



Using a no-take zone to assess the impacts of fishing: Sessile epifauna appear insensitive to environmental disturbances from commercial potting

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ABSTRACT

Aside from effects on target populations and/or by catch, there is widespread interest in the physical impacts of fishing activity on marine systems. Frequently, it is the effect of mobile gear such as dredges and trawls which are of principal concern. The ecological consequences of passive/static gear such as pots and traps on benthic communities are far less understood. The establishment of a Sanctuary Zone/No-Take Zone at Lundy Island in 2003 allowed us to test the idea that if potting for crustaceans has an impact on benthic assemblages, then on the cessation of potting, assemblages should become different from those where potting continues. A concurrent study using experimental potting within the No-Take Zone allowed a separate test of this idea, but with a controlled fishing effort. This is the first longer-term study into the effects of lobster potting on benthic assemblages in the northern Atlantic. Multivariate analyses on patterns of change in assemblages over the 4 years of the study indicated no differences in how assemblages changed over time between areas subject to potting and those not fished. From the results, we conclude that potting for lobsters and crabs in inshore waters seems to have no detectable effects over the timescale of the experiment. In addition to being management goals/tools in their own right, strongly protected areas such as Sanctuary/No-Take Zones provide an opportunity to test experimental hypotheses about the effects of fishing. Our study shows there were no conservation benefits, in terms of the investigated species, arising from exclusion of potting in this Marine Protected Area (MPA). For some MPAs it may be possible to permit limited potting to allow small-scale artisanal fisheries to persist.

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1. Introduction

Investigations into the physical impact of fishing gear on the benthic ecology have focused almost exclusively on mobile gears, such as trawls and dredges (see Auster and Langton, 1999; Hall, 1999; Hinz et al., 2011; Kaiser and de Groot, 2000 for review). Potential damage from static gear such as baited pots and traps, have received relatively little attention. This is unsurprising given that the potential for disturbance per unit deployment is clearly very much less with pots and traps, but their cumulative effects on the benthos may be substantial nevertheless.

Baited pots and traps are widely used throughout the World to catch a variety of crustaceans, molluscs and finfish (Gabriel et al., 2005). Approximately half of the vessels in the UK fishing fleet (i.e. ~3,600 vessels) and a further 7,000 other EU vessels use pots and traps as their main fishing gear (Anon, 2011). Within the north-eastern Atlantic,

the principal species targeted using these gears include lobster (*Homarus gammarus*), crabs (e.g. the brown or edible crab, *Cancer pagurus*, spider crab, *Maja squinado* and velvet crab *Necora puber*), Norway lobster (*Nephrops norvegicus*), prawns (*Palaemon serratus*), whelks (*Buccinum undatum*), cuttlefish (*Sepia officinalis*) and a variety of finfish. It is notable that fishing effort involving static gears has increased dramatically in recent years. For example, whilst the overall size of the UK fishing fleet has declined in the last 20 years (from ~9900 to ~6500 vessels) the number of vessels using pots and traps as their main gear has doubled from ~1700 to ~3500 (<http://ec.europa.eu/fisheries/fleet/index.cfm>, accessed 5/8/11). Using the UK as an example for the NE Atlantic, it is noted that the majority of these vessels are engaged in fisheries for lobster and brown crab, which are by far the most extensive and valuable pot or trap fisheries in UK waters (Marine and Fisheries Agency, 2009).

Any direct ecological effects of pots or traps will be a consequence of how the gears are used. The usual arrangement is to deploy multiple (~10–20) pots or traps spaced evenly along a common 'back-rope', forming a 'string' of gear. A buoy-line is attached to each end of the string to mark its position at the surface and facilitate its retrieval. When fishing in areas subject to powerful waves and/or tidal currents,

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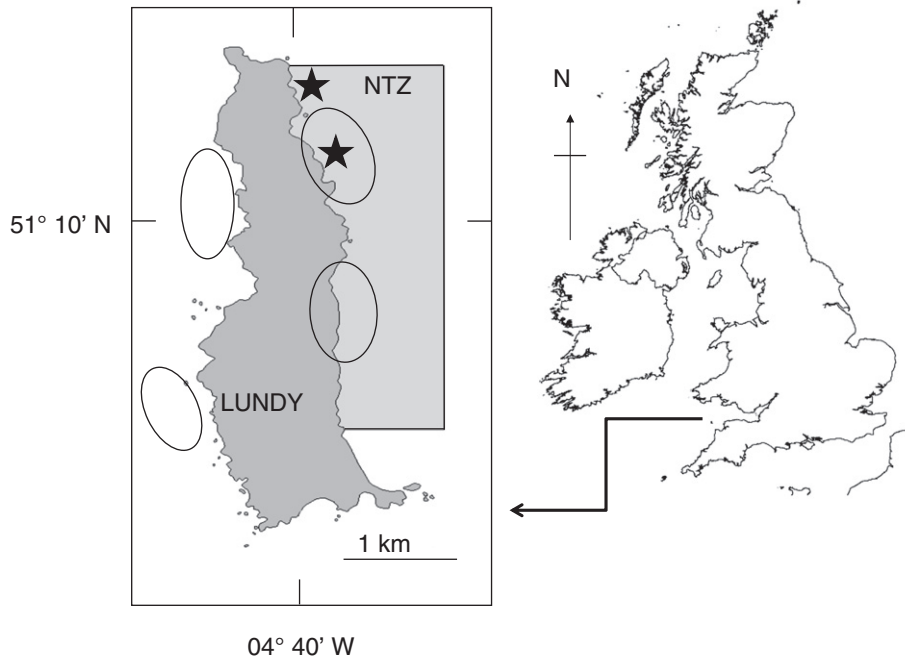


