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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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Abstract

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 after fishing activity had commenced. No significant differences were observed for sediment parameters (grain size, permeability and percentage organics) between plots in either survey. Macrofauna assemblage, abundance and diversity were not significantly different between treatments in the baseline survey or after fishing had commenced. *C.edule* size and abundance were not significantly different between plots in the baseline survey and no significant changes occurred after fishing had begun. Due to the limited time of the study and the time of year that the study was carried out, it is not possible to conclude if a negative impact is occurring as a result of the mechanical fishing method used.

KEY WORDS: Macrofauna- Disturbance-Eco-elevator harvester-Fishery.

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, and 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). 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-Smythe 2009 Personal communication).

Macrofauna is defined as those animals which are retained on a 0.5-1mm sieve, with marine macrofauna at phylum level being the most diverse assemblage (Eleftheriou & Holme 1984). Marine macrofauna is a widely used ecological indicator for a variety of impact surveys due to their high diversity of phyla, the relative ease of sampling and the potential to indicate stress or disturbance and its intrinsic relationship with sediment (Salas et al. 2004, Patrício et al. 2009). Macrofauna consists of a wide variety of taxa both mobile and sessile, each with its own tolerance levels to a variety of stresses (Dauer et al. 1993). Macrofaunal organisms are crucial for ecosystem health as they are a food source for higher trophic species, help maintain sediment stability, remove pollutants from the water and regulate organic matter (Snelgrove 1998, Bolam & Fernandes 2002)

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. (2002) 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. Suction dredging can cause significant negative impact on cockle abundance, non-target fauna abundance, diversity, 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. Elliott & Quintino (2007) go further with the theory of Estuarine Quality Paradox in which "*dominant estuarine faunal and floral community is adapted to and reflects high spatial and temporal variability in naturally highly stressed areas similar to those found in anthropogenically stressed areas thus making it difficult to detect anthropogenically-induced stress in estuaries.*" They concluded that due to this paradox environmental management strategies based on fauna changes could be questioned. A combination of biological and physical parameters, therefore, could be more suitable in determining overall ecological interactions.

Despite extensive research into the mechanical effects of cockle harvesting there has been little investigation into the effects of the eco-elevator dredges on macrofauna assemblage composition, abundance and diversity, the disturbance effects on the sediment (grain size, organic concentration and penetration) and cockle abundance. This gap in knowledge has lead to this study being commissioned by Natural England in conjunction with Devon Sea Fisheries Committee as the initial investigation of a 12 month monitoring program into the impact as a result of mechanical cockle fishing in the Exe estuary, South West England.

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) 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). 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). 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).
Further investigation into this fishing method is needed to identify any ecological implications related to its application upon estuarine sand flats.

The aim of this study was therefore to test the hypothesis that the eco-elevator harvester would not cause as significant reductions in macrofauna assemblage, cockle populations and sediment parameters as seen with other types of mechanical cockle harvesters.

(a)

(b)

(c)

Figure 1: (a) Mesh of elevator dredge(photo S.Hulme). (b) Track marks immediately after fishing 13th July (photo S.Hulme). (c) Track marks 10 days after fishing occurrence 23rd July (photo S.Clarke).

Materials and Methods

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 2). 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).

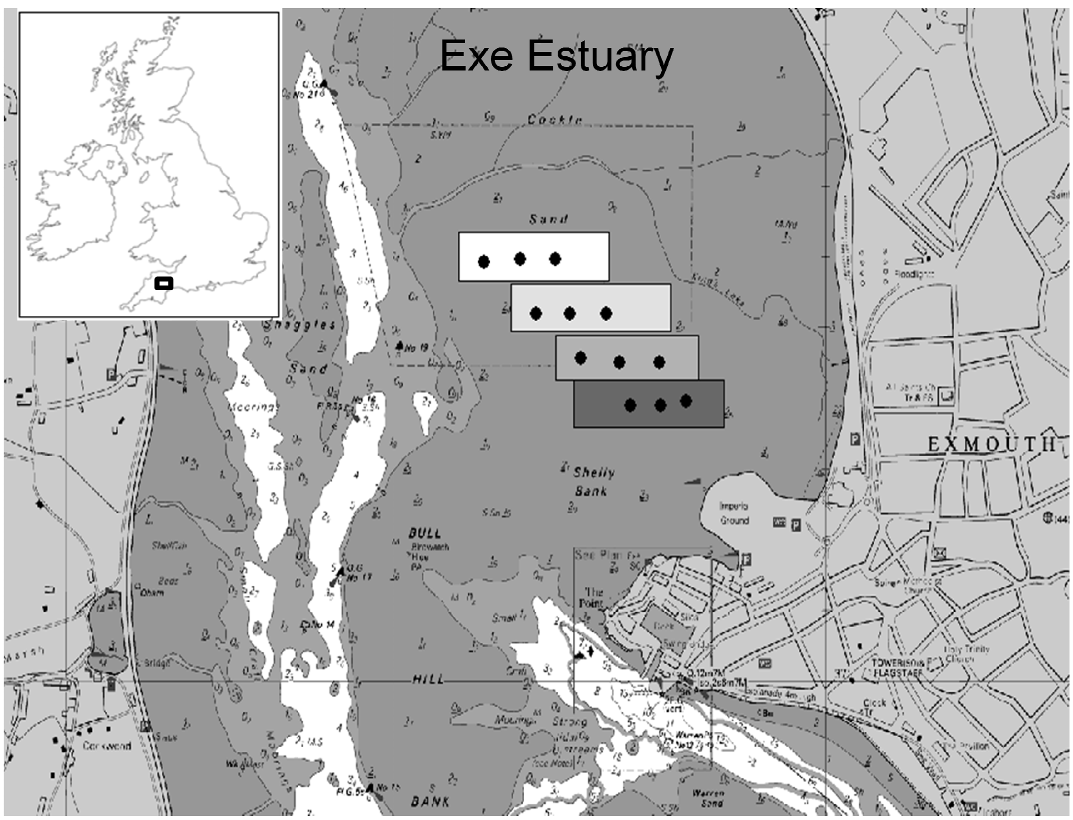


Figure 2.Map of Exe Estuary (Devon, UK) showing plots where baseline samples were taken. (Not to scale) =control plot 1 = fished plot 1 =control plot 2 = fished plot 2. • = Sampling sites for baseline survey.

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, 34 days after fishing had started.

Macrofauna samples were sieved onsite where possible through a 1 mm mesh before being preserved in 70% alcohol to transport to the laboratory; samples were then fixed 24 hours later in 4% formalin.

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 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 and size recorded to the nearest millimetre.

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 conducted by weighing samples prior to 24 hours combustion at 450 ^{o}C; samples were reweighed after combustion to establish dry ash weight. Total organic content percentage was then calculated from these weights.

