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1 **Decline in oyster populations in traditional fishing grounds; is habitat damage by static fishing**
2 **gear a contributory factor in ecosystem degradation?**

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11
12 **Running Page Head:** "Oyster decline in traditional fishing grounds"

13
14 **ABSTRACT**

15 The territorial waters of Qatar once supported dense assemblages of the pearl oyster *Pinctada radiata*.
16 The oysters settled on a patchy network of limestone platforms (*hairât*) and provided a suite of
17 ecosystem services to the surrounding marine environment. Commercially important fish species are
18 associated with *hairât* and as a result, industrial fishing with traps focused on these areas. This study
19 has shown that heavily-fished areas are presently in a state which can be considered non-favorable to
20 conservation while areas closed to fishing are recovering. It is probable that an increase in fishing
21 activity using traditional Gargoor traps and grapple retrieval are responsible for the current ecological
22 status of the *hairât*. The intensity in trap fishing appears to be having a detrimental effect on species
23 such as corals, sea grasses and oysters. The decline in the standing stock of oysters is dramatic with

24 an estimated reduction ratio of 580:1 between 2002 and 2016. As fishing damage appears to be a
25 significant contributor to these losses, measures such as spatial protection of productive shallow
26 offshore habitats and restriction on fishing effort are urgently required to address the decline. Strategic
27 oyster stock enhancement through the re-seeding of selected areas could boost the recovery of
28 damaged *hairât* as *P. radiata* ecosystem services return.

29 **Keywords** ecosystem providers; habitat associated fishery; habitat recovery; oyster population decline;
30 trap fishing

31 INTRODUCTION

32 Fishing techniques which make contact with the seafloor have the potential to produce damaging
33 impacts on the benthos (Hinz et al., 2009). On heavily fished grounds, habitat features can be reduced
34 or removed with seafloor topography and substrate composition changed. The use of trawls and
35 dredges can have an almost instantaneous effect (Kaiser et al., 2002; Rice, 2011; Shester and Micheli,
36 2011). Rice (2006) in a comprehensive review on the environmental impacts of bottom-tending gear
37 concluded that severe and in some instances irreversible damage can be caused to benthic habitat
38 complexity if fishing pressure is intense.

39 In the Middle East the Gulf State of Qatar was one of the first countries to recognise the ecological
40 impact that industrial scale trawling could have on the marine environment and as a result all trawling
41 within its territorial waters were banned in 1992 (Al-Abdulrazzak et al., 2015). The Qatar Fisheries
42 Department (QFD) introduced legislation whereby bottom fishing could only be undertaken using
43 artisanal passive traps known as Gargoor (De Yonge, 2006; Sheppard et al., 2010; Al-Abdulrazzak et
44 al., 2015). Gargoor are semi-circular domed creels which have been fished for centuries throughout the
45 region. They were traditionally constructed from a woven cane mesh and fished in a similar fashion to
46 that of the European lobster pot (Grandcourt et al., 2004). The Gargoor works on the bottle neck
47 principle whereby fish are enticed inside the baited trap through a large opened mesh channel which
48 tapers into the main capture chamber. The QFD promoted the use of Gargoor as similar passive
49 stationary gears like the Atlantic cod pot and European lobster pot were shown to be considerably less
50 destructive than trawl fishing (Jennings and Kaiser, 1998; Bradshaw et al., 2001; Pauly et al., 2002).

51 Gargoor vessels today no longer fish wooden creels but use a lighter more robust galvanised wire
52 version. Once on site > 200 traps will be deployed in strings of three to six each spaced approximately
53 one meter apart. In the past three sizes were fished to target specific species; a small 165cm trap for
54 crab, squid and cuttlefish, 205cm for Sweetlips (*Diagramma picta*) Goatfish (*Parupeneus margaritatus*)
55 and Rabbitfish (*Siganus canaliculatus*) and large 225cm traps for Grouper (*Epinephelus coioides*)

56 White-cheek shark (*Carcharhinus dussumieri*) and Cobia (*Rachycentron canadum*). However, the
57 225cm trap is currently the most commonly used as it maximises capture potential while reducing
58 under-sized fish and by-catch (Grandcourt, 2012).

59 The Qatari Gargoor fleets fish year-round although a QFD limit on fishing days is enforced for larger
60 vessels. Fishing generally takes place on historically productive grounds with site location handed down
61 through generations (De Yonge, 2006; Al-Abdulrazzak et al., 2015). Theft from unattended traps is
62 common and the majority of vessels do not mark trap deployments in an attempt to avoid interference.
63 If accurate bearings have not been taken, finding shot trap lines can become a challenging task and
64 typically a grapple search is employed over the deployment site. A heavy 30 Kg multi-hooked grapple
65 known as a “*Manshal*” is used to snag the Gargoor lines which are then dragged on-board by hand.
66 Owing to its weight and design the *Manshal* can have considerable impact on the benthic environment
67 during deployment and recovery. The QFD estimate a *Manshal* is commonly shot and dragged three
68 times over a distance of approximate 150 m before a line of Gagoor is located (Personal
69 communication Al-Mohammadi 2016).

70 Fishing activities which use artisanal gears are generally considered as low impact when compared to
71 more industrial scale techniques. However, they can still cause a substantial amount of habitat
72 destruction if their use is not policed (Pauly et al., 2002; Althaus et al., 2009; Giraldes et al., 2015).
73 Within complex biogenic habitats such as coral reefs, oyster beds and sea grass meadows the impact
74 of trap fishing can be rapid if controls are absent (Hirst et al., 2012; Strain et al., 2012).

75 In Qatari waters oysters and corals have formed extensive biogenic structured benthic habitats which
76 can accommodate dense assemblages of molluscs, polychaetes, crustaceans, and other habitually
77 exclusive invertebrates (Lenihan et al., 2001; Rothschild et al., 1994; Wells, 1961). When these
78 ecosystems are in a good state of ecological health they will augment tertiary productivity, as juvenile
79 fish and mobile crustaceans will utilize the assemblages for refuge and foraging (Coen and

80 Luckenbach, 2000; Harding and Mann, 2003; Soniat et al., 2004; Luckenbach et al., 2005; Tolley and
81 Volety 2005; Rodney and Paynter 2006).

82 In the Arabian / Persian Gulf the historical instigator of the regions rich biogenic reef structures was the
83 bivalve *Pinctada radiata* or Arabian Pearl Oyster (Mohammed and Yassien, 2003; Smyth et al., 2016a).
84 It is considered highly fecund and Al-Ansari et al., (1994) estimated an average of between 0.95 and
85 1.7 million eggs could be produced during a single spawning event, of which there could be several
86 throughout a year. Larval settlement is gregarious in nature and influenced by adult conspecifics;
87 subsequently the highest attachment densities are recorded on live and dead shell of its own species
88 (Gosling, 2003). It was these *P. radiata* life cycle traits which were responsible for the large offshore
89 oyster beds along the western coast of the Arabian Gulf. The oyster beds were known as *hairât* and are
90 almost exclusively located on subtidal limestone pinnacles. Documented references to the scale of
91 these *hairât* can be found as far back as AD 32 when the Roman Scholar and naturalist Pliny describes
92 the richness of the Gulf in terms of "its beds of Pearls and bounties of fishes which stretch from Sharjan
93 (in the United Arab Emirates) to Qatif (in Saudi Arabia)", a distance of > 1050 Km (Lorimer, 1915;
94 Carter, 2005).