Fig. 1. Location of Lundy in the UK. Inset shows the boundary of the Lundy No-Take Zone where commercial fishing was stopped (NTZ). Ellipses indicate the study areas for the longer-term investigation, the stars show the approximate position of the experimental potting study, the northern star being the place subject to experimental potting.

anchor-weights are usually attached to the ends of the string to prevent it dragging. A string of pots is deployed by dropping the first buoy-line and anchor-weight into the water and allowing the sequence of pots, followed by the second anchor-weight and buoy-line, to be pulled overboard as the vessel transits the target fishing ground. Once pots have been baited and deployed for fishing, they are generally left to 'soak' for 1–3 days before harvesting. Retrieval is simply the reverse process.

Direct impacts from the potting gear could arise in several ways. First, pots and/or end-weights make strike and damage benthic organisms during deployment. Second, once gear has been deployed, movements of pots and ropes under the influence of waves or tidal currents may damage the benthos via abrasion. Finally, retrieving potting gear may also disturb the benthos, but only if the gear is dragged laterally as it is being lifted (Eno et al., 2001). Fishers generally avoid doing this because it increases wear and tear and the risk of snagging. Lateral drag would generally only occur where some combination of wind, tide or navigational hazard prevented a direct vertical lift.

Conservation concerns about the potential impacts of pots and traps are greatest for long-lived sessile fauna, particularly those with erect body-forms. Such taxa are most diverse and abundant in rocky-reef habitats, which is where almost all lobster potting and a significant amount of crab potting takes place. Potentially vulnerable species in the NE Atlantic include the pink sea fan (*Eunicella verrucosa*), dead men's fingers (*Alcyonium digitatum*), ross coral (the bryozoan *Pentapora fascialis*) and various erect branching sponges (e.g. *Axinella* spp., *Raspalia* spp.). In UK waters, these species and many like them are recognised as being rare and threatened, and concerns about their perceived vulnerability to potting gear regularly feature in official guidance on marine habitat management for nature conservation e.g. guidance published under the Convention on Biological Diversity and the EC Habitats Directive (e.g. Hartnoll, 1998; Jones et al., 2000).

There are few published experiments on the potential impacts of lobster and crab potting on sessile epifauna in rocky habitats. In an attempt to determine the effects of potting, Eno et al. (2001) failed to detect any impacts. There were problems with this study, however, which reduced its capacity to detect such impacts. First, this was only a

short-term study in which the potting treatment comprised 10 repetitions of deployment and retrieval over one month. Second, the levels of sampling replication and methods of analysis led to issues of power (Eno et al., 2001). Finally, there were problems in their design that may have confounded the experiment so as to make it appear that some species had significantly increased in abundance in response to the potting treatment (this was not considered to be plausible given the life-histories of the species concerned). In contrast to Eno et al. (2001), a study by Lewis et al. (2009) detected impacts of lobster traps on sessile epifauna (stony corals, octocorals and sponges) on coral reef habitats in the Florida Keys, USA. This fishery was for the Caribbean spiny lobster (*Panulirus argus*). Unlike European lobster and crab fisheries, traps in this fishery are typically deployed singly, rather than in strings, so there are no concerns about abrasion by ropes linking traps. The study concentrated on the benthic impact of movements of traps during winter storms. Lewis et al. (2009) found that single buoyed traps left on the reef for 4–7 months caused visible disturbance to the

Table 1

Species of long-lived sessile epifauna in circalittoral rocky habitats that were selected for test for the effects of potting.

Phyla	Species	Common name (or description)
Porifera (Sponges)	<i>Axinella dissimilis</i> (Bowerbank, 1966)	branching sponge
	<i>Axinella infundibuliformis</i> (L.)	funnel-shaped sponge
	<i>Axinella damicornis</i> (Esper, 1794)	erect, non branching sponge
	<i>Homaxinella subdola</i> (Bowerbank, 1966)	branching sponge
	<i>Raspalia ramosa</i> (Montagu, 1818)	branching sponge
	<i>Polymastia boletiformis</i> (Lamarck)	cushion sponge
	<i>Polymastia mammillaris</i> (Müller)	cushion sponge
	<i>Cliona celata</i> (Grant, 1826)	Boring sponge
	<i>Alcyonium digitatum</i> (L.)	Dead men's fingers
	<i>Alcyonium glomeratum</i> (Hassal, 1843)	Red fingers
Cnidaria	<i>Eunicella verrucosa</i> (Pallas, 1766)	Pink sea-fan
	<i>Anemonia viridis</i> (Forskål, 1775)	Snakelocks anemone
	<i>Aiptasia mutablis</i> (Gravenhorst, 1831)	Trumpet anemone
	<i>Pentapora fascialis</i> (Pallas, 1766)	Ross coral
Bryozoa	<i>Pentapora fascialis</i> (Pallas, 1766)	Ross coral
Chordata (class Ascidiacea)	<i>Stolonica socialis</i> Hartmeyer, 1903	

benthos over a mean area of approximately 4.7 m² when placed at 4 m depth, declining to approximately 1.1 m² at 12 m. Unbuoyed traps (simulating lost traps) caused substantially less disturbance, indicating that it was drag from the buoy-line that moved the traps rather than drag on the trap itself. Within the disturbed areas created by traps, the percentage cover of sessile epifauna showed an average reduction of 10%.