Statistical analysis

Univariate analysis: Analysis was conducted using GMAV5 software package (Underwood et al. 1998). Three way analysis of variance (ANOVA) was carried out for all macrofauna data using treatments, location (plot) and site as factors;- two way ANOVA was conducted on all other data with 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:

Two surveys examining possible effects of a mechanical fishing method for cockles on sediment characteristics and target and non target species were successfully carried out. In total 30 different taxa were identified with *Cerastoderma edule* and *Angulus tenuis* juveniles recorded separately; most of the taxa were successfully identified to species level.

Environmental parameters: The baseline survey did not show any significant difference (P>0.05), for abiotic factors (grain size, permeability and organic content) between locations or treatment (Table 1) this showed that the sediment was the same across the sandbank area before fishing had begun.

Table 1. Two way ANOVA summary of abiotic variables with treatment (Tr) and plots(Pl) as factors. Permeability (cm), grain size (phi), and organics (%), test statistic (F), associated probability (P).

		Baseline survey			**After fishing activity**			
Phi size								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	0.017	0.35	0.616	1	0.102	0.66	0.501
Pl x Tr	2	0.049	1.37	0.307	2	0.154	87.66	**<0.001**
Residuals	8	0.036			8	0.002		
Total	11				11			
		Baseline survey			**After fishing activity**			
Permeability								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	4.441	13.03	0.069	1	8.841	8.13	0.104
Pl x Tr	2	0.341	0.46	0.647	2	1.088	1.03	0.399
Residuals	8	0.741			8	1.052		
Total	11				11			
		Baseline survey			**After fishing activity**			
% organics								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	0.003	0.31	0.632	1	0.000	0.00	0.993
Pl x Tr	2	0.011	1.10	0.377	2	0.129	18.01	**0.001**
Residuals	8	0.010			8	0.007		
Total	11				11			

Neither particle size, sediment permeability nor sediment organic concentration were affected by the eco-elevator harvester (P>0.05) when treatments were compared against controls. A significant interaction was observed for organic concentration and phi size after fishing activity between plot and treatment (P<0.01) (Table 5 Figure 7).

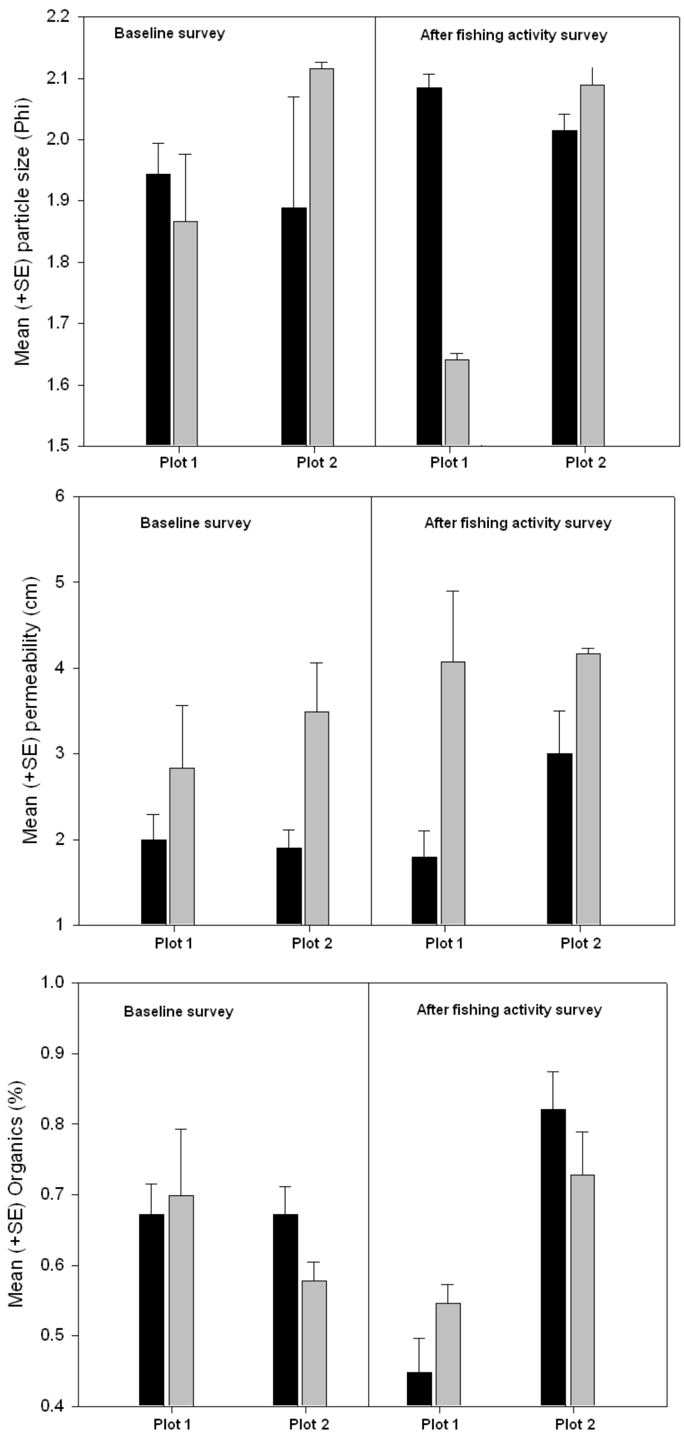


Figure 3. Environmental parameters (mean +1SD) for different treatment areas (■) treatment and (□) control.

SNK test showed that phi size in treatment plot 1 was greater than control plot 1($P<0.01$), whilst organic concentration was higher in control plot 1 than the treatment plot 1, with the reverse occurring for plot 2 with treatment having a higher organic concentration than the control ($P<0.05$) (Table 1Figure 7).

Univariate Analysis: Macrofauna assemblage: Infaunal samples, both before and after fishing, were largely dominated by *Hydrobia ulvae* which accounted for 62.4% of all the fauna samples, although the highest abundances were found in the second control and fished plots. The polychaete *Pygospio elegans* was the next most dominant species comprising 20.7% of all samples taken. The rest of the community was comprised of bivalves, other polychaetes, amphipods, oligochaetes and nematodes.

Analysis of species number showed no significant differences ($P>0.05$) between proposed treatment plots and controls in the baseline survey (Table 2 Figure 4). The secondary survey also did not show a significant reduction in macrofauna species number ($P>0.05$) between the treatment and control plots (Table 2 Figure 4) after fishing had been commenced.

Species diversity (H') did not show variation in the baseline survey ($P>0.05$) between proposed treatment and control plots (Table 2 Figure 4). Macrofauna species diversity (H') did not vary significantly ($P>0.05$) after fishing had commenced (Table 2 Figure 4). There was however an observed increase in species diversity (H) between the baseline survey and the survey carried out after fishing had begun (Figure 4).