95 High density oyster assemblages in the Middle East are now rare and therefore estimating their
96 associated species diversity is difficult. However, similar biogenic structures can be highly effective in
97 augmenting and enhancing biodiversity. For example, a comparable surface topography and 3-D matrix
98 complexity can be found when examining the European Horse mussel (*Modiolus modiolus*) reefs; the
99 associated diversity of *M. modiolus* assemblages in the United Kingdom is high, with > 900 species
100 recorded at the most pristine sites (Sanderson et al., 2008). Assemblages supported by habitat-forming
101 byssal-attached species like *M. modiolus*, *Mytilus edulis* and *P. radiata* are extremely susceptible to
102 physical disturbance and can be dislodged during the retrieval and deployment of static and passive
103 fishing gears (Lokrantz et al., 2009; Strain et al., 2012). Similarly, deployment of static gear can
104 entangle soft and hard corals damage seagrass meadows (Coll et al., 2012).

105 Cury et al. (2003) suggested that the majority of these vulnerable complex habitats should be
106 considered as ecologically distinct landscape features or ecotopes in their own right in recognition of
107 their valuable associated ecosystem services. The destruction of these ecotopes can have a serious
108 impact on the biological functionality of the wider ecosystem. It is therefore essential that habitat
109 forming species should be considered a conservation priority when management plans are being
110 designed (Lokrantz et al., 2009). A responsibility also rests with fisheries managers to recognise that
111 the species creating these ecologically distinct features warrant protection equivalent to that afforded to
112 the associated fishery which they support (Hall, 2002; Rice, 2006; Parker et al., 2009).

113 The large oyster bed ecotopes of the western Gulf supported huge pearl and demersal fisheries for
114 centuries. The pearl fishery is now considered non-commercially viable with many of the *hairāt* being
115 reported as barren as far back as the mid-1930s (Burdett, 1995; Carter, 2005). The exploitation during
116 the 1700-1800s which resulted in the collapse of the Gulf beds was not confined to the region but was
117 mirrored in other global oyster fisheries; in Europe *Ostrea edulis*, in Asia *Crassostrea gigas* and in the
118 USA *Crassostrea virginica* were all fished beyond a state favourable to conservation (Botsford et al.,
119 1997; Jackson et al., 2001; Beck et al., 2011).

120 The loss of ecosystem services (water column filtration, sediment stabilisation, substrate provision and
121 benthic pelagic coupling) provided by a functional oyster *hairāt* can have drastic and rapid ecological
122 effects, in regards to; water quality, benthic biodiversity and fish habitat which lead to decreases in reef-
123 associated demersal fish (Bouma et al., 2009). Intrinsic fish-habitat associations between the Qatari
124 *hairāt* and a number of commercially important demersal species were recently confirmed in
125 hydroacoustic surveys of sites which were once renowned as prolific during the pearl fishing epoch
126 (Egerton et al., 2018) and wider habitat surveys (Walton et al 2017). Planning future management
127 strategies for fish stocks should recognize the essential habitual niche of these fish, as many spend
128 their entire life cycle associated with specific reefs (Egerton et al., 2018) while the Gargoor fleets
129 consistently target traditional *hairāt* sites (Stamatopoulos and Abdallah, 2016).

130 Although the pearl sites in Qatar have been considered as barren for years (First author pers.
131 observation) Smyth et al., (2016b) discovered fragmented assemblages of mature *P. radiata* during
132 biotope surveys, however no obvious signs of large-scale recruitment were detected. This raises the
133 question as to why recruitment is failing despite the high fecundity of *P. radiata* and its non-
134 discriminatory settlement. We set out to test the hypothesis that disturbance by the combined effects of
135 retrieving static fishing gear using grapples and the seabed drag of strings of Gargoor could interfere
136 with recruitment by causing newly-settled oysters to become detached from the substratum, as well as
137 reducing the available areas for settlement by damaging habitat-forming biogenic reefs (Shester and
138 Micheli, 2011). We revisited sites previously surveyed by Al-Madfa et al., (1998), Al-Khayat and Al-Ansi,
139 (2008) and Smyth et al., (2016b), to compare changes in the ecological status and population density
140 of *P. radiata* over the last two decades in both fished and protected areas.

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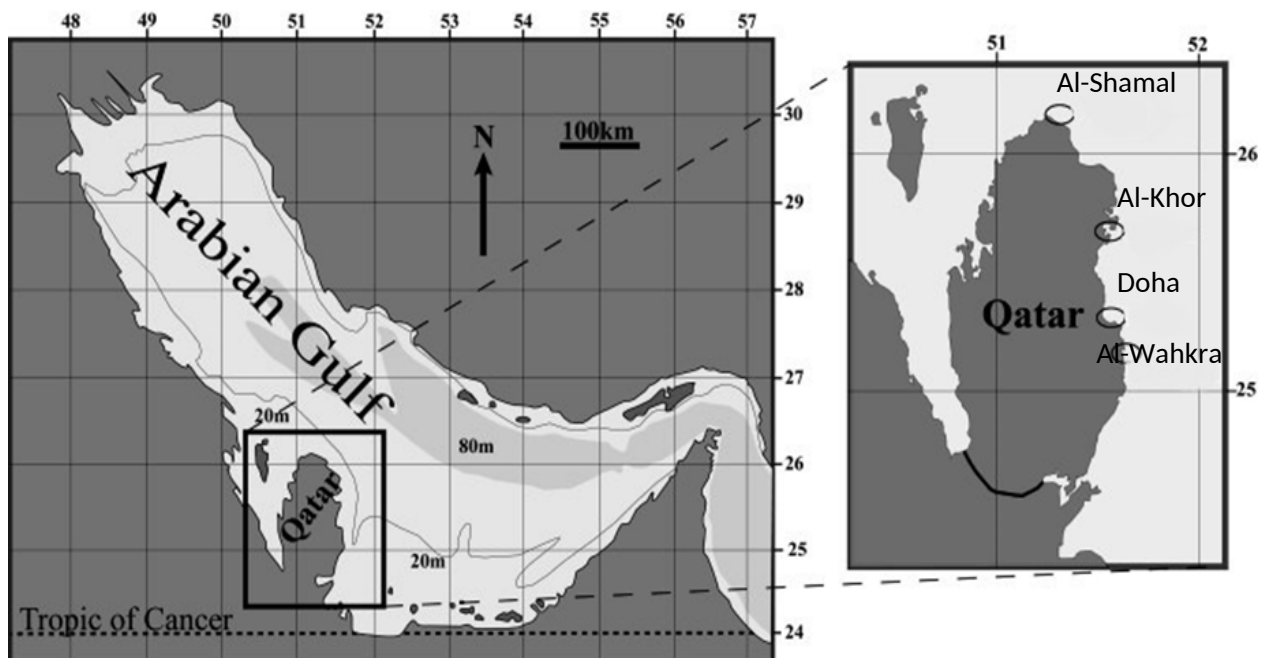
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150 **Fig 1.** Qatar and its main fishing ports within the Arabian / Persian Gulf

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152

153 MATERIALS AND METHODS

154 Study area

155 Qatar is situated on the west coast of the Arabian Gulf on the peninsula bordering Saudi Arabia and the
156 United Arab Emirates at 25°30'N and 51°15'E (Fig 1). It has a total coastline of 563 km with a
157 hydrodynamic regime typified by a south easterly surface current. Sea water surface temperatures
158 range from 18.7 to 35.0 °C. Where depth is < 15 m, temperatures remain relatively constant between 15
159 to 20 °C throughout the year. Salinities above the thermocline fluctuate between 35.5 to 44.5 ppt
160 depending on season (Kampi and Sadrinassab, 2006).