In 2003, designation of the UK's first statutory no-take zone (NTZ) at Lundy island in the Bristol Channel (Fig. 1) to conserve circalittoral fauna presented a unique opportunity to examine both the potential impacts of potting on sessile reef epifauna and their potential recovery, post-designation. As well as supporting an important potting fishery (primarily for lobster, but brown, spider and velvet crabs are also harvested; Hoskin et al., 2011), Lundy also has various statutory conservation designations in recognition of the national and international significance of its marine habitats and species; i.e. Marine Nature Reserve (MNR) under the Wildlife & Countryside Act 1982 and Special Area of Conservation (SAC) under the EU Habitats Directive (Anon, 1992). Notable amongst Lundy's important features are the bed-rock and boulder reef habitats surrounding the island, which are the main habitats that are targeted by the local potting fishery. An important reason for the designation of the Lundy NTZ was concern for the potential impact of lobster potting on Lundy's reef habitats. Since it was thought that the physical disruption caused by potting gear is likely to be detrimental to assemblages of sessile epibiota then if potting was stopped then there should be changes in assemblages in the closed areas. This expectation was underpinned by the large body of evidence from NTZs elsewhere in the World indicating that sanctuary level protection can trigger lasting, often rapid, increases in the abundance, productivity and diversity of marine organisms (e.g. Côté et al., 2001; Dugan and Davis, 1993; Halpern and Warner, 2002; Jennings, 2001; Lubchenco et al., 2003; Roberts and Polunin, 1991; Sánchez-Lizaso et al., 2000). Experimental results indicated that the lobster populations at Lundy responded to the designation of the NTZ with one of the fastest increases ever noted (Hoskin et al., 2011), and it is possible that fishing activity could have been displaced from the NTZ to outside, thus increasing the potting pressure in such areas.

Whilst reefs at Lundy support important algal assemblages, including several species that are nationally rare (e.g. *Zanardinia prototypus* and *Carpomitra costata*), their annual life-cycles and relatively fast growth rates should make them much less sensitive to disturbance from potting. If potting had the impacts suggested then it would be expected that the assemblage of sessile epifauna in circalittoral reef habitats in places where potting had stopped (inside the NTZ) will change more over time since these are now only subject to natural environmental variation whereas the sites subject to continued fishing effort would show much less temporal variation since the imposition of the NTZ since these assemblages will be subject to pressures from potting and from natural variations (Hypothesis 1). It would also be expected that those species of sessile epifauna most susceptible to impacts of potting will become relatively more abundant inside the NTZ over the 5 years of the study, compared to similar habitats where fishing continues over the same time-scale if (Hypothesis 2).

An ideal experimental study of the effects of potting on epifauna would apply a known, standardised level of potting. This was not possible in any comparison of the closed areas (inside the NTZ) with commercially fished areas. A separate study on the effects of potting on the populations of fished crustaceans (Hoskin et al., 2011) provided the opportunity for such a standardised study with the area closed to commercial fishing. This is because in that study, we had locations within the closed area which were experimentally potted and locations which were not and thus had not been potted since the designation of the NTZ. It should be noted that although the experimental used a commercial methodology and sample size (Hoskin et al., 2011), the fact that it only happened once per year may mean effects were less than in the NTZ contrast. The hypotheses tested were that the

Table 2

(a) Permutational MANOVA (PERMANOVA) examining differences in assemblages in places where potting for crustaceans (lobsters and crabs) had stopped versus those where potting continued. There were two locations for each treatment and at each location, six sites were used and $n = 12$ quadrants sampled. Samples were $\sqrt[4]{x}$ transformed after a dummy variable of 0.0001 was added to avoid the weighting by zeroes, prior to the construction of a Bray–Curtis resemblance matrix. (b) Results of pairwise tests for the factor $Ye \times Si$ ($Lo \times Po$) showing the frequency of occurrences of assemblages differing among years 2–4 for each treatment of potting, out of 12 possible tests.

(a)						
Source	df	SS	MS	Pseudo-F	P (perm)	Unique permutations
Year Ye	2	17834	8917	0.728	0.692	9944
Potting Po	1	264560	264560	1.880	0.333	3
Location Lo (PO)	2	281460	140730	9.346	<0.001	9953
Ye × Po	2	23299	11649	0.951	0.509	9950
Site Si (Lo(Po))	20	301150	15058	7.568	<0.001	9823
Ye × Lo(Po)	4	48987	12247	1.577	0.085	9925
Ye × Si(Lo(Po))	40	310630	7766	3.903	<0.001	9789
Residual	792	1575700	1990			

(b)					
Treatment	Differences in years				
	2 = 3 = 4	2 = 3 ≠ 4	2 ≠ 3 = 4	2 = 4 ≠ 3	2 ≠ 3 ≠ 4
Potting Stopped	6	3	2	0	1
Fished area	4	4	1	0	3

assemblage of sessile epifauna in experimentally potted locations in an area otherwise protected from any other form of fishing would show greater divergence when compared to control sites also within the NTZ where there had been no experimental potting (Hypothesis 3); we also predicted that individual species of sessile epifauna would be less abundant in experimentally potted locations sites compared to control sites (Hypothesis 4).

2. Materials and methods

To test the hypotheses, we needed an assemblage of epifauna that were most likely to be impacted, which meant organisms with an erect growth form, so rather than just sample the benthos we examined the responses of a suite of susceptible taxa similarly to Eno et al. (2001). We selected a set of sessile epifauna that were reasonably abundant around the island, this was necessary to be confident that we could detect impacts that may be present. The taxa monitored were decided upon from a pilot study at Lundy during May 2004 and are given in Table 1.

