2016

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Cole, R. (2016) 'Effects of dredging disturbance on seagrass coverage, sediment composition and infaunal assemblages within a SW England Zostera marina bed', The Plymouth Student Scientist, 9(1), p. 83-104.
http://hdl.handle.net/10026.1/14117

The Plymouth Student Scientist
University of Plymouth

*The Plymouth Student Scientist, 2016, **9**, (1), 83-104*

Effects of dredging disturbance on seagrass coverage, sediment composition and infaunal assemblages within a SW England *Zostera marina* bed

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Abstract

A previously surveyed *Zostera marina* (L.) bed in South West England was mapped using SCUBA in September- October 2011 to assess changes in bed extent, environmental and faunal characteristics following disturbance from a scallop dredger in 2006. Surveys indicated a continued absence of seagrass within the dredged area, suggesting no recovery five years after the dredging damage occurred, but new growth was apparent at the shoreward edges of the bed. Within the surveyed area, five conditions were sampled: bare sand (BS), the dredge scar (DS), low density seagrass (LOW: 0-50 shoots $25cm^{-2}$), medium density (MED: 50-100) and high density (HIGH: >100). Seagrass and sediment parameters were measured within each condition, and core samples for infaunal invertebrates were obtained using SCUBA divers. Seagrass and epiphyte biomass increased with higher densities of seagrass, as did the abundance and diversity of associated infauna; univariate and multivariate analyses indicated significant differences in seagrass and epiphytic biomass, sediment parameters and infaunal assemblages between conditions. Whilst assemblages within the dredge scar had lower diversity and abundance than in dense seagrass, these measures were higher than in bare sand. The assemblage composition was more similar to seagrass conditions than bare sand, potentially due to fine sediment material present within the DS condition, indicating the existence of seagrass fauna within the scar. Continued disturbance of this site through anchoring may be reducing the rate of recovery, however, implying the need for protection and future management strategies for this seagrass bed.

Introduction

Seagrass systems are among the most productive autotrophic communities in the world (Orth et al., 2006a). Seagrass is known to play an important role in biodiversity enhancement and provides physical structures on relatively featureless bottoms, enhancing epifaunal community diversity, biomass and primary and secondary production (Duffy, 2006). Higher abundance of infauna and epifauna are associated with greater densities and biomass of seagrass (Webster et al., 1998; Matilla et al., 1999; Fredriksen et al., 2010). Also, microscopic epiphytic organisms and alga grow on seagrass blades supplying a valuable food source for epifaunal grazers e.g. crustaceans and fish (Duffy, 2006). Other key ecological roles include creating nurseries for commercial fisheries species (Heck et al., 1989; Matilla et al., 1999), improvement of water quality through accumulation of contaminants and heavy metals and the release of oxygen (Francois et al., 1989; Moore 2004). Sediment stabilisation using complex rhizome mats and turbidity reduction through canopies (Short & Wyllie-Echeverria, 1996; Newell & Koch, 2004) and contributions to carbon nutrient cycles and facilitating organic matter (Duarte et al., 2004; Barron et al., 2006) have also been documented.

Seagrasses have shown a global decline of 7 % yr^{-1} since 1990 (Waycott et al., 2009) and extensive research has shown the majority of seagrass loss to be human induced through a variety of direct (e.g. dredging) and indirect impacts, e.g. eutrophication (Short & Wyllie-Echeverria, 1996). Direct anthropogenic impacts actively remove, disturb and damage seagrass affecting the complex interactions between the seagrass and associated fauna and sediment within the ecosystem and surrounding areas (e.g. Bell et al., 2001; Collins et al., 2010). Anchoring is an example of a direct impact which can damage and remove seagrass biomass. This causes increased sediment erosion that can lead to the deterioration of the bed (Creed & Filho, 1999; Fancour et al., 1999). Certain fishing practices such as trawling and dredging have similar effects on seagrass with the addition of damage and removal of fauna (Guillén et al*.,* 1994; Bishop et al., 2005; see review Erftemeijer & Lewis, 2006). Fonseca et al., (1984) and Bishop et al., (2005) demonstrated that dredging for scallops in *Zostera marina* reduces plant density and biomass. Scallop dredging has been reported in seagrass beds in Fishcombe Cove, Torbay (part of Lyme Bay, SW England).

Seagrass in Torbay is a 'National and Regional Priority Habitat' under the Habitat and Species Action Plan (Torbay Coast and Countryside Trust, 2004). This action plan initiated annual surveys to map and asses the status of seagrass in Torbay. Torbay Coast and Countryside Trust (TCCT) carried out a previous survey of the Fishcombe Cove *Zostera marina* bed in 2006, labelling it at "very high risk" from anchoring (Flint, 2006), leading a map being provided to boat users to promote a voluntary 'no anchoring' zone. Following the surveys, a commercial scallop dredger was reported dredging in Fishcombe Cove (Flint, 2006). Subsequent dive surveys revealed extensive above and below ground rhizome damage. Annual survey results showed a gradual decrease in seagrass density and extent in and around the dredged area in the middle of the bed, and an increase in extent at the east and west edges (Flint, 2008). Although density and cover are fundamental methods in assessing effects of disturbance in seagrass beds, sediment composition (e.g. particle size and organic carbon content) and infaunal assemblages (e.g. abundance and diversity) can often give a wider view of the status of a seagrass bed and how it has responded to disturbance (Fonseca et al., 1984; Collins et al., 2010; Fredriksen et al., 2010).

The aims of this study were to investigate any continued effects of the scallop dredger disturbance on the Fishcombe Cove *Zostera marina* bed by analysing: 1) the extent of the *Zostera marina* bed compared to the extent prior to dredging in 2006; 2) The difference in seagrass and epiphytic biomass between different seagrass densities; 3) Variation in sediment composition (particle size and homogeneity, organic carbon content) between seagrass, bare sand and the dredging scar and 4) infaunal abundance, diversity and assemblages composition across the conditions. It is hypothesised that little or no recovery will be shown for this seagrass bed and the highest shoot density will be further away from the damaged area. Greater seagrass density and biomass is expected to show greater epiphytic biomass (Gullström et al., 2012) and a higher abundance and diversity of associated infauna (Webster et al., 1998; Fredriksen et al., 2010) compared to bare sand. The sediment in the seagrass conditions are expected to be finer and hold more organic carbon than bare sand (Webster et al., 1998; Collins et al., 2010). It is hypothesised that the physical and biological characteristics within the dredging scar will be most like bare sand.

Materials and Methods

Study site, field surveys and sample collection

The study site location of field surveys and sample collection was Fishcombe Cove in Torbay, SW England (50° 24'10" N, 003° 31'20" W; Figure 1a). The small cove, approximately 220 m from west to east has a depth range of 1-3 m chart datum (C. D) and is sheltered from prevailing winds (at risk from NE storms) by surrounding steep rock and woodland. The seabed is mixed substrate of sand, gravel, pebble, algal turf and *Zostera marina* seagrass. There are two small sandy pebble beaches on the south and west sides and an approximately 160 m wide northerly cove entrance.

The area is very popular with recreational boat users and fishermen, and a 3 hectare mussel farm is situated approximately 200 m adjacent to the cove entrance. A small café on the southerly beach and a nearby holiday park make the cove a very popular tourist site which gets particularly busy in the summer months.

