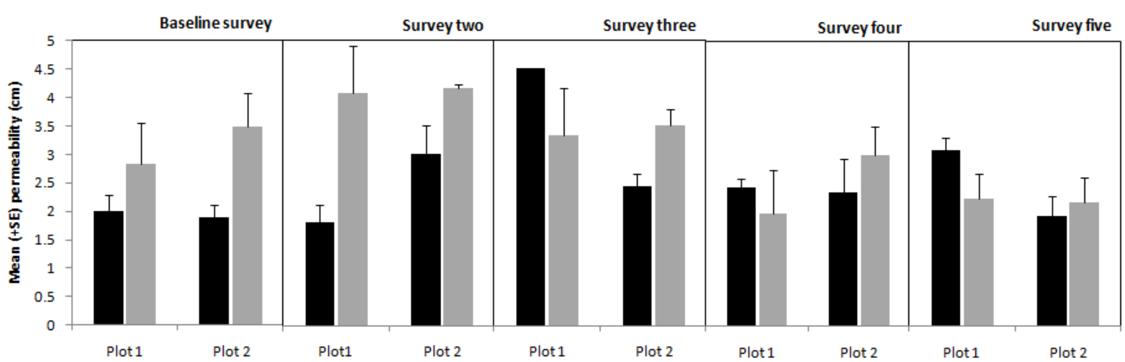
Multivariate analysis: PRIMER v5 software package (Clarke & Warwick 1994) was used for multivariate statistical analysis. Data were fourth root transformed with Bray-Curtis similarity measure used to create similarity matrixes. A two way nested analysis of similarity (ANOSIM) permutation test was applied to investigate for differences between the factors of treatment and location. Non-parametric multidimensional scaling (MDS) was used for ordination of the data. The similarities percentage procedure (SIMPER) was also adopted to examine species contribution between samples.

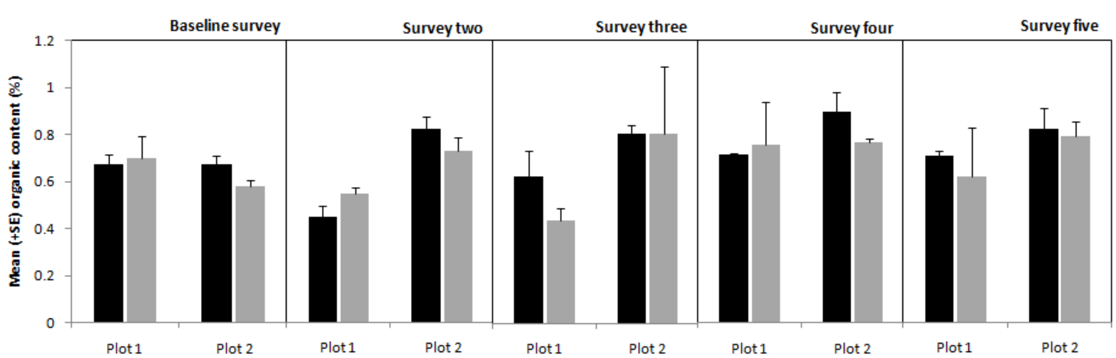
Results

Five surveys examining the effects of the eco-elevator harvester fishing method for cockles on sediment characteristics and target and non target species were successfully carried out.

Environmental Parameters

Phi size did not vary significantly between surveys, however a significant interaction was seen between plots (P=0.0291), between survey and plot (P=0.0013), treatment and plot (P=0.0061) and survey, treatment and plot (P=0.0125).





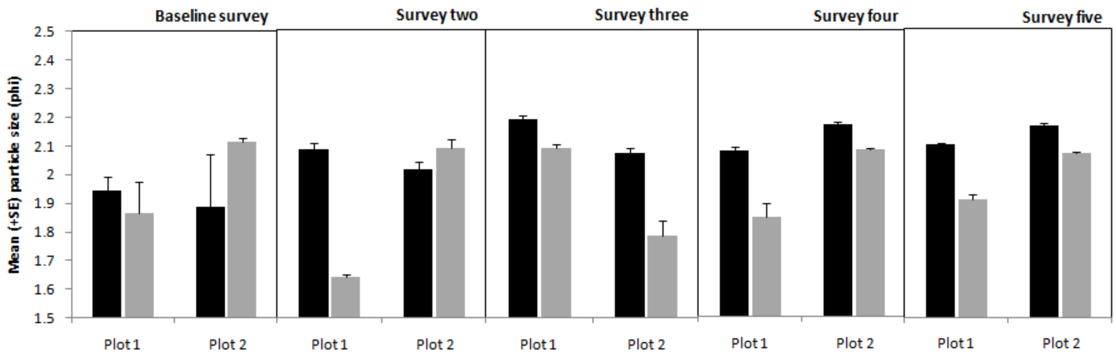


Figure 3. Environmental parameters (mean +1SD) for different treatment plots for each survey (■) treatment and (□)control.

Further statistical testing (SNK) (Table 2) showed that plot 2 had a higher phi size than plot 1 ($P<0.05$) however, the classification of phi size stayed the same as medium sorted sand throughout survey 1, 2, 3 and 4. Treatment plot 1 had a greater phi size than control plot 1 ($P<0.05$), however there was also significant differences between control plot 1 and 2 with control plot 2 having a higher phi size ($P<0.05$). Closer examination showed that the difference between control plots occurred in survey 1, 4 and 5 ($P<0.05$), and treatment plot 1 had a greater phi size then control plot 1 in survey 1, 2 and 4 ($P<0.05$). Survey 3 showed a slight anomaly in that treatment plot 2 had a significantly higher phi size then control plot 2 ($P<0.05$) which wasn't seen in any of the other surveys. This is clearly shown when observing the mean particle size in Figure 3.