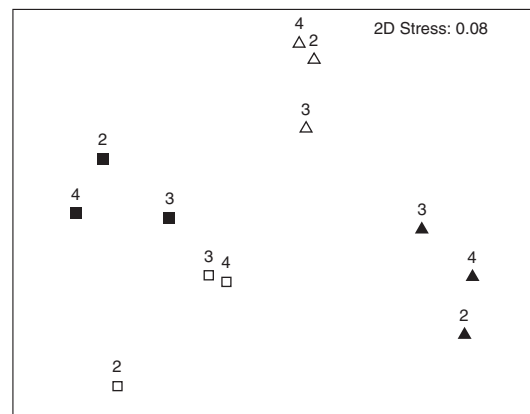


Fig. 2. nMDS showing differences in assemblages in each of two locations (unfilled shapes = location 1 and solid shapes = location 2) at Lundy where potting had been prohibited for 4 years (triangles) and where potting had continued (squares = controls). The shapes indicate positions of centroids of sites and quadrats for each location in each year. Samples were taken every year for 3 years, indicated by numbers next to the shapes.

Table 3

Results of SIMPER analysis showing which species contributed the greatest differences in samples over the final three years of the study comparing the areas where commercial potting had been stopped in the NTZ and that where it had continued. Axinellid sponges have been grouped together since these were difficult to distinguish in the field.

Species	NTZ		Commercially potted		Dissimilarity/SD	Percentage contribution	Cumulative percentage
	Average abundance	Average abundance	Average abundance	Average dissimilarity			
Axinellid sponges	0.92	0.64	21.92	0.95	26.72	26.72	
<i>Pentapora fascialis</i>	0.06	0.5	13.15	0.69	16.03	42.76	
<i>Alyconium digitatum</i>	0.01	0.49	11.09	0.58	13.53	56.29	
<i>Cliona celata</i>	0.1	0.42	10.83	0.64	13.2	69.49	
<i>Aiptasia mutabilis</i>	0.5	0	9.08	0.58	11.07	80.56	
<i>Raspalia ramosa</i>	0.32	0.21	8.42	0.66	10.27	90.82	

For the tests of hypotheses 1 and 2, we used two locations on the west coast of Lundy and 2 locations within the NTZ on the east coast. For each sampling year, we used the same location but randomly selected six different sites. The 2004 pilot study was also used to select locations for sampling epifauna that were broadly comparable in terms of environmental conditions and the species of epifauna present (Hoskin et al. unpubl. data). As the NTZ occupies most of the east coast of Lundy, there was no alternative to having control sites on the west coast of the island (Fig. 1). This meant that there were at least three types of physical environmental difference between NTZ and control locations: (i) the west coast was more wave-exposed since it faced the prevailing weather; (ii) control sites on the west coast were ~5 m deeper than NTZ sites within the NTZ (~20–25 m versus 15–20 m average depth respectively); and (iii) the seabed within west coast control sites was mainly bed-rock, whilst NTZ sites were boulder-dominated. It was recognised that these natural environmental differences were likely to have caused differences in epifaunal assemblages between NTZ and control locations and that this would complicate efforts to detect potential changes due to the NTZ. Ideally, assemblages in NTZ and control locations would have been similar initially, in which case effects of the NTZ, if they occurred, would increase differences between NTZ and fished locations. Here, epifaunal assemblages in NTZ and fished sites were expected to differ from the outset, so potential effects of cessation of potting might modify differences in assemblages already present; this is essentially an interaction between time, spatial variation and treatment effects. The tests for hypotheses 3 and 4 used two locations

inside the NTZ not previously subject to potting and two locations inside the NTZ which had not been potted for 4 years.

Within each location, epifauna were sampled by scuba divers in six random sites. In each plot, divers recorded epifaunal abundances in each of 12 quadrats (75×75 cm) placed at predetermined random distances along a transect (giving a total of 72 quadrat counts per location). Only quadrats falling on 90–100% rock habitat were sampled. Where this condition was not met, the diver moved onto the next random position and so on. All transects were laid out parallel to the shore, and adjacent quadrats were 1–5.5 m apart. All quadrat counts were converted to abundances m⁻² prior to statistical analyses.

Analyses of data from pilot work at Lundy (M.G. Hoskin et al., Unpubl. data) suggested that 72 samples per site was sufficient to maximise the precision of mean estimates of epifaunal abundances per 0.57×0.75 m quadrat. Sampling was done in late summer each of years 1–4 (August–September 2004, July–August 2005, 2006 and 2007).

The hypothesis of any compositional divergence among closed locations versus the fished locations (hypothesis 1) was tested by examining the interaction term of Year×Treatment in a permutation multivariate analysis of variance (PERMANOVA; Anderson, 2001) using PRIMER 6 with PERMANOVA+ (Primer-E, Plymouth) (Clark & Warwick, 2001), where location and site were random factors and the contrast of in vs. out of the NTZ was a fixed factor; the analysis used counts per quadrat as the response variable. Multivariate variability among samples was quantified using the Bray–Curtis index

Table 4

(a) PERMANOVA examining differences in abundances of Axinellid sponges, *Pentapora fascialis* and *Cliona celata* in places where potting for crustaceans (lobsters and crabs) had stopped versus those where potting continued. There were two locations for each treatment and at each location, six sites were used and n=12 quadrats sampled. Samples were ^{1/4}x transformed after a dummy variable of 0.0001 was added to avoid the weighting by zeroes, prior to the construction of a Bray–Curtis resemblance matrix. (b) Results of pairwise tests for the factor Ye×Si (Lo×Po) showing the frequency of abundances of Axinellid sponges and *Pentapora fascialis* differing among years 2–4 for each treatment of potting.

