The Impacts of commercial anchoring on maerl beds in Falmouth Bay

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Abstract

Maerl beds are slow growing biogenic habitats, providing refuge for a high diversity of organisms similar to sea grass beds. They act as nurseries for a variety of commercially important fish and shellfish species. However, maerl beds are highly susceptible to mechanical damage, such as scallop dredging or anchoring. Falmouth Bay contains large areas of maerl bed, which are heavily anchored on by large commercial vessels. This study has compared areas of anchored and unanchored maerl bed in Falmouth Bay, UK to assess the affects of commercial vessels on the maerl beds. Remarkably high amounts of live maerl were found at both sites, 8.45% surface cover on unanchored sites and 4.2% surface cover on anchored sites, though anchoring does not statistically decrease live maerl cover (p=0.961). Abundances of individuals did differ significantly between sites (t=3.6, p=0.033). However, ideal sample sizes were not achieved. As many small differences were observed, due to the natural variation in maerl beds it is likely other significant differences were not identified. Possible links are discussed, with suggestions made for improvements on future studies.

Introduction

Bays and estuaries often provide protection from powerful storm systems, allowing fragile habitats such as seagrass and maerl beds to exist within. These bays can also act as excellent natural harbors providing protection for large cargo and cruise vessels. The anchoring of large vessels, with the various associated activities, provides port towns with significant income and many jobs. However, several impacts to local biodiversity have been identified. Anchoring by its very nature mechanically interferes with the seabed, altering benthic complexity and community structure (Francour, et al. 1999; Backhurst and Cole 2000; Lloret, et al. 2008). There has been much research into small recreational boat anchoring and its respective mechanical effects on seagrass beds (Backhurst and Cole 2000; Milazzo, et al. 2004; Lloret, et al. 2008; Montefalcone, et al. 2008). Several studies have also investigated the anchoring of cruise liners on coral reefs (Smith 1988; Rogers and Garrison 2001). However, data on the effects of commercial vessel anchoring on other benthic communities, in particular maerl are limited.

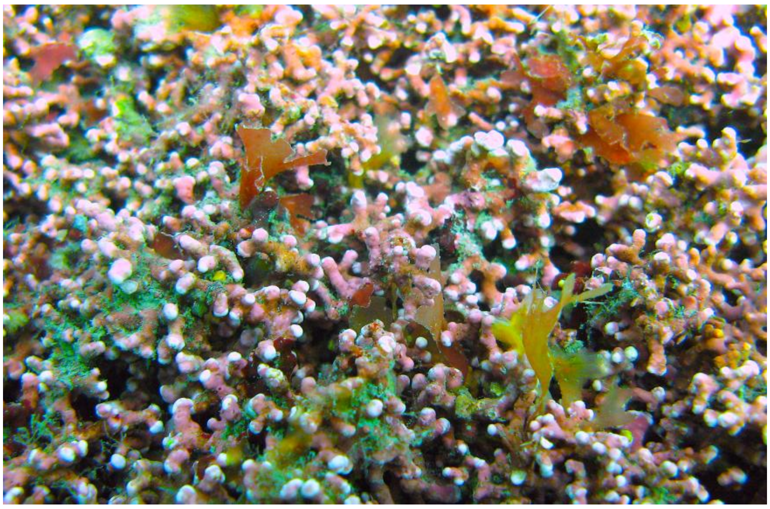

Plate 1 Live maerl from St Mawes Bank, showing open lattice-like structure. Photo: Sam Coombs

Maerl is the common name for several species of calcareous red marine algae that form free-living nodules, which are known as rhodoliths (plate 1). Rhodoliths have an exceedingly slow growth rate in temperate waters, 0.1 – 1.0 mm y^{-1} (Bosence & Wilson, 2003), though over centuries or millennia the dead skeletons of maerl accumulate to form thick beds (Hall-Spencer, et al. 2003). Depths of these accumulations vary, the deepest being >10 meters, covered with a fine layer of live rhodoliths on the surface. However, Maerl beds require specific environmental conditions. Maerl is a member of the Rhodophycaea, and is limited to shallow depths (>32 m) in the relatively turbid waters of the European continental shelf (Hall-Spencer, et al. 2003). They are generally found in areas with a moderate amount of water flow to provide adequate sunlight and prevent burial by sediments. In addition, maerl requires shelter from intense wave action in order to prevent burial or dispersal into deeper water (Hall-Spencer 1998). Due to the complex nature of rhodoliths and their skeletons, highly complex habitats are created supporting high biodiversities of benthic epifauna and infauna (Hall-Spencer 1998), which are comparable to seagrass beds (Grall & Hall-Spencer, 2003).

Maerl has limited legal protection, though the species *Lithothamnium coralloides* and *Phymatolithon calcareum* are protected under Annex V of the Habitats Directive. Maerl, however, can be protected as a habitat under Annex I of the EC Habitats Directive. In addition to supporting high biodiversities, maerl play other important ecological roles that should be considered for conservation. The complexity of maerl beds provides ideal habitat for juvenile cod, King *(Pecten maximus*) and Queen Scallops (*Aequipecten opurcularis*) and other important commercial species (Kamenos, et al. 2004)[a]. Maerl beds increase the holding capacity of localized nursery areas, so destruction of these habitats would decrease the numbers of juvenile Cod and Pollack restocking local inshore areas (Kamenos, et al. 2004)[b]

Current research into mechanical damage to maerl beds concerns dredging methods. Both hydraulic dredging for deep infauna and scallop dredging reduce structural complexity by homogenizing the area; breaking, burying and dispersing live maerl thalli. Species diversity of the immediate area decreased following dredging as a direct result of the loss of habitat structure and complexity (Hall-Spencer, et al. 2003; Hauton, Hall-Spencer, & Moore, 2003). Resulting sedimentation occurs up to 15 m away from each dredge track. These sediments smother and kill live maerl, which further reduces long term biodiversity (Hauton, Hall-Spencer, & Moore, 2003). Live maerl thalli are very vulnerable to being smothered, mortality usually caused by a combination of lack of sunlight and interactions with the sediment (Hall-Spencer & Moore, 2000; Wilson, et al, 2004; Hall-Spencer, et al. 2006). After mechanical disturbance occurs to maerl beds, quick recovery has not been observed (Hall-Spencer & Moore, 2000). Due to the increase in fine sediments, community shifts occur towards species poor muddy sediments, with a dominance of opportunistic omnivores instead of the characteristic filter feeding long-lived bivalves (Grave and Whitaker 1999).

