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Title: Experimental Potting Impacts on Common UK Reef Habitats in Areas of High and Low Fishing Pressure.

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14 **1. Abstract**

15 Impacts of mobile fishing gears on habitat and benthos have been well documented;
16 in contrast, less studied physical impacts of static fishing gear on benthic habitats are
17 still debated. Pot fishing, is a growing sector in the UK and evidence of any impacts
18 is needed to inform management. This study simulated high intensity experimental
19 pot fishing on the epibenthos of two common UK reef habitats in Northumberland,
20 UK. Single tethered pots were fished in intensively and lightly fished areas over the
21 course of 2 months. Within each area, 3 experimental sites and control sites were
22 surveyed before and after fishing using photoquadrats ($n= 240$ per 290m^2 site)
23 collected by SCUBA divers. PERMANOVA analysis indicated no evidence of
24 epibenthic species abundances decreasing due to physical crushing or abrasion from
25 potting on either intensively or lightly fished reefs. A shift in community composition
26 over time was detected but was attributed to natural change as epibenthos in control
27 sites shifted similarly. Experimental pot impacts far exceeded those of the local
28 commercial pot fishery, providing relevant evidence for statutory governing bodies
29 revisiting current fisheries management. Results are applicable across Europe, due
30 to the selection of habitats with abundant and commonly distributed benthic species.

31 Key words: *Fishing impacts; European Marine Site; Natura 2000; Photoquadrats;*
32 *Pots; Temperate reefs; Traps.*

33 **Introduction**

34 Habitat quality and health have been recognised as integral to the sustainable
35 management of fish stocks as part of an ecosystem based approach to fisheries
36 management (EBFM) (Pikitch *et al.*, 2004; Armstrong and Falk-Petersen, 2008;
37 Salomidi *et al.*, 2012). Temperate reefs are some of the most biodiverse and
38 productive habitats in the world, and their epifauna are key ecosystem components
39 (Hiscock and Tyler-Walters, 2006) providing food and shelter for other species,
40 including commercially important ones (Lambert *et al.*, 2011). In light of the
41 importance of fishery-habitat interactions for successful EBFM, effects of high impact
42 mobile fishing gears on habitat and benthos have been well documented; epifauna
43 prove sensitive to both direct (physical disturbance) and indirect (e.g. changes of
44 food web structure) impacts of fishing (Kaiser *et al.*, 1996; MacDonald *et al.*, 1996;
45 Collie *et al.*, 1997; Collie *et al.*, 2000a; Collie *et al.*, 2000b; Kaiser *et al.*, 2006;
46 Lambert *et al.*, 2011). Static fishing gears, such as baited-pot fishing, have been
47 assumed to be relatively benign (Eno *et al.*, 2001; Lewis *et al.*, 2009; Shester and
48 Micheli, 2011; Coleman *et al.*, 2013) yet there is limited evidence to support this (Eno
49 *et al.*, 2013).

50 Static gear fisheries are important components of the UK fishing industry, landing
51 86,600 tonnes worth £173.6 million per annum and landing 41% of the shellfish in the
52 UK in 2014, including high value species such as lobster and crab which are primarily
53 caught inshore using pots or traps (MMO, 2015). As demersal and pelagic fish
54 landings decline, the use of static gears may increase (Acheson and Brewer, 2003;
55 Turner *et al.*, 2012; Molfese *et al.*, 2014). Although, these are likely to have lower
56 disturbance per unit effort than mobile gear, cumulative effects could be considerable
57 in light of increasing use (Shester and Micheli, 2011; Coleman *et al.*, 2013). The

need to address the lack of information on static gear fishing impacts has become increasingly relevant due to the requirements to assess the effects of fishing activities in UK designated conservation areas: European Marine Sites (EMSs) and Marine Conservation Zones (MCZs). Many of which are located inshore and currently fished with static gears (MMO, 2016).

The present research aims to investigate direct impacts of potting – a commonly used static gear in the UK– on epibenthos of common reef habitats, using marine protected areas in the North East of England as a case study. Pot fishing occurs worldwide (Coleman *et al.*, 2013) and in Northumberland, targets European lobster (*Homarus gammarus*), edible crab (*Cancer pagurus*), and velvet crab (*Necora puber*). Potting generally consists of deploying 10-30 baited pots attached to a ‘mainline’ (termed a ‘string’ or ‘fleet’) to the seafloor, held in place by anchor-weights on each end of the fleet in order to prevent wave or current induced dragging, and marked by buoys at each end. Deployed pots are left to ‘soak’ (fish) for 1–3 days before retrieval. Direct physical impacts of potting gear on the benthic environment may occur: during deployment, crushing epibenthic organisms when pots land on the seafloor (Eno *et al.*, 2001); during soaking, where underwater pot movement by strong tides and/or waves may cause abrasion (Jones *et al.*, 2000; Lewis *et al.*, 2009); and during retrieval, where dragging gear along the seafloor and lifting may damage epibenthos (Eno *et al.*, 2001; Coleman *et al.*, 2013).