(a)													
Source	df	Axinellids				<i>Pentapora fascialis</i>				<i>Cliona celata</i>			
		MS	Pseudo-F	P (perm)	Unique Perm.s	MS	Pseudo-F	P (perm)	Unique Perm.s	MS	Pseudo-F	P (perm)	Unique Perm.s
Year Ye	2	0.13	0.079	0.921	9953	0.22	0.166	0.860	9944	1.36	7.914	0.050	9944
Potting Po	1	17.02	1.141	0.670	3	41.84	4.210	0.332	3	20.83	267.270	0.331	3
Location Lo (Po)	2	14.91	6.598	0.005	9959	9.94	12.329	<0.001	9959	0.08	0.053	0.950	9949
Ye×Po	2	5.31	3.147	0.146	9948	0.25	0.191	0.827	9953	0.43	2.514	0.206	9949
Site Si (Lo×Po)	20	2.26	6.629	<0.001	9904	0.81	5.202	<0.001	9924	1.48	9.111	<0.001	9907
Ye×Lo (Po)	4	1.69	1.317	0.280	9952	1.33	2.345	0.069	9954	0.17	0.333	0.858	9964
Ye×Si (Lo×Po)	40	1.28	3.757	<0.001	9895	0.57	3.668	<0.001	9889	0.51	3.173	<0.001	9856
Residual	792	0.34				0.15				0.16			

(b)						
Taxon	Treatment	Differences in years				
		2 = 3 = 4	2 = 3 ≠ 4	2 ≠ 3 = 4	2 = 4 ≠ 3	2 ≠ 3 ≠ 4
Axinellid sponges	Potting Stopped	7	2	2	0	1
	Control	7	0	4	1	0
<i>Pentapora fascialis</i>	Potting Stopped	12	0	0	0	0
	Control	8	0	2	2	0
<i>Cliona celata</i>	Potting Stopped	12	0	0	0	0
	Control	9	3	0	0	0

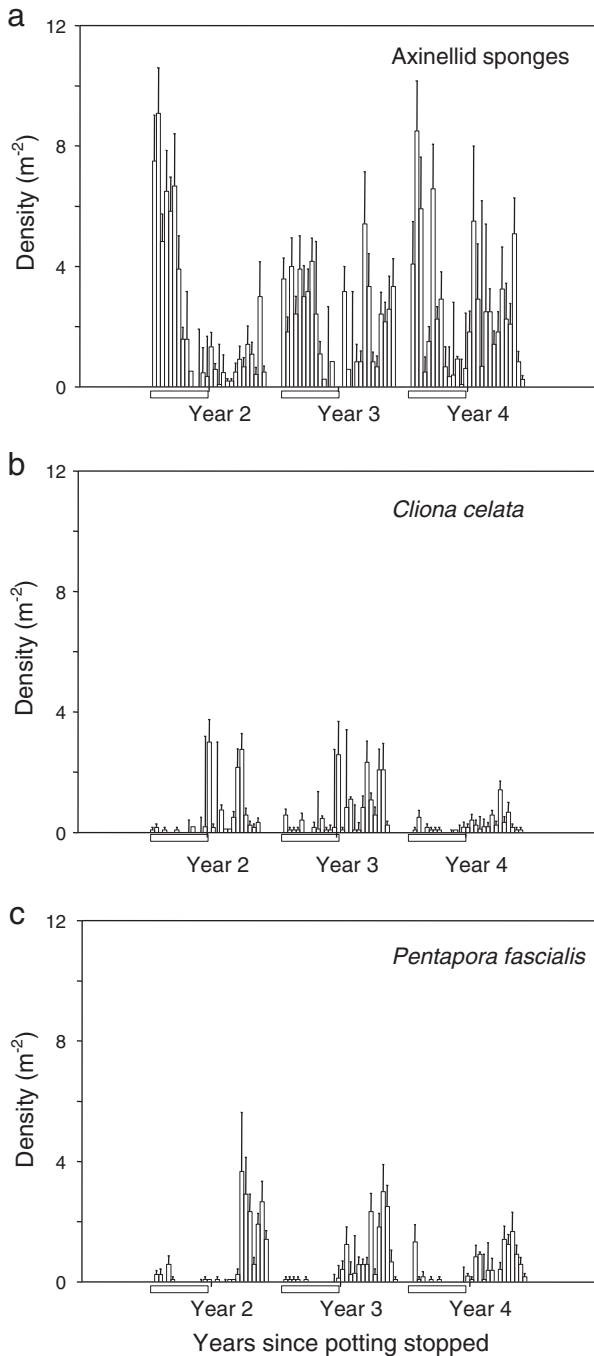


Fig. 3. Differences in abundances of key taxa identified from SIMPER analyses (see Table 2) from areas in Lundy protected from commercial potting for 4 years and areas continued to be fished. (a) Axinellid sponges, (b) *Cliona celata* and (c) *Pentapora fascialis*. The grey lines under the abscissa indicate the plots in areas where potting was stopped, correspondingly, the absence of a grey line indicates those places where potting continued. Each bar is the mean of 12 quadrats from a site (\pm SEM); the first six bars represent different sites in location 1 and the next six bars indicate sites in location 2. Since these show spatial variation which is treated as a random factor, the sites and locations are not identified.

of dissimilarity, with a dummy weighting of 0.0001 to reduce the effect of zero counts (Clarke et al., 2006). Dissimilarities were calculated using 4th-root transformed abundances of epifaunal species and one composite variable – total Axinellid sponges. A 4th-root transformation reduces the effect on Bray–Curtis dissimilarity values of differences in the overall abundances of different taxa such that the most abundant taxa do not distort the dissimilarity values. The contribution of each species to the overall Bray–Curtis dissimilarity for each

Table 5

Permutational MANOVA (PERMANOVA) examining differences in assemblages in places subject to experimental potting every year for 4 years versus control places where no potting occurred. The experiment was set in the NTZ where any form of fishing was prohibited. There were two locations for each treatment and at each location, six sites were used and $n = 12$ quadrats sampled. Samples were $\sqrt[4]{x}$ transformed after a dummy variable of 0.0001 was added to avoid the weighting by zeroes, prior to the construction of a Euclidean distance resemblance matrix.