Table 2: Three way ANOVA summary of abiotic variables with survey (su) treatment (Tr) and plots(Pl) as factors. Permeability (cm), grain size (phi), and organics (%), test statistic (F), associated probability (P).

Phi Size				
Factor	DF	MS	F	P
Survey	4	0.0026	0.39	0.8083
Treatment	1	0.0188	1.84	0.4047
Plot	1	0.0063	5.12	**0.0291**
suXtr	4	0.0032	0.72	0.6212
suXpl	4	0.0067	5.47	**0.0013**
trXpl	1	0.0102	8.37	**0.0061**
suXtrXpl	4	0.0045	3.66	**0.0125**
Residuals	40	0.0012		
Total	59			

Permeability				
Factor	DF	MS	F	P
Survey	4	302.7667	2.35	0.2144
Treatment	1	336.0667	1.31	0.457
Plot	1	1.0667	0.01	0.9056
suXtr	4	277.4833	2.52	0.1959
suXpl	4	128.9833	1.72	0.1638
trXpl	1	256.2667	3.42	0.0716
suXtrXpl	4	109.9333	1.47	0.2297
Residuals	40	74.8333		
Total	59			
% Organic				
Factor	DF	MS	F	P
Survey	4	0.1104	0.81	0.5797
Treatment	1	0.1314	19.79	0.1408
Plot	1	0.7832	12.98	**0.0009**
suXtr	4	0.0307	0.66	0.654
suXpl	4	0.1368	2.27	0.0789
trXpl	1	0.0066	0.11	0.7418
suXtrXpl	4	0.0468	0.78	0.5475
Residuals	40	0.0603		
Total	59			

Sediment permeability did not differ significantly (P>0.05) between treatment and control areas in all surveys (Table 2). A significant interaction was seen for organic concentration between plots with plot 1 (control and fished) having a higher concentration than plot 2 (P<0.05), however no differences were observed between surveys or treatment and control plots (P>0.05).

<u>Macrofauna</u>

A total of 31 species were identified, with samples largely dominated by *Hydrobia ulvae* accounting for 58.9% of all the fauna samples, although the highest abundances were found in survey one and four. The polychaete *Pygospio elegans* was the next most dominant species comprising 13.1% of all samples taken. The rest of the community was comprised of bivalves, other polychaetes, amphipods, oligochaetes and nematodes.

Species index (H) varied significantly between surveys (P<0.05) with SNK showing that survey 1 had a higher species than survey 5 (Table 3). Analysis of treatment and survey showed a significant interaction (P<0.05) with survey 5 and 1, 5 and 3, 5 and 2, 4 and 1, 4 and 2 and 4 and 3 for treatment plots compared to controls. However the species index in treatment plots increased (Figure 5) although there was not a significant difference in control plots between surveys.

Table 3: Four way ANOVA summary of Species index (Shannon-Wiener diversity (H)) with survey (Su) treatment (Tr), plot (Pl) and site (Si) as factors. Test statistic (F), associated probability (P),

Species Index (H)				
Factor	DF	MS	F	P
Survey	4	0.105	6.53	**0.0123**
Treatment	1	0.000	0.01	0.9332
Plot	1	0.750	10.45	0.0838
Site	2	0.039	2.58	0.0783
suXtr	4	0.063	4.51	0.0337
suXpl	4	0.016	0.36	0.8310
suXsi	8	0.016	1.07	0.3830
trXpl	1	0.346	23.48	**0.0401**
trXsi	2	0.033	2.2	0.1138
plXsi	2	0.072	4.81	**0.0092**
suXtrXpl	4	0.004	0.09	0.9845
suXtrXsi	8	0.014	0.93	0.4937
suXplXsi	8	0.045	2.99	**0.0036**
trXplXsi	2	0.015	0.99	0.3748
suXtrXplXsi	8	0.050	3.37	**0.0013**
Residuals	180	0.015		
Total	239			

Significant difference was also observed between treatment and plot areas (P<0.05) (table 2), however SNK showed that control plot 1 had a higher species index than control plot 2 (P<0.05) but there was not a significant difference between treatment and control. Species index was also higher for plot 1 site 2 and 3 compared to plot 2 site 2 and 3 (P<0.05), closer examination showed that this was observed in all surveys with the exception of the baseline (P<0.05). When treatment was factored in, a significant result was seen in surveys one, two three and five, with control plot 1 having a higher species index at site 2 and 3 then control plot 2(P<0.05). There was however no difference between control and treatment plots in these surveys. Survey four on the other hand had a significant difference between treatment plot 1 and 2, with treatment plot 1 having a higher species index than plot 2 at site 2 and 3, the same was also observed between the control plots(P<0.05), however there was not a significant interaction when examining treatment plots against control plots.