It has been suggested that anchoring on maerl has a similar effect, though more localized, to that of dredging (Pers. Comms. Hall-Spencer). Anchors cause damage to the seabed in several different way: upon deploying the anchor and the subsequent dragging and locking in, chain abrasion to the surround area whilst at anchor and upon recovery (Walker, et al, 1989; Francour, Ganteaume, & Poulain, 1999; Milazzo, et al. 2004; Montefalcone, et al.2008). On hard substrates such as coral, anchoring significantly reduces structural complexity, as corals are damaged. This results in a loss of biodiversity, similar to the effect of scallop dredging on maerl beds (Smith, 1988). As it falls, the anchor itself crushes coral heads (Rogers & Garrison, 2001). Anchors then penetrate significantly deeper in softer sediments. On substrates like maerl, anchor penetration depths are estimated to a maximum of 2.5m (Allan, 1998). On maerl beds, the deep penetration depths are likely to interact with the characteristic deep-burrowing organisms found in the maerl beds (Hall-Spencer & Atkinson, 1999). On coral reefs, anchor chains cause further damage as the ship moves on the tides. Live coral tissue is scraped off, and entire coral heads can be removed (Rogers & Garrison, 2001). In softer sediments such as sea grass beds, circular scours from moorings in Australia have been observed, from $3m^2$ to $300m^2$ in diameter (Walker, et al. 1989). Community shifts are observed on coral reefs after anchoring occurs, with fast growing algae species taking over the area. The original coral species often do not recover or recruit to the area. For coral reefs, if recovery is to occur, the estimated time after anchoring takes place is 50+ years (Rogers & Garrison, 2001). Soft sediment communities are able to recover within 14 months of mechanical disturbance (Currie & Parry, 1996). However, maerl recovery after dredging is again estimated to be similar to coral reef times, if it is able to recover at all. (Hall-Spencer & Moore, 2000). Therefore, recovery after anchoring may also be limited.

Falmouth bay supports one of England's most pristine maerl beds, St Mawes Bank, situated in ≈7 m of water off St Mawes headland (plate 2). Here, as much as 100% live maerl cover is present, with a large biodiversity of associated flora and fauna (Perrins, et al. 1995). However, much of the surrounding area, including Falmouth Bay, is composed of reported "dead maerl" deposits (Davies & Southeran, 1995). Anchoring on St Mawes bank is prohibited, and is naturally protected by the close proximity to the Carrick roads shipping channel. Shipping, anchoring and associated activities within the bay have taken place for hundreds of years. Activities such as scallop dredging and extraction have taken place since the turn of the century, but have been restricted or prohibited in recent decades (pers. comms. FHC).

The area of interest for this study is within the area of the Fal & Helford Special Area for Conservation (SAC), under the authority of Falmouth Harbors Commissioners (FHC). Falmouth harbor commissioners are required by law to protect the interests of the SAC, and historically have always responded to environmental issues. However, the FHC are now on a new initiative to proactively invest in research into areas of concern. Data provided by the FHC from Automated Information System (AIS) transponders installed on every cargo vessel, which relays to local harbor authorities GPS location and ship information, have allowed tracking of anchoring sites within the bay (fig 1). After data suggesting that the dead maerl beds around Falmouth Bay support relatively high biodiversities, there were concerns that anchoring may have negative ecological effects on the maerl deposits. Using anchoring data provided by the FHC, it is now possible to determine whether cargo vessel anchoring has a negative impact on the dead maerl beds.

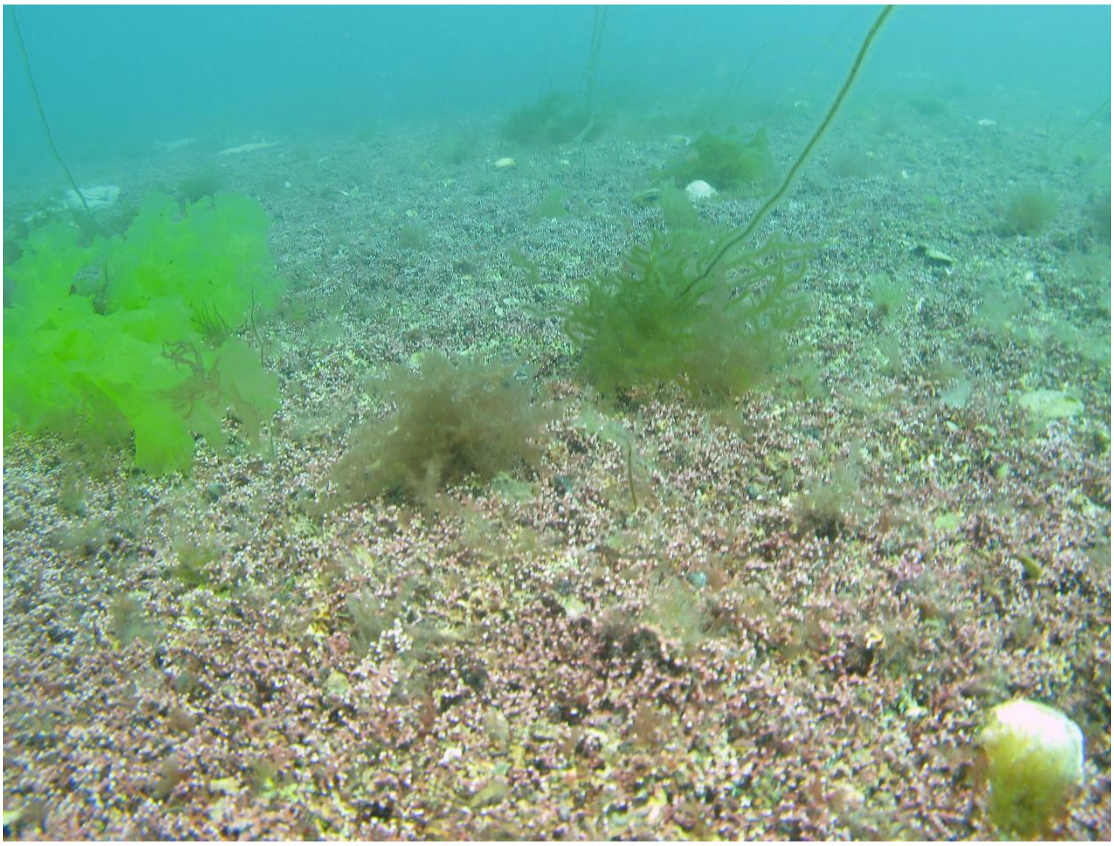

Plate 2 Example of a 'pristine' maerl bed. St Mawes Bank, Falmouth. Photo: Joseph Newton, taken during study.