Similarly the number of individuals found in samples in the baseline survey and the survey after fishing had commenced did not show significant interactions ($P>0.05$) between treatment and controls (Table 2 Figure 4). High variability observed in the number of individuals in the baseline survey for treatment plot 2 could potentially mask any significant interactions in this survey (Figure 4).

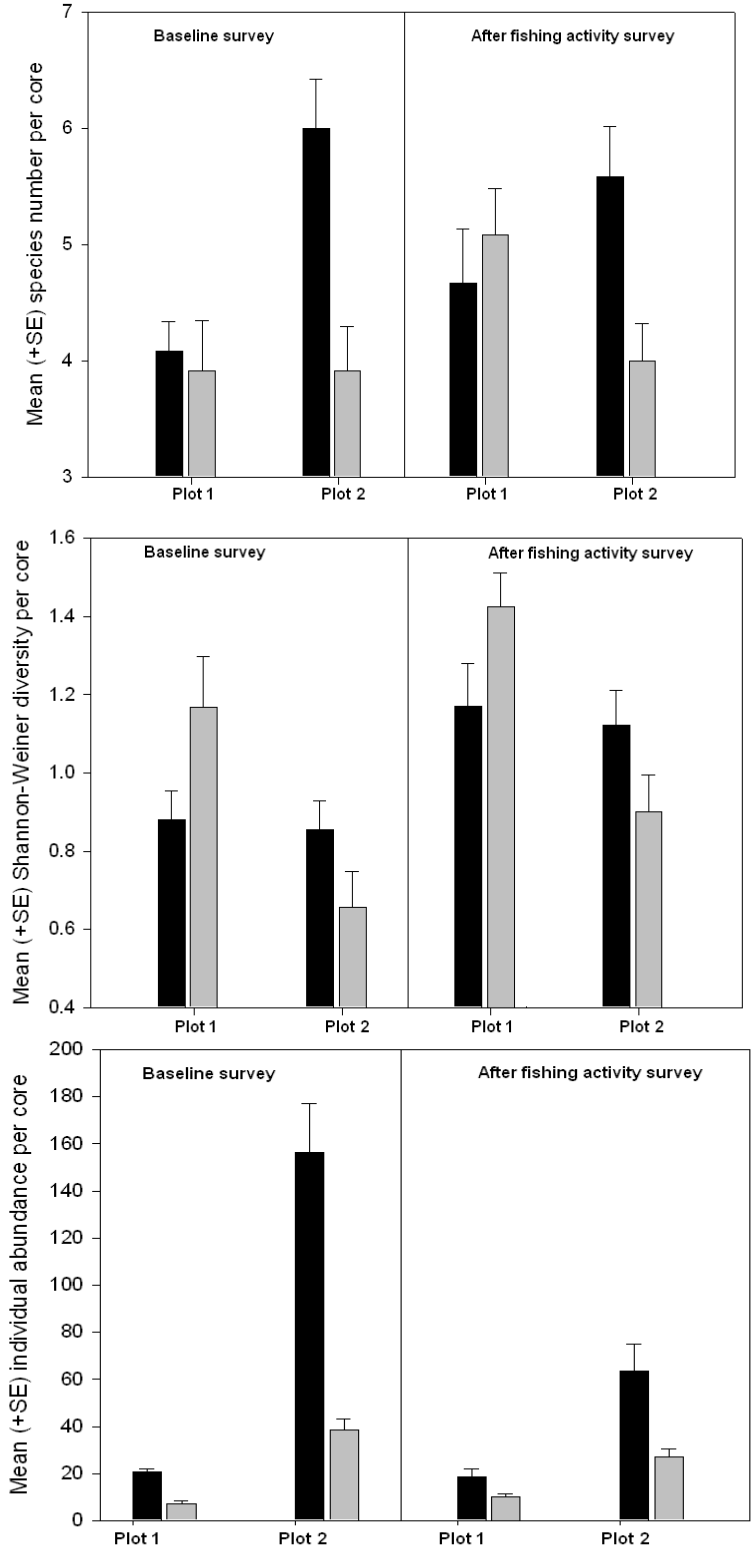


Figure 4. Species number, Shannon-Weiner diversity (H') and number of individuals (abundance) for ( ) control and ( ) treatment plots.

There were, however, significant interactions between plot and treatment for species diversity (H’) in the baseline survey, post-hoc analysis (SNK) showed that control plot 2 had significantly higher H’ than control plot 1 ($P<0.05$).

Abundance of individuals was significant between plot and treatment in both surveys, with a greater number of individuals in treatment plots when compared with controls ($P<0.05$), Plot 2 also had significantly higher number of individuals when compared to plot 1($P<0.05$) (Table 2 Figure 4). The same relationship between plots occurred with the number of species in the baseline survey ($P<0.05$) (Table 2 figure 4).

A significant interaction was also observed between site, plot and treatment for species diversity (H’) species number after fishing activity (Table 2). Further analysis showed that site 3 had a greater number (abundance) of species than site 1 ($P<0.05$) in both treatment plots. Whilst post-hoc analysis on diversity index (H’) showed a greater number in control plot 2 site 3 when compared to site 1 and 2 ($P<0.05$).

Table 2 Three factor ANOVA summary of macrofauna assemblage with treatment (Tr), plot (Pl) and site (Si) as factors. Test statistic (F), associated probability (P), Shannon-Wiener diversity (H).

	Species number baseline				**Species number after activity**			
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	15.19	1.38	0.3613	1	4.08	0.68	0.4974
Pl x Tr	2	7.05	13.33	**0.0028**	2	6.04	1.15	0.3637
Si x Tr x Pl	8	0.53	1.70	0.1313	8	5.25	4.06	**0.0016**
Residuals	36	1.69			36	1.29		
Total	47				47			
	Diversity (H') baseline				**Diversity (H')after activity**			
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	0.025	0.03	0.8752	1	0.0034	0	0.955
Pl x Tr	2	0.7886	7.36	**0.0154**	2	0.8426	4.22	0.0562
Si x Tr x Pl	8	0.1072	0.99	0.4624	8	0.1998	2.31	**0.0414**
Residuals	36	0.1086			36	0.0866		
Total	47				47			
	Number of individuals baseline				**Number of individuals after activity**			
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	20.6548	0.94	0.4343	1	3.8961	0.55	0.5348
Pl x Tr	2	21.9426	72.74	**<0.001**	2	7.0549	13.33	**0.0028**
Si x Tr x Pl	8	0.3017	1.23	0.3086	8	0.5294	1.7	0.1313
Residuals	36	0.2447			36	0.3108		
Total	47				47			

Key species: Several key species were analysed to assess disturbance on a species level. The species examined were *H.ulvae*, as it was the dominant species, *P.elegans* as the second dominant species and an opportunistic species (Grassel and Grassel 1974) and *Corophium arenarium* as the species is a short term coloniser (Conlan 1994).