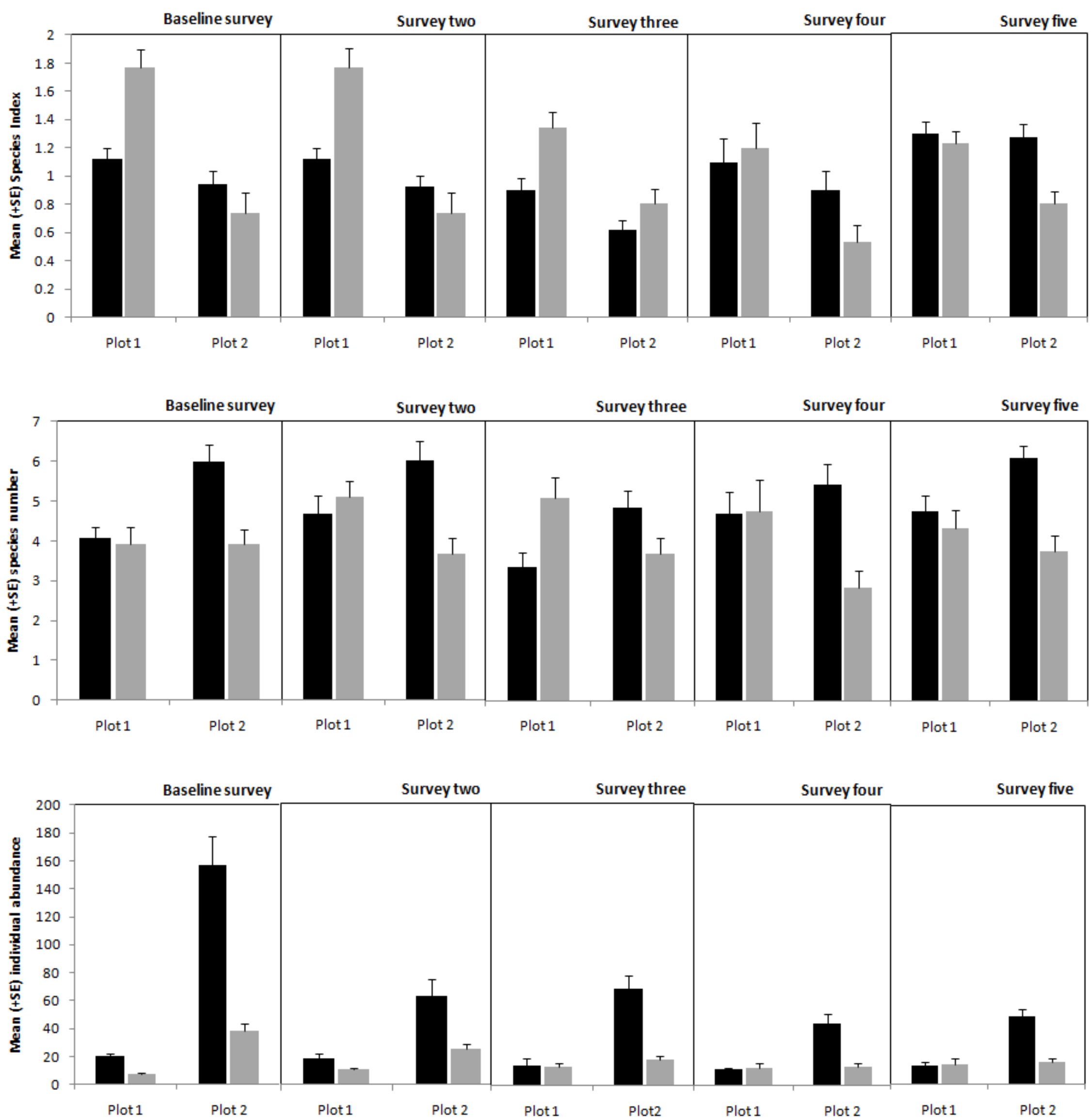


Figure 4. Species index (H), species number and individual abundance for each survey (■) treatment and (□)control.

The number of species present did not change significantly between surveys or between control and treatment areas (P>0.05), significance was however observed between treatment and plots. Nevertheless, SNK test showed that this was a result of higher species abundance in control plot 1 compared to control plot 2 ($P<0.01$), no difference occurred between treatment plots or between control and treatment plots ($P>0.05$). The four way ANOVA (Table 4) also showed a significant interaction for survey, treatment, plot and site for species abundance. SNK test showed significance between survey 1 and survey 4($P<0.05$), treatment plot 1 site 2 and treatment plot 2 site 3. This is due to a low species abundance in survey 4 compared to survey 1 (mean 0.44 compared to 0.86). The same as above with plot difference with plot 1 having higher abundance then plot 2. The significant interactions were not found through all the effect surveys (2-5).

Table 4 Four way ANOVA summary of species number with survey (Su) treatment (Tr), plot (Pl) and site (Si) as factors. Test statistic (F), associated probability (P),

Species Number				
Factor	DF	MS	F	P
Survey	4	0.2325	2.15	0.1663
Treatment	1	1.4797	12.93	0.0694
Plot	1	0.0867	0.11	0.7682
Site	2	0.1749	1.88	0.1549
suXtr	4	0.2179	1.64	0.2549
suXpl	4	0.1307	0.78	0.5669
suXsi	8	0.1084	1.17	0.3212
trXpl	1	3.7142	60.8	0.0161
trXsi	2	0.1144	1.23	0.294
plXsi	2	0.7635	8.22	**0.0004**
suXtrXpl	4	0.0512	0.15	0.9561
suXtrXsi	8	0.1327	1.43	0.1868
suXplXsi	8	0.1669	1.8	0.08
trXplXsi	2	0.0611	0.66	0.5191
suXtrXplXsi	8	0.3339	3.6	**0.0007**
Residuals	180	0.0928		
Total	239			

The number of individuals showed significant changes on a number of levels as seen in table 5 including between treatments, plots, sites, survey and treatments, and survey and plots ($P<0.05$). Further analysis showed that over the whole survey, treatment areas had a higher number of individuals then control areas ($P<0.05$) (Figure 4). Breakdown of treatment and survey showed that number of individuals rose significantly ($P<0.05$) from the baseline survey and subsequently throughout the surveys in the treatment areas. Whilst the opposite was found in the controls with a reduction of number of individuals, all surveys showing a significant reduction compared to the baseline survey only ($P<0.05$) (Figure 4).

Significant interactions were also seen between plots with plot 2 having a higher number of individuals then plot 1 (P 0.05) with a mean of 5.50 compared to just 3.99 in plot 1. This was found in surveys 2, 3, 4 and 5. When treatment was factored in however, it showed that treatment plot 2 was significantly higher than control plot 2, treatment plot 2 was also greater than treatment plot 1 ($P<0.05$). No differences in the number of individuals were observed for plot 1 between treatment and control, nor was there a difference between control plots.