161 Survey site selection was based on information provided by the Qatar Fisheries Department (QFD),
162 findings from the Qatar University biotope mapping programme (Smyth et al., 2016b) and availability of
163 previous survey data from 1992 (Al-Madfa, 1998) and 2002 (Al-Khayat and Al-Ansi, 2008). All the
164 selected sites were once considered as prolific oyster beds or *hairāt*, as confirmed from archaic maps
165 held by the QFD showing pearl fishing activities between 1830-35. Historical descriptions of *hairāt* refer
166 to dense assemblages of oysters settled on raised limestone platforms found at depths of between 8-
167 25m typically surrounded by a deep sand-mud plateau (Carter, 2005; Walton et al., 2017).

168 Fishing intensity also influenced survey site location and was provided by the QFD in the form of days
169 at sea and landings records for the Gargoor fleets in proximity to an associated port (Fig 1). The
170 majority of sites were located within designated open fishing zones (Fig 2). Only one northerly site was
171 accessible during the survey, as *hairāt* in this region are located either on or straddling a contentious
172 maritime border with Bahrain (Fig 2). The areas with *hairāt* closed to fishing were under the jurisdiction
173 of the Qatar Coastguard. Coordinates identifying these sites were plotted and presented in Fig 2 and
174 are referred to as 'protected areas / potential fishing grounds'. This terminology has been used to
175 emphasise that although marked as closed on nautical charts the sites could be potentially open for

176 fishing in the future. Also the sites are not strictly policed and illegal fishing activity cannot be
 177 completely ruled out.

178 The location and areal extent of *hairāt* ($4740.15 \times 10^6 \text{ m}^2$) was predicted from depth-habitat
 179 relationships (Walton et al., 2017; Egerton et al., 2018) and calculated using Arc GIS[®] software and
 180 bathymetry data from the NOAA National Center for Environmental Information. All mapping was to a
 181 resolution of one minute. The total available fishing area on *hairāt* was calculated using results as per
 182 Arc GIS[®] pixel allocations in m^2 . Coverage within restricted sites was subtracted from the total
 183 accessible area to provide the amount of available open fishing grounds within each zone (Table 1)
 184 presented in Fig 2.

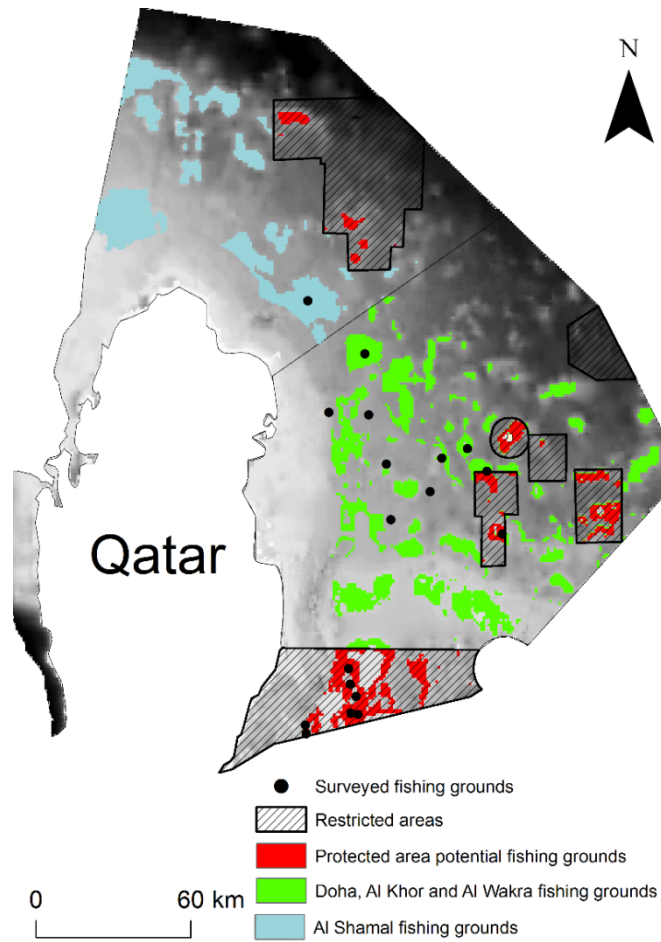
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186 **Table 1.** Dimensions of survey regions (surface area).

Region	Total Hairāt area m^{-2} ($\times 10^6$)	Closed Fishing area m^{-2} ($\times 10^6$)	Total Fishing area m^{-2} ($\times 10^6$)
North			
Al-Shamal	1719.109	0.109	1719.00
East			
Al-Khor	3020.486	0.332	3020.153
Doha			
Al-Wakra			
South			
Protected zone	0.556.000	0.556	0
Total	4740.15	0.9977	4,739.15

187

188



189

190 Fig 2. Historically renowned *hairāt* oyster sites at 8-25m depth as per Carter (2005), showing survey
 191 sites and restricted and protected area potential fishing grounds. Grey scale background represents
 192 bathymetry.

193

194 **Methodology**

195 A team of scientific divers was used to collect quantitative and qualitative data from the 18 sites. Divers
 196 carried out video and digital still surveys along 4x100 m transect lines separated by a 10 m gap as
 197 outlined in Smyth et al., (2009) and Giraldes et al., (2015). On commencement and completion of the
 198 transect travel a diver deployed surface marker buoys from which longitudinal and latitudinal co-
 199 ordinates were recorded using a Garmin® GPS plotter. Using underwater cameras divers recorded
 200 substratum type, environmental damage, dominant key species and *P. radiata* density within a 0.25 m²

201 frame fixed quadrat randomly positioned x 25 along the transect length. A third diver recorded substrate
202 type *in-situ* by touch, noting consistency of sedimentary mix and allocating a substrate biotope code as
203 per descriptions based on Joint Nature Conservation Committee standards (Connor et al., 2004) and
204 listed in Smyth et al., (2016b).