A significant effect did not occur between treatment or control for all three key species examined (P>0.05) in the baseline survey or after fishing activity had occurred (Table 3). This was despite a higher abundance seen for treatment plot 2 for *H.ulvae* and *P.elegans* in the baseline survey, and higher abundance of *C.aranarium* for treatment plot 2 after fishing activity had occurred (Figure 5). There was, however, an interaction evident in both surveys for *H.ulvae* and *P.elegans* between plots and treatment (Table 3); further analysis showed that abundance of both species was significantly greater in plot 2 when compared to plot 1 ($P<0.05$) (Figure 5).

Table 3. Two way ANOVA summary for key species, treatment (Tr) and plot (Pl) as factors Test statistic (F), associated probability (P).

		Baseline survey			**After fishing activity**			
H.ulvae								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	27600.02	0.99	0.424	1	21.68	0.34	0.617
Pl x Tr	2	27751.27	32.35	**<0.001**	2	63.07	25.64	**<0.001**
Residuals	44	857.95			44	2.46		
Total	47				47			
		Baseline survey			**After fishing activity**			
P.elegans								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	13.91	1.83	0.309	1	526.69	2.9	0.231
Pl x Tr	2	7.60	7.73	**0.001**	2	181.77	4.8	**0.013**
Residuals	44	0.98			44	37.88		
Total	47				47			
		Baseline survey			**After fishing activity**			
C.arenarium								
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	0.02	1	0.4226	1	2.52	1.49	0.346
Pl x Tr	2	0.02	1	0.3761	2	1.69	2.61	0.085
Residuals	44	0.02			44	0.65		
Total	47				47			

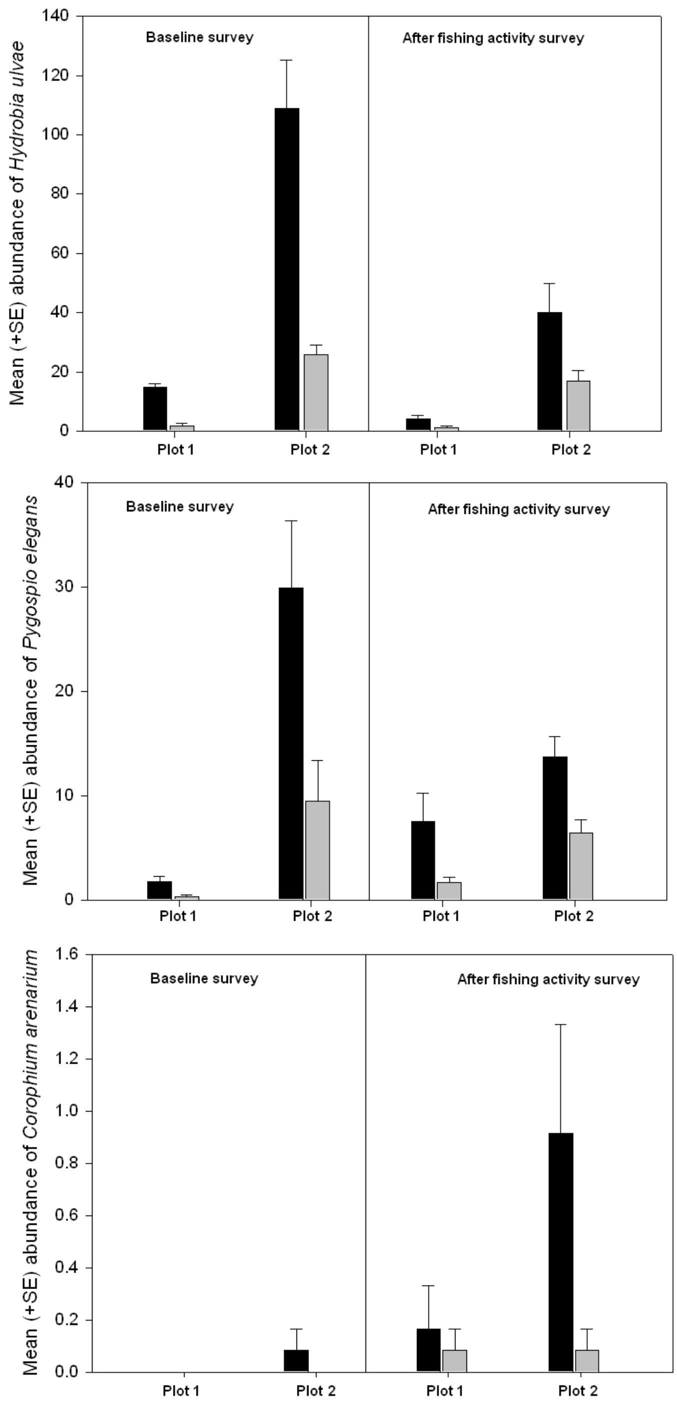


Figure 5. Mean abundance of key species for ( ) control and ( ) treatment plots.

Multivariate analysis: Macrofauna assemblage

MDS ordination (Figure 6 a) of macrofauna for the baseline survey showed that infauna communities in control plot 1 was clustered away from those found control site 2 and the fished treatment areas. A similar occurrence is also observed with the MDS after fishing (figure 6 b) with the first control plot; the species in the remaining plots however, are clustered tighter together than in the baseline survey. This is most likely to be attributed to the low faunal abundance within the core samples found at this location in both surveys.