Analysis of site showed that significant differences between all site 3 locations when compared to site 1 and 2 ($P<0.05$) with site 3 having a higher mean number of individuals then site 1 and 2 at a mean of 5.13 compared to 4.57 and 4.54 respectively. Further analysis including survey, plot and site showed this was the case for survey 4 in plot 1, survey 3 showed the interaction in plot 1 also but only between site 3 and 1($P<0.05$). Plot 2 had a significantly higher number of individuals then plot 1 for all sites for all surveys with the exception of the baseline.

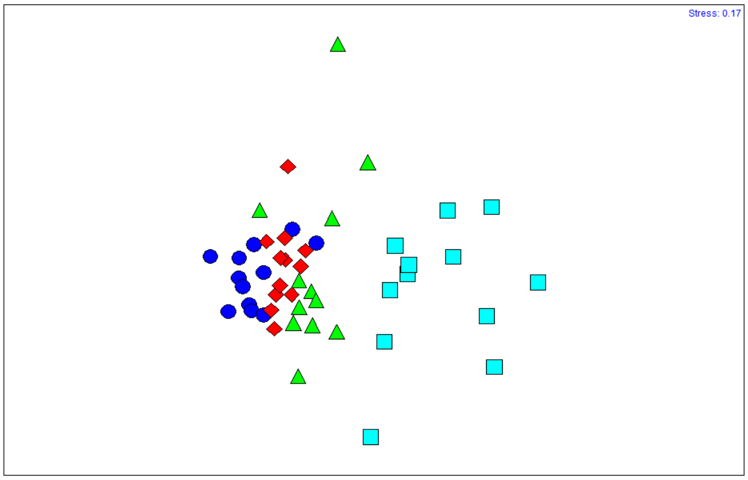
The SNK also showed that survey 1 had significantly higher individuals for plot 1 site 1 and 2 than the other surveys with a mean number of individuals at 5.79 compared to means of 2 and 3 for the other surveys. However no difference was seen for site 3. Significant variation was also seen for plot 2 site 1 between all surveys. This was due to an initial increase in mean number of individuals from survey 1-3 followed by a reduction in survey 4 and recovery in survey 5. Plot 2 site 2 on the other hand produced significant results ($P<0.05$) as a result of a peak in the mean number of individuals in survey 2 followed by a reduction in survey 3 and a recovery in survey 4 and 5. When treatment was also factored in significant differences were seen between surveys, survey 1 (baseline) control plots 1 and 2 all sites had a higher number of individuals. However after fishing had commenced for surveys 2, 3 and 4 all sites within treatment plot 2 had a higher number of individuals than controls and in survey 5 this was seen for plot 1 and 2.

Table 5 Four way ANOVA summary of number of individuals with survey (Su) treatment (Tr), plot (Pl) and site (Si) as factors. Test statistic (F), associated probability (P),

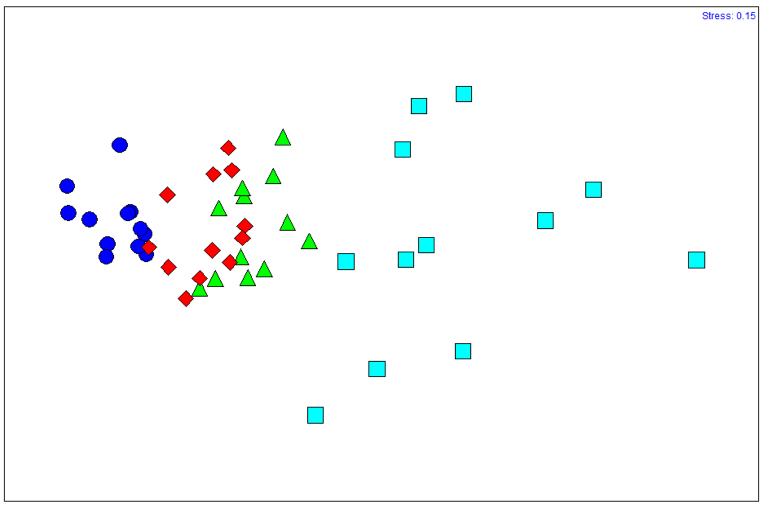
Number of Individuals				
Factor	DF	MS	F	P
Survey	4	5.7471	3.26	0.0728
Treatment	1	33.8049	23.86	**0.0395**
Plot	1	136.7069	144.93	**0.0068**
Site	2	8.8535	5.4	**0.0053**
suXtr	4	39.7191	82.23	**0.0000**
suXpl	4	24.579	6.46	**0.0127**
suXsi	8	1.7642	1.08	0.3814
trXpl	1	59.8974	24.44	0.0386
trXsi	2	1.417	0.86	0.4229
plXsi	2	0.9433	0.58	0.5634
suXtrXpl	4	16.2399	2.47	0.1289
suXtrXsi	8	0.483	0.29	0.967
suXplXsi	8	3.8068	2.32	**0.0214**
trXplXsi	2	2.4511	1.5	0.2268
suXtrXplXsi	8	6.5825	4.02	**0.0002**
Residuals	180	1.6386		
Total	239			