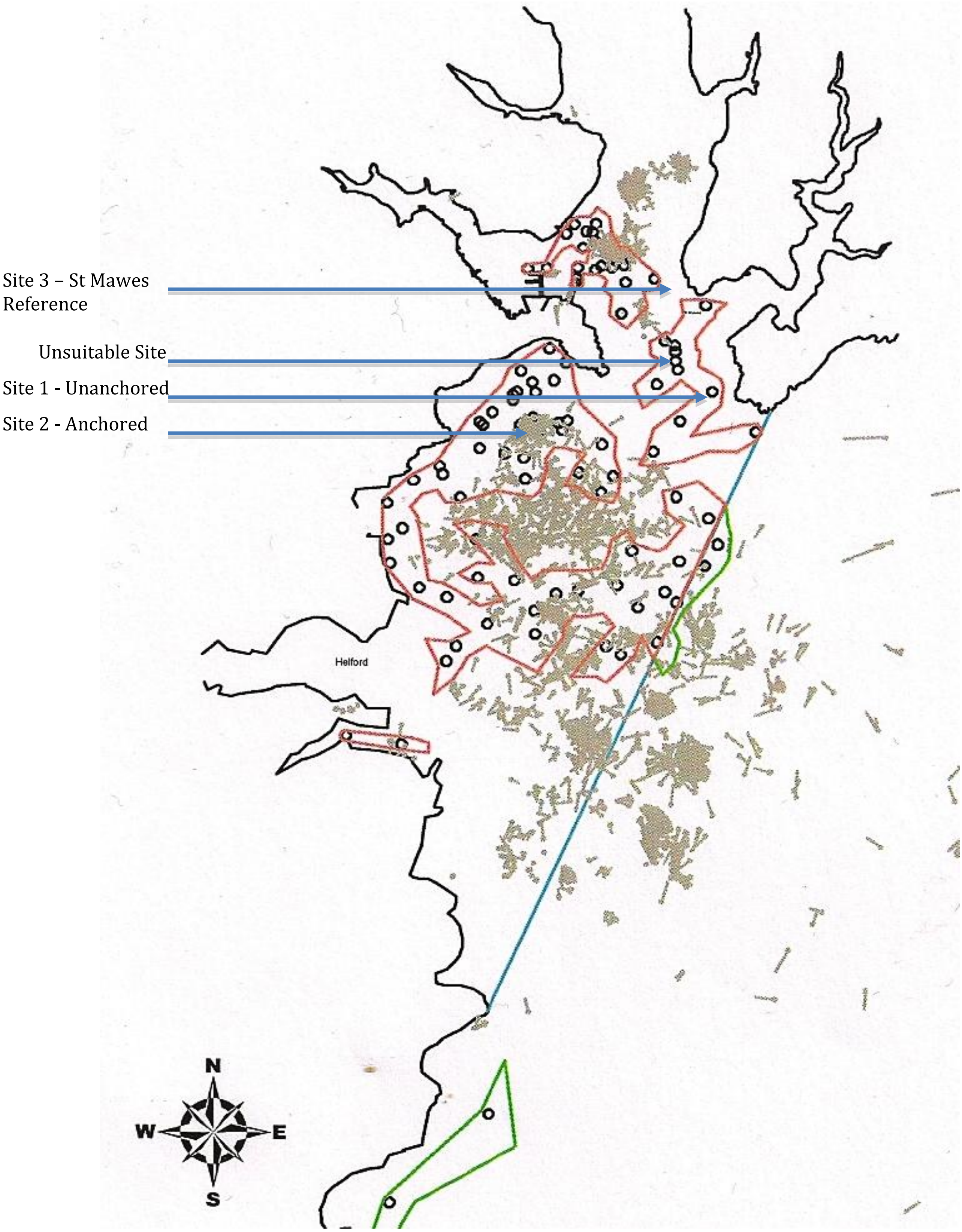

Figure 1 GIS Map of Falmouth Bay, with anchoring sites of all commercial vessels for 2008 marked in grey. Red polygons indicate known “dead maerl" deposits; white dots indicate grab samples with dead maerl, within SAC boundary (Davies & Southeran, 1995), indicated by blue line. Sites used in this study are also marked.

Methods

Diving operations took place 1st-3rd of June 2010, onboard the diver charter "Patrice II", skippered by Mike Tuffrey. Five sites were identified from FHC records as having suitable maerl substrate types (see appendix table 3). Two sites, one anchored and one unanchored, were to be studied in detail, with the further 3 sites as reserve sites if the primary sites proved unsuitable. However, after preliminary dives, the original anchored site (50° 08.' 5494N , 005° 01.'2178W) was found to be unsuitable, with a sandy bottom type. The reserve site also proved unsuitable, so a 6th site was identified as a replacement "anchored site". A 7th site, St Mawes, was also added as a control for seabed surface affects.

For the purposes of this report, site 1 is unanchored (50°07.'94N, 005° 04'. 1028 W), site 2 anchored (50°08.' 1388N, 005° 03.'4518W) and site 3 the reference site of St Mawes (50°07'.'24N, 005° 01.'7830W) (Fig. 1)

Both unanchored (1) and anchored (2) sites had similar environmental factors, both situated in the bay at 10 m chart datum (CD). However, the reference site (3) was subject to several different environmental factors, with a depth of CD 5m, gradually rising to the CD 2m. It is also protected from predominate weather and wave patterns. Therefore, it was used to provide divers with an insight into pristine unanchored maerl beds.

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Effect of the Epibenthos

At all 3 sites, diver recorded quadrats and photoquadrats were employed. One dive pair conducted preliminary dives on each site. Physical features were recorded on waterproof slates: depth, temperature, visibility, bottom type and bottom topography. Biological factors were also recorded. Live maerl cover was estimated using 0.5m^2 metal framed quadrats. Species and abundance of live and dead bivalve shell were recorded using the SACFOR scale (Super-abundant, Abundant, Common, Frequent, Occasional, Rare), as was live epifauna. These recordings were conducted along a 5m transect from a GPS marked shot.

0.5m^2 Metal framed quadrats were haphazardly lain within a 5m radius of the shot. Using a Cannon G7 camera with Ikelite housing and Ikelite DS50 strobe, photographs of each quadrat were then taken. Close up photographs of each corner were taken to ensure easy identification later

Photographs were uploaded onto a Mac desktop. Photographs were then analyzed on Preview, species were identified and abundance of each was recorded. Live maerl, intact shell and broken shell were used as indicators of the intensity of mechanical disturbance areas experienced (Pers. Comms Hall-Spencer). Cover was estimated by personal observation, as was broken and intact shell cover.

Effect on infauna

At site 1 and site 2 diver-operated 0.1m^2 core samples were taken. 10 cores were taken from each site, which is reportedly sufficient to show statistical significances in Falmouth Bay (Dyer & Worsfold, 1998). PVC pipes 30cm long and 10cm wide were worked by hand into the maerl as far as physically possible. This was to at least 10 cm in depth, marked on the outside of the core. If cores were unable penetrate to a minimum of 10cm, they were removed and a new location was chosen to core. Cores were then dug out by hand, capped by purpose made wooden caps, held together with large heavy-duty elastic bands and each sealed in 2 heavy-duty polythene bags. These were placed in a crate on the seabed upright, to maintain vertical stratification, and manually hauled up to the surface. To reduce storage space, half the cores were sieved through 0.5mm mesh bags and placed in self-sealing polythene bags. All cores were treated with 10% Formaldehyde and labeled once on shore. COSHH regulations were followed: anyone handling cores, bags and formaldehyde wore protective gloves, and the formaldehyde along with treated samples were stored in designated containers in a suitable safe location.