Physical impacts of potting on epibenthos have received relatively little attention globally. Although potential impacts are regularly mentioned in official guidance on habitat management as a threat to fragile epibenthic species (Roberts *et al.*, 2010; Natural England *et al.*, 2012; Eno *et al.*, 2013; Grabowski *et al.*, 2014), only two UK (Eno *et al.*, 2001; Coleman *et al.*, 2013) and two US studies exist (Lewis *et al.*, 2009;

Shester and Micheli, 2011). These did not find evidence that potting impacted habitat or epibenthos, however these had limitations. Eno *et al.* (2001) conducted experimental fishing in South West England, but low replication and the analytical methods used reduced statistical power and led to sampling bias and uncertain conclusions (Coleman *et al.*, 2013). For example, some species appeared to significantly increase in abundance in response to the potting treatment although these species could not have settled and grown within the time period and their presence was attributed to sedimentation prior to the experiment (Eno *et al.*, 2001). Coleman *et al.* (2013) compared circalittoral assemblages in sites both in fished areas and a protected no-take zone (NTZ) in south west England, showing no change in sessile epifauna composition over four years between sites. This highlighted a major difficulty in assessing localised impacts of potting. The small physical footprint of pots ($\approx 1\text{m}^2$), and consequently, the small scale of actual interactions with the seafloor, compared to the large areas covered by potting gear (fleet of 12 pots $\approx 800\text{m}^2$). Therefore, the low sample size compared to the large experimental area, coupled with control sites of differing depths (5–10m), exposures (controls areas more wave exposed) and substrates (boulder reefs compared to bedrock) (Coleman *et al.*, 2013), may have resulted in small-scale impacts being overlooked. No studies to date have tried to control for habitat differences when examining potting impacts. This may be particularly important to the detection of low level, small-scale impacts. Experts suggest that certain rocky reef habitats such as those dominated by erect and branching species, fast growing faunal turfs and kelp, are sensitive to high levels of potting activity (Eno *et al.*, 2013), but more experimental evidence is required to confirm this.

The few studies of direct potting impacts in Europe have only been conducted in circalittoral habitats. In Florida, single pots impacted a greater area of benthos in

109 shallow than in deep water due to increased pot movement from wave action (Lewis
110 *et al.*, 2009). Here, research focuses on shallower infralittoral and circalittoral
111 habitats, for this reason. Pots were fished experimentally in specific habitats, with
112 proximate control sites exposed to the same environmental conditions. It also aimed
113 to address issues of statistical power with high-density sampling. This allowed
114 cumulative potting impacts to be investigated robustly.

115 The two most common subtidal habitats within the Berwickshire and North
116 Northumberland Coast European Marine Site (BNNC EMS) (Mercer *et al.*, 2003;
117 Mercer, 2012) were selected for investigation. Defined in the 'Marine Habitat
118 Classification for Britain & Ireland version 15.03', these are: 'Faunal and algal crusts
119 on exposed to moderately wave-exposed circalittoral rock' (FaAlCr) and '*Laminaria*
120 *hyperborea* park with foliose red seaweeds on moderately exposed lower infralittoral
121 rock' (Lhyp.Pk). These habitats are found across Europe (Connor *et al.*, 2006;
122 MESH, 2008), and are important within the BNNC EMS (Regulation 33(2) advice,
123 English Nature & Scottish Natural Heritage 2000), as they are characterised by erect
124 and presumably fragile species, likely to be impacted by even low levels of
125 disturbance (Roberts *et al.*, 2010). In addition, Lhyp.Pk has high biodiversity and
126 biomass (Connor, 1989; Connor *et al.*, 2004) and may be particularly prone to impact
127 due to higher levels of wave action (Lewis *et al.*, 2009).

128 Abundances of erect and potentially fragile species are expected to decline due to
129 physical abrasion from pot fishing, with bare rock and percentage cover of encrusting
130 species increasing (hypothesis 1). Areas which are lightly fished by mobile gears
131 have higher biomass, larger individuals and greater abundances, especially of erect
132 fauna and flora, than intensely fished sites (Collie *et al.*, 1997; Collie *et al.*, 2000b;
133 Kaiser *et al.*, 2002; Lambert *et al.*, 2011); thus here, lightly fished sites were expected

to be more sensitive to fishing impacts. Potting pressure for the NIFCA district was modelled using a combination of vessel sightings and landings data (Turner *et al.*, 2015; Stephenson, unpublished) and experimental sites within both intensively and lightly fished areas (herein referred to as intensively and lightly fished sites respectively) were subjected to the same amount of experimental potting. Lightly fished areas were expected to have greater abundances of erect long-living species (hypothesis 2), which would exhibit greater change in assemblage structure than high fished areas (hypothesis 3).

2. Methods

2.1. Study site

Bathymetry, substrate hardness, fishing pressure and habitat data were combined in GIS to select intensively and lightly fished rocky areas less than 16m deep (Fig 1). Bathymetry and hardness layers were created for the Northumberland inshore fisheries district using Olex echosounder data (2012–2014) from the NIFCA patrol vessel *St Oswald* (Fig 1, C and D). Summer fishing effort maps for 2012-2013 were produced for the district following Turner *et al.* (2015), combining fishing vessel sightings and landings data (Fig 1, E). Fishing effort was divided in three categories using the natural jenks function (Esri ArcGIS v. 10.1): low (0–139 pots in the sea month⁻¹.km⁻²), medium (140–187 pots in the sea month⁻¹.km⁻²) and high (188–265 pots in the sea month⁻¹.km⁻²) (Fig 1, E).