All survey work was undertaken from mid-September to early October 2011. An initial snorkel survey was completed to acquire co-ordinates of the extent of the bed using GARMIN e-trex hand held GPS (WGS84) in a waterproof case, for mapping and to plan where the subsequent dive surveys would take place. All boat dives were carried out by Plymouth University HSE Commercial PRO SCUBA qualified divers. Three 100 m transects were set from the eastern seagrass bed edge to the western edge and marked at 2 m increments (T1, T2 and T3; Figure 1a). The transects were laid approximately 10 m apart, ensuring the northerly transect ran through the deeper edge to acquire a more detailed survey of the bed edge, as logistical limitations prevented this in the snorkel survey. Divers performed a primary survey of the density and area of the seagrass, recording a density score of low (0- 50 shoots), medium (50- 100 shoots) and high (> 100 shoots) every 2 m within three 25 x 25 cm quadrats (625 cm^2 area), similar to Webster et al., (1998). From this survey it was possible to locate sample sites for collection of seagrass, sediment and infaunal cores, and use density location records to construct a density map of the bed (Figure 1b). The sampled conditions were: bare sand around the bed (BS), the dredge scar in the middle of the bed made up of mostly algal turf (DS; Figure 1a), low density (LOW), medium density (MED) and high density (HIGH) seagrass bed. Five samples of seagrass were collected from within the 3 varying density conditions by placing a 25 x 25 cm quadrat with an attached 0.5 mm mesh bag to cover seagrass and epiphytes. Leaves were cuts at the ligula (blade-sheath interface of the shoots) to

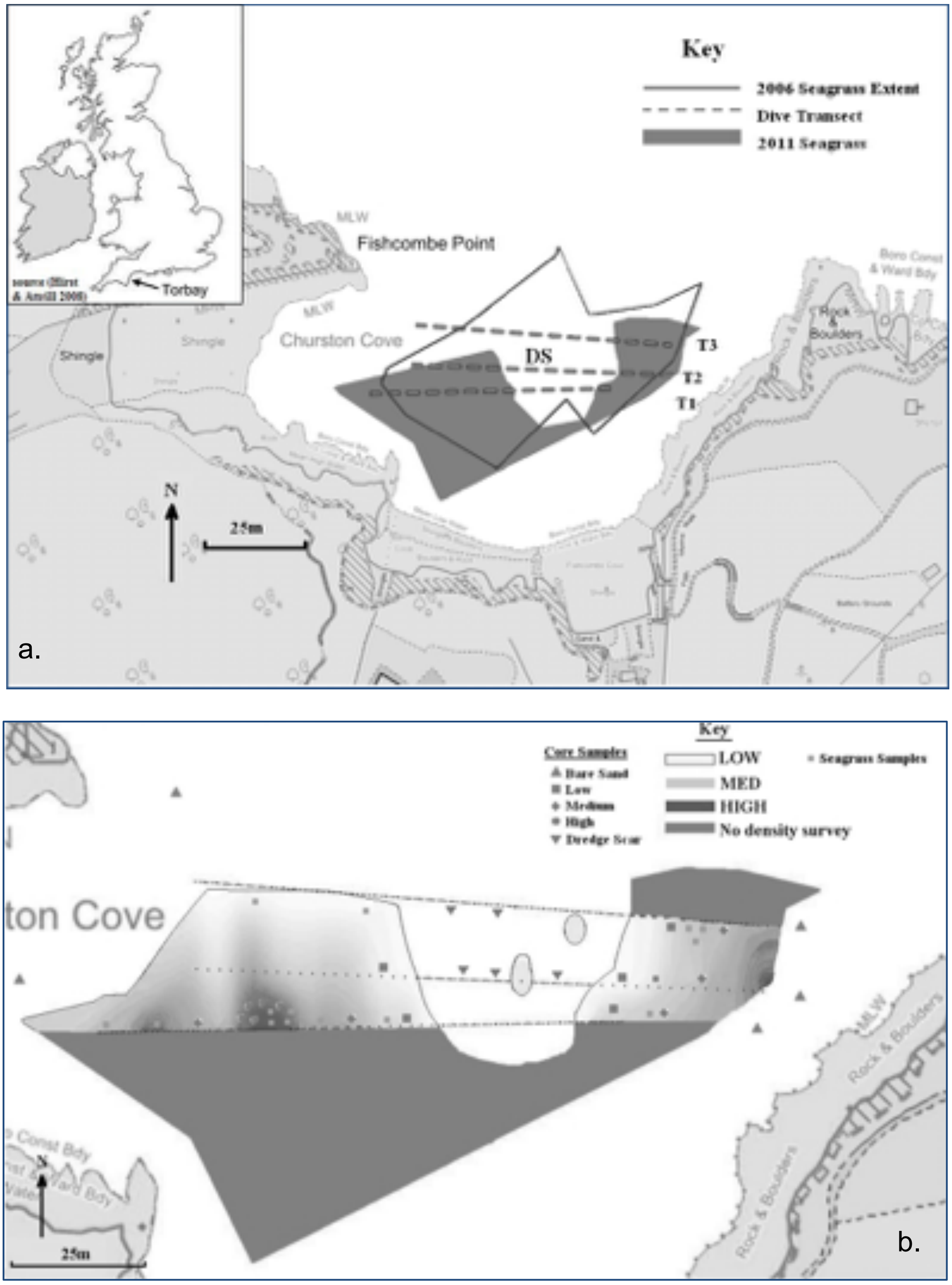

Figure 1: a. Extents of *Zostera marina* in Fishcombe Cove showing the dredge scar site (DS) and the three dive survey transects (T1, T2 and T3). b. Density of seagrass within the surveyed dive transects, showing approximate locations of where samples were taken (see key). Mean above ground seagrass biomass (dry weight g/ 25cm^2) values were used to

represent density of LOW, MED and HIGH conditions and plotted on the map according to dive survey records of density location within the bed. Density was not recorded outside of the transects.

reduce damage to the slow growing rhizome, and collected in the bag. At the surface, samples were rinsed into large polythene bags with sea water resulting in 15 samples. PVC plastic cylindrical corers 10 x 20 cm (1570cm volume) were used to collect 5 infaunal cores randomly within each of the five conditions (Figure 1b). Corers were pushed 20 cm deep into the sediment, or as close to 20 cm as possible when encountering rocks or thick rhizome mat. Divers held the core in place while transferring into large polythene bags underwater, securing them and returning to the surface. Three sediment samples were collected in 120 ml pots randomly within the conditions. All samples were stored in a cool place on the boat until transport to the laboratory. In the laboratory, each core and seagrass sample was rinsed on a 500 µm sieve to remove seawater and silt while obtaining fauna (Hirst & Attrill, 2008) and stored in 70 % ethanol. 1 ml of 0.1 % Rose Bengal in 70 % ethanol was added to infaunal core samples staining DNA proteins and ensuring only recently living tissue was counted (Stoner, 1980). Empty mollusc shells and incomplete unidentifiable animals were not counted. Sediment sample pots were frozen until analysis.

Data collection and analysis

Each core sample was rinsed over a 500 µm sieve to remove the ethanol. Infauna were picked out of sediment by hand, numerated and identified as close to species level as possible using low power microscopy and Hayward & Ryland (1995). Species were later checked with the World Register of Marine Species for up to date accepted species names. Specimens of each species were placed in 70 % ethanol and stored. Seagrass samples were rinsed over a 500 µm sieve to remove the ethanol and any remaining silt residue. Each blade was counted, measured (length and width) and scraped with a razor blade to remove the epiphytes (Libes, 1986) onto filter paper. Epiphytes included epiphytic algae and small and microscopic organisms attached to the blades. The seagrass and epiphyte filter paper were weighed, dried at 50 °C in a drying oven for 24 hours and weighed again, determining the dry weight biomass per five 625 cm^2 area samples of each seagrass density.