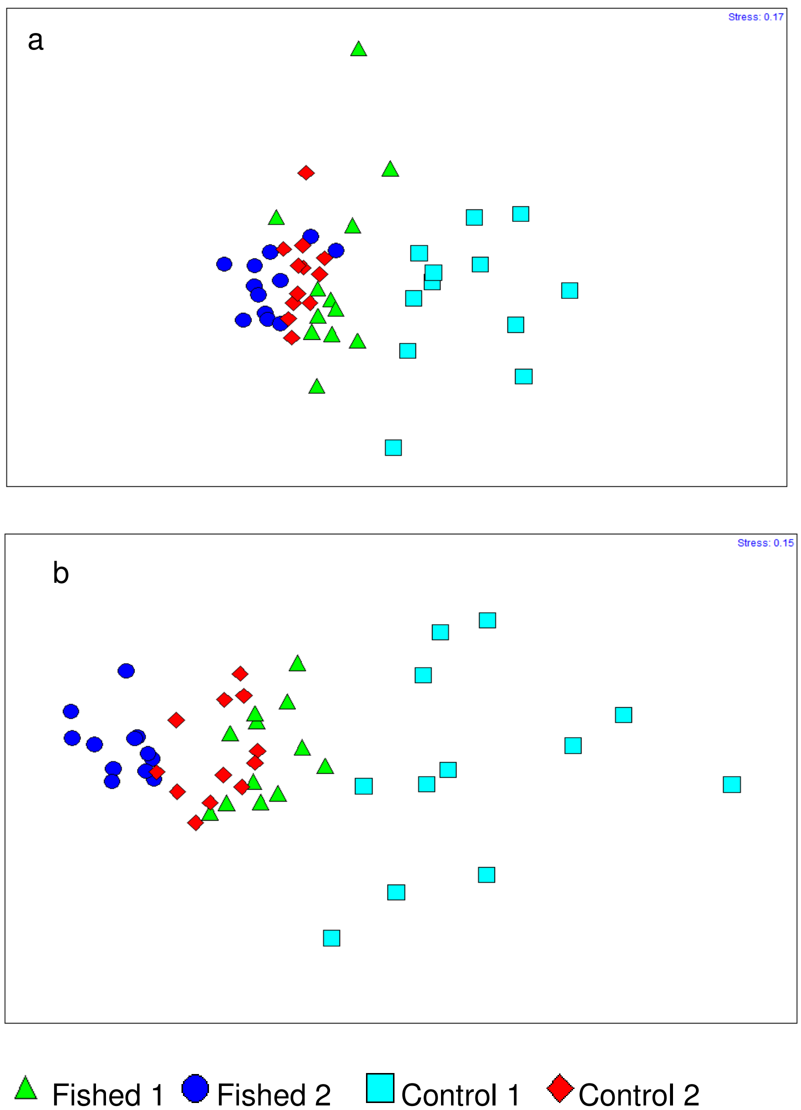


Figure 6. 2 dimensional non-metric multi-dimensional scaling configuration of 4th root transformed macrofauna data from control and treatment locations before fishing activity had been carried out (a) and after fishing activity (b).

Multivariate analysis showed significant difference between the location of the samples and the macrofauna community in both the baseline survey and after fishing activity had commenced (ANOSIM R=0.564 P=0.001 and R=0.345 P=0.001 consecutively). There was, however, no significance in assemblage composition treatments in either survey (ANOSIM P>0.05 R=-0.25 baseline R=0.25 after fishing activity). SIMPER analysis showed that all samples were dominated by similar species (Table 4).

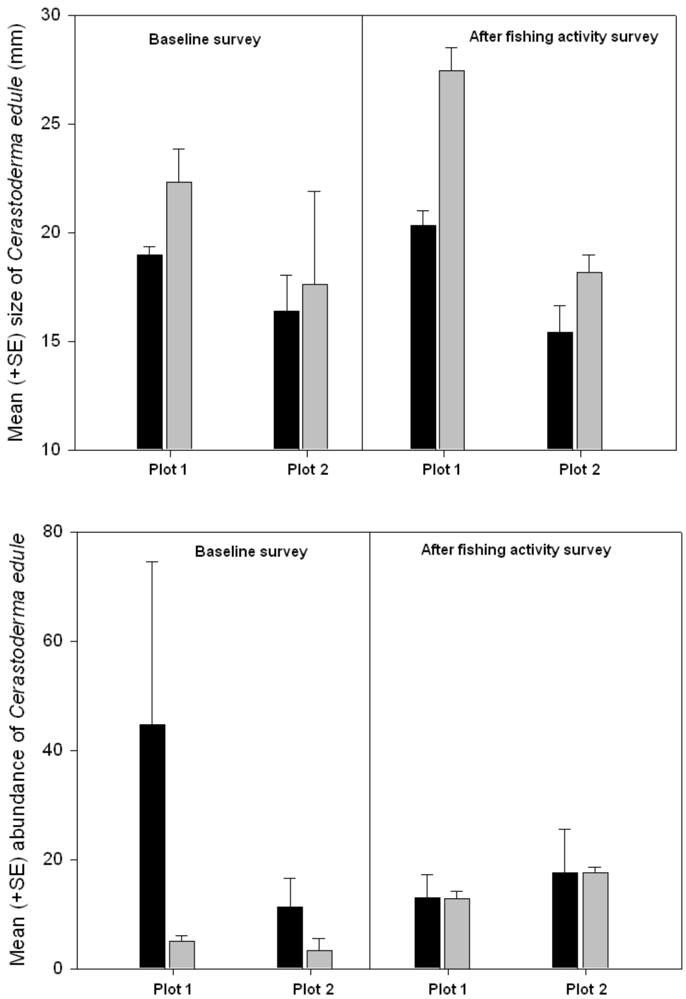
Table 4. Average similarity (SIMPER) between treatment and controls before fishing had commenced and afterwards.

	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		
	After fishing activity		
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 juvenille*	6.75
Cerastoderma edule juvenille	0.61	*Cerastoderma edule*	5.84
Bathyporeia pilosa	0.34		

Cockle abundance and size:

The cockle abundance and size in the baseline survey showed no significant interaction between treatment and control (P>0.05) (Table 5). This did not alter as a result of fishing activity as the survey carried out after fishing had commenced produced similar results (P>0.05) (Table 5). Mean abundance of *C.edule* in fished plot 1 in the baseline survey was, however, greater than other plots by 31 individuals (Figure 7). This was a result of an anomaly of 104 individuals at site 1 within the plot.

The only significant interaction which occurred (P<0.05) was for the size of *C.edule* in the secondary survey between plot and treatment (Table 5), further analysis showed that that this was due to a larger cockle sizes being found at plot 1 in comparison with plot 2 (Figure 7).



<u>Figure 7. Mean size (mm) and mean abundance of C.edule for ( ) control and ( ) treatment plots.</u>

Table 5. Two way ANOVA summary for abundance and size (mm) of *C.edule* treatment (Tr) and plot (Pl) as factors Test statistic (F), associated probability (P).

	***C.edule* abundance baseline**				***C.edule* abundance after activity**			
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	1704.08	2.04	0.289	1	252.08	12.35	0.072
Pl(tr)	2	835.42	1.20	0.349	2	20.42	0.32	0.735
Residuals	8	694.50			8	63.92		
Total	11				11			
	***C.edule* size (mm) baseline**				***C.edule* size (mm) after activity**			
Factor	Df	MS	F	P	Df	MS	F	P
Treatment	1	0.026	0.33	0.6261	1	73.4085	0.88	0.4463
Pl(tr)	2	0.08	1.57	0.2658	2	83.0061	30.05	**0.0002**
Residuals	8	0.0509			8	2.762		
Total	11				11			

A comparison of size classes taken from the intertidal area show the greatest percentage for 19mm with a range of sizes from 10mm-28mm(Figure 8a).The samples from the boat however peak at 25mm with a size range of 20-39mm(Figure 8b). The two-sample Kolmogorov-Smirnov test however, did not produce a significant result ($\alpha>0.05$) when comparing size distribution of intertidal samples to those collected from the eco-elevator harvester on the boat.