Areas of target benthos (Lhyp.Pk and FaAlCr), matching the selection criteria, were identified using previous ecological monitoring data (Edwards, 1983; Foster-Smith and Foster-Smith, 1987; Connor, 1989; Holt, 1994; Brazier *et al.*, 1998; Mercer *et al.*, 2003; Mercer, 2012; Fitzsimmons *et al.*, 2015), and their presence confirmed at target sites during ground truthing by SCUBA divers. Areas large enough for

experimental potting were not found for lightly fished Lhyp.Pk areas, as these habitats are heavily fished during the summer months (NIFCA, pers. comm.).

2.2. Design and layout of experimental fishing

Potting impacts were investigated by fishing experimentally and measuring epibenthos diversity and abundance in photoquadrats collected by scuba divers. Three replicate sites were selected for each fishing pressure level (Fig 1, F) at similar depths and distances from each other (Van Rein *et al.*, 2012). FaAICr sites were surveyed between July and September 2014 and Lhyp.Pk sites were surveyed between July and August 2015 (Fig 1, F). Experimental sites comprised two areas: a 25x10m impact area and a smaller 5x10m control area (Fig 2), delimited by buoys at the East and West sides (Fig 2).

Prior to experimental fishing, transects were deployed 2.5m South and North of the anchor weights respectively (Fig 2). *In-situ* sampling was undertaken by SCUBA divers using a 625cm² photoquadrat (25 x 25cm) (Canon Powershot s120, Ikelite underwater housing and two Sea&Sea YS-02 Underwater Strobes) (Fig 3, a) to record epibenthos and habitat type, to provide a baseline (Fig 2). Photoquadrats were then taken haphazardly on both sides of the tape, up to 2.5m away from it, once every metre. A total of 50 photographs were collected per transect. A further 20 photoquadrats were similarly collected in the control area (Fig 2). As the small quadrat size did not allow for easy sampling of kelp abundance (*Laminaria hyperborea*, >50cm), a crucial element of Lhyp.Pk community composition and structure, kelp abundance was recorded in a 1m² area on either side of transects every 5m ($n=18$ per site).

Due to the spatial heterogeneity of temperate subtidal rocky reef habitats (Birkett *et al.*, 1998; Hartnoll, 1998; Van Rein *et al.*, 2012), a large number of small quadrats

were deemed to provide a greater accuracy, precision and efficiency when describing communities compared to fewer large quadrats (Sayer and Poonian, 2007). The use of small photoquadrats (25x25cm) produced high resolution imagery of the area, and the short focal length (40cm) reduced any negative influence of water column turbidity on species identification (Leujak and Ormond, 2007). The simple deployment and ease of operation meant that *in-situ* task loading was minimised, increasing sampling replication for a given bottom time (Van Rein *et al.*, 2012).

Disturbance was then generated using a single standard parlour pot (0.66x0.46x0.38m, with 10mm steel frame) attached via a 3m 'leg', tethered to an anchored mainline (Fig 2). This was deployed and left to soak in the 'impact area' of each site for a minimum of 48 hours (FaAlCr mean soak time : 72h in intensively fished areas, and 65h in lightly fished areas; Lhyp.Pk mean soak time: 169h in intensively fished areas) (Fig 2), then hauled North-South, against the tide, to mimic local commercial practice. The experimental area was resurveyed, a further 120 photoquadrats being collected per site. Soaking, surveying and hauling was repeated a total of three times for each site.

2.3. Data extraction

In order to maintain efficient sampling, communities were treated as two-dimensional (Meese and Tomich, 1992; Van Rein *et al.*, 2011). Epibenthos from the images collected from FaAlCr habitats was identified to the lowest possible taxonomic level using up-to-date identification manuals (Cornelius, 1995; Hayward and Ryland, 1995; Bunker *et al.*, 2012; Wood, 2013b; Wood, 2013a) and expert knowledge. Epibenthos from the images collected from Lhyp.Pk habitats was identified to broad benthic categories of coarse taxonomic resolution to represent their structural role within the community. This was due to difficulty identifying individuals to species level without a

microscope and small algal species often being partially obstructed by larger overlapping algae. Groups selected (Van Rein *et al.*, 2012) were: red foliose algae >1cm above the substratum, non-red foliose algae >1cm above the substratum, red algal turf <1cm above the substratum, non-red algal turf <1cm above the substratum, encrusting algae, mixed hydrozoan/bryozoan turf <1cm, feather hydroids, and encrusting bryozoans.

Benthos which could not be identified because it was too far away, too dark (e.g. in a crevice) or was not biological material was excluded from the analysis. Mobile species were also excluded from the analysis. Percentage cover was assessed using a point method (Aronson *et al.*, 1994), which is time efficient, unbiased by observers, and more sensitive to changes in community composition than visual estimation and frequency occurrence methods (Aronson *et al.*, 1994; Drummond and Connell, 2005; Van Rein *et al.*, 2012). Images were overlaid with 100 evenly placed points (Drummond and Connell, 2005), using photoQuad (v1.0) (Trygonis and Sini, 2012), epibenthos under each point was identified and taken to represent 1% of the image area (Fig 3, b).