Source	df	SS	MS	Pseudo-F	P (perm)	Unique perms
Experimental Potting EP	1	18360	18360	0.381	> 0.999	3
Location Lo (EP)	2	96364	48182	5.120	0.001	9943
Site Si (Lo(EP))	20	188200	9410	5.039	<0.001	9845
Residual	264	492970	1867			

comparison was determined using the SIMPER routine. These SIMPER analyses were used to identify species whose variations in abundance warranted significance testing by ANOVA. The species selected for ANOVA were those that contributed > 15% to the multivariate difference for at least one temporal or spatial comparison. Permutational analyses of variance (PERMANOVA; Anderson, 2001) on univariate data using Euclidean distances produce results identical to standard univariate ANOVA, this meant we could use the same analytical structure from above to test the patterns of differences in individual species/taxa that contributed to the observed patterns. *Post-hoc* separation of significant means was done using the pairwise procedure in PERMANOVA.

3. Results

Poor weather prevented data collection in 2004.

The PERMANOVA (Table 2a) showed much variation among sites and locations for each year and treatment, showing that sites at each location differed from each other substantially, with no consistent pattern through time since sites and locations are random factors these are not interpreted further. The prediction that temporal changes in assemblages would be different in those areas still open to commercial potting compared to those areas closed to commercial potting (Hypothesis 1) was not supported since the term $Ye \times Po$ was not significant (Table 2a). The distribution of pairwise differences between years did not vary for each treatment (Table 2b) indicating that the pattern of differences in assemblages between years two, three and four was similar for sites listed in locations in potted and non-potted areas. This lack of difference between treatments in respect of spatio-temporal variation is further illustrated by an nMDS

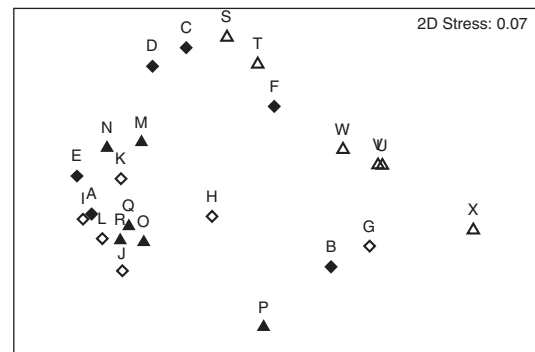


Fig. 4. nMDS indicating differences in assemblages associated with experimental potting in six sites in each of four locations at Lundy where there had been no commercial fishing for 4 years. The shapes indicate positions of centroids of quadrats ($n = 12$) for each site in location in each year; diamond shapes represent areas subject to experimental potting for 5 days every year for 4 years and the triangle areas those places which have not been potted over the same time period. The random effect of location is indicated filled vs. unfilled shapes and the random effect of sites indicated by letters.

Table 6

Results of SIMPER analysis showing which species contributed the greatest differences in samples in the places within the NTZ that were experimentally potted versus those places not subject to experimental potting. Axinellid sponges have been grouped together since these were difficult to distinguish in the field.

Species	Potted		Unpotted		Average dissimilarity	Dissimilarity/SD	Percentage contribution	Cumulative percentage
	Average abundances	Average abundances	Average abundances	Average abundances				
<i>Aiptasia mutabilis</i>	1.03	0.57	21.37	1.01	30.18	30.18		
Axinellid sponges	1.04	0.83	20.97	0.9	29.63	59.8		
<i>Raspalia ramosa</i>	0.48	0.32	10.65	0.83	15.04	74.84		
<i>Polymastia mammillaris</i>	0.25	0.08	5.08	0.53	7.17	82.01		
<i>Pentapora fascialis</i>	0.17	0.07	4.98	0.42	7.03	89.04		
<i>Eunicella verrucosa</i>	0.11	0.06	3.46	0.33	4.89	93.93		

plot of the centroids of sites and quadrats for each location in each treatment (Fig. 2). This plot indicates that although locations differed from each other in every year, there was no variation in assemblages between locations for each treatment consistent with the hypothesis. The SIMPER analysis indicated that 90% of the variation in assemblages came from six taxa, there were three taxa which each contributed more than 15% of the variation in patterns of assemblages between samples, these were Axinellid sponges, *Pentapora fascialis* and *Cliona celata* (Table 3). There was no support for the hypothesis (2) that temporal variation in the abundance of individual taxa would differ between those areas with no commercial potting in those areas where such potting continued (Table 4). The PERMANOVA analyses on the abundance of each of the taxa identified above indicated that there was much variation between locations and sites but there was no link with the cessation of potting (Fig. 3). Since sites and locations represent random variation for which we had no hypotheses, these terms are not interpretable.