Multivariate analysis: Macrofauna assemblage

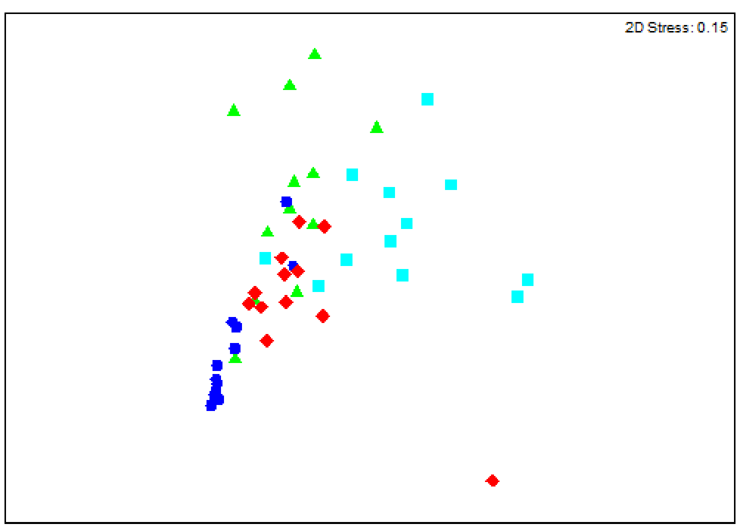
MDS ordination (Figure 5) of macrofauna for the baseline survey showed that infauna communities in control plot 1 were clustered away from those found in control site 2 and the fished treatment areas. A very similar pattern was seen throughout all the surveys compared to the baseline with control plot 1 species clustered away from all other plots. This suggests that the species found at this particular location differ to the rest of the sandbank. Treatment and control areas did become more dispersed through the surveys, however fished plot 2 and control plot 2 are still closely mirrored and grouped, showing little change in species.



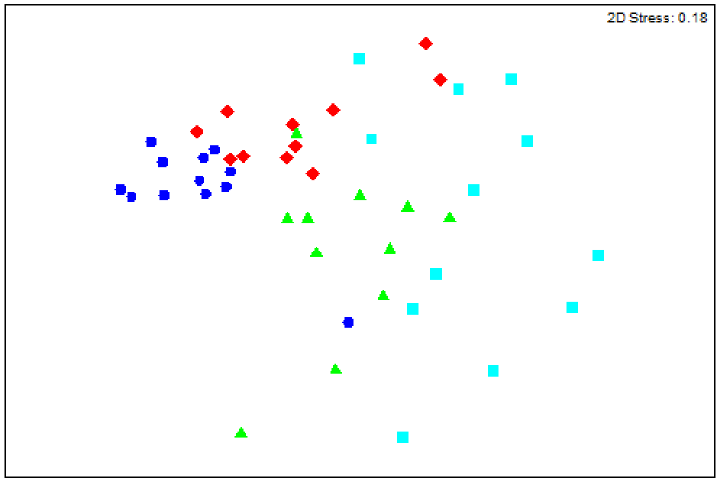
(a)



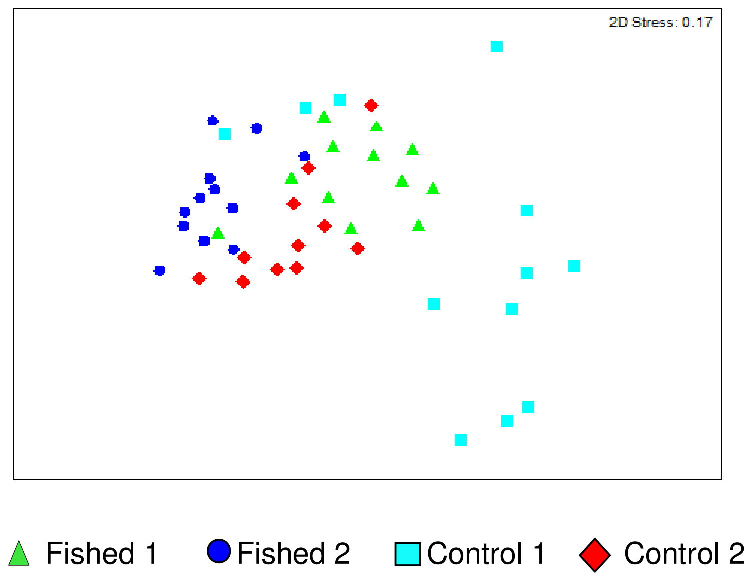
(b)



(c)



(d)



(e)

Figure 5. 2 dimensional non-metric multi-dimensional scaling configuration of 4th root transformed macrofauna data from control and treatment plots, (a) baseline survey (b) survey 2 (c) survey three (d) survey four and (e) survey five

SIMPER analysis showed that species with the highest percentage contribution were the same in all surveys (*Hydrobia ulvae* and *Pygospio elegans*) with the exception of treatment areas in survey 3 with a reduction seen with *Pygospio elegans* not seen. However the species contribution returned to previous levels in survey 4 and 5.

Table 6. Average similarity (SIMPER) between treatment and controls

Baseline			
Control		Fished	
Species	%contribution	Species	%contribution
Hydrobia ulvae	70.98	*Hydrobia ulvae*	82.05
Bathyporeia pilosa	10.42	*Pygospio elegans*	10.99
Pygospio elegans	7.91		
Cerastoderma edule juvenille	0.37		
Survey two			
Control		Fished	
Species	%contribution	Species	%contribution
Hydrobia ulvae	39.76	*Hydrobia ulvae*	43.39
Pygospio elegans	30.81	*Pygospio elegans*	37.91
Angulua tenuis	11.35	*Cerastoderma edule juvenile*	6.75
Cerastoderma edule juvenille	0.61	*Cerastoderma edule*	5.84
Bathyporeia pilosa	0.34		
Survey three			
Control		Fished	
Species	%contribution	Species	%contribution
Hydrobia ulvae	65.49	*Hydrobia ulvae*	0.31
Cerastoderma edule	9.27	*Cerastoderma edule*	12.34
Pygospio elegans	6.24	*Cerastoderma edule juvenile*	7.76
Eteone sp.	4.96		
Corophium arenarium	3.27		
Bathyporeia pilosa	3.12		
Survey four			
Control		Fished	
Species	%contribution	Species	%contribution
Hydrobia ulvae	69.17	*Hydrobia ulvae*	55.06
Bathyporeia pilosa	5.77	*Pygospio elegans*	13.82
Pygospio elegans	5.59	*Cerastoderma edule*	10.37
Eteone sp.	3.99	*Corophium arenarium*	6.99
Corophium arenarium	3.19	*Cerastoderma edule juvenile*	6.89
Survey five			
Control		Fished	
Species	%contribution	Species	%contribution
Hydrobia ulvae	49.19	*Hydrobia ulvae*	46.54
Pygospio elegans	15.13	*Pygospio elegans*	24.94
Bathyporeia pilosa	11.71	*Cerastoderma edule*	9.92
Eteone sp.	8.08	*Eteone sp.*	8.77

Cockle abundance and size distribution

Analysis of cockle abundance showed that there were no significant differences in the number of cockles found along the Exe Estuary in any treatment area or plot whilst the surveys were being carried out (P>0.05) (Table 7).