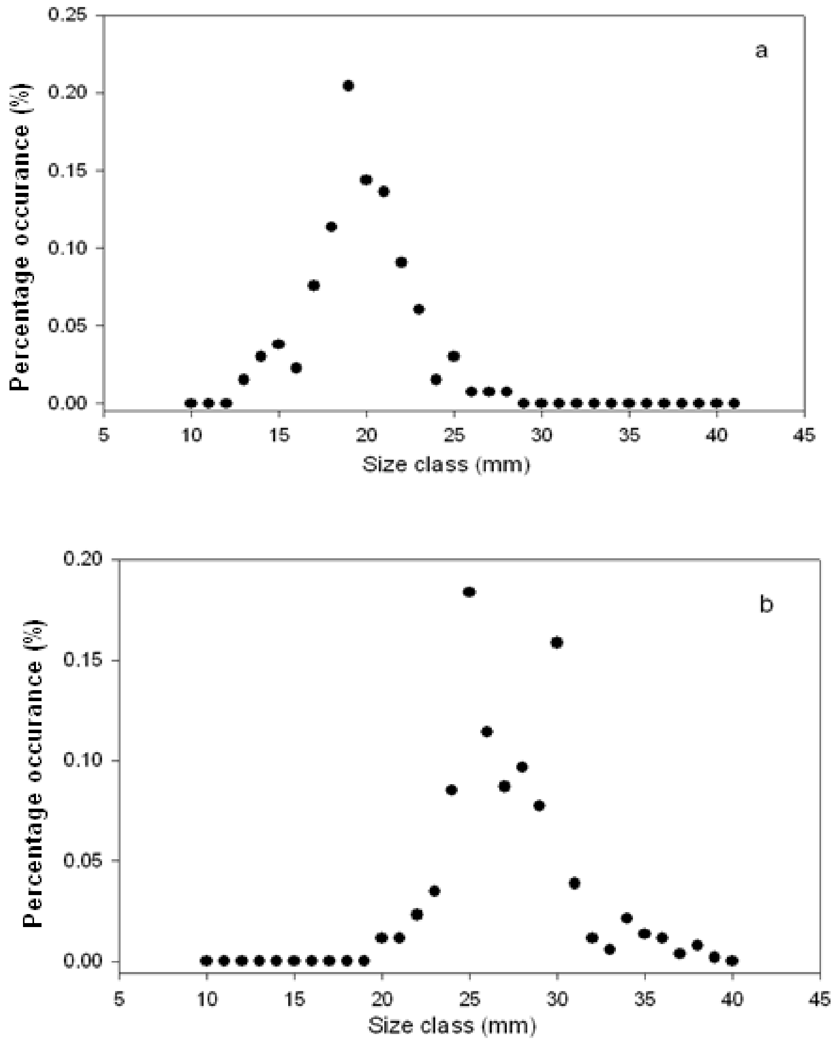


Figure 8. Size distribution graphs for cockle size (width mm) (a) sampled intertidally before fishing activity and (b)during fishing activity from the eco-elevator harvester.

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 physical disturbance of the sediment could be seen with the presence of tracks directly after fishing (figure 1b), although there was a notable reduction in the prominence of the track marks after just 10 days (figure 1c). The visual disturbance however, did not correlate with any significant change in particle (phi) size in the treatment plots compared to controls as a result of fishing activity. The sediment found across the experimental area was medium "well sorted" sand; sand flat habitats have been demonstrated to recover quicker from disturbance then other substratum (Collie et al. 2000). Reductions in grain size have however, been shown to result in a negative relationship with the abundance of settling bivalve larvae, macrofauna diversity and abundance (Piersma et al. 2001, Thrush et al. 2003).

The remaining abiotic factors indicated that organic concentration and permeability were the same throughout the plots in the baseline survey. Furthermore fishing activity did not cause a change in organic concentration within the sediment or the permeability of the sediment. There was however an observed increase in the permeability of the sediment in the control plots and in the second treatment plot after cockle fishing had been occurring in comparison with the baseline survey. 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 probably attributed to the weather conditions which occurred during the month of July.

Significant interactions were, however, recorded for organic concentration between plot and treatment, with control plot 1 having a greater organic concentration than the fished plot and the reverse occurring at plot 2 after fishing activity. Estuaries are characterised by high natural spatial and temporal variation (Ysebaert & Herman 2002), the significant differences in organic concentrations are therefore a result of this variation. These interactions between plot and treatment, and in the case of macrofauna composition between site plot and treatment, were seen throughout the study. However they were not a result of fishing activity and were therefore not of relevance to the study.

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). Although it does not occur in this study, fishing activity could potentially reduce the organic concentration by altering the topography of the sediment which in turn could have a negative effect on macrofauna recruitment (Dernie et al. 2003).

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). More detailed sediment analysis would be beneficial to provide more pronounced detail into the potential impact of the eco-elevator harvester on the interaction between sediment and macrofauna assemblage, however due to time constraints this was not feasible with the present study.

Macrofauna assemblage 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, diversity or the number of individuals (abundance). The eco-elevator harvester did not result in negative widespread changes (e.g. reduction in diversity or abundance) in the intertidal sandy habitat in the sample time period unlike other types of mechanical harvesting methods. Studies into other mechanical methods, such as suction dredging, have resulted in high mortality of macrofauna, loss of species diversity and abundance in similar time periods (Hall & Harding 1997, Spencer et al. 1998). However intertidal sand habitats have been demonstrated to withstand disturbances without a significant negative effect to the macrofauna assemblage in comparison with other substrates and can tolerate three fishing disturbances a year (Collie et al. 2000). The intertidal sand flat at the Exe estuary is exposed to frequent disturbance, both naturally from currents and tidal forces and from other mechanisms, such as anglers digging for bait worms. The macrofauna species therefore could have adapted to regular disturbance as suggested by the intermediate disturbance hypothesis (Connell 1978), and thus not be negatively effected by the fishing activity. Alternatively, there is the possibility that the macrofauna composition could exist in an altered state prior to the cockle fishing due to the regular disturbance, such as trampling, by other users (Collie et al. 2000).

The baseline survey indicated that there was no significant difference in the macrofauna diversity, number of individuals or species number prior to fishing activity however PRIMER provided an insight into any interactions occurring. 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 species diversity and abundance 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 species. Alternatively reduction could be a result of smothering of organisms by finer sediment deposited from the channel (Meridet et al. 1996).

Key species were examined to assess any impact at a species level. The intertidal sand bank was mainly dominated by the species *Hydrobia ulvae* in both of the surveys, with the baseline producing a visibly greater abundance than that after cockle fishing. This could be an indication of a negative interaction between *H.ulvae* abundance and fishing; however there were no significant differences between the control and treatment in the second survey which ruled out fishing as a probable cause. Therefore, reduction in abundance is probably a result of other factors such as predation by birds. *H.ulvae* is a highly mobile species which will actively seek out food sources and protection (Patrício et al. 2009). Caution has been highlighted in previous research when examining dominant and mobile species in the context of impact. Dominance by one or two species in estuaries are more common than in coastal habitats as inter-specific competition is reduced as a result of natural environment variability adaptation (Elliott & Quintino 2007, Patrício et al. 2009).