2.4. Data analysis

The locations of the pot and anchor weights were recorded during every dive. Sites were divided into 5 sections of 5m running north-south along the transect. For each site, images from the two most heavily impacted sections (10x10m) were used for analysis. Data in the experimental area before fishing (E_B) were compared to the same areas after the three fishing events (E_A). To ensure any changes detected were potting impacts not natural variation, data for control sites were analysed before (C_B) and after (C_A).

233 A total of 1440 images were analysed. 480 images for each habitat (40 images for
234 each E_B, E_A, C_B and C_A per site). Benthic community data were square root
235 transformed to reduce dominance of common taxa (Martin *et al.*, 2012) and Bray
236 Curtis similarity matrices produced.

237 Differences in community data were investigated using mixed models in
238 PERMANOVA v. 1.0.5 (type III sum of squares, under a reduced model with 999
239 permutations) following a 4 factor design with interaction. Habitat was used as a fixed
240 factor with 3 levels: FaAlCr intensively fished, FaAlCr lightly fished site, Lhyp.Pk
241 heavily fished; site was nested within habitat as a random factor with 3 levels (1, 2
242 and 3). Fishing was entered as a fixed factor with 2 levels (experimental (E) and
243 control (C) areas) and time as a random factor with 2 levels (before (B) and after (A)
244 fishing). Tests of homogeneity of dispersion were conducted using PERMDISP in
245 PERMANOVA to test the null hypothesis of no difference in dispersion among *a priori*
246 groups. Post-hoc analysis between treatments E_B-C_B, E_B-E_A, C_B-C_A and E_A-C_A using
247 the pairwise function in PERMANOVA v. 1.0.5 investigated which factor levels were
248 responsible for significant interactions (type III sum of squares, reduced model, 999
249 permutations).

250 The sensitivity of this PERMANOVA mixed model to changes in community
251 composition was investigated by modifying the community data collected from
252 experimental areas after fishing (see supplementary materials). Modified data for
253 experimental area after fishing (M_A) were created based on expert knowledge of
254 potential potting impacts. Benthos likely to be impacted by physical impacts due to
255 physical characteristics, i.e. erect, fragile species, was proportionally reduced by a
256 total of 5%, and that expected to thrive was proportionally increased by a total of 5%
257 (Eno *et al.*, 2001; Coleman *et al.*, 2013; Eno *et al.*, 2013). Species which were

258 expected to be unaffected by potting were not modified. E_A was then replaced by M_A
 259 in the mixed model to investigate whether known 5% changes in species would be
 260 detected.

261 The components of variation were estimated to provide a measure of the multivariate
 262 variability between factors (Anderson *et al.*, 2008; Martin *et al.*, 2012; Van Rein *et al.*,
 263 2012). The use of Bray-Curtis similarity matrices, allowed estimates of the
 264 components of variation to be directly interpreted as percentage dissimilarity of
 265 conditions within experimental factors (Anderson *et al.*, 2008). Variability associated
 266 with factor 'site' indicates general spatial variability, that associated with factors E_B-E_A
 267 and E_A-C_A, indicates temporal variability, and that associated with 'residuals' indicates
 268 the residual variability among replicate photoquadrat samples (Anderson *et al.*,
 269 2008).

270 To visualize multivariate patterns, principal coordinate ordination (PCO) was used
 271 (PERMANOVA v. 1.0.5). As the variation explained by the axes of all plots was high,
 272 these best captured the high-dimensional structure, more closely reflecting the
 273 resemblance values used in the portioning methods for PERMANOVA, than a non-
 274 metric MDS plot (Anderson *et al.*, 2008). There were too many samples to visually
 275 examine differences in assemblages between treatments E_B, E_A, C_B and C_A for each
 276 habitat in a single ordination (n=480) therefore centroids of each treatment and site
 277 were produced and plotted using PCO (Terlizzi *et al.*, 2005)

278 A total of 72 kelp abundance recordings were obtained from Lhyp.Pk habitats.
 279 Differences in kelp abundance between treatments were tested for significance in
 280 PERMANOVA v. 1.0.5 (type III sum of squares, under a reduced model with 999
 281 permutations) following a 2 factor design with interaction (site as a random factor with

282 3 levels: site 1, 2, 3, and treatment type as a fixed factor with 2 levels: E_B-C_B, E_B-E_A,
283 C_B-C_A and E_A-C_A).

284 3. Results

285 3.1. Site features

286 Intensively and lightly fished, rocky FaAlCr sites had similar assemblages dominated
287 by faunal (e.g. encrusting bryozoan, *Spirobranchus* spp and short hydroid turf) and
288 algal (Corallinaceae) crusts (Fig 3, B and C). Sites tended to have a grazed
289 appearance, potentially due to the abundance of *Echinus esculentus* (Connor *et al.*,
290 2004). Erect species such as *Alcyonium digitatum* and other feather hydroids were
291 frequently encountered (Fig 5, A and B). There were some differences in
292 assemblages between intensively and lightly fished areas. Generally, the intensively
293 fished sites were slightly shallower (~12m compared to ~13.5m for lightly fished
294 sites), and had lower abundances of fleshy erect red algae (Fig 5, A and B).

295 Intensively fished sites also had a greater diversity of Cnidaria (*Tubularia indivisa* and
296 *Caryophyllia smithii*) and Chordata (*Ciona intestinalis* and *Clavellina lepadiformis*). All
297 differences in assemblages were for species with low abundance (Fig 5, A and B).