In the experimental potting study, we examined differences in assemblages in areas subject to experiment reporting every year for 4 years (Hoskin et al., 2011) and places where potting was discontinued after the designation of the NTZ in 2003. The PERMANOVA on differences in assemblages showed no effect of the experimental potting treatment and further indicated that variation in assemblages was associated with the existing spatial differences which were independent of fishing (Table 5). This spatial variation is shown in the nMDS plot of the centroids for each site (Fig. 4) where there is no apparent clustering of treatments. SIMPER analyses indicated that six taxa contributed 95% of the observed dissimilarity (Table 6) with three taxa each contributing more than 15% (*Aiptasia mutabilis*, Axinellid sponges and *Raspalia ramosa*; Table 6). Individual PERMANOVA on these taxa were consistent with the patterns observed for the whole assemblage, in that there was no effective treatment and most of the variation was due to spatial differences at the smallest scale (Table 7). As before, since this variation comes from random effects for which we had no hypotheses, they are not interpreted further.

4. Discussion

During 4 years of sampling there were no important changes (potential effects of the NTZ or otherwise) in the assemblage of sessile epifauna that was monitored for this study, either in terms of multivariate composition or the abundances of individual species. The two main

findings to come from quantifying the responses of circalittoral epifauna were (i) consistent differences in the abundances of certain species between the NTZ on the east coast of Lundy and the control location on the west coast (e.g. dead men's fingers *Alcyonium digitatum*, ross coral *Pentapora fascialis* and the boring sponge, *Cliona celata*); and (ii), for a different set of species, consistent differences between replicate sites within the NTZ (e.g. the trumpet anemone *Aiptasia mutabilis*, the sponge *Raspalia ramosa* and axinellid sponges). These spatial differences were unchanging throughout the monitoring programme, so they almost certainly reflect the influence of perennial environmental differences among the locations concerned (e.g. differences in wave energy, depth, etc.). These results were consistent with those obtained by Eno et al. (2001), although it should be noted that their study was a much smaller spatial and temporal scale than the one presented here (Fig. 5). The fact that no epifaunal species increased in abundance within the NTZ or declined in abundance in areas outside the NTZ suggests that these species are all generally insensitive to commercial shellfish potting and that consequently, commercial potting probably has little impact on benthic assemblages on contrast with benthic trawling (Thrush and Dayton, 2002).

There are a few possible alternative explanations for a lack of effect of potting. The first is that all commercial potting effort at Lundy stopped contemporaneously with the establishment of the NTZ, such that both the NTZ and non-NTZ areas were unfished. Evidence from commercial landings (Devon Sea Fisheries Committee; <http://www.devonandsevernifca.gov.uk/>) and our own observations from the island, in addition to those of the Lundy wardens would indicate that this was not the case; on many occasions we saw boats legitimately potting outside the NTZ. Alternatively, if the NTZ was not complied with, then there would be no difference between the NTZ and the commercially fished areas. There was though, a high degree of compliance with respect to the Lundy NTZ (Hoskin et al., 2011). Since the pots are deployed with floats/buoys at each or either end of a string of pots, and have to be soaked for a minimum of 24 hours then, as the wardens patrolled the island everyday we judged that the NTZ had a high degree of compliance by those fishing with pots. Another possible model is that the level of potting at Lundy was not substantive enough to be detectable upon cessation. The fact that compared to the fished area at Lundy, the population of lobsters rapidly and significantly increased (Hoskin et al., 2011) would indicate that there really was an impact of lobster fishing. A fourth possibility is that the experiment was not run long enough. This could be true of all ecological experiments when no

Table 7

Permutational MANOVA (PERMANOVA) examining differences in abundances of three key taxa in places subject to experimental potting for crustaceans every year for 4 years versus control places where no potting occurred. The experiment was set in the NTZ where any form of fishing (commercial or recreational) was prohibited. There were two locations for each treatment and at each location, six sites were used and $n = 12$ quadrats sampled. Samples were $\sqrt[4]{x}$ transformed after a dummy variable of 0.0001 was added to avoid the weighting by zeroes, prior to the construction of a Euclidean distance resemblance matrix.

Source	Df	<i>Aiptasia mutabilis</i>				Axinellid sponges				<i>Raspalia ramosa</i>			
		MS	Pseudo-F	P (perm)	Unique perms	MS	Pseudo-F	P (perm)	Unique perms	MS	Pseudo-F	P (perm)	Unique perms
Experimental Potting EP	1	15.694	0.511	0.672	3	2.028	0.407	> 0.999	3	24.034	0.593	0.666	3
Location Lo (EP)	2	30.724	8.513	0.003	9947	4.978	3.274	0.067	9957	40.521	4.606	0.006	9943
Site Si (Lo(EP))	20	3.609	9.977	<0.001	9925	1.520	6.727	<0.001	9916	8.798	6.639	<0.001	9809
Residual	264	0.362				0.226				1.325			