Sediment analysis was performed in accordance with NMBAQC best practice (Mason, 2011). Sample pots were defrosted and dried for 48 hours in a 60 °C drying oven. The dried sediment was sieved through a series of sieves (<1 mm- 16 mm) to separate particle sizes and weighed. It was noted when small seagrass fragments were present in the sediment, but were not removed in accordance with the NMBAQC guidance. Sediment <1 mm was split into three subsamples and used for particle size analysis using a Malvern Mastersizer 2000 with wet sample unit Hydro-G software version 5.6. Total organic carbon content was measured using the Loss On Ignition method (Hirst & Attrill, 2008) where approximately 5 g of each sample was weighed, heated to 400 °C for 24 hours to burn off organic matter and weighed again. The change in weight determined the percentage organic carbon.

Data were analysed using both univariate and multivariate techniques. Seagrass, sediment and infauna data were tested for normality using Kolmogorov-Smirnov test. Data were transformed (√) where appropriate for analysis. Univariate analysis was performed using one-way ANOVA on the seagrass, sediment and infauna data within and between conditions using MINITAB 16 software. In cases where ANOVA indicated significant differences, post hoc analysis of equal variance was investigated using Tukey's test. The seagrass data were classified as mean leaf length (mm.25 cm^{-2}), mean leaf area (mm^2.25 cm^{-2}), mean above ground biomass dry weight (g.25 cm^{-2}) and mean epiphyte biomass dry weight (g.25 cm^{-2}) for each seagrass density. Sediment components were analysed as mean particle size (µm), mean sorting coefficient (µm), mean percentage fines (combining silt and clay content) and % total organic carbon content for each condition. Infauna data were analysed as mean total number of individuals, mean total number of taxa present and Shannon-Weiner (H') Diversity Index for all samples and conditions. The three most dominant phyla (Annelida, Mollusca and Arthropoda) were analysed using mean total abundance and mean total number of taxa. Further univariate analyses were undertaken using linear regression investigating the effect of above ground seagrass biomass dry weight on the epiphyte biomass dry weight.

Multivariate analyses were carried out using PRIMER V6 (Plymouth Routines in Multivariate Ecological Research version 6) package (Clarke & Warwick, 2001). To test for differences in infaunal assemblages between all five conditions, an ordination plot was produced by multi-dimensional scaling (MDS) using the ranked similarity matrix based on the Bray-Curtis similarity coefficient using fourth rooted (to balance the rare and dominant species) infauna abundance data. Significant differences in infaunal assemblages between samples within conditions were tested using one-way analysis of similarity (ANOSIM; Clarke & Green, 1988). Investigation into species contribution to similarity within samples from the same condition, and dissimilarity between samples in different conditions were achieved using the similarity percentages procedure SIMPER (Clarke, 1993) with cut off for low contributions at 90%.

Results

Seagrass components

Co-ordinates taken with the hand held GPS during the snorkel survey, and co-ordinates of the 2006 TCCT survey area were plotted on a map (Figure 1a) using RockWorks™ v. 15 software package and Digimap marine charts (http://edina.ac.uk/). The map shows the location of the three dive transects (T1, T2 and T3) and gives the approximate bed extent in 2006 and 2011 to show the change in bed shape and extent. Assuming a bed of continuous cover within mapped edges, calculations using RockPlot software within RockWorks, showed the approximate area of the bed in 2006 to be ~ 1950 m^2. The area of the bed in 2011 was shown to be only ~ 1216 m^2, despite new seagrass coverage evident towards the shore in the north east and south west areas equivalent to ~ 438 m^2. Absence of seagrass within the dredge scar in the middle and north of the bed compared to the 2006 extent would be equivalent to a ~ 296 m^2 loss. The net difference in seagrass area between 2006 and 2011 was therefore a loss of ~ 734 m^2, or 37% of the pre-existing area.

The dive surveys acquired 441 quadrat recordings of the density and presence of the seagrass within the surveyed area. The majority of the area was absent of seagrass (59 % of 441 quadrat records) and, when present, seagrass was mainly at low density (27 %). Medium and high density areas were recorded as the least frequent (10.6 %, 3.4 %). The bed was found to be clearly fragmented, with frequent damaged areas of substratum (~ 50 cm wide) within the seagrass, with the appearance of anchor scars. In keeping with the previous Torbay Coast and Countryside Trust surveys 2006-2008, the middle of the bed is less dense than the east and west edges. A map of the current density of the surveyed seagrass was constructed using the mean above ground biomass (dry weight g/ 25cm^2) values for each LOW (1.1 g/ 25cm^2), MED (1.9 g/ 25cm2) and HIGH (4.3 g/ 25cm^2) seagrass conditions and plotted on the dive transects according to quadrat scoring records (Figure 1b). The density was higher further away from the dredge scar, with the largest high density area surveyed located in the more sheltered western side of the bed.

Table 1: ANOVA results. Effect of seagrass density (low, medium, high) on (a) mean leaf area (mm^2/ 25 cm^2), (b) mean leaf length (mm/ 25 cm^2), (c) mean above ground biomass dry weight (g/ 25 cm^2) and (d) mean epiphyte biomass dry weight (g/ 25 cm^2). Significant p values in bold font.

Variable Source	**df**	**MS**	***F***	**p**
(a) Leaf area				
Density	2	389765	0.70	0.516
Error	12	558038		
Total	14			
(b) Leaf length				
Density	2	5483	1.15	0.350
Error	12	4773		
Total	14			
(c) Above ground biomass				
Density	2	1.4077	15.53	**< 0.001**
Error	12	0.0907		
Total	14			
(d) Epiphyte biomass				
Density	2	0.4035	13.29	**0.001**
Error	12	0.0304		
Total	14			

The seagrass samples indicated no significant difference in leaf area or leaf length between low, medium and high densities (Table 1a and b, Figure 2). The mean densities of blades collected per 25 cm^2 were: Low, 27.2; Medium, 49.4; High, 110.8. The mean above ground biomass data were shown to be not normal ($p < 0.05$) by

Kolmogorov-Smirnov test and so were transformed (√). There were significant differences in mean above ground biomass dry weight and mean epiphyte biomass dry weight between the sample conditions (Table 1c and d, Figure 2). Linear regression analysis showed that above ground biomass was a significant predictor of epiphyte biomass ($R^2_{adj} = 0.387$, $F_{1,13} = 9.85$, $p = 0.008$). The presence of many seed-bearing leaves within the samples was noted.

Sediment components

According to GRADISTAT results, fine-medium sand (180- 280 μm) dominates the sampled area. ANOVA showed significant differences in the analysed sediment components between the sample conditions (Table 2, Figure 3). Mean particle size data were shown to be not normal ($p < 0.05$) by Kolmogorov-Smirnov test and so were transformed (√). Mean particle size (Figure 3a) was significantly different between all conditions except DS and HIGH, which had the smallest particles; BS