Table 7 way ANOVA summary for abundance of *C.edule* Survey (Su)treatment (Tr) and plot (Pl) as factors Test statistic (F), associated probability (P).

C.edule Abundance				
Factor	DF	MS	F	P
Survey	4	0.5705	0.44	0.7752
Treatment	1	34.804	38.27	0.102
Plot	1	0.0914	0.16	0.6879
SuXTr	4	1.0993	1.19	0.4349
SuXPl	4	1.2884	2.31	0.0747
TrXPl	1	0.9094	1.63	0.2093
SuXTrXPL	4	0.9234	1.65	0.1799
Residual	40	0.5585		
Total	59			

Although there were no statistical differences between cockle abundance, mean abundance of *C.edule* in fished plot 1 in the baseline survey was, however, greater than other plots by 31 individuals (Figure 6). This was a result of an anomaly of 104 individuals at site 1 within the plot, more cockles were found in treatment plots across all surveys.

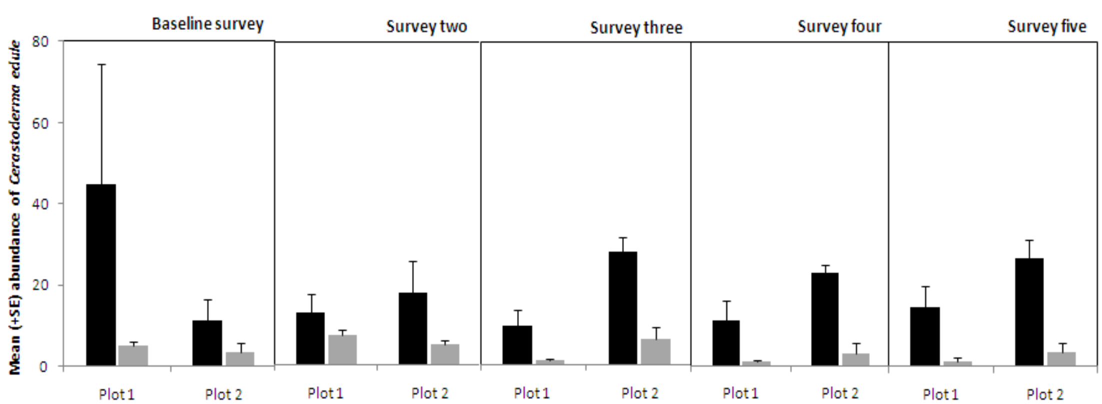


Figure 6 (mean +1SD) for different treatment plots for each survey (▬) treatment and (▭)control.

A comparison of cockle size classes taken from the intertidal area and boat samples can be seen in Table 8. Boat sampling was not carried out during survey four due to time constraints. The two-sample Kolmogorov-Smirnov test did not produce a significant result ($\alpha>0.05$) when comparing size distribution of intertidal samples from survey two to those collected from the eco-elevator harvester. Comparisons of cockle widths between intertidal samples and boat samples from survey three and survey five however, did produce a significant difference ($\alpha <0.05$) with samples taken from the boat being significantly larger than those sampled from intertidal.

The range of cockle sizes can be seen in table 8, it is notable however that in survey 3 and 5 there were some cockles sized 10cm and 13 cm when the mesh of the eco-elevator harvester size is 20mm. However this was during a period when the mesh was under repair, it is also worth noting that there was frequency of class size 13mm in survey 3 and 10mm in survey 5 was low, and the mean size from the boat for these surveys was infact 24mm and 23mm consecutively.

<u>Table 8. Range of cockle sizes and greatest % of cockle width within samples taken from the eco-elevator harvester and intertidally.</u>

Survey	Intertidal sample range (mm)	Boat sample range (mm)	Greatest % cockle width (mm)	
			Intertidal	Boat
2	10 - 28	20 – 39	19	25
3	11 - 28	13 – 37	18	26
4	11 - 26	-	20	-
5	10 - 29	10 – 31	10	25

<u>Landings</u>

Records of total landings before each survey were also carried out as well as the number of days fishing activity before surveying (Figure 8). As seen the amount of fishing effort is directly linked to the total catch amount, with fishing before survey 3 correlating to the highest catch total. However this was also the greatest time period between survey 2 and 3, rather then an increase in fishing effort.