Core samples were taken to the University of Plymouth Laboratories, where they were washed with running water in a 0.5mm mesh sieve to remove the formaldehyde. Again, COSHH regulations were followed, with protective gloves used and washing taking place in a fume cupboard with sediment trap. Samples were placed in storage containers and preserved with 70% IMS. Samples were scanned under a low magnification. Fauna were identified to the lowest taxonomic level possible, recorded and removed.

Data analysis

All data were first input into Microsoft Excel, where Shannon-Wiener Indexes were calculated. Data were then copied to SPSS 18 statistical software for further statistical analysis. For both photoquadrat data and core data, Shannon-Wiener index, number of Taxa and total individual abundance were compared between sites with independent-sample t-tests. For photoquadrats, live maerl cover, intact shell and broken shell were compared with one-way ANOVA's. Tukey's test was then used to determine where the differences occurred. Due to the cryptic nature of many organisms living on the maerl of the reference site (site 3), which would otherwise give inaccuracies, only live maerl cover, broken shell and intact shell cover were recorded. For core samples, data were explored further. Between sites, abundance within Phyla were compared using independent-sample t-tests. Due to small sample sizes and large variability, equality of variances were not assumed for all factors. Graphs of mean and standard deviation (SD) were constructed using Sigmaplot.

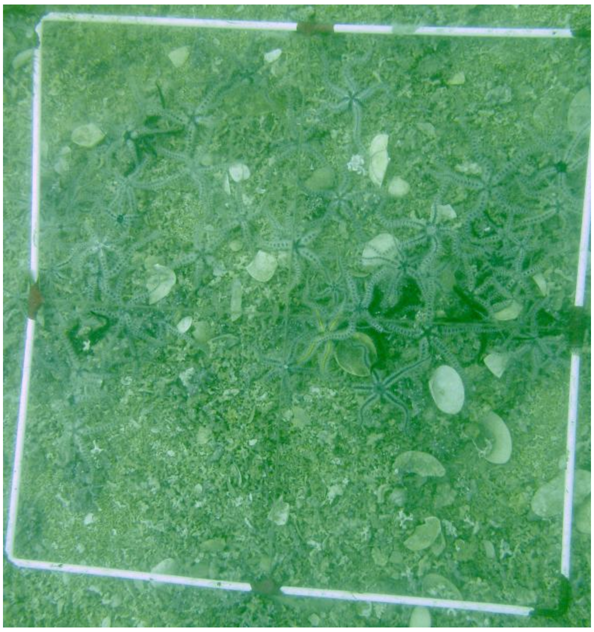
Results

Depth measured by divers at sites 1 and 2 ranged between 16 and 19m. Little current was experienced throughout diving operations. Visibility was estimated by divers to be 5-7m. Water temperature at the seabed was recorded at 12°C. Divers noted higher numbers of large epifauna, mostly echinoderms *Ophiothrix fragilis* and O*phiocomina nigra* in the unanchored sites, with King Scallop (*Pecten maximus)* and Starfish (*Marthasterias glacialis)* encountered frequently and occasionally respectively. Where maerl had been formed into megaripples by wave/tidal action (Hall-Spencer & Atkinson, 1999), brittle stars were found in the troughs of such gullies (Plate 3). At site 3 (St Mawes reference), depth measured by divers was 7m; decreasing to 5 m. Visibility was estimated at 7m+. However, water temperature at the seabed remained at 12°C.

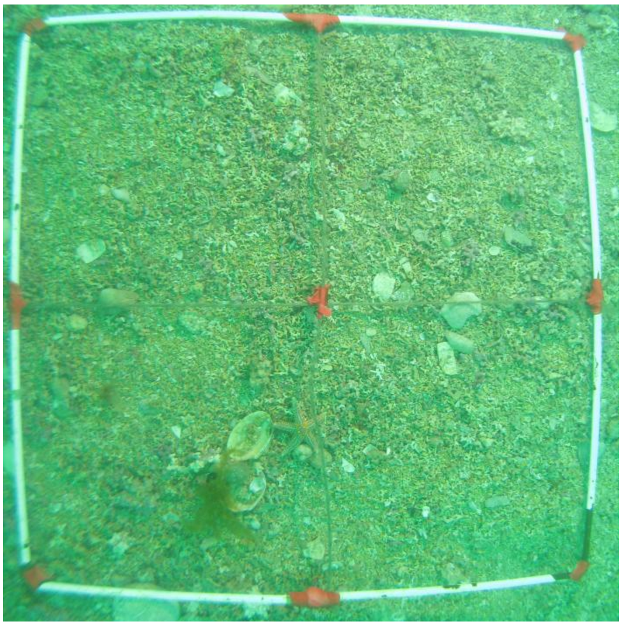
Plate 3 Photo of megaripples in the seabed of Falmouth Bay formed by wave action, taken during study. Note aggregations of brittle stars and heavy debris in the trough. Photo: Kat Brown

Photoquadrat Data

After uploading onto the Apple Mac desktop, 5 photoquadrats from the unanchored site, and 9 from the anchored site had visible identifiable features and organisms.



(a)



(b)

Plate 4 Example of complete $0.5m^2$ photoquadrats for site 1 (a) and site 2 (b). Photographs of individual corners for all photoquadrats were also taken to aid in identification and cover estimate4s

Live maerl cover

Finding live maerl in Falmouth Bay in such relatively high amounts was remarkable. As expected, there were statistically higher amounts of live maerl cover ($F=30.432$, $p<0.001$) on the reference site of St Mawes compared to both anchored ($p<0.001$) and unanchored sites ($p<0.001$) within the bay itself. However, no significant difference in live maerl cover was found between anchored and unanchored sites ($p=0.961$).

% Cover

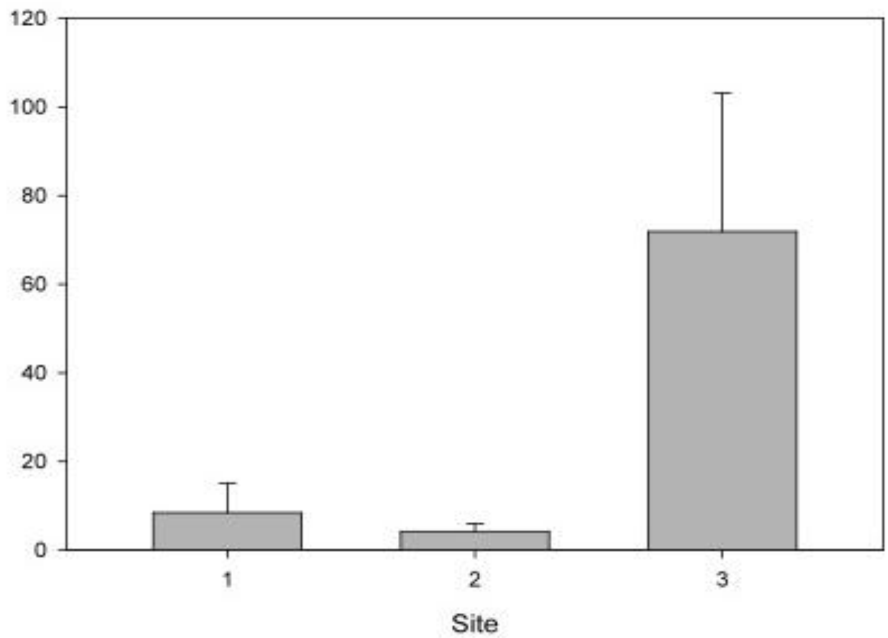


Figure 2. Comparison of mean +SD live maerl percent cover in replicate $0.5m^2$ photoquadrats between unanchored (1) (n=5) anchored (2) (n=9) and reference (3) (n=50) sites, with standard deviation