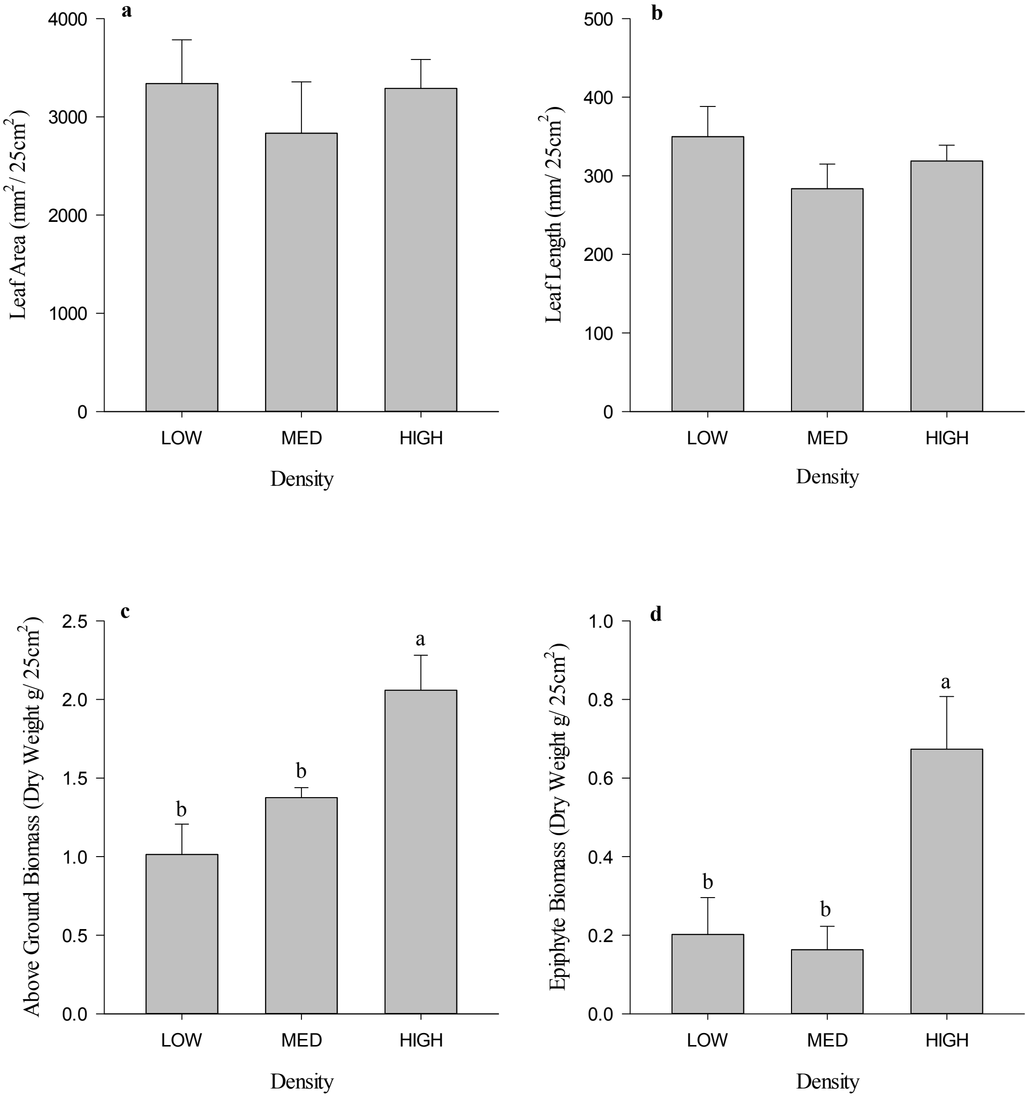

Figure 2: Effect of density on seagrass components (±SE) for samples taken within low (LOW), medium (MED) and high (HIGH) density seagrass. (a) mean leaf area (mm^2/ 25cm^2), (b) mean leaf length (mm^2/ 25cm^2), (c) mean above ground biomass dry weight (g/ 25cm^2) and (d) mean epiphyte biomass dry weight (g/ 25cm^2). Letters indicate significant groupings following post hoc tests.

had the largest. The sorting coefficient showing the level of heterogeneity within the sediment samples (Figure 3b) was significantly different between BS and LOW sediment with these having the highest values, whereas DS, HIGH and MED sediment share a small sorting value showing them to be more homogenous.

Table 2: ANOVA results. Effect of condition on (a) mean particle size (µm), (b) mean sorting coefficient (µm), (c) mean percentage fine (%) and (d) percentage of total organic carbon content (%). Significant p values in bold font.

Variable Source	df	MS	*F*	p
(a) Particle Size				
Condition	4	200.19	24.21	**< 0.001**
Error	10	8.27		
Total	14			
(b) Sorting coefficient				
Condition	4	1092842	7.37	**0.005**
Error	10	148200		
Total	14			
(c) Percentage fine				
Condition	4	9.909	20.79	**< 0.001**
Error	10	0.477		
Total	14			
(d)Organic carbon content				
Condition	4	0.604	5.16	**0.012**
Error	10	0.108		
Total	14			

The percentage of fine sediment consisting of clay and silt (Figure 3c) was significantly different between BS, LOW and HIGH sediment, but DS and MED sediment share a similar value. LOW and HIGH sediment had the greatest percentage of silt and clay, with BS recording the least. Significant differences of percentage of organic carbon content (Figure 3d) were apparent between DS and MED sediment. BS, LOW and HIGH sediment all shared a similar high value and differed from DS and MED. DS had the least organic carbon.

Infaunal assemblages

A total of 1226 individuals were counted and identified as close to species levels as possible. Some taxa were not in a suitable condition to enable identification to species level due to the mechanical processing of core samples. A total of 80 taxa were identified including 49 species, 17 to genus level, 10 to family level, 2 to class level and 2 to phylum. The most common phyla were Annelida (37 taxa), Mollusca (27 taxa) and Arthropoda (12 taxa). Three other phyla were identified: Sipuncula (1 taxon), Cnidaria (1 taxon) and Echinodermata (2 taxa). Overall, the most abundant taxa were the polychaete *Lepidonotus* sp. (146 individuals), the crustacean *Leptochelia* sp. (138) and Ostracoda sp. (89). Kolmogorov-Smirnov test revealed mean total arthropod abundance and mean total number of arthropod taxa data to be not normal ($p < 0.05$) and so were transformed (√). ANOVA results (Table 3) show significant differences in mean number of individual organisms (a), mean number of taxa (b) and the Shannon-Weiner diversity index (c) between conditions.