205

206 **Seafloor imagery**

207 Quadrat imagery was calibrated to the known distance at which the photos were taken (1m) based on
208 methodologies from Neumann and Kroncke, 2011 and Smyth et al., 2016b. Still images were analysed
209 using Coral Point Count[®] software with a Microsoft Excel[®] extension (CPCe[®]). The CPCe[®] software
210 randomly overlaid 50 points onto each image and identification labels were assigned by a benthic
211 taxonomist. Identification encompassed indicators of benthic disturbance such as; seabed gouges,
212 substratum scraps / trails, key dominant epibenthic species associated with *hairāt* as listed previously in
213 Al-Khayat and Al-Ansi (2008). This methodology was adopted from Carleton and Done (1995) and
214 Bento et al., (2017) as it was particularly comprehensive with a mandatory 5,000 individual
215 observations per site and total of 90,000 for the complete survey.

216 The image overlay points acted as non-biased observation markers from which quantitative data of
217 disturbance and fishing impact could be categorised as:

- 218 • “RD” recent damage; evidence of scraped or gouged substratum, broken non-bleached coral,
219 smashed fresh shell and lost gear.
- 220 • “OD” old damage; encrusted broken shell, bleached broken coral and encrusted lost gear.
- 221 • “SR” successional recovery; identified by newly established corals, juvenile oysters, algae and
222 sponges.

- 223 • “ND” no damage; mature oysters (2-5yr), well established corals, large sponges, dense algae
224 and rich epibiotic cover.

225 CPC[®] quantified quadrat disturbance observations into the above categories, with epibionts identified to
226 class level as per Kohler and Gill (2006). The total counts of specific observations per site were
227 averaged using and presented in a proportionally fractioned chart. The fractioned representation was
228 plotted in relation to site and mapped using ArcGIS[®] 9.3.

229 Univariate and multivariate analysis was used to investigate differences in the ecological status
230 between survey sites. Initial analysis of the sites and the four fractionally weighted habitat
231 categorisations was undertaken using repeated measures ANOVA. An overall comparison of the
232 complete CPC[®] habitat observational data within closed or open fishing zones were analysed by
233 PERMANOVA in PAST[®] 3.14.

234 In order to investigate the relationship between site, fishing activity and individual components further
235 analysis was undertaken using multivariate techniques in PRIMER[®] 6 and Past 3.14[®]. Firstly, a
236 Multidimensional Scaling (MDS) programme subjected data to 2-D ordination whereby Bray-Curtis
237 coefficients between replicates at each site were used to produce a plot showing any possible
238 relationships. The relationship between data was presented as a “Stress” value in the top right hand
239 corner of the plot with < 0.05 considered an excellent expression, 0.1 regarded as good and between
240 0.1 and 0.2 useful (Clarke and Warwick, 1994). The data for each site was then subjected to ANOSIM
241 and SIMPER tests.

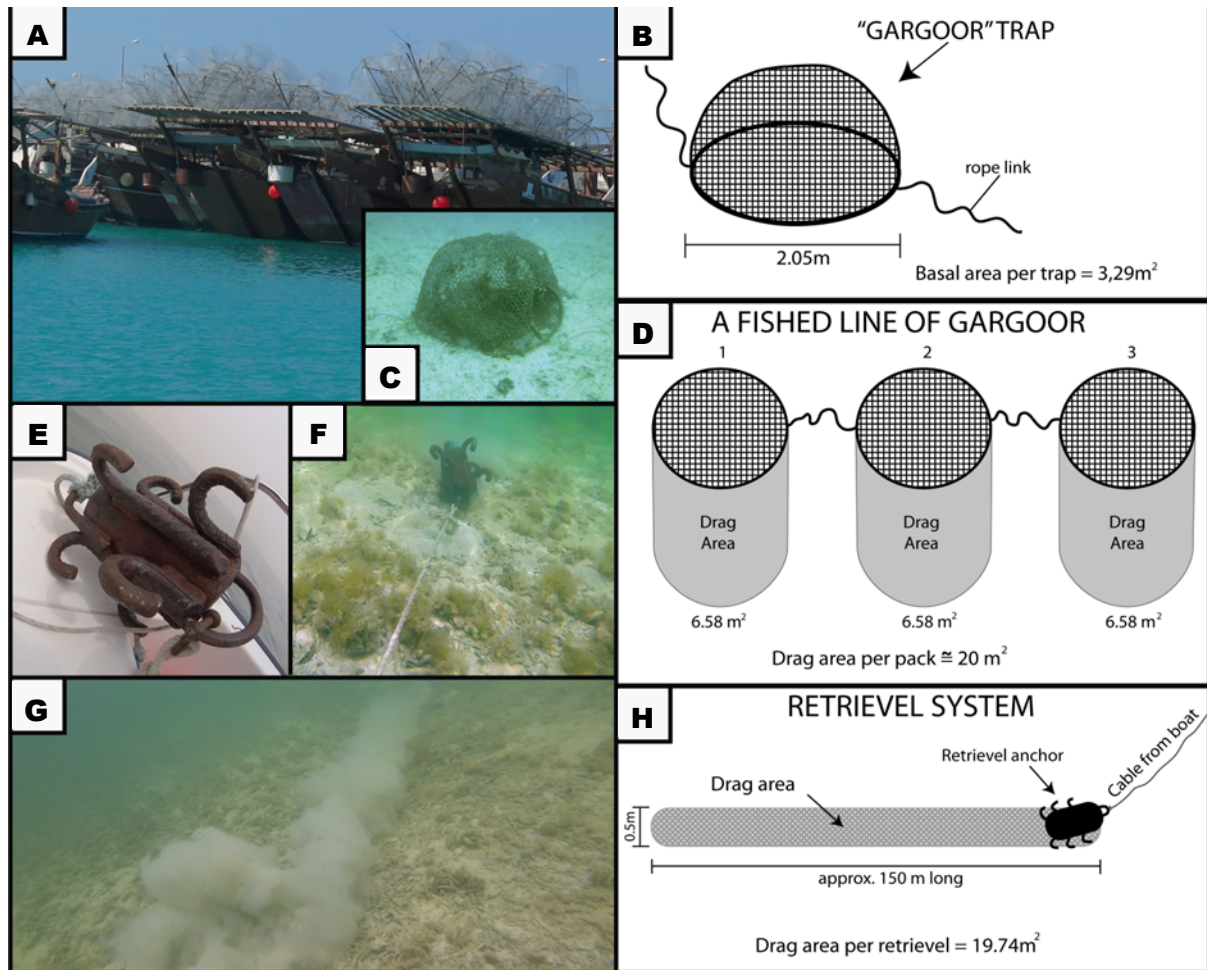
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243 **Analysis of fishing intensity**

244 Fishing activity in the region was considered intense (Fig 3 A-G). *Hairāt* were visited on a regular basis
245 by > 300 vessels > 15 m long each fishing 400 - 600 (225 cm) Gargoor. The *hairāt* closer to shore had
246 the additional pressure of 350 smaller vessels < 15m which fished 50 to 150 traps (Al-Abdulrazzak et

247 al., 2015; Pauly and Zeller, 2016). The QFD provided statistical data in relation to the number of
 248 licensed vessels per port, annual active fishing days and number of licensed Gargoor per vessel for
 249 2014 and 2015 (Fig 3 G). This information was used to produce a fishing intensity score based on a
 250 similar formula used by MacDonald et al., (1996).