The polychaete *Pigospio elegans* was also examined in terms of abundance as it is an opportunistic species with the ability to quickly re-colonise disturbed areas (Grassle & Grassle 1974), therefore, polychaete *Pigospio elegans* are good indicators for impact. However no difference was recorded for the species between treatment and control plots after fishing had commenced.

P.elegans is also 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).

The greater abundance of *C.arenarium* after fishing activity than in the baseline survey could suggest that disturbance is occurring. Highly mobile amphipods can re-colonise an area quickly in the short term before superior competitive species by brooding offspring rather than planktonic reproduction (Conlan 1994). However the lack of significance between the control and treatment plots suggests that the higher abundance of *C.arenarium* is most likely attributable to their reproductive cycle with the species being more mobile during mating season which occurred during sampling time (Conlan 1994).

The lack of impact by the eco-elevator harvester on macrofauna composition, number of individuals, species and diversity in the second survey could be the result of movement into the fished area either actively for mobile species such as *H.ulvae* or passively as a result of the mixing of the disturbed and undisturbed sediment known as the "dilution effect" (Hall et al. 1990). The adverse weather conditions seen for the month of July could make this hypothesis plausible as weather conditions increase the mixing of sediments. However Hall & Harding (1997) noted that storm events would be needed for large scale sediments which despite unfavourable weather conditions, did not occur during the study.

Another explanation for lack of impact is the effect of seasonality on recovery time. The secondary survey was carried out in peak summer when many species are at the peak of their recruitment cycles such as the species *C.arenarium* (Conlan 1994) therefore recovery could be occurring in "real time" (Hall and Harding 1997). Although abundances in general were lower in the secondary survey making this inconclusive.

The study was limited by time and activity with fishing only being carried out once before sampling in the second treatment plot due to uncontrolled circumstances. Although some previous studies 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), the techniques used were more intensive than in this study. To obtain a true reflective picture of any effects of the eco-elevator harvester and any interactions on species diversity, abundance or macrofauna composition, the study area needs to be examined after the 12 month trial.

Cockle size and abundance

The cockle fishing did not have an impact on cockle abundance or size, although fishing had only occurred on a few occasions by the second sampling date therefore the long-term implication is still unclear. 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. The significant result between treatment and plot with a greater size being found at plot 1 than plot 2 is due to spatial variability as explained previously, rather than a result of fishing activity as treatment plot 1 was fished more intensively than plot 2 therefore a reduction in size would be expected in plot 1 rather than an increase.

Examination of size frequency showed that 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). However significance was not recorded for size distribution between samples taken intertidally and those collected from the boat, despite percentage distribution graphs showing a clear difference. This is most likely attributable to the type of test used rather than there being no difference between distributions, a statistical test showing distribution tails would have been better suited to the data (Foggo 2009 personal communication).

The conclusions of this study are limited by the amount of fishing activity which took place before the second survey was undertaken, with treatment plot 1 being fished on 2 separate occasions and treatment plot 2 being fished on one occasion. This was a result of adverse weather conditions causing a delay to the fisherman as the vessel used (the *Alibi*) cannot be launched if waves are above a certain height (Blood-Smythe 2009 personal communication). The experimental design of haphazard sampling meant that there was the possibility in the second survey of some core samples in treatment plots not being taken from areas directly affected by the fishing activity. However due to the nature of the fishing method used, which does not occur in parallel lines, results obtained would be reflective of interactions occurring in the area as not all areas within the treatment plots would be fished.

Conclusion

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. Sediment parameters of grain size, permeability and organic concentration were not adversely affected despite clear visual disturbances. Due to the limited time and season that this study was carried out, it is not possible to conclude that the eco-elevator harvester is not having any impact on the Exe Estuary.

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References

Atkinson PW, Clark NA, Bell MC, Dare PJ, Clark JA, Ireland PL (2003) Changes in commercially fished shellfish stocks and shorebird populations in the Wash, England. Biological Conservation 114:127-141

Billerbeck M, Werner U, Polerecky L, Walpersdorf E, deBeer D, Huettel M (2006) Surficial and deep pore water circulation governs spatial and temporal scales of nutrient recycling in intertidal sand flat sediment. Mar Ecol-Prog Ser 326:61-76

Blood-Smythe M (2009) Personal communication

Blott SJ, Pie K (2001) GRADISTAT: A Grain size distribution and statistics packagage for the analysis of unconsolidated sediments. Earth Surface Processes and Landforms 26:1237-1248

Bolam SG, Fernandes TF (2002) Dense aggregations of tube-building polychaetes: response to small-scale disturbances. Journal of Experimental Marine Biology and Ecology 269:197-222

Cammen (1982) Effects of particle size on organic content and microbial abundance within four marine sediments. Marine Ecology Progress Series 9:273-280

Camphuysen CJ, Ens BJ, Heg D, Hulscher JB, VanderMeer J, Smit CJ (1996) Oystercatcher Haematopus ostralegus winter mortality in The Netherlands: The effect of severe weather and food supply. Ardea 84A:469-492

Charette MA, Sholkovitz ER, Hansel CM (2005) Trace element cycling in a subterranean estuary: Part 1. Geochemistry of the permeable sediments. Geochimica et Cosmochimica Acta 69:2095-2109

Clarke KR, Warwick RM (1994) Changes in marine communities: An approach to statistical analysis and interpretation., Vol 2nd edition. Bourne Press

Coffen-Smout SS (1998) Shell strength in the cockle Cerastoderma edule L. under simulated fishing impacts. Fisheries Research 38:187-191

Coffen-Smout SS, Rees EIS (1999) Burrowing behaviour and dispersion of cockles Cerastoderma edule L. following simulated fishing disturbance. Fisheries Research 40:65-72

Collie JS, Hall SJ, Kaiser MJ, Poiner IR (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. Journal of Animal Ecology 69:785-798

Conlan KE (1994) Amphipod crustaceans and environmental disturbance:a review. Journal of natural history 28:519-554

Connell JH (1978) Diversity in tropical rainforests and coral reefs. Science 99:1302-1310

Cotter AJR, Walker P, Coates P, Cook W, Dare PJ (1997) Trial of a tractor dredger for cockles in Burry Inlet, South Wales. ICES J Mar Sci 54:72-83

Crothers J (1997) A key to the major groups of British marine invertebrates. Field Studies 9:1-177

Dauer D, Luckenbach M,.Rodi A (1993) Abundance biomass comparison (ABC method)- effects of an estuarine gradient, anoxic hypoxic events and contaminated sediments. Marine Biology 116:507-518