298 The kelp dominated rocky Lhyp.Pk habitat (Connor *et al.*, 2004) was only surveyed in
299 intensively fished areas at depths of between 3.5 and 4m (chart datum). Sites were
300 dominated by erect species such as sparse kelp (*Laminaria hyperborea* and
301 *Saccharina latissima*), with a dense turf of foliose red seaweeds (*Callophyllis laciniata*,
302 *Plocamium cartilagineum*, *Delesseria sanguinea*, *Hypoglossum hypoglossoides* and
303 *Phycodrys rubens*) as well as coralline and faunal crusts (bryozoans and hydroids,
304 *Spirobranchus* spp and *Balanus* spp.) (Fig 5, C). Feather hydroids and sponges were
305 also frequently encountered although in low abundances (Fig 5, C).

3.2. *Multivariate analysis of potting impacts*

The non-significant interaction Si(Ha)xFixTi indicated that assemblages did not differ between experimental treatments at sites within habitats ($p=0.06$, Table 1). As this value was marginal, post hoc analysis was undertaken to investigate if communities differed between treatments for individual habitats.

The same overall changes were observed in percentage benthos cover between experimental treatments in different habitats and fishing pressures (Post hoc analysis,

Table 2). The non-significant interaction SitexE_B-C_B (

Table 2) indicated assemblages did not differ between Experimental versus Control treatments for all habitats before fishing (all $p>0.179$), bar one (site 1 intensively fished Lhyp.Pk, $p=0.032$).

Significant interactions were observed between SitexE_B-E_A for all habitats and fishing pressures (all $p>0.002$,

Table 2). Post hoc testing confirmed assemblages differed before and after experimental fishing at all intensively fished FaAlCr and Lhyp.Pk sites (all $p<0.048$), none of which were affected by dispersion (all $p>0.06$). Dispersion was detected for FaAlCr lightly fished habitats ($p<0.014$). Assemblages also differed between C_B and C_A at all sites for all habitats and fishing pressures (all $p<0.049$, no dispersion effects, all $p>0.094$). However, assemblages did not differ between E_A and C_A at any sites for any habitats and fishing pressures (all, $p>0.145$), indicating that temporal change in community composition between E_B and E_A is not due to potting impacts. As dispersion effects were detected for lightly fished FaAlCr habitats, it is unclear whether differences were due to location or dispersion (Anderson et al., 2008). The

PCO centroids were used to further investigate the direction and magnitude of changes in assemblages between treatments (Fig 4).

Changes in multivariate space between treatments were similar for all habitats and fishing pressures. E_B and C_B samples showed clustering by site as did E_A and C_A (Fig 4, A-C). Directions and distances between E_B and E_A were similar to C_B and C_A at individual sites. This reinforces PERMANOVA results (**Error! Reference source not found.**) that community assemblages changed in similar ways in experimental and control areas, suggesting experimental potting impacts were not responsible.

3.3. Sensitivity of multivariate analysis to modified community data

Significant interaction confirms the model's ability to detect fixed differences in assemblages of $\pm 5\%$ between experimental treatments within habitats ($Si(Ha) \times FixTi$, after experiment, $p=0.022$) (Table, A.4, Supplementary materials). Post hoc analysis revealed community assemblages differed between E_B-M_A at all sites for all habitats and fishing pressures (all $p<0.025$). M_A-C_A also differed at sites for all habitats and fishing pressures bar one (all, $p<0.041$, except site 1 FaAlCr intensively fished, $p=0.16$) indicating that modified community data differed from the control areas. Simulated changes in community assemblages (5%) were detected by the PERMANOVA mixed model.

3.4. Species composition

Only small differences in individual species' abundance were observed between treatments in FaAlCr and Lhyp.Pk (Fig 5, A-C). Samples from intensively and lightly fished FaAlCr habitats had high cover of faunal and algal turf, *Spirobranchus* spp, bare rock and gravel (Fig 5, A and B). Still frequently occurring but with lower percentage cover were encrusting algae and encrusting bryozoans (Fig 5, A and B).

Samples from intensively fished Lhyp.Pk had high cover of red algae turf (<1cm), red algae (>1cm), non-red algae (>1cm) and short faunal (hydroid, bryozoan) turf (Fig 5, C). Frequently occurring but with lower percentage cover were encrusting algae, encrusting bryozoan, bare rock and *Spirobranchus* spp (Fig 5, C).

Species percentage cover did not differ greatly pre- and post-experimental fishing in either experimental or control areas (Fig 5, A-C). SIMPER highlighted individual species' contribution to significant differences between treatments at site level.

Consistent indicators (those contributing >10% dissimilarity for all sites, with a dissimilarity:standard deviation ratio >1 (Clarke, 1993; Terlizzi *et al.*, 2005)) were the same for each site, within habitats, and contributed to differences between treatments in similar ways (Table A.1, A.2 and A.3, supplementary materials).

Kelp abundance (*Laminaria hyperborea* and *Saccharina latissimi* >50cm) was highly variable (**Error! Reference source not found.**, supplementary material) with no differences found between sites (all $p>0.08$) or treatments (all $p>0.17$).