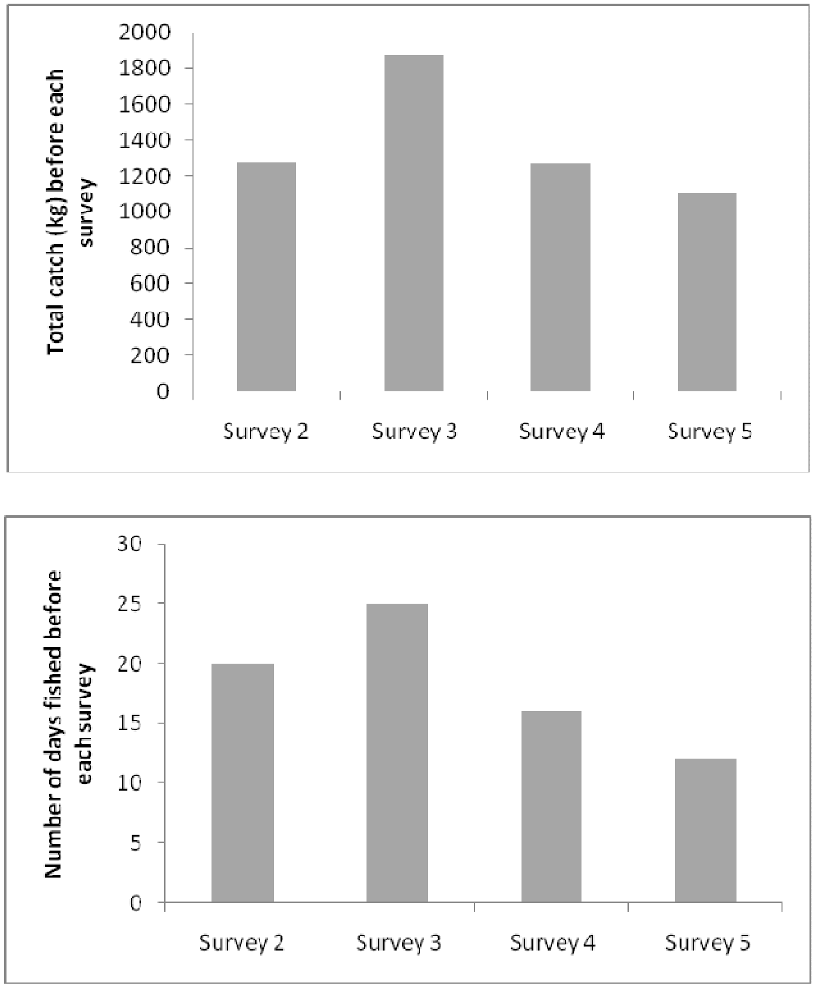


Figure 7. Total catch by the eco-elevator harvest (kg) and numbers of days fishing

Track marks

The presence of tracks could be seen directly after fishing (Figure 8a and 8c); however, there was a notable reduction in the prominence of the track marks after just 10 days (figure 8b and 8d).

(a) (b)

(c) (d)

Figure 8: (a) Track marks immediately after fishing 13th July (photo S.Hulme). (b) Track marks 10 days after fishing occurrence 23rd July (photo S.Clark) (c) Track marks immediately after fishing 5th November (photo V.Lee) (d) Track marks 10 days after fishing occurrence 15th November (photo S.Clark)

Discussion

Previous studies into the effects of cockle fishing has shown a direct negative relationship between fishing activity and non-target macro-fauna, often taking over a year for some long lived species to fully recover (Ferns et al. 2000 & Kaiser et al. 2006).

Sediment characteristics

The relationship between macrofauna assemblage and sediment characteristics is closely interlinked with benthic species having considerable interactions with biogeochemical and bioturbation processes (Mermillod-Blondin et al. 2004), monitoring sediment changes is therefore vital in assessing potential impact to the target and non target macrofauna species.

The presence of tracks from the eco-elevator harvester caused a visual disturbance; however they did not correlate with any significant change in particle (phi) size on the sandbank over the course of the study as a result of fishing activity, despite significant changes being seen between treatment and plots. The significance between treatment and plots was seen between control plots as well as the fished areas and was not consistent with a difference between control and fished plot 1 only being seen in survey 1,2, and 4 but not in survey 3 or 5. There was also significance between control plots in these surveys. The sediment found across the experimental area was medium “well sorted” sand which did not change throughout the survey; sand flat habitats have been demonstrated to recover quicker from disturbance then other substratum (Collie et al. 2000). Estuaries are characterised by high natural spatial and temporal variation (Ysebaert & Herman 2002), the significant differences in phi size are therefore a result of this variation rather than result of fishing activity and were therefore not of relevance to the study.

Despite an observed change in permeability (Figure 3) over the 5 surveys, no significant differences were seen throughout the study. The permeability of sediment is related to its grain size, shape and distribution (Soulsby 1997), therefore weather and tidal conditions play an important factor. The observed difference is therefore attributal to natural spatial variation.

Analysis of organic concentration only showed significant differences between plots, however this did not factor in treatment, which showed no difference, therefore is not a result of fishing activity. Over the study as a whole organic concentration was low, low organic carbon concentration can reflect low nutrient availability to macrofauna and low activity. However with permeable sands such as those found at the Exe estuary, the low organic matter is a result of high turnover rates as a result of wave and tidal action (Huettel et al. 1996, Charette et al. 2005). Anoxic conditions in permeable sands are reduced by the rapid mineralization of organic matter by advective flushing (Huettel et al. 1998), with metabolic products from the sediment rapidly removed when inundated (Billerbeck et al. 2006).

The reliability of sediment analysis data in impact surveys has however been questioned in previous research due to the core technique showing the interaction over the whole sample but possibly masking finer interactions in the top surface layer (Dernie et al. 2003). Ideally sediment chemistry and water quality parameters would also be measured to rule out other contributing factors to sediment parameters such as terrestrial runoff or eutrophication (Cammen 1982, Philips & Walling 1999, Patrício et al. 2009) especially due to the natural channels running alongside the sandbank.

Macrofauna Composition

The analysis of macrofauna assemblage composition in this study indicated that the mechanical fishing used did not have a significant effect upon the number of species or diversity. Although significant differences were seen, these were a result of differences between survey attributal to natural seasonal variations or as a result of differences found at control plot 1. Throughout analysis control plot 1 was significantly different to the rest of the test area with a greater diversity of species found. This was the only location where the polychaete *Ophelia* was found, which is not common around the South West Coast (Marlin 2010), PRIMER provided an insight into species variation between plots and examined any possible interactions. The MDS indicated that the first control plot had a different species composition than the other plots in both surveys. This plot was located next to a channel and was the closest to the shore; therefore the low abundance (number of individuals) could be attributable to two factors. The proximity to the shore could result in greater disturbances from public activity (e.g. trampling) and therefore a reduction in numbers. The different species composition could also be caused by this with a notable difference in this location with a bank largely consisting of shell debris. The bank is a slightly higher ridge than the rest of the sandbank and therefore could be an altered habitat when compared to the other areas due to regular disturbances such as trampling, by other users (Collie et al. 2000). Alternatively reduction could be a result of smothering of organisms by finer sediment deposited from the channel (Meridet et al. 1996). As it is a control area the differences are not attributable to fishing.