Despite no statistical difference between anchored and unanchored sites, small differences were noticeable (fig 6), with unanchored areas having a slightly higher cover of live maerl. There was considerable variation of cover on all sites. As the reference site suggests, healthy live maerl beds exhibit large natural variability in live maerl cover.

Abundance and diversity

Unanchored sites had higher abundance (fig. 3) of individual organisms than anchored sites did ($t=3.6$, $p=0.033$). Again assuming unequal variances, there were no significant differences between sites for number of Taxa ($t=1.709$, $p = 0.126$) (fig 4) or Shannon-Weiner diversity indexes ($t=-0.45$, $p=0.965$) (fig. 5).

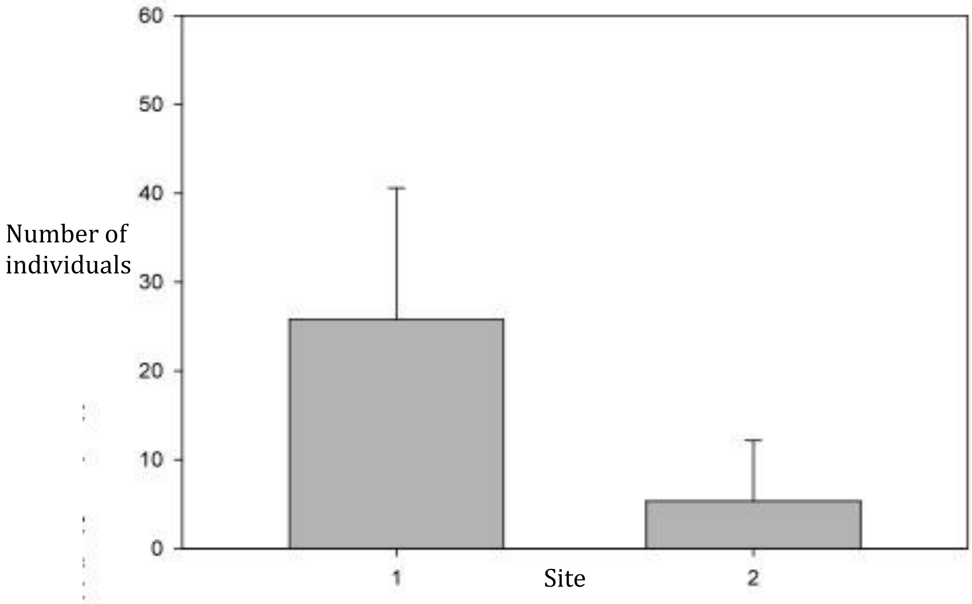


Figure 3. Comparison of mean +SD of abundance of individual organisms in $0.5m^2$ photoquadrats between unanchored (1) (n=5) and anchored (2) sites (n=9). P=0.033

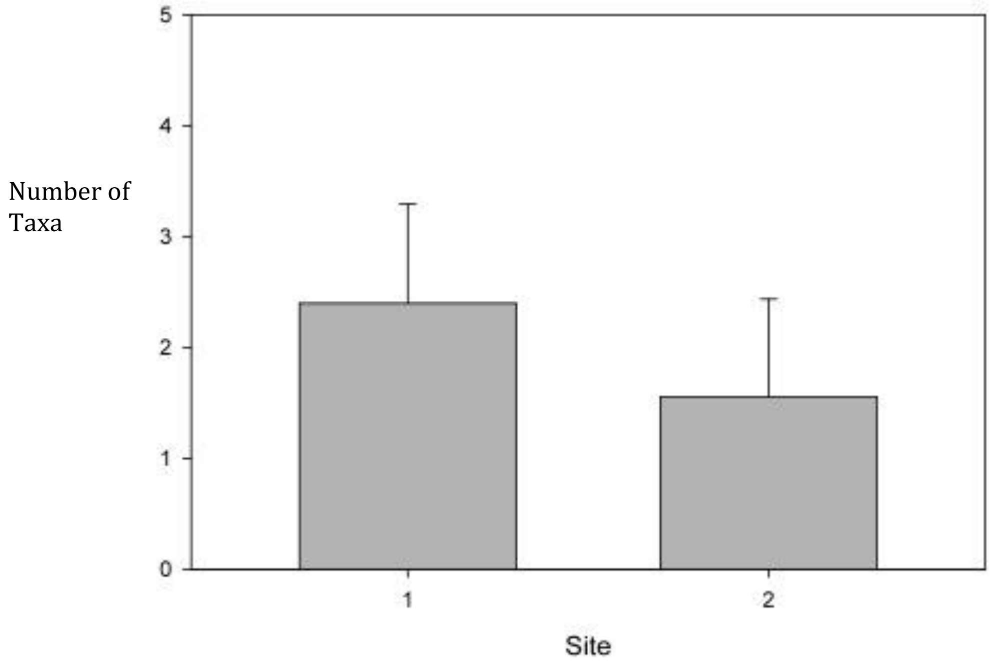


Figure 4. Comparison of mean + SD of number of species in $0.5m^2$ photoquadrats between unanchored (1) (n=5) and anchored (2) (n=9) sites.