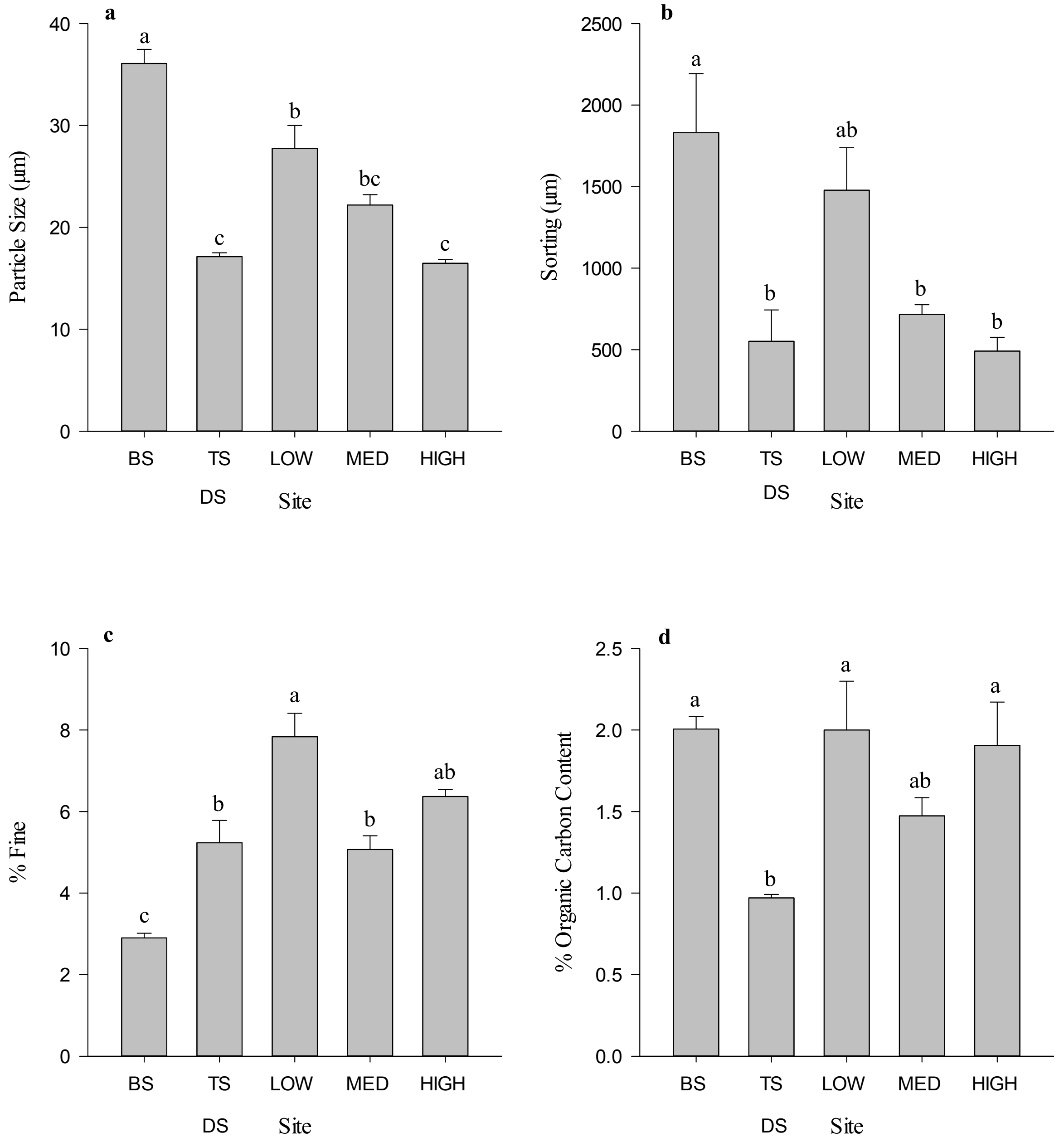

Figure 3: Effect of condition on sediment components (±SE) for samples taken within bare sand (BS), dredge scar (DS), low density seagrass (LOW), medium density (MED) and high density (HIGH). (a) mean particle size (μm), (b) mean sorting coefficient (μm), (c) mean percentage fine (%) and (d) percentage of total organic carbon content (%). Letters indicate significant groupings following post hoc tests.

There was a constant increase in abundance and number of taxa from BS-HIGH (Figure 4a, b) and Shannon-Weiner Diversity Index values showed significant differences between conditions (Figure 4c), with the greatest differences overall evident between BS and HIGH.

Table 3: ANOVA results. Effect of condition on (a) mean number of individual organisms per core, (b) mean number of taxa identified per core and (c) the Shannon-Weiner Diversity Index.

Variable Source	**df**	**MS**	***F***	**p**
(a)Number of individuals				
Condition	4	3699	6.40	**0.002**
Error	20	578		
Total	24			
(b) Number of taxa				
Condition	4	228.3	10.96	**< 0.001**
Error	20	20.8		
Total	24			
(c)Shannon-Weiner diversity index				
Condition	4	0.6277	7.51	**0.001**
Error	20	0.0836		
Total	24			

ANOVA results for the three most common phyla (Table 4) showed significant effects of condition type on abundance and number of taxa, except for mean number of mollusc taxa. Significant differences in abundance of phyla between conditions can be seen in Figure 5. There was a constant increase in annelid abundance from BS-HIGH (Figure 5a). A general increase from BS-HIGH in number of annelid taxa (Figure 5b) showed the greatest difference between BS and all seagrass conditions. Mollusc abundance showed an overall increase from BS-HIGH with some variation (Figure 5c), while no significant difference between conditions was found for number of Mollusca taxa (Figure 5d). Arthropod abundance showed a constant increase from BS-HIGH (Figure 5e) while number of arthropod taxa showed significant differences in MED and HIGH but not BS-LOW (Figure 5f). There was an overall trend of an

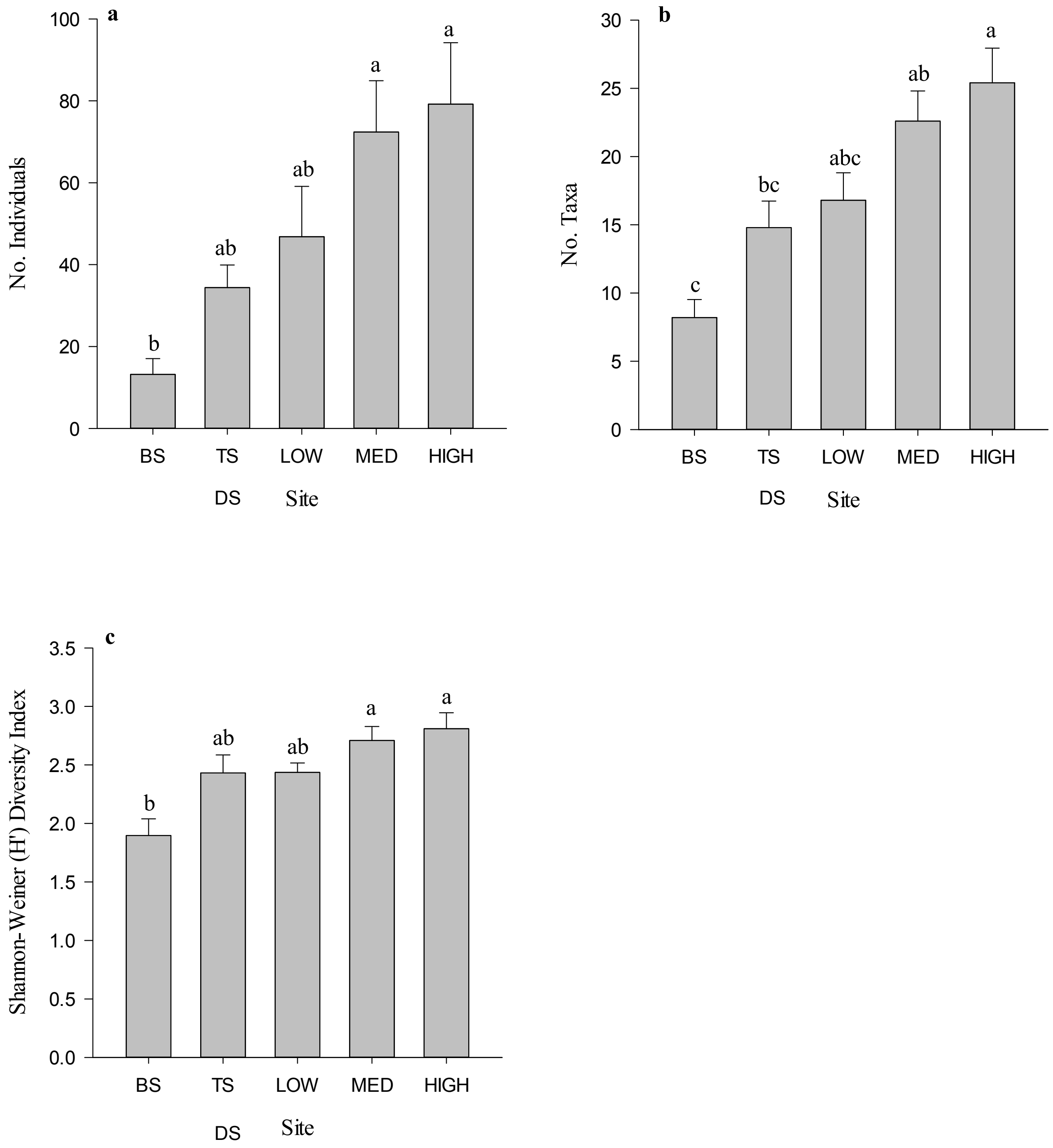

Figure 4: Effect of condition on infaunal components (±SE). Mean number of individual organisms per core (a), mean number of taxa identified per core (b) and the Shannon-Weiner (H') Diversity Index (c). Letters indicate significant groupings following post hoc tests