Examination of number of individuals did show significant differences between treatment and control plots as a result of fishing activity. However unlike previous studies which have shown a dramatic negative short term impact on macrofauna composition and abundance as a result of shellfish harvesting (Hall and Harding 1997, Kaiser et al. 1996), numbers of individuals increased in treatment plots. This could potentially be linked to the Pearson and Rosenberg (1978) Successional Model in which larger sensitive species are replaced with smaller opportunistic species as intensity of disturbance increases. However, diversity and species number did not vary, and Simper showed no shift in the benthic community therefore it cannot be attributable or explained by this model. It could be however that the community had reached an equilibrium state despite being in a changeable habitat before fishing was carried out (Ellis 2004). After the disturbance the community could be attempting to regain a balance which is why abundance increased as juveniles then had the opportunity to colonise. Gilikinson et al (2005) showed a similar increase in abundances of macrofauna two years after dredging activity and concluded that the community was still in recovery period. However, once again this study saw a change in species composition with opportunistic species increasing in abundance and equilibrium species decreasing. This was not the case in this current study, although the disturbance is allowing more individuals into the area, suggesting that space within the community is being created by the removal of more established individuals. The species composition was not negatively effected by the fishing with *Hydrobia ulvae* and *Pygospio elegans* still being the dominant species. *Hydrobia ulvae* is an important food source for the wetland birds. Whilst *Pygospio elegans* is important in the larger ecological context of the estuary as the species help stabilise the sediment by building tubes and therefore minimise the effects of large scale disturbances (Bolam &Fernandes 2002).

Cockle Size and Distribution

The cockle fishing did not have an impact on cockle abundance or size with the size ranges on the intertidal area staying the same throughout the study. Previous research into other mechanical harvesting methods (suction dredging, hydraulic dredging and tractor dredging) has shown a negative correlation between fishing activity and target species abundance and a positive correlation between fishing activity and cockle mortality (Hall 1994, Hall & Harding 1997, Norris et al. 1998 & Piersma et al. 2001). Although this negative interaction was not recorded for the eco-elevator harvester a cautionary approach needs to be adopted due to the sensitive nature of the Exe estuary and its importance for migratory birds. The collapse of cockle stocks and the subsequent mortality of migratory birds due to loss of a food source as a result of mechanical harvesting is well documented in the Dutch Wadden Sea (Swart and Andel 2008), highlighting the importance of careful monitoring of cockle stocks.

Examination of the harvester (figure 1) showed that the fishing gear allowed for cockles of a smaller size class (<20mm) to pass through the fishing gear and return to the seabed. Therefore they can reburrow and re-establish themselves whilst still covered by the tide, leaving them less vulnerable to predation as they are not exposed (Coffen-Smout & Rees 1999). However when survey 3 and 5 occurred from the boat, the harvester was slightly damaged, therefore some undersized cockles were recorded as seen in table 6 with a range starting at 13mm for survey 3 and 10mm for survey 5. Despite this, cockles sampled from the boat and eco-elevator harvester were significantly greater in size than those found in the intertidal. The difference was not seen for survey 2, however this is linked to the test and small abundance found in the intertidal rather than it not occurring as the size distribution graphs show. The fishing gear allowed for cockles of a smaller size class (<20mm) to return to pass through the fishing gear and return to the seabed. Therefore they can reburrow and re-establish themselves whilst still covered by the tide, leaving them less vulnerable to predation as they are not exposed (Coffen-Smout & Rees 1999).

Cockle Landings

Analysis of data provided by the fisherman showed that fishing effort mirrored catch amount with the highest catch being from survey 2 to survey 3. However this was the greatest time difference with 97days in between sampling which is reflected on by the amount caught.

Future Recommendations

There are limitations to the study carried out, such as the size of the fished area, the Exe estuary treatment areas were only 200m^2 with low fishing intensity of twice a week only on Spring tides. With this low intensity little impact was seen, however it is recommended that the size of areas or intensity should not be greatly increased. Hall and Harding (1997) showed rapid recovery in a 7 hector area, despite original high macrofauna mortality. One plot was also favoured by the fisherman with plot one being fished more intensively than plot 2. This would have to be monitored so that fishing was not concentrated solely on this area. More detailed sediment analysis could also be beneficial to provide more pronounced detail into the potential impact of the eco-elevator harvester on the interaction between sediment and macrofauna assemblage.

Control plot one varied significantly throughout the survey, with particle and macrofauna differences. Examining these differences in this area would be beneficial especially for the protection of species and considering the significance of the area to birds. Considering the importance of the Exe Estuary, future monitoring of the impact of the fishing activity on target species and macrofauna is of continual importance. With collapses of cockle stocks and the subsequent mortality of migratory birds due to loss of a food source as a result of mechanical harvesting such as in the Dutch Wadden Sea (Swart and Andel 2008), a cautionary approach and the importance of careful monitoring of cockle stocks needs to be highlighted.

Conclusions

The current study found no significant impact upon non target macrofauna abundance, assemblage or diversity, nor any impact upon the target species *C.edule* abundance or size as a result of fishing activity by the eco-elevator harvester. Although significant differences were seen for number of individuals, there was in fact an increase. Sediment parameters of grain size, permeability and organic concentration were not adversely affected despite clear visual disturbances.

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