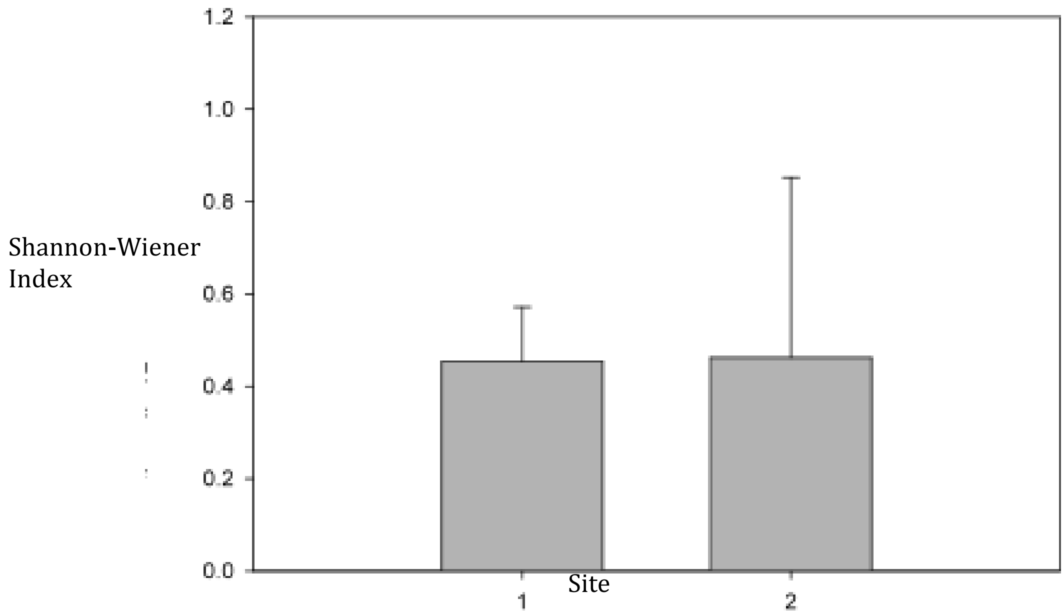


Figure 5. Comparison of mean + SD Shannon-Wiener Biodiversity index per $0.5m^2$ photoquadrat between unanchored (1) and anchored (2) sites

In terms of abundance, *O. fragilis* dominated unanchored sites, averaging 22 individuals per $0.5m^2$. *O. nigra* were present in most quadrats from the unanchored site in low abundances (≈2 m^{-1}). However, both *O. fragilis* and *O. nigra* were in much lower abundances in quadrats from the anchored site (≈5 m^{-1}) than unanchored sites. Despite very low abundances, other scavengers (*A. rubens, M. glacialis and P. bernhardus*) were observed in anchored sites that were not seen in unanchored sites. Total Taxa in the unanchored site were dominated by mostly the brittle stars, with few other species present, whereas the anchored site had fewer brittle stars (fig. 4).

Intact and broken shell

There were statistically significant differences in broken shell ($F=36.074$ $p<0.001$) cover between the reference site and both anchored (Tukey's $p<0.001$) and unanchored sites (Tukey's $p<0.001$), with less on the reference site in both cases (fig. 7). There were statistical differences for intact shell cover ($F=5.436$, $p=0.007$) (fig 6). The reference site had significantly higher percentage cover of intact shell than the anchored site ($p=0.01$). However, no statistical significance was observed in the differences between anchored and unanchored sites of percentage cover of both intact shell ($p= 0.847$) and broken shell ($p=0.353$), and of intact shell coverage between the reference site and unanchored site ($p=0.223$).

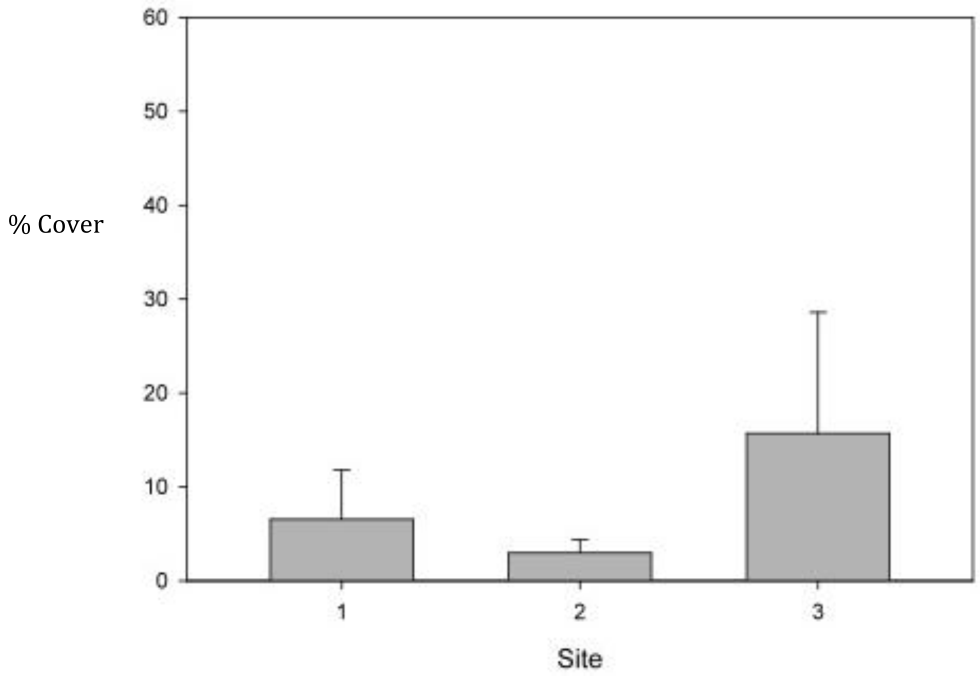


Figure 6. Comparison of mean percentage + SD of intact shell cover in $0.5m^2$ between unanchored (1) (N=5), anchored (2) (N=9) and reference (3) (N=50) sites

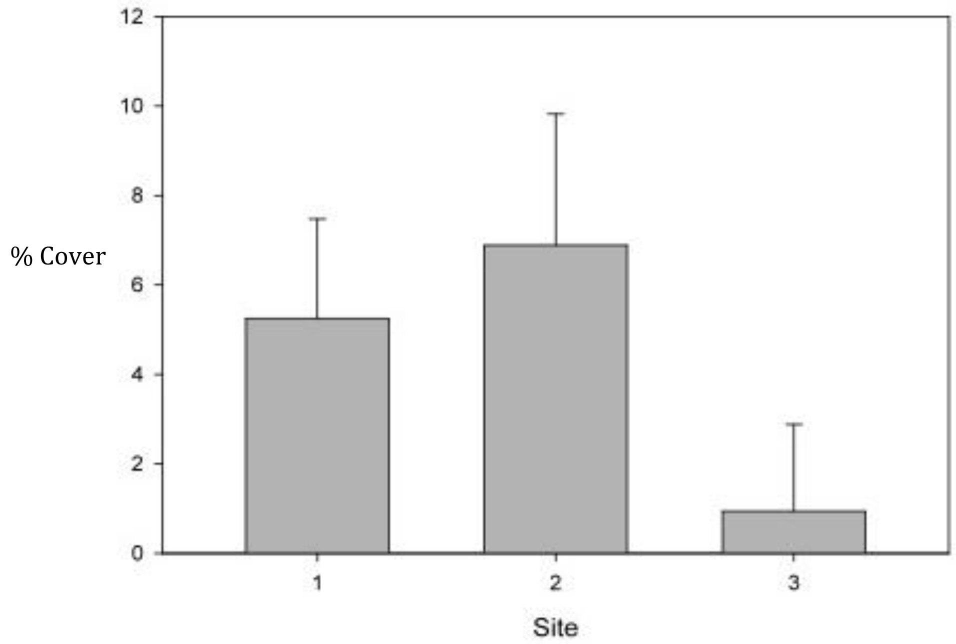


Figure 7. Comparison of mean percentage + SD of broken shell cover per 0.5m2 photoquadrat between unanchored (1) (N=5), anchored (2) (N=9) and reference (3) (N=50) sites

It can be seen that the unanchored site has a higher percent cover of intact shell than the anchored site, and a lower percent cover of broken shell. However, due to the large variation within the anchored site and small sample size, any significant difference may be missed.