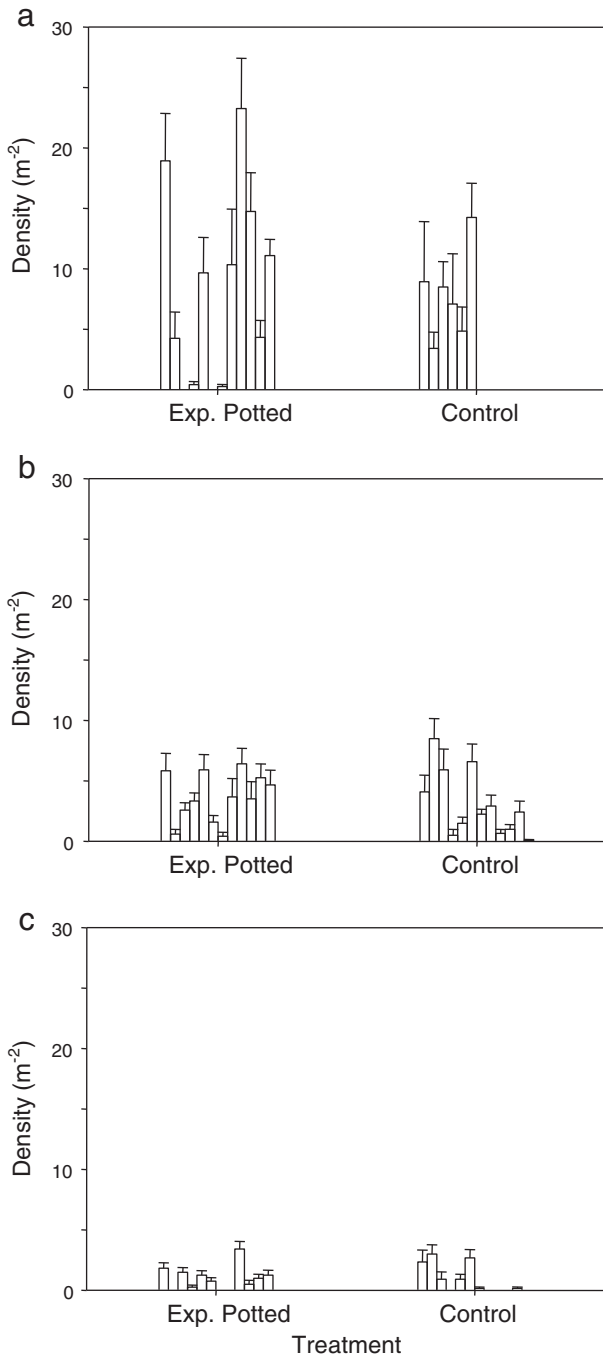


Fig. 5. Differences in abundances of key taxa identified from SIMPER analyses (see Table 4) associated with experimental potting in six sites in each of four locations at Lundy where there had been no commercial fishing for 4 years, (a) *Aiptasia mutabilis*, (b) Axinellid sponges and (c) *Raspalia ramosa*. 'Experimental potting' refers to places where experimental pots were set for 5 days in June/July every year for 4 years and 'controls' are places where there had been no potting. Each bar is the mean of 12 quadrats from a site (\pm SEM); the first six bars represent different sites in location 1 and the next six bars indicates sites in location 2. Since these show spatial variation which is treated as a random factor, sites and locations are not identified.

effects have been observed, and is often used as a criticism, or reason not to publish (Underwood, 1997, 1999). Our results were based on a hypothesis of a detectable effect after 4 years; such a period has been sufficient to be able to detect temporal changes in similar assemblages in other temperate rocky subtidal systems (Ambrose, 1986; Bulleri and Benedetti-Cecchi, 2006; Schiel et al., 2004). It must be noted that in some systems, recovery of directly fished populations is not as fast as the Lundy lobsters (Beukers-Stewart et al., 2005). If the habitats

have been exposed to a sustained impact of many years, the recovery may well take much longer than the 4 years that was available to this study. In the work presented here, the hypothesis was not of a population increase but of greater variability in the populations not impacted by potting, such changes would be apparent before experiments would detect an increase in populations of individual taxa. It is not unreasonable then to consider that the lack of effect of potting is probably representative of what really happens on rocky reefs at Lundy and is consistent with a previous shorter-term experiment (Eno et al., 2001). Some habitats such as biogenic reefs may respond differently to the impact of potting and are likely to respond differently when potting stops. Research into the effect of MPAs should consider the context of the habitats affected.

Both the commercial potting, and our experiment, used standard commercial methods of shooting and recovering the pots. This involves the vessel steaming slowly or drifting in the wind whilst the fisher throws the first pot in. This being heavy, drags the next part of the string in and so on until the entire string is deployed. Hence, damage if it occurs, would be a result of (a) the arrival of the pots on the benthos, (b) residence on the benthos whereby any sideways movement of the pots and attaching ropes would damage biota, and (c) recovery from the benthos. If there is substantial wave action during a 'soak' such that the pots move during the residence phase, damage may occur and it may be only during the circumstances that potting may exert a damaging effect on benthic assemblages. The experimental study by Lewis et al. (2009) indicated that damage from pots occurred during storms and this effect could be magnified in systems where fishers deploy pots in strings. Experimental potting should be done whereby lobster/crab pots are agitated during a soak simulating wave events, or pots could be deployed immediately prior to storms and the resulting damage on benthic assemblages evaluated.

Potting for Crustacea on rocky habitats in inshore waters may well be a benign fishery with limited impact on benthic assemblages (this work and Eno et al., 2001). There may of course be indirect effects of such fishing associated with the removal of predators such as lobsters (Babcock et al., 2010; Siddon and Witman, 2004) but these need to be further evaluated in places where lobster populations are either expanding or contracting significantly. In the context of integrating conservation strategies with sustainable use of marine resources, it is suggested that management needs to be directed at high impact activities such as benthic trawling dredging rather than potting so that in any zoning of marine parks, potting for Crustacea may be subject to less management intervention than other, more destructive, activities. This is particularly true in marine parks where activities are zoned, such that fishing co-exists alongside strong sanctuary-level protection. Here managers may add a new category of habitat protection where ecologically benign or neutral fisheries can be permitted but destructive fishing practices prohibited.

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