251



252

253

254 Fig 3. A) Dhow loaded with Gargoor; B-D) Gargoor *in-situ*, trap and line of drag area; E-G) *Manshal* /
 255 multi-grapple line retrieval; H) Multi-grapple drag area. (Drag area estimated from information provided
 256 by QFD officials in relation to length of trap line attachment rope and average retrieval rope length).

257

258

259 Fishin intensity was estimated from the area of annual trap coverage inclusive of seabed drag from
260 both traps and *Manshall* (Fig 3 B - H) relative to the total area of *hairāt* open to fishing.

261 (*A*) annual area fished per vessel

262 (*N_v*) number of traps fished per vessel

263 (*D*) trap diameter (Fig 3 B)

264 (*L*) length of Gargoor drag during retrieval (Fig 3 D)

265 (*G*) area of grapple retrieval drag (Fig 3 D&H)

266 (*N_d*) number of trap collection days per annum.

267 Area of fished m²: (*N_d* x *N_v* x *G*)/3 + (*N_v* x *N_d* x *L* x *D*) (*i*)

268 Fleet intensity was then calculated for each port (*F*) by dividing the area fished per vessel (*A*) x the
269 number of vessels per fleet (*V*) by the total available fishing area per sector; *j* (*TA_j*) (Table 1).

270
$$F = \frac{AxV}{TA_j} \quad (ii)$$

271 Fleet intensity was gauged in on the number of times the area of *hairāt* was covered by the drag area of
272 Gargoor fished per annum. An impact score of 1 was considered to reflect a high intensity of activity. As
273 the area of Gargoor fished would have equated covering the total *hairāt* in one year.

274

275 ***Pinctada radiata* standing stock estimates**

276 In order to determine the standing stock of oysters (Fig 4 A-F) since the first documented surveys in
277 1992 (Al- Madfa et al., 1998) and 2002 (Al-Khayat and Al-Ansi, 2008), a comparison of oyster density
278 per m² within the replicated surveyed areas was undertaken using *P. radiata* observations gathered

279 from the CPCe[®] data for each of the 100 m transects. Estimates were made of the current standing
 280 stocks of oysters within the fished and protected offshore zones using the following mathematical
 281 model adapted from Gunderson (1993):

$$282 \quad P = \sum_{i=1}^h (R_i \cdot a) C_i$$

283 (iii)

284 Where; P = Total population resident in full survey area.

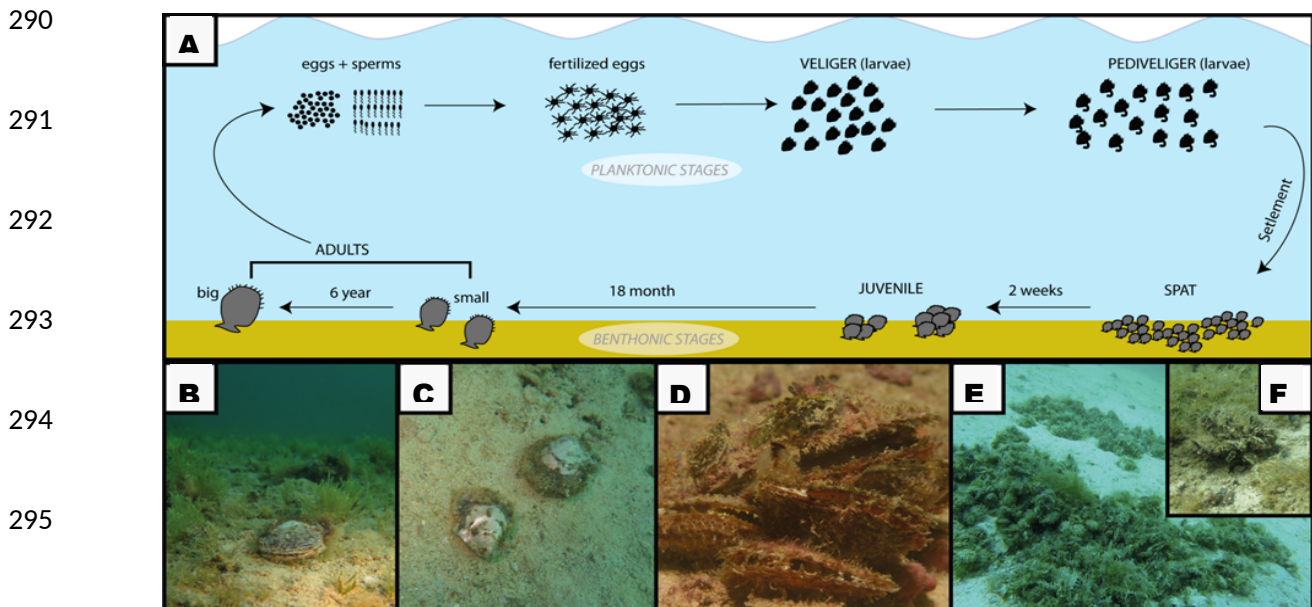
285 R_i = Area of region i in m^2 .

286 a = Area sampled within a single sampling unit.

287 C_i = Mean no. of oysters observed per sample unit in region i based on

288 n = Samples.

289 h = Number of regions composing the survey.



297 Fig 4. A) Life cycle of *Pinctada radiata*; B-D) Adult *P. radiata*; E-F) Juvenile *P. radiata*.

298 The total population resident in the entire survey area, 'P', was determined using an estimate of survey
299 region in m². The surface area, 'Ri', for the regions was estimated using scaled images of the *hairāt*
300 from Arc GIS[®] with a pixel value (924.33 m²) per calculated area (Table 1). Value 'a' is a constant which
301 refers to the area sampled within a single sampling unit (9 m). 'Ci' refers to the mean number of oysters
302 observed per sampling unit in region 'i' based on 'n' samples.