4. Discussion

No declines in erect species abundances were found under intensive potting, supporting previous UK potting studies (Eno *et al.*, 2001; Coleman *et al.*, 2013) (rejection of hypothesis 1). Direct impacts were not observed in either intensively or lightly fished FaAlCr, or intensively fished Lhyp.Pk habitats; a shift in community composition was found but attributed to natural change when controls were considered (rejection of hypothesis 2). Sensitivity analysis of the models ensures that community change would have been detected at this small-scale. In the experiments, hauling three pots in a 10x10m area over 1 month at each site, was comparable to a

fishing pressure of ca. 30,000 pot hauls month⁻¹.km⁻², which exceeds more than threefold even the highest estimated fishing pressure in the NIFCA district (Table 3). The physical footprints of static gears such as shellfish pots are much smaller than mobile gears such as trawls and dredges (Nielsen *et al.*, 2013), making it very unlikely that pots fished commercially would land, soak and be hauled, in exactly the same location on successive fishing trips (Eno *et al.*, 2001).

Although experimental fishing differed from commercial practices (a single pot, compared to fleets of 10 or more pots, both tethered by two weights), it assured accurate placement within the small study area. This enabled the collection of a large number of photoquadrat samples in the small area, ensuring images were of epibenthos which were likely to have been in contact with the trap, providing reliable information needed for small-scale impacts. This fishing method was deemed more damaging than commercial practice, due the higher weight:pot ratio. Overall, this experiment far exceeds the maximum likely direct impact from commercial potting over short periods of time (1–2 months). Experimental studies provide useful insights into direct impacts, and relative severity of these, as well as investigating habitat–fishing gear interactions (Kaiser *et al.*, 2006; Hinz *et al.*, 2009). However, experimental results are difficult to extrapolate to the ecosystem scale ideally required for EBFM (Hiddink *et al.*, 2006; Hinz *et al.*, 2009). Given the limited research on potting impacts to date, robust and focussed experimental evidence remains an important first step (Eno *et al.*, 2001; Gray *et al.*, 2006). Long-term impacts were outside of the scope of this research, although previous research suggests that these are unlikely (Coleman *et al.*, 2013)co.

Erect species found in intensively and lightly fished FaAlCr and intensively fished Lhyp.Pk habitats, such as *Laminaria hyperborea*, red and brown algae >1cm,

403 *Delessaria sanguinea*, *Flustra foliacea*, *Alcyonium digitatum* and feather hydroids
404 (e.g. *Nemertesia ramosa*) are all expected to recover between 6 and 36 months from
405 mechanical interference, based on crushing, physical blows, or rubbing and erosion
406 of a single passage of a standard scallop dredge across the organism (Jackson,
407 2004; Tyler-Walters, 2006; Tyler-Walters and Ballerstedt, 2007; Budd, 2008). Thus, it
408 is likely that given time between repeat potting in any give location by commercial
409 fishers, and the lower level of impact from potting compared to scallop dredging
410 (Thrush and Dayton, 2002; Coleman *et al.*, 2013; Eno *et al.*, 2013), these species will
411 not be severely damaged even at the high potting pressures mimicked here, and are
412 expected to recover between fishing events. Encrusting and smaller species such as
413 encrusting bryozoans, sponges, *Balanus* spp and *Spirobranchus* spp, are unlikely to
414 be damaged by pots, their shape and size enabling them to withstand physical
415 disturbance and abrasion.

416 It is equally unlikely, given the lack of evidence of short-term direct impacts, that
417 longer-term direct potting impacts occur in these habitats (Coleman *et al.*, 2013).
418 However, rocky reef habitats in Northumberland may already be in a degraded state
419 due to decades of fishing (Scheffer *et al.*, 2001; Hiddink *et al.*, 2006; Nyström *et al.*,
420 2012; Sciberras *et al.*, 2013; Caveen *et al.*, 2014; Laugen *et al.*, 2014) and current
421 potting impacts may not be detectable. Habitats in intensively fished FaAICr areas
422 were dominated by short mixed faunal and algal crust, with few large, slower
423 growing, erect fauna and flora, suggesting harsh physical conditions (Hartnoll, 1998).
424 Whether these are due to long-term potting, environmental conditions, other impacts
425 or a combination of these is unclear. However, local knowledge of the area indicates
426 that abundant sediment, which can cause scour damage to species, and high energy
427 from waves and currents, are likely to contribute to determining species composition.
428 Lightly fished FaAICr habitats were also composed of hardy benthic species further

suggesting that environmental conditions may be driving assemblage structure in these areas. Overall, intensively fished FaAICr areas had a greater overall diversity and higher abundance of large erect species than lightly fished areas (Fig 5). This contrasts with predictions from trawl impacts (Collie *et al.*, 1997; Collie *et al.*, 2000b; Kaiser *et al.*, 2002; Lambert *et al.*, 2011) that intensively fished habitats would exhibit lower species diversity, lower abundances, smaller individuals and/or fewer erect and/or slow-growing species compared to lightly fished habitats. Here, there was no evidence that differences in community composition between intensively and lightly fished habitats were due to potting. However, as FaAICr has somewhat variable community composition even between areas separated by only hundreds of metres (Connor *et al.*, 2004), definitive conclusions on the 'health' of these biotopes and historic impacts of potting would require much larger samples over a greater spatial and temporal extent. Investigation of interactions between environmental variables and anthropogenic uses on resultant community compositions using predictive modelling (Lambert *et al.* (2011) would be useful in future.