Core Data

After processing cores, 8 samples from unanchored site (1) and 7 samples from the anchored site (2) were in a suitable condition to be used. Other cores, despite being double bagged, had been punctured and had shown considerable degradation in the bag beyond the point where even families were unidentifiable. From all cores used, 1329 individual organisms were collected from cores from 55 Taxa, identified to 6 Phyla with individuals identified to 29 Families, 2 Classes, 8 Orders and 7 Species.

No statistical differences in biodiversity index (t=0.612, p=0.551)(fig. 8), number of identifiable Taxa, and abundance within sediment cores were found between anchored and unanchored areas (table 1). There were no differences between abundances of individual phyla (table 2). Errant annelids dominated all core samples at both sites. Neridae and Syllidae were the highest in abundance, followed by the nematodes. There was no statistical difference in abundance of individual phyla between sites. No species varied greatly in abundance between sites.

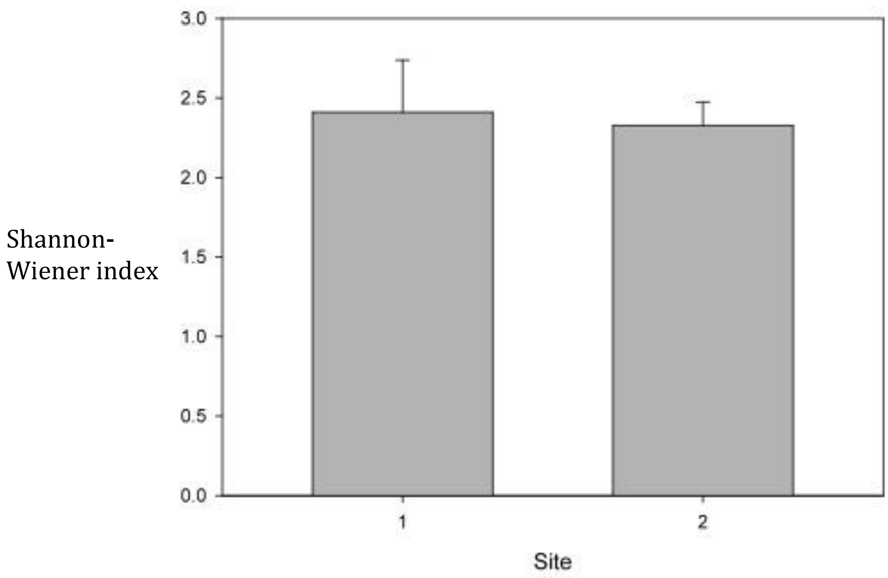


Figure 8. Comparison of mean Shannon-Wiener biodiversity index of core samples between unanchored (1) (n=8) and anchored sites (2) (=7)

	T	P (2 tailed)
No. Taxa	1.650	0.115
No. individual organisms	1.911	0.72

Table 1. T-test statics for core samples

Phyla	T	P (2 tailed) Equal variances not assumed
Polychaete	1.651	0.123
Nematode	1.348	0.216
Crustacea	1.172	0.283
Molusca	-0.402	0.695
Echinodermata	1.963	0.84

Table 2. T-test results for abundance of Phyla

Discussion

With such a long history of anchoring and other activities, and as sites were situated within the main bay unprotected from south-southwest weather, it is remarkable that both sites had such high levels of live maerl present. Contrary to previous reports (Davies & Southeran, 1995), the sediment cannot be regarded as completely dead maerl deposits. Lower figures, as little as 0% live cover, have been recorded in areas that have had a history of repeated disturbance or undergone experimental scallop dredging (Hall-Spencer & Moore, Scallop dredging has profound, long-term impacts on maerl habitats, 2000).

Anchoring may decrease, though not significantly reduce the live maerl cover on the seabed surface. Low concentrations of maerl have been found in areas that are regularly disturbed by dredging activities (Hall-Spencer & Moore, 2000). However, the lack of any significant difference in live maerl cover could be due to any combination of factors. Sources of possible damage, such as chain abrasion may not cause similar high amounts of damage as seen to seagrass beds or corals, or by scallop dredging on maerl, due to the small, unattached nature of the thalli. As Falmouth Bay is also not protected from South and Southeasterly weather, unlike like the banks off St Mawes, environmental conditions such as storm activity may be unsuitable to allow it to reach higher percent coverage, as seen on St Mawes bank. As suggested by the presence of the megaripples at unanchored and anchored site (plate 3), the seabed occasionally experiences relatively high-energy wave action (Hall-Spencer & Atkinson, 1999). This by itself may increase sedimentation of the surrounding area; therefore limiting live maerl growth and cover of both anchored and unanchored areas. The high level of ship activity seen in Falmouth Bay would also increase vertical mixing, and increase turbidity, decreasing available light and so growth rates of the entire bay (Lindhom, et al. 2001). Sites 1 and 2 reside in deeper water compared to St Mawes (reference site 3). Perrins et al. (1995) noticed significantly reduced maerl cover with depth along transects on St Mawes bank. It was suggested that the maerl here is vulnerable to even slight increases in turbidity. However, maerl is adapted to low light levels (Wilson, et al. 2004). Therefore, whether low cover is a result of lower light level in Falmouth Bay could be further investigated. Most likely, light intensity at depth as a result of increased turbidity is just one of many combining factors, such as wave action and depth, which affects cover. Therefore, direct physical disturbance by anchoring may be only a small contributing factor to the cover of live maerl seen, and any small differences observed.

Despite these suggestions, low sample numbers of photoquadrats for anchored and unanchored sites may have affected the outcome of this study. With the larger sample size for St Mawes reference site, it is clear that even unaffected 'pristine' sites have a high natural variability in live maerl cover. It is very likely that the maerl beds of Falmouth bay show similar natural variability, suggested by the standard deviation. Therefore, with more photoquadrats of anchored and unanchored sites, significant difference in live maerl cover may become apparent. As can be seen in fig 2, there were differences observed between all sites. However, possibly through large natural variations of live maerl cover for all sites, this study found no significant differences.