303

304 **RESULTS**

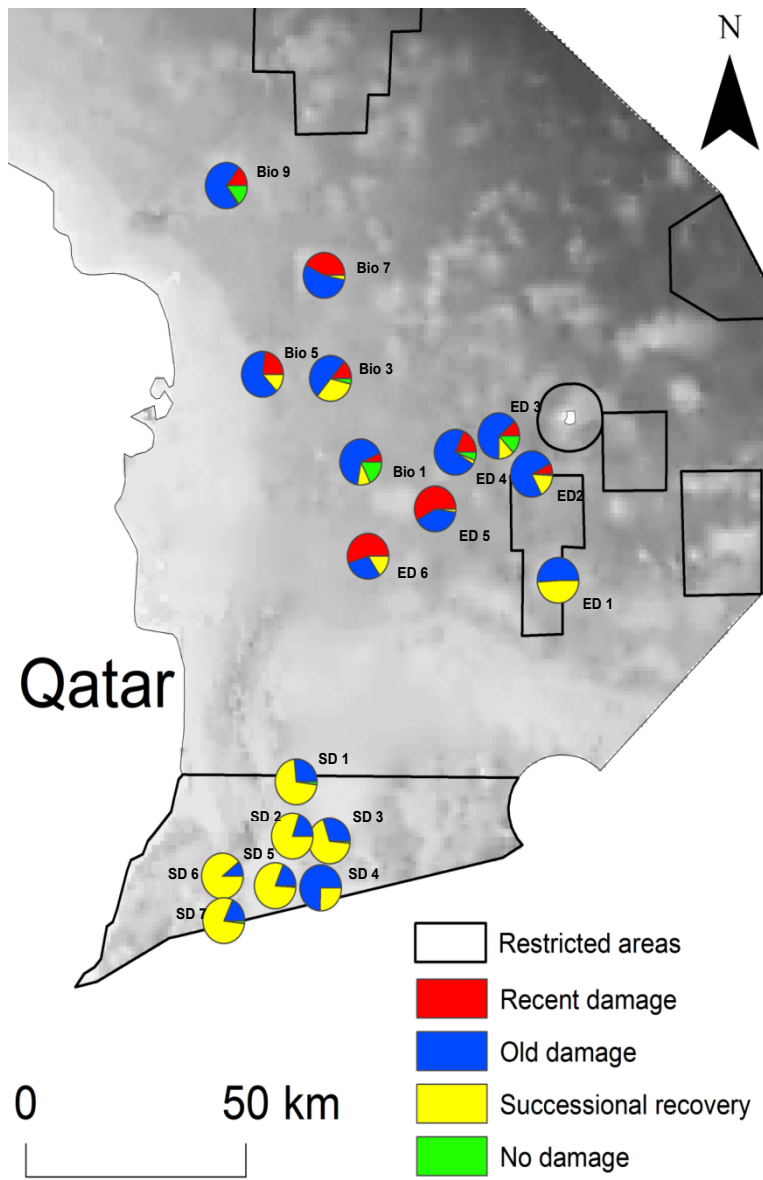
305 **Habitat Imagery analysis**

306 Analysis revealed eight sites which could be considered in a state of conservational recovery whereby
307 they had *in-situ* observational incidents which included epibiota vulnerable to mechanical disturbance;
308 sponges, ascidians, green, brown, red algae, *juv. Pinctada radiata* and coral buds. All of these sites
309 (SD 1-7) were within closed fishing areas, with seven located in the southern marine closed zone and
310 one (ED1) within the boundaries of a protected oil installation (Fig 5). The remaining 10 sites recorded
311 varying categories of disturbance which indicated a poor conservational state (Fig 5) such as; mixed
312 sand substrate, clean broken shell, broken coral, abandoned gear debris and substrate gouges. At
313 eight sites (Bio 1-9 & ED2-4) > 50% of the habitat observations could be considered as "Old Damage",
314 at (Bio 7) observations displayed > 40% "Recent Damage". An additional two sites (ED5 & ED6)
315 displayed > 50% "Recent Damage" with the majority of seafloor void of epibenthic species (Fig 5). A
316 repeated measure ANOVA of the habitat categorisations and sites revealed a significant difference with
317 ($F= 2.847, P < 0.001$). A Tukey's Pairwise Post-Hoc identified significant differences ($p < 0.005$)
318 between ED5 and Bio 5. Bio 5 was also significantly different ($p < 0.05$) from SD6.

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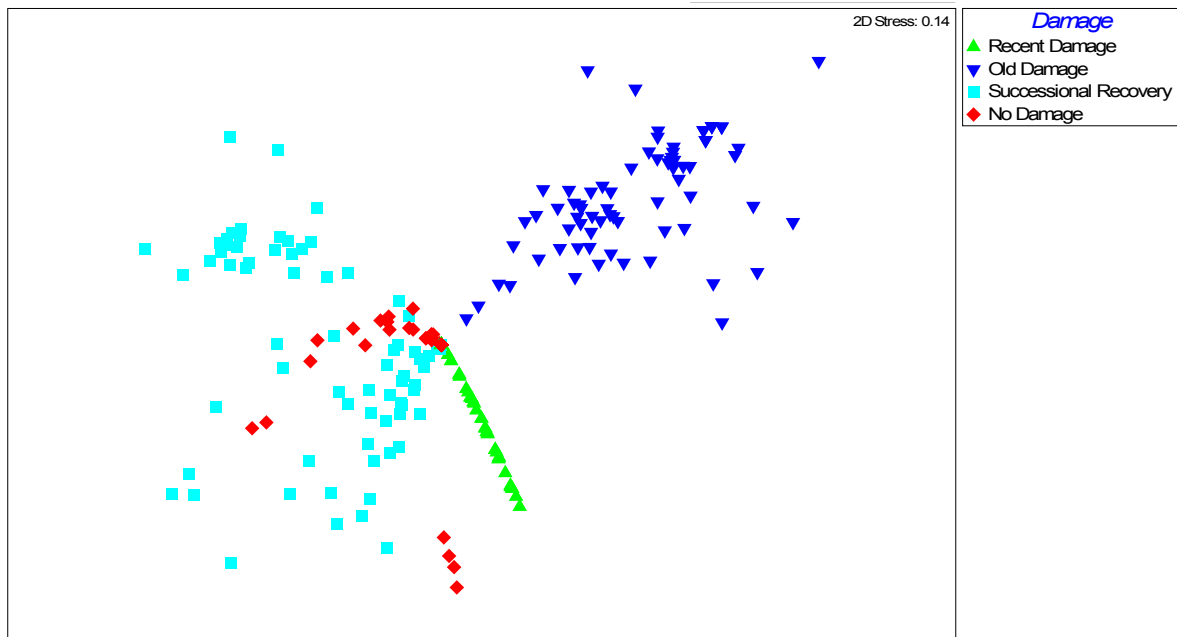
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Fig 5. Fractionally proportioned habitat categories per surveyed site.

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338
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Primer6[®] was used to subject the categorised data to multivariate analysis this resulted in the plotting of a 2-dimensional MDS chart of epibiotic species and habitat categorisation per replicate per site (Fig 6). The comparisons produced a stress value of 0.14, which was considered a valuable assessment. The MDS plot revealed clear groupings within observed incidents in relation to the “OD” and also among the

340 “RS” data. The labels and site groupings associated with “SR” were predominantly separated but some
341 overlap was observed within the “ND” categorised sites.



342
343 Fig 6. MDS of differences between CPCe observational categorisations indicative of epibionts.

344 SIMPER analysis revealed that the highest average dissimilarity (*av. dis.*) 28.72 was between “OD”
345 sites and those showing signs of “SR”. The categories of “RD” and “OD” displayed the second highest
346 average similarity (23.04) while “RD” and “ND” revealed the lowest *av. dis* (6.40). Similarity scores
347 displayed the most ecologically different categories in “RD” observations as being small fragments of
348 live corals and oysters. These were observed within fissures and depressions in the *hairât* topography
349 (Table 2).

350

351

352

353 Table 2. SIMPER analysis, *av. dis.* of "Habitat Category". Data were standardised and fourth root
 354 transformed comparison based on epibota and substrate observations for each site, listed below in rank
 355 importance. (OD; Old Damage, SR; Successional Recovery; RD; Recent Damage, ND; New Damage).