Laminaria hyperborea (>50cm) provides habitat and substratum for many organisms, with high biodiversity recorded in kelp habitats (Birkett *et al.*, 1998). A loss of this species could result in loss or degradation of the associated community (Tyler-Walters, 2007), as well as changing the biotope classification. Here, potting did not affect kelp abundance (Fig 6), however, this was variable within transects and between treatments (Fig 6). This variability was expected due to the dynamic nature of these habitats, where competition for light, space and food result in a species rich but patchy distributions of flora and fauna (Jones *et al.*, 2000), but it may be masking small changes due to potting. However, even with high sample numbers, potting impacts on epibenthic assemblages clearly cannot be detected against the

background of natural variability (Sciberras *et al.*, 2013), suggesting that these are not pushing the ecosystem outside natural limits (De Backer *et al.*, 2010).

4.1. Experimental limitations

Impacts on the two most frequently occurring biotopes in Northumberland were investigated (Mercer, 2012). FaAICr habitats are actively targeted by fishers over others (Stephenson, unpublished data). These were deemed good experimental habitats as they are representative of local fishing practices, and present across many parts of Europe (MESH, 2008). Rarer habitats with smaller spatial extents, may be more sensitive to potting impacts, for example *Sabellaria spinulosa* reefs, which are sensitive to fishing and may be present in Northumberland in small areas (Holt *et al.*, 1995; Vorberg, 2000; Roberts *et al.*, 2010; Eno *et al.*, 2013).

This research sought to examine whether potting impacts were detectable in shallower habitats, which may be subject to higher wave action (Lewis *et al.*, 2009). There was no evidence that shallow habitats in Northumberland were more sensitive to potting impacts even though all habitats at all fishing pressures were subject to sustained periods of bad weather (swell >1.5m over >7 days) which could have caused high pot movement during fishing. In addition, pots left in shallow water during bad weather did not follow commercial practice as fishers move fleets of pots to deeper water areas in order to avoid damage to their fishing gear (Turner *et al.*, 2012; Coleman *et al.*, 2013), and as such habitats surveyed in this research are rarely fished in winter and spring. This study, with high levels of fishing effort maximising possible damage, demonstrated that there was no detectable impact of potting on reef benthos of two common habitats.

5. Conclusion

Experimental potting did not damage epibenthos of intensively and lightly fished rocky FaAlCr or Lhyp.Pk habitats over the short-term, in a range of weather conditions (swell: 0-2.5m; current: 0-1.3km.h⁻¹; wind: 0-53km.h⁻¹). Given, also the quick recovery of dominant and characteristic benthic species in both FaAlCr and Lhyp.Pk habitats and the intensity of fishing in the NIFCA district, it is also unlikely that these habitats are impacted in the longer-term. Results from this study suggest that fishing effort may be a sufficient proxy for likely potting impacts at other temperate locations with similar habitats. However, no conclusions are made with regards to the sustainability of the current exploitation levels of target species stock or any in-direct effects of potting on habitats and ecosystems. The present findings are relevant to similar habitats elsewhere in the UK and Europe, where similar fishing practices and biota occur.

Supplementary material

The following supplementary material is available at ICESJMS online: methods and results of SIMPER analysis used to identify the percentage similarity that benthos contributed to the measure of Bray-Curtis Similarity for treatments (E_B-E_A and C_B-C_A). Benthos modified for the simulation of potting impacts and results of the PERMANOVA model using modified community assemblages. Mean kelp abundance (m²) across three Lhyp.Pk sites at different experimental treatments.

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744 **Tables**

745 Table 1. PERMANOVA based on Bray-Curtis similarity (square root transformed) of
746 multivariate data. Highlighted grey cells show significant p-values. Results are shown
747 for habitat (Ha), Site nested in habitat (Si(Ha)), Fished (Fi) and Time (Ti).

Source of variation	df	SS	MS	Pseudo F	p
Ha	2	1.017E6	5.0848E5	44.063	0.005
Fi	1	2102.4	2102.4	1.6066	0.238
Ti	1	18497	18497	4.2674	0.033
Si(Ha)	6	69507	11584	15.725	0.001
HaxFi	2	4219.5	2109.7	1.6101	0.196
HaxTi	2	20228	10114	2.3279	0.045
FixTi	1	1117.5	1117.5	1.0384	0.37
Si(Ha)xFi	6	7876	1312.7	1.7819	0.09
Si(Ha)xTi	6	26157	4359.5	5.9179	0.001
HaxFixTi	2	1377	688.5	0.63921	0.673
Si(Ha)xFixTi	6	6471.2	1078.5	1.4641	0.06
Res	1306	9.621E5	736.67		

748
749

750 Table 2. PERMANOVA based on Bray-Curtis similarity of square root transformed
751 multivariate data collected in intensively and lightly fished FaAlCr and intensively
752 fished Lhyp.Pk areas (999 permutations per test). Highlighted grey cells show
753 significant p-values. Results are shown for treatments E_B, E_A, C_B and C_A.