increase of infauna abundance with seagrass cover, while number of taxa varied between conditions from BS-HIGH (Figure 10). Separation of conditions was evident within the MDS ordination plot of infauna abundance data (Figure 6); ANOSIM indicated this to be a significant difference in infaunal assemblages between conditions (Global R = 0.526; p = 0.001). Pairwise tests showed a significant difference between all conditions (Global R = 0.216 to 0.848; p = 0.008 to 0.04, Figure 6). SIMPER showed high average dissimilarity values between BS and the other conditions (minimum 74.54 %), but low dissimilarity values between LOW, MED and HIGH (58.48 % to 63.85 %) instigated the investigation into analysing all seagrass conditions together (SG) with BS and DS (Table 5). This analysis showed higher average dissimilarity values between BS and DS (76.61 %), BS and SG (76.11 %) and DS and SG (64.81 %). The three species with the largest contribution to dissimilarities were the bivalve mollusc *Moerella donacina* (7.68 %), the annelid Oligochaete sp. a (6.38 %) and the polychaete *Magelona filiformis* (6.17 %), all contributing to the differences between BS and DS. The largest contribution to dissimilarity between BS and SG was 4.24 % and between DS and SG was 4.48 %, both contributed by the polychaete scale worm *Lepidonotus* sp., the most abundant overall taxon.

Table 4: ANOVA results showing effect of condition on common phyla. (a) mean annelid abundance per core, (b) mean number of annelid taxa per core, (c) mean mollusc abundance per core, (d) mean number of mollusc taxa per core, (e) mean arthropod abundance per core and (f) mean number of arthropod taxa per core.

Variable Source	**df**	**MS**	***F***	**p**
(a)Annelid abundance				
Condition	4	706	3.26	**0.033**
Error	20	217		
Total	24			
(b)Number of annelid				
Condition	4	70.3	5.34	**0.004**
Error	20	13.2		
Total	24			
(c)Mollusc abundance				
Condition	4	202.5	3.51	**0.025**
Error	20	57.8		
Total	24			
(d)Number of Mollusc				
Condition	4	14.50	1.75	0.179
Error	20	8.30		
Total	24			
(e)Arthropod abundance				
Condition	4	9.89	4.91	**0.006**
Error	20	2.01		
Total	24			
(f)Number of arthropod				
Condition	4	1.158	5.78	**0.003**
Error	20	0.200		
Total	24			

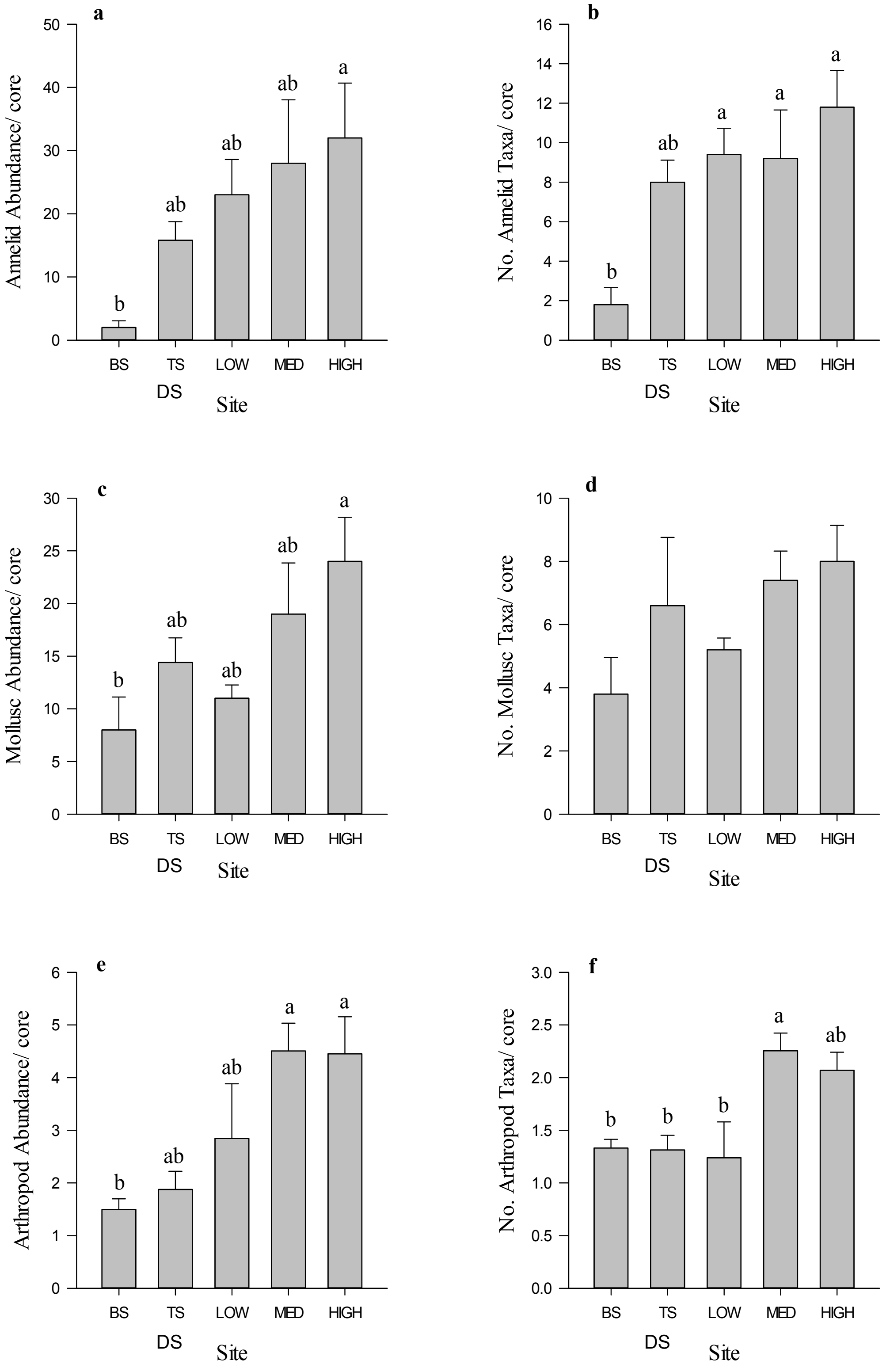

Figure 5: Effect of condition on abundance and diversity of common phyla (±SE). (a) mean annelid abundance per core, (b) mean number of annelid taxa per core, (c) mean mollusc abundance per core, (d) mean number of mollusc taxa per core, (e) mean arthropod abundance per core and (f) mean number of arthropod taxa per core. Letters indicate significant groupings following post hoc tests.