<i>CPCe</i> [®]	"OD"	"SR"	<i>CPCe</i> [®]	"RD"	"ND"
<i>Incident label</i>	<i>Av.Abund</i>	<i>Av.Abund</i>	<i>Observation label</i>	<i>Av.Abund</i>	<i>Av.Abund</i>
<i>Encrusted</i> (Broken shell, Coral fragment and fishing gear)	1.41	0.30	<i>Substrate Disturbance</i> (scars, gouge and lost gear)	0.97	0.30
<i>Rubble</i> (Shell and coral debris)	1.31	0.30	<i>Mussidae</i> (Live Coral)	0.30	0.45
<i>Shell</i> (Shell fragments and empty valves)	1.19	0.32	<i>P. radiata</i> (Live Oyster)	0.30	0.41
<i>Dead Coral</i>	1.17	0.30	<i>Montastrea</i> (Live Coral)	0.30	0.40

356

357 The categorised groupings were then examined as percentage coverage of individual constituents per
 358 site. Individual constituent components were treated as separate entities within the 100m transect
 359 replicates. This defragmentation of the data sets was undertaken to produce a clearer interpretation of
 360 site related fishing activity. The site related transect data was investigated using ANOSIM and SIMPER.
 361 The ANOSIM revealed a significant difference for individual constituent contributions within all sites with
 362 $p \leq 0.005$. *R* – Values of > 0.65 were displayed throughout suggesting similarities between the % of
 363 constituent components within all sites with the exception of Site Bio 5, which displayed an even
 364 distribution of factors and an *R*- value 0.04 (Tables 3 a-c).

365 SIMPER analysis revealed that five sites within the fished zone had no living epibiota recorded within
 366 the four highest ranking contributing factors, instead all contributing constituents were recognised signs
 367 of fishing activity; "Broken Coral, Broken Shell, Seabed Gouge / Scrape and Dragged Rubble".

368 Site ED 1 which straddles a fished and protected zone also had no live epibiota listed (Table 3 a).

369 The majority of observations were made in Trans 2 the first central transect that lay on the border of the

370 protected zone suggesting most fishing activity was taking on the perimeter of protection. One totally

371 protected site (SD 3) also had no live epibiota recorded and indeed the data confirmed that fishing

372 incursions must have been taking place as Broken Coral, Broken Shell and Seabed Gouge / Scrape

373 were all recorded as the highest ranking average abundances within factors (Table 3 c). A SIMPER

374 average dissimilarity value of < 27.00% was revealed for the majority of sites showing a relatively

375 constant percentage representation of individual constituent components within transects at each site.

376 Tables 3 (a-c)

3 a) Site Bio. 1 Fished Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.73			
Species / Substrate	Av. Dis	Contrib. %	Cumulative %	Trans 1 Mean Ab	Trans 2 Mean Ab	Trans 3 Mean Ab	Trans 4 Mean Ab
Encrusted Debris	3.81	17	17	0	2.24	0	0.28
<i>Pinctada radiata</i>	3.68	16.42	33.42	0.5	0.31	2.04	0
Broken & Bleached Coral	3.01	13.42	46.84	0.1	1.83	0	0.3
Broken Shell	2.65	11.82	58.66	0	1.65	0.29	0
SIMPER Average Dissimilarity 22.41 %							

Site Bio. 3 Fished Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.97			
Broken Coral	2.86	11.89	11.89	0	1.71	0.5	0
Dragged Rubble	2.84	11.83	23.72	0	1.7	0	0.3
Encrusted Debris	2.22	9.25	32.96	0	1.4	0.32	0.27
Broken Shell	2.08	8.66	41.62	0.35	1.33	0	0.29
SIMPER Average Dissimilarity 24.02 %							

Site Bio. 5 Fished Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.04			
Broken Shell	1.44	19.55	19.55	0	0.87	0	0
Encrusted Debris	1.43	19.46	39	0	0.85	0	0
Algae	1.37	18.54	57.54	0.2	0.24	0.83	0.27
Healthy Coral	0.87	11.6	69.14	0	0	0.63	0
SIMPER Average Dissimilarity 7.39 %							

Site Bio. 7 Fished Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.83			
Recent Damage – Gouge, Scrape	4.01	25.56	25.56	1.96	0.35	0	0.28
Broken Shell	3.02	19.22	44.77	0	1.67	0	0.37
Encrusted Debris	2.82	17.97	62.75	0	1.57	0	0
Dragged Rubble	2.0	12.4	75.13	0.3	1.18	0	0

SIMPER Average Dissimilarity 15.07%							
Site Bio. 9 Fished Zone	ANOSIM p-value 0.0001			ANOSIM R-value 1			
Broken Shell	4.19	18.39	18.39	0	2.35	0.6	0.22
Recent Damage – Gouge, Scrape	3.67	16.12	34.5	1.92	0	0.28	0
Recent Broken Coral	3.44	15.08	49.58	0.28	1.98	0	0
Coral - Mussidae (live)	3.32	14.64	64.23	0	0.29	0	1.83
SIMPER Average Dissimilarity 22.83%							
Site ED. 1 Fished / Protected Zone	ANOSIM p-value 0.0001			ANOSIM R-value 0.82			
Encrusted Debris	2.63	20.53	20.53	0	1.43	0	0
Broken Shell	1.77	13.82	34.36	0	1.06	0	0.34
Dragged Rubble	1.74	13.56	47.92	0	1.05	0	0.28
Recent Broken Coral	1.72	13.39	61.31	0.2	1.04	0	0
SIMPER Average Dissimilarity 12.81%							

377

378

3 b) Species / Substrate	Av. Dis	Contrib. %	Cumulative %	Trans 1 Ab Mean	Trans 2 Ab Mean	Trans 3 Ab Mean	Trans 4 Ab Mean
Site ED. 2 Fished /Protected	ANOSIM p-value 0.0001			ANOSIM R-value 0.67			
Encrusted Debris	2.64	16.5	16.5	0.3	1.48	0	0.2
Recent Broken Coral	2.01	12.56	29.06	0	1.2	0	0.1
Gorgonian <i>Ellisella sp.</i>	2	12.52	41.58	0	0	1.15	0
Recent Damage-Gouge, Scrape	1.88	11.82	53.31	1.06	0.43	0	0
SIMPER Average Dissimilarity 15.98 %							

Site ED. 3 Fished Zone	ANOSIM p-value 0.0001			ANOSIM R-value 0.9			
Encrusted Debris	3.42	21	21	0	1.86	0.32	0.21
Recent Damage – Gouge, Scrape	3.01	18.04	39.04	1.51	0.38	0	0
Recent Broken Coral	2.73	16.84	55.88	0.62	1.55	0	0
Broken Shell	1.7	10.34	66.22	0.32	0.35	0.22	0.3
SIMPER Average Dissimilarity 16.26 %							

Site ED. 4 Fished Zone	ANOSIM p-value 0.0001			ANOSIM R-value 0.67			
Recent Damage – Gouge, Scrape	4.19	22.81	22.81	2.13	0.34	0	0
Dragged Rubble	3.75	20.46	43.27	0.52	2.08	0	0.84
Encrusted Debris	3.26	17.7	60.97	0	1.86	0	0
Recent Broken Coral	1.81	9.9	70.86	0.4	1.19	0	0.3
SIMPER Average Dissimilarity 18.32 %							