Habitat	Treatment	Source of variation	df	SS	MS	Pseudo F	p	Estimates of components of variation
FaAlCr intensively fished	E _B - C _B	Site	2	19170	9585.1	17.061	0.001	11.3
		Site x E _B - C _B	2	2018.5	1009.3	1.797	0.072	3.5
		Residuals	206	1115730	561.8			23.7
	E _B - E _A	Site	2	13724	6862.1	14.121	0.001	9.2
		Site x E _B - E _A	2	6562.3	3281.2	6.752	0.001	8.6
		Residuals	221	107390	485.93			22.0
	C _B - C _A	Site	2	15200	7599.9	13.125	0.001	9.7
		Site x C _B - C _A	2	3837.7	1918.8	3.3139	0.005	6.0
		Residuals		125650	579.03			24.0
	E _A - C _A	Site	2	17899	8949.4	15.304	0.001	10.2
		Site x E _A - C _A	2	1812	906.01	1.5493	0.145	2.8

		Residuals		135670	584.79			24.1
FaAlCr lightly fished	E _B - C _B	Site	2	2832.1	1416	5.9586	0.001	3.8
		Site x E _B - C _B	2	585.88	292.94	1.2327	0.305	1.6
		Residuals	239	56797	237.65			15.4
	E _B - E _A	Site	2	5235.4	2617.7	11.272	0.001	5.4
		Site x E _B - E _A	2	2439.1	1219.5	5.2515	0.001	4.9
		Residuals	237	55038	232.23			15.2
	C _B - C _A	Site	2	4168.5	2084.3	9.6399	0.001	4.9
		Site x C _B - C _A	2	2005.2	1002.6	4.6371	0.001	4.5
		Residuals	230	49728	216.21			14.7
	E _A - C _A	Site	2	9065.2	4532.6	21.89	0.001	7.5
		Site x E _A - C _A	2	285.44	142.72	0.68926	0.7	1.3
		Residuals	228	47211	207.07			14.4
Lhyp.Pk intensively fished	E _B - C _B	Site	2	20 828	10 414	6.2	0.001	10.9
		Site x E _B - C _B	2	5 296	3 447	1.6	0.139	5.1
		Residuals	218	36 8260	1 689			41.1
	E _B - E _A	Site	2	10 875	5 437	3.2	0.002	7.6
		Site x E _B - E _A	2	10 220	5 109	3.0	0.007	10.3
		Residuals	195	32 7830	1 681			41.0
	C _B - C _A	Site	2	26 518	13 259	10.0	0.001	13.2
		Site x C _B - C _A	2	4 092	4 046	2.8	0.045	4.6
		Residuals	206	273 120	1 326			36.4
	E _A - C _A	Site	2	21 381	10 691	8.4	0.001	12.3
		Site x E _A - C _A	2	3 391	1 695	1.3	0.238	3.7

Table 3. Estimated maximum number of pots in the sea month⁻¹.km⁻² (Fig 1), the number of hauls month⁻¹ (estimated conservatively) and fishing pressure (pot hauls month⁻¹.km⁻²) in lightly and intensively fished areas in Northumberland compared to the experimental fishing undertaken.

	Maximum number of pots in the sea month ⁻¹ .km ⁻²	Number of hauls month ⁻¹	Fishing pressure (pot hauls month ⁻¹ .km ⁻²)
Lightly fished	139	30	4170
Intensively fished	265	30	7950
Experimental fishing	10,000	3	30,000

761
762 **Figures**

763

764 Fig 1. A) Location of Northumberland in Great Britain. B) Location of Berwickshire
765 and North Northumberland Coast European Marine Site and the Coquet to St Mary's
766 recommended MCZ within the NIFCA district. C) Depth below chart datum (m). D)
767 Olex percentage substrate hardness. E) Spatial distribution of mean summer fishing
768 effort in 2012–2013 (pots in the sea month⁻¹.km⁻²). F) Areas matching selection
769 criteria (red: <16m, >35% hardness, high fishing pressure; yellow: <16m, >35%
770 hardness, low fishing pressure). i) Final sites for experimental potting of Lhyp.Pk in
771 intensively fished areas, ii) FaAICr in intensively fished areas, iii) lightly fished areas.

772

773 Fig 2. Diagram of example study site. Experimental area (25x10m) and a control area
 774 (black squares, 5x10m) with haphazardly distributed sample areas (grey squares,
 775 0.625m² each), anchor weights (black anchors), transect tapes (dotted black line)
 776 and the experimental potting fleet: 20kg weights (black circle), rope (black lines,
 777 including 3m 'leg') and parlour pot (hashed black rectangle).

778

779 Fig 3. Photoquadrat with camera housing and strobes (A). Photoquadrats of typical
 780 species from: FaAlCr lightly fished sites (B); FaAlCr intensively fished sites (C); and
 781 Lhyp.Pk intensively fished sites (D).

782

783 Fig 4. Principal coordinate ordination (PCO) of square root transformed percentage
 784 cover grouped according to treatment (Experimental areas, closed symbols; control
 785 areas, open symbols) and site (site 1, triangle; site 2, square; site 3, circles) for
 786 FaAlCr with high fishing pressure (A), low fishing pressure (B) and Lhyp.Pk with high
 787 fishing pressure (C). Lines show the direction of change in assemblage structure
 788 before and after fishing (E_B-E_A , and C_B-C_A).

789

790 Fig 5. Square root transformed percentage cover of benthos from E_B (dark grey
 791 columns), C_B (white column), E_A (light grey) and C_A (white) experimental areas in
 792 intensively (A) and lightly fished (B) FaAlCr, and intensively fished Lhyp.Pk (C)
 793 habitats.