Discussion

This investigation achieved the initial aim of comparing the extent of Fishcombe Cove *Zostera marina* bed between 2006 and 2011. Disturbance by the scallop dredger, and possibly other disturbances such as anchoring, have caused a loss of cover which has continued since 2006. The presence of some new seagrass areas, however, is a positive sign the bed is extending to areas of lower disturbance closer to the shore and towards Churston Cove (Figure 1a). The Torbay Biodiversity Action Plan 2004-2012 Marine Survey Records mention a dense *Z. marina* bed just off Fishcombe beach but not Churston Cove beach, suggesting it has changed growth direction since 2004 (Torbay Coast and Countryside Trust, 2004). Although it was possible to compare the extent of the bed between 2006 and 2011, it was noted at the beginning of this study it would not be possible to compare the exact change density of *Z. marina* since 2006 due to differing methods of density scoring (Flint, 2008). Despite this, the same trend was apparent where density was lower near to the dredged area and higher at the east and west edges; there has been no significant signs of re-growth of seagrass in the dredged area five years after the disturbance event. Overall, the surveyed Fishcombe Cove *Zostera marina* bed was observed as mainly low density and fairly fragmented. The density has shown a similar trend to that reported by the previous surveys by TCCT (Flint, 2008).

No significant difference was found in leaf length or blade area between different densities, but as expected, seagrass biomass increased with density and epiphyte biomass increased with greater seagrass biomass. The observation of larger epiphytes within samples from HIGH may contribute to the increased epiphyte biomass. It is documented that greater seagrass complexity, i.e. greater density, benefits small epifaunal organisms in the reduction of predation from larger epifauna (Heck & Orth, 2006), denser beds providing a greater amount of shelter and leaf area for epiphytes to reside (Connolly et al., 1995; Gullström et al., 2012). Attrill et al (2000), however, demonstrated that any seagrass biomass increases related with seagrass density is simply an area effect, so the same mechanism is likely to be operating to elevate epiphyte biomass where more seagrass exists.

Fine- medium sand dominated the conditions within the sampled area. Particle size decreased with increasing seagrass density, with bare sand having the largest particles (Figure 3a). Seagrass is known to trap small sediment particles between shoots and within the underlying complex rhizome mat, so our results are in keeping with those from other studies comparing sediment between bare sand to seagrass (Collins et al., 2010; Fredriksen et al., 2010). A similar trend was found with the sorting coefficient (Figure 3b) showing a decrease in heterogeneity with an increase in seagrass density also found by Webster et al., (1998). This previous study shows that percentage of silt-clay material (% fine) increases with seagrass density. In contrast, this study showed a variation between densities with LOW containing the most silt-clay (Figure 3c) possibly transported from outside areas. Similar values were found for organic carbon content within the seagrass so this may indicate a relationship between the two (Figure 3d). Unlike some other studies, e.g. Barron et al., (2006) and Collins et al., (2010), samples from BS had high organic carbon content similar to that of the seagrass sediment. However, this has been reported before (Fonseca et al., 1984; Fredriksen et al., 2010). It’s suggested that spaces between larger sediment particles in bare sand can accommodate more organic material. Seagrass beds produce abundant organic carbon matter which can be transported from the bed by water movements (Hemminga & Duarte, 2000).

The DS sediment was significantly grouped with that sampled from the seagrass sites but never with sediment from bare sand (Figure 3) showing a similarity in the measured characteristics. The DS and BS sediment was sampled only ~ 2-3 m from the main seagrass bed yet the differences were significant, so the sampling location of the DS sediment may not justify its similarity with that from the seagrass. Assuming the seagrass sediment characteristics are those favourable for growth and survival of the seagrass, similarities shown by the DS sediment suggests there could still be some potential for future growth in this area.

Compared to Hirst & Attrill (2008), who found 54 taxa at similar levels of identification, this study found a fairly wide range of diversity (80 taxa) for a small Torbay seagrass bed, indicating diversity possibly differs between intertidal and subtidal beds in this area. More taxa may have been present but the poor condition of some individuals made it impossible to identify to species level. A consistent increase in abundance and diversity followed the increase in seagrass density (Figure 4). Significant differences between bare sand and high density seagrass has been documented by several investigations showing that sediment vegetated by seagrass supports greater densities of infauna than unvegetated sediments (e.g. Stoner, 1980; Webster et al., 1998; Hirst & Attrill, 2008; Collins et al., 2010; Fredriksen et al., 2010). Post-hoc tests revealed that DS faunal abundance and diversity was grouped with both BS and seagrass conditions, presenting values between the two (Figure 4). The most common taxa in DS were *Dosinia* sp. and *Magelona filiformis,* which burrow in sandy-muddy bottoms (Hayward & Ryland, 1995) and were not found in abundance in other sites. The most common phyla found were similar to those found by other studies of this nature (Webster et al., 1998; Hirst & Attrill, 2000; Fredriksen et al., 2010). These phyla are often associated with seagrass (Hayward & Ryland, 1995) and most measures of faunal abundance and diversity increased with seagrass density; this was not apparent for Mollusca diversity, however, which showed no significant difference between conditions (Figure 5d, Table 4d), perhaps due to a high variation between samples and high diversity within all conditions.

Multivariate analysis confirmed there are significant differences in infaunal assemblages between surveyed conditions (Figure 6) with the greatest difference between BS and DS, and the least between DS and seagrass (SG); which was to be expected considering DS has shown more similarities to seagrass in this study. The most significant contributing species to dissimilarities (Table 5) were *Morella donacina,* Oligochaeta sp. A and *Magelona filiformis,* which were all abundant in DS. *M. donacina* was the only one also abundant in any other condition (HIGH). This species is a burrowing deposit and suspension feeder (Appeltans et al., 2011) so may prefer the smaller particle size of the DS and HIGH conditions (Figure 3a) where it was found most abundant. This is possibly aided by the ability to burrow into rhizomes matrixes with ease due to its slender shape. *Lepidonotus* sp. was the contributed most to the difference between SG and the two other conditions. *Lepidonotus* sp. and *Leptochelia* sp. were by far the most abundant taxa overall (146 and 138 individuals), with much greater abundances in the HIGH seagrass condition, perhaps benefiting from the higher structural complexity due to elevated seagrass biomass and increased prey for *Lepidonotus*. The crustacean genus *Leptochelia* sp. has been found to be most abundant in long-established *Zostera* beds (Posey, 1988), which is the case for the HIGH, less disturbed seagrass within this study.

Table 5: SIMPER analysis results for top ten contributing species to most dissimilarity between samples taken from bare sand (BS), dredge scar (DS) and collective seagrass (SG). Av. Abundance: average abundance per core; Av. Dissimilarity: average dissimilarity; Av. D/SD: ratio of average dissimilarity to the standard deviation of dissimilarity for the taxa.