The only marked differences between sites were with abundance on the surface. Epifauna are likely to be affected by anchor drag upon deployment, and chain abrasion whilst anchored. Differences in epifauna were observed as chains affect much larger areas of the seabed than anchors themselves (Walker, et al. 1989). On unanchored sites, large aggregations of brittle stars dominated. These aggregations could be broken up and individuals killed by anchoring activities, such as chain swing and abrasion, which is suggested by their low abundance on the anchored site. It is already well known that anchor chains severely damage seagrass beds, and scour marks from moorings suggest that no fragile organisms such as fragile brittle stars would be able to survive (Walker, et al. 1989; Montefalcone, et al. 2008)

As with live maerl cover however, small sample size and high variability within sites may have affected the results. A false positive may have been found. However, divers observed similar differences in epifauna on dives, with higher abundances of brittle star on unanchored sites. Therefore, it is likely that there are higher abundances of fragile, slow-moving fauna in unanchored areas, though more replicates should be taken for stronger evidence.

The high numbers of intact shell on the St Mawes bank reference site (3) suggest a large population of burrowing bivalves is present. Many *M. glacialis* were observed in the process of opening live bivalve shells. St Mawes bank is also very flat (see plate 1), with no megaripples, indicative of interaction with waves as seen in sites 1 and 2. These factors combined demonstrate that the St Mawes bank area is protected from both mechanical disturbance and weather disturbance, therefore providing the lowest broken shell cover. Though no statistical difference was detected between site 1 and 2, and 2 and 3, the differences in intact shell suggest all 3 sites experience different levels of physical disturbance: St Mawes reference site experiences the least, and the anchored site experiences the most. However, no differences in broken shell cover between anchored and unanchored sites could be attributed to shell movement between areas by tides, currents and storms. However, insignificant differences may again be attributed to small sample size, as suggested by the observed minute difference in both intact and broken shell cover between anchored and unanchored sites.

Shannon-Wiener biodiversity figures for core samples are similar to areas of Falmouth Bay unaffected by previous physical disturbance, found in previous studies for the impacts of dredging (Dyer & Worsfold, 1998). The small area that anchor penetration would effect, combined with not finding any obvious scars within the sample sites, may have resulted in no differences being found in infaunal communities between anchored and unanchored site. Core samples varied in Taxa present, though all with similar number of different Taxa. Both sites had samples with high numbers of bivalve spat, or where particular Taxa dominated. This supports the diversity within maerl beds, and the patchiness of faunal distribution throughout maerl beds (Grall et al.2006). However, due to samples being spoilt in transport by leaks in bags, sample numbers where not of the number recommended by Dyer and Worsfold (1998) to reliably show statistical differences. Another factor, which was not investigated here is health and abundance of long-lived deep burrowing organisms. Airlifting, the optimal way to explore deeper sediments (Hall-Spencer & Atkinson, 1999), was advised against for safety and logistical reason. Therefore, it cannot be concluded whether anchoring has any affect on infauna. We suggest that a better-equipped study include investigating identified anchor scars, with the addition of airlifting equipment.

Due to limited time and resources, ideal sample sizes were not achieved. This will have had a marked effect on the results, so perhaps contributing to the insignificant differences observed. However, the small insignificant differences observed have given us an insight into the possible true effects of anchoring. In this study, core samples took priority, with less time given to photoquadrats. However, it now seems apparent that emphasis must be given to the surface changes, due to the larger area of impact from chains. Further investigation is needed, with increased numbers of photoquadrats taken per site. Anchor scars should be sought out, and core samples and airlift samples of the area affected by anchor taken for further comparison with unanchored sites, and surrounding 'anchored' sites. In addition to this, increased numbers of sites for both anchored and unanchored areas should be studied, with a range of depths included to successfully link anchoring and affects to the seabed.

Conclusion

It is remarkable to find live maerl in such relatively high quantities, even in areas that are anchored intensely. Maerl beds naturally display high variability in characteristics such as live maerl cover, species present and abundance of organisms. Due to the size of area affected, chains are most likely to cause damage to the seabed surface. However, significant differences in maerl cover were not detected suggesting that anchoring does not have a marked effect on cover. The anchoring of commercial vessels on Falmouth Bay's maerl beds is likely to reduce the abundance of slow moving aggregating species, and increase the abundance of motile scavengers. Moving vessel anchorages is at present most likely not required. However, further research and monitoring of the areas would be the best course of action.

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Appendix

Site	Latitude	Longitude	Condition
1	50°08'. 5494 N	005° 01'. 2178 W	Anchored
2	50°08'. 1388N	005° 03'. 4518 W	Unanchored
3	50°08'. 0789N	005° 01'. 5694 W	Unanchored
4	50°07'. 3318N	005° 04'. 1028 W	Anchored
5	Approx site of 20 m – not used		
6	50°07'. 94	005° 04'. 1028 W	Anchored
7	50°07'. 324	005° 01'. 7830 W	Reference

Table 3. Original locations of sites to be used. Sites marked in red not used, as had unsuitable substrates.

Site	Shannon-Wiener Index	Number Taxa	No. Individuals	No. polychaetes	No. nematodes	No. Crustacea	No. Mollusca	No. Echinodermata
1	1.695659	11	72	44	17	8	3	0
1	2.233465	14	125	56	38	19	12	0
1	2.322905	15	49	28	12	3	6	0
1	2.660041	17	103	69	13	10	10	1
1	2.525335	20	116	51	7	54	1	3
1	2.602667	24	186	112	11	47	14	2
1	2.603589	21	112	58	15	26	12	0
1	2.638766	24	113	50	10	34	16	3
2	2.146352	12	87	48	5	13	21	0
2	2.533056	16	74	42	5	18	9	0
2	2.326633	13	95	72	12	11	0	0
2	2.329374	16	82	23	9	32	18	0
2	2.43472	21	103	47	20	24	11	1
2	2.127914	12	40	12	13	6	9	0
2	2.396241	13	57	34	6	11	6	0

Table 4. Core sample data, number of each Phylum found within each core, total individuals and total Taxa and Shannon-wiener Biodiversity for Site 1 unanchored, site 2 anchored.

	Anchored		Unanchored	
	Mean	Std. Deviation	Mean	Std. Deviation
Shannon-Weiner	2.53994213	.257263530	2.41557614	.200705209

Table 5. Mean with standard deviation of core data

Quadrat	*O. fragilis*	*O. nigra*	*P. pictus*	*T. rhomboides*	*Total No. Species*
1	24	1	2	1	4
2	22	3			2
3	5	0	2		2
4	18	3			2
5	43	5			2

Table 5. Unanchored photoquadrat species data

Quadrat	*O. fragilis*	*O. nigra*	*A. rubens*	*M. glacialis*	*P. bernhadus*	*Total No. Species*	*Total Individuals*	*Shannon Index*
1	2	1				2	3	0.636514168
2	1					1	1	0
3	1	1				1	2	0.693147181
4	9	0				1	9	0
5	19	3				2	22	0.398307114
6						/	/	N/A
7			1		1	2	2	0.693147181
8	4				2	2	6	0.636514168
9							0	N/A
10	1	1		1		3	3	1.098612289
	4.625	0.75						0.461804678

Table 6. Anchored site photoquadrat species data