Dernie K, J.Kaiser M, A.Richardson E, M.Warwick R (2003) Recovery of soft sediment communities and habitats following physical disturbance. Journal of Experimental Marine Biology and Ecology 285-286:415-434

Eleftheriou A (2000) Marine Benthos Dynamics: Environmental and Fisheries Impacts. Journal of Marine Science 57:1299-1302

Eleftheriou A, Holme, N. A. (1984). Macrofauna techniques.In: Holme, N. A., McIntyre. A. D. (eds.) Methods for the study of marine benthos. Blackwell. Oxford, p. 140-216

Elliott M, Quintino V (2007) The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin 54:640-645

Ferns PN, Rostron DM, Siman HY (2000) Effects of mechanical cockle harvesting on intertidal communities. Journal of Applied Ecology 37:464-474

Foggo A (2009) Personal communication

Goss-Custard JD, Stillman RA, West AD, Caldow RWG, P.Triplet, Durell SEAIVd, McGrorty S (2004) When enough is not enough: shorebirds and shellfishing. Proceedings - Royal Society Biological sciences 271:233-237

Grassle JP, Grassle JF (1974) Opportumstic-life histories and genetic systems in marine benthic polychaetes. Journal of Marine Research 32:253-284

Hall S, Basford D, Robertson M (1990) The impact of hydraulic dredging for razor clams *Ensis sp* on an infaunal community. Netherlands journal of sea research 27:119-125

Hall SJ (1994) Physical disturbance and marine benthic communities: life in unconsolidated sediments. Oceanography and Marine Biology Annual Review 32

Hall SJ, Harding MJC (1997) Physical disturbances and marine benthic communities: the effects of mechanical harvesting of cockles on non-target benthic infauna. Journal of Applied Ecology 34:497-517

Hartley R (2009) Personal communication

Hayward PJ, Ryland JS (1995) Handbook of the marine fauna of North West Europe, Vol. Oxford University Press

Hinndink J (2003) Effects of suction-dredging for cockles on non-target fauna in the Wadden Sea. Journal of Sea Research 50:315-323

Howard A (1999) Cockle farming takes off. Shell fish news 8:5-6

Huettel M, Ziebis W, Forster S (1996) Flow-induced uptake of particulate matter in permeable sediments. Limnol Oceanogr 41:309-322

Huettel M, Ziebis W, Forster S, Luther GW (1998) Advective transport affecting metal and nutrient distributions and interfacial fluxes in permeable sediments. Geochimica et Cosmochimica Acta 62:613-631

Kaiser M, D.Edwards, B.Spencer (1996) Infaunal community changes as a result of commercial clam cultivation and harvesting. Aquatic living resources 9:57-63

Kaiser M, K.R.Clarke, H.Hinz, M.C.V.Austen, P.J.Somerfield, I.Karakassis (2006) Global analysis of response and recovery of benthic biota to fishing. Marine Ecology

Kraan C, Piersma T, Dekinga A, Koolhaas A, Meer JVd (2007) Dredging for edible cockles (*Cerastoderma edule*) on intertidal flats: short-term consequences of fisher patch-choice decisions for target and non-target benthic fauna. ICES Journal of Marine Science 64:1735-1742

Meridet L, M.Philippe, J.G.Wasson, J.Mathieu (1996) Spatial and temporal distribution of macroinvertebrates and trophic variables wihtin the bed of 3 streams differing by their morophology and riparian vegetation. Archiv Hydrobiologie 136:41-64

Mermillod-Blondin, R.Rosenberg, F.Francois-Carcaillet, K.Norling, L.Mauclaire (2004) Influences of bioturbation by three benthic infauna species on microbial communities and biogeochemical processes in marine sediments. Aquatic Microbial Ecology 36:271-284

Norris K, R.C.A.Bannister, P.W.Walker (1998) Changes in the number of oyster catchers *Haematopus ostralegus* wintering in the Bury Inlet in relation to the biomass of cockles *Cerastoderma edule* and its commercial exploitation. Journal of Applied Ecology 35:75-85

Patrício J, Neto JM, Teixeira H, Salas F, Marques JC (2009) The robustness of ecological indicators to detect long-term changes in the macrobenthos of estuarine systems. Marine Environmental Research 68:25-36

Philips JM, Walling DE (1999) The particle size characteristics of fine-grain channel deposits in the River Exe Basin, Devon, UK. Hydrological Processes 13:1-19

172-239

Piersma, A.Koolhaas, A.Dekinga, J.J.Beukema, R.Dekker, K.Essink (2001) Long-term indirect effects of mechanical cockle-dredging on intertidal bivalve stocks in the Wadden Sea. Journal of Applied Ecology 38:976-990

Probert P (1984) Disturbance, sediment stability, and trophic structure of soft-bottom. J Mar Res 42:893-921

Robbins T (2009) Personal communication

Rusch A, Huettel M, Forster S (2000) Particulate Organic Matter in Permeable Marine Sands--Dynamics in Time and Depth. Estuarine, Coastal and Shelf Science 51:

Salas F, Neto JM, Borja A, Marques JC (2004) Evaluation of the applicability of a marine biotic index to characterize the status of estuarine ecosystems: the case of Mondego estuary (Portugal). Ecological Indicators 4:215-225

Sheehan EV, Thompson RC, Coleman RA, Attrill MJ (2008) Positive feedback fishery: Population consequences of 'crab-tiling' on the green crab Carcinus maenas. Journal of Sea Research 60:255-261

Snelgrove PVR (1998) The biodiversity of macrofaunal organisms in marine sediments. Biodiversity and Conservation 7:1123-1132

Soulsby R (1997) Dynamics of marine sands: a manual for practical applications, Vol. Thomas Telford, London

Spencer BE, Kaiser MJ, Edwards DB (1998) Intertidal clam harvesting: benthic community change and recovery. Aquaculture Research 29:429-437

Swart JAA, Andel Jv (2008) Rethinking the interface between ecology and society. The case of the cockle controversy in the Dutch Wadden Sea. Journal of Applied Ecology 45:82–90

Thrush SF, J.E.Hewitt, A.Norkko, P.E.Nicholls, G.A.Funnell, J.I.Ellis (2003) Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. Marine Ecology Progress Series 263:101-112

Underwood AJ, Clapham MG, Richards SA, Sage MB (1998) GMAV5 for windows. Institute of Marine Ecology University of Sydney

Wynberg RP, Branch GM (1997) Trampling associated with bait-collection for sandprawns Callianassa kraussi Stebbing: effects on the biota of an intertidal sandflat. Environ Conserv 24:139-148

Ysebaert T, Herman PMJ (2002) Spatial and temporal variation in benthic macrofauna and relationships with environmental variables in an estuarine, intertidal soft-sediment environment. Mar Ecol-Prog Ser 244:105-124