	Condition		**Av.**		**Av.**		**Contributio**
	a	***b***	***a***	***b***			
Moerella	BS	DS	0.00	1.44	5.88	3.37	7.68
Oligochatete sp.	BS	DS	0.00	1.22	4.89	5.75	6.38
Magelona	BS	DS	0.20	1.40	4.73	2.11	6.17
Magelona alleni	BS	DS	0.00	0.91	3.92	1.71	5.11
Nephtys sp.	BS	DS	0.00	0.94	3.58	1.85	4.67
Ampelisca sp.	BS	DS	0.40	1.16	1.16	1.37	4.12
Tharyx sp.	BS	DS	0.00	0.64	2.60	1.09	3.39
Arabella sp.	BS	DS	0.00	0.64	2.44	1.13	3.19
Tapes sp.	BS	DS	0.70	0.92	2.40	1.07	3.13
Sipunculun sp.	BS	DS	0.60	0.00	2.38	1.13	3.10
Lepidonotus sp.	BS	SG	0.40	1.36	3.23	1.51	4.24
Moerella	BS	SG	0.00	1.07	3.15	1.67	4.14
Ostracod sp.	BS	SG	0.40	1.30	2.89	1.59	3.80
Leptochelia sp.	BS	SG	0.88	1.48	2.53	1.52	3.32
Tubificoides sp.b	BS	SG	0.00	0.90	2.41	1.60	3.17
Rissoa	BS	SG	0.00	0.66	2.23	0.98	2.93
Cirratulid sp.a	BS	SG	0.00	0.79	2.21	1.29	2.90
Tapes sp.	BS	SG	0.70	1.00	2.13	1.16	2.80
Phoronis sp.	BS	SG	0.00	0.72	2.11	1.16	2.77
Abra nitida	BS	SG	0.40	0.77	1.99	1.16	2.62
Lepidonotus sp.	DS	SG	0.20	1.36	2.91	1.61	4.48
Ostracod sp.	DS	SG	0.28	1.30	2.70	1.65	4.16
Magelona	DS	SG	1.40	0.39	2.62	1.55	4.04
Leptochelia sp.	DS	SG	0.70	1.48	2.33	1.42	3.60
Tubificoides sp.b	DS	SG	0.00	0.90	1.98	1.59	3.06
Nephtys sp.	DS	SG	0.94	0.31	1.82	1.36	2.81
Oligochatete sp.a	DS	SG	1.22	0.57	1.77	1.13	2.73
Rissoa	DS	SG	0.00	0.66	1.75	0.99	2.69
Abra nitida	DS	SG	0.00	0.77	1.72	1.16	2.65
Cirratulid sp.a	DS	SG	0.24	0.79	1.71	1.22	2.64

Dense seagrass promotes high production of organic carbon, making seagrass beds sites of elevated microbial activity leading to high heterotrophic activity (Hemming &

Duarte, 2000). So considering BS showed high levels of organic carbon, it would be expected to find an abundance of infauna here. This was not the case in this study, as with many similar studies (Webster et al., 1998; Bell et al., 2001; Hirst & Attrill., 2008; Collins et al., 2010), highlighting the structural value of seagrass above and below ground.

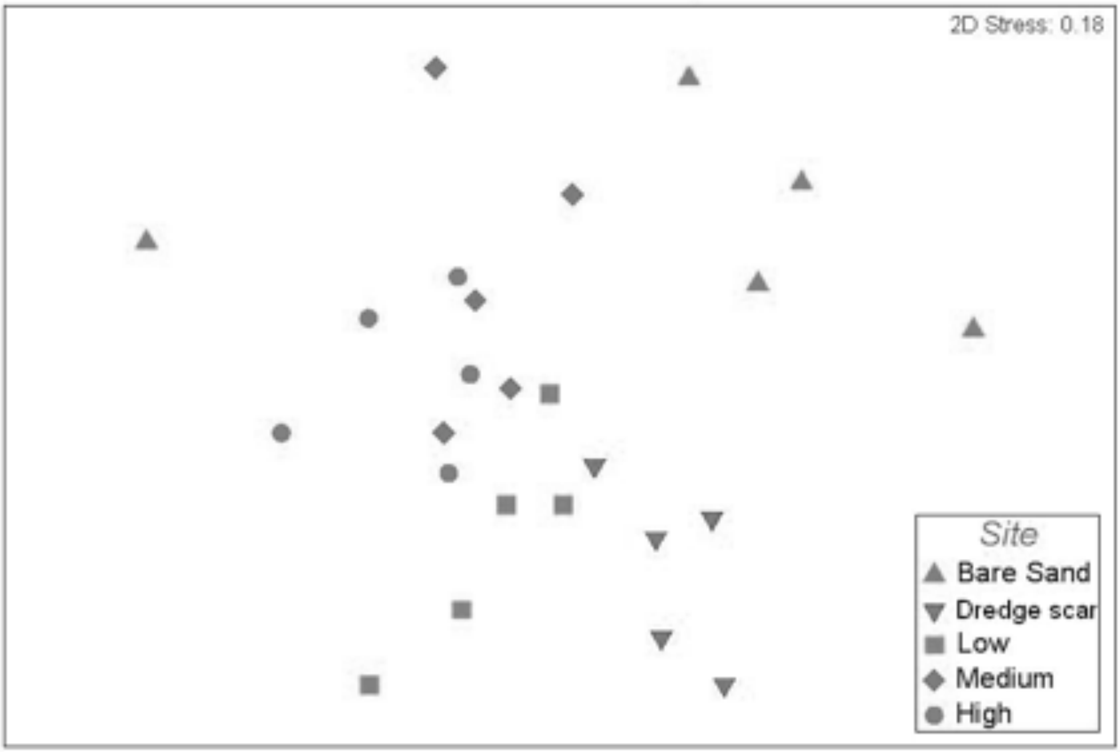

Figure 6: MDS ordination illustrating similarity between invertebrate assemblages in samples taken from different conditions within the seagrass beds (Bare Sand, Dredge Scar, Low, Med, High Seagrass).

Overall, similar significant groupings between the DS and SG conditions suggest this area would be favourable for seagrass growth, however the results from previous surveys by Flint, (2008) and this study have shown the seagrass has continued to decline in this area since 2006. Due to its location at the cove entrance where it is relatively sheltered and has a minimum depth of 1- 3 m C. D, it is an optimum site for anchoring, which has been observed to increase here in the busy summer months (TCCT and personal observation). Although there is a lack of raw data to support this; with no records of anchoring activities kept by the Brixham Harbour Master, TCCT surveys recorded this site to be at “very high risk” from anchoring compared to other seagrass beds in Torbay (Flint, 2006). It has been shown that anchoring has negative impacts of seagrass beds by reducing shoot density and disturbing or damaging underlying rhizomes (Creed & Filho, 1999; Francour et al., 1999; Milazzo et al., 2004). The majority of anchoring occurs during the summer months when tourism and boat use is high. These activities coincide with *Zostera marina*’s peak growing period of May- September. Although anchoring can cause small scale impacts, high frequency of small disturbances can increase fragmentation which can lead to coalescence of bare substrate (Montefalcone et al., 2008). This disturbance is a possible reason the seagrass has shown a lack of growth into the dredge scar from the surrounding area after 5 years. The initial event of removing the seagrass by the scallop dredger in 2006 and subsequent continual disturbance has prevented the recovery of this bed. No further reports of dredging activities have been reported in this cove since 2006, but anchoring still persists.

Recovery of *Zostera marina* has been reported to be slow (Fonseca et al., 1984; Neckles et al., 2005) so protection from disturbance and long term monitoring may increase recovery. Increased reproductive effort and extending the fertile season into autumn has been documented in *Zostera* species as a sign of stress or effort to re-colonise damaged areas (Alexandre et al., 2005; Wisehart et al., 2007). The presence of many seed bearing leaves in the samples taken at beginning of autumn suggests the seagrass in Fishcombe Cove may take part in this process. Permanent dive transects are an effective method for implementing consistent annual surveys which could aid in documenting recovery rate and changes (Kirkman et al., 1996). This study, the first of its kind on the *Zostera marina* in Fishcombe Cove and the first subtidal survey of its kind in Torbay, has provided baseline measurements which will allow for future investigations into any future changes and thus inform monitoring strategies (Kirkman et al., 1996).

Acknowledgements

Many thanks to Dominic Flint of Devon Wildlife Trust and Torbay Coast and Countryside Trust for information on previous surveys, Roger Crook and student divers of Plymouth University Marine Station for their professional diving work, Rick Parker, skipper of the Jennifer Ann survey boat and finally, the helpful technicians of Plymouth University.

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