Site ED. 5 Fished Zone	ANOSIM p-value 0.0001			ANOSIM R-value 0.98			
Recent Damage – Gouge, Scrape	4.99	18.05	18.05	2.63	0	0	0
Sponge	2.95	10.64	28.69	0	0	1.8	0
Dragged Rubble	2.85	10.33	39.01	0.22	1.79	0	0
Recent Broken Coral	2.83	10.24	49.25	0	1.77	0	0
SIMPER Average Dissimilarity 27.63 %							

Site ED. 6 Fished Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.75			
Dragged Rubble	3.59	19.38	19.38	0	1.7	0	0.3
Encrusted Debris	2.95	15.85	35.23	0	1.65	0	0
Recent Damage - Gouge, Scrape	2.08	11.22	46.45	1.28	0	0	0
Algae	1.74	9.6	55.81	0	0	1.06	0
SIMPER Average Dissimilarity 18.57 %							
Site SD. 1 Protected/ Fished	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.73			
Encrusted Debris	2.63	20.53	20.53	0.33	1.43	0	0
Broken Shell	1.77	13.83	34.36	0	1.06	0	0.23
Dragged Rubble	1.73	13.56	47.92	0	1.05	0	0
Recent Broken Coral	1.71	13.39	61.31	0	1.04	0	0
SIMPER Average Dissimilarity 15.08 %							

379

3 c) Species / Substrate	Av. Dis	Contrib. %	Cumulative %	Trans 1 Mean Ab	Trans 2 Mean Ab	Trans 3 Mean Ab	Trans 4 Mean Ab
Site SD. 2 Protected Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.75			
Green Algae	4.55	25.17	25.17	0.15	0	2.38	0
Dragged Rubble	2.49	13.78	13.78	0	1.42	0	0
Encrusted Debris	2.09	11.56	50.51	0	1.24	0	0
<i>Pinctada radiata</i>	1.62	8.95	59.45	0	0	1.05	0.38
SIMPER Average Dissimilarity 18.08 %							

Site SD. 3 Protected Zone	ANOSIM <i>p</i> -value 0.005			ANOSIM <i>R</i> -value 0.66			
Encrusted Debris	3.06	18.92	18.92	0	1.72	0	0.1
Broken & Bleached Coral	2.67	16.05	35.42	0	1.54	0	0
Dragged Rubble	2.53	15.54	50.95	0	1.47	0	0
Broken Shell	2.02	12.51	63.46	0	1.24	0.2	0
SIMPER Average Dissimilarity 16.17 %							

Site SD. 4 Protected Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.76			
Green Algae	4.97	30.5	30.5	0.13	0	2.48	0.3
Encrusted Debris	2.25	13.8	44.3	0	1.31	0	0.4
Dragged Rubble	2.21	13.54	57.83	0	1.29	0	0
Broken Shell	1.53	9.36	67.19	0.4	0.98	0	0
SIMPER Average Dissimilarity 16.29 %							

Site SD. 5 Protected Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.86			
Green Algae	4.28	17.64	17.64	0.63	0	2.41	0.37
Dragged Rubble	2.93	12.07	29.71	0	1.8	0	0
Recent Broken Coral	2.91	11.97	41.69	0	1.77	0	0
<i>Pinctada radiata</i>	2.73	11.26	52.94	0	0	1.67	0
SIMPER Average Dissimilarity 24.27 %							

Site SD. 6 Protected Zone	ANOSIM <i>p</i> -value 0.0001			ANOSIM <i>R</i> -value 0.729			
Green Algae	3.94	24.04	24.04	0	0	2.11	0.3
Dragged Rubble	3.15	19.2	43.24	0	1.65	0	0

Juv. <i>Pinctada</i>	2.6	15.8	59.02	0.3	0	1.5	0
<i>Pinctada radiata</i>	2.58	15.78	74.81	0	0	1.5	0
SIMPER Average Dissimilarity 16.41 %							
Site SD. 7 Protected Zone							
ANOSIM <i>p</i> -value 0.0001				ANOSIM <i>R</i> -value 0.75			
Green Algae	4.08	29.24	29.24	0	0	2.02	0
Dragged Rubble	2.51	17.06	46.84	0	1.36	0	0.2
Broken Shell	1.73	12.33	59.16	0	1.05	0	0
Sponge	1.6	11.16	70.32	0	0	0.95	0
SIMPER Average Dissimilarity 13.95 %							

380

381 The relationship between grouped observations and fishing intensity data was analysed using
 382 PERMANOVA where fishing intensity and closed and open fishing zones were the fixed factors and
 383 habitat categorisations groups were the source of variation. Significant differences were revealed for;
 384 observational identifications which were indicative of recent damage and which represented hard and
 385 soft epibionts (Tables 4 a-c).

386

387 Tables 4 (a-c). Two-way PERMANOVA where fishing intensity and closed and open fishing
 388 zones were the fixed factors and habitat categorisations groups were the source of variation for
 389 observational identifications which were indicative of recent damage and hard and soft epibionts.

390

PERMANOVA examining observational variations of Recent Damage between Fishing Intensity and Closed and Open fishing zones

Source of variation	df	MS	Pseudo- F	<i>P</i>
Fishing Intensity	2	0.05	1.86	< 0.005
Closed and Open zones	1	0.09	3.58	< 0.005
Residual	12			
Total	17			

391 a) *Recent Damage observational components included; clean broken shell, broken coral, gear debris and
 392 substrate gouges*
 393

PERMANOVA examining observational variations in Soft Epibiota between Fishing Intensity and Closed and Open fishing zones

Source of variation	df	MS	Pseudo- F	<i>P</i>
Fishing Intensity	2	0.04	3.52	< 0.005
Closed and Open zones	1	0.06	5.33	< 0.005
Residual	12			
Total	17			

394 b) *Soft Epibiota observational components included; sponges, ascidians, green, brown and red algae*

395

PERMANOVA examining observational variations in Hard Epibiota between Fishing Intensity and Closed and Open fishing zones

Source of variation	df	MS	Pseudo- F	<i>P</i>
Fishing Intensity	2	0.01	2.4	< 0.01
Closed and Open zones	1	0.02	3.09	< 0.0005
Residual	12			
Total	17			

396 c) *Hard Epibiota observational components included; the corals *Acropora*, *Montastrea*, *Porites* and *Sidestrea*
397 and bryozoans*

398

399 Oyster standing stock comparisons

400 The Gunderson model of standing stock comparisons of oyster densities between 2002 and 2016 using
401 equation (*v*) revealed a total standing stock reduction ratio of 580:1 in relation to the total area of *Hairāt*
402 in the Qatar EEZ over a 14 year period (Table 5). Density data for the 1992 survey could not be
403 included as site locations could not be reliably corroborated.

404

405

406

407 Table 5. Comparisons of total standing stocks of *Pinctada radiata* between 2002 and 2016 for *Hairāt* in
 408 three regions in Qatar waters, estimated from the Gunderson (1993) stock density model.

Region	<i>hairāt</i> area ($\times 10^6 \text{ m}^2$) Fishing zone Area	Standing Stock ($\times 10^6$) 2002 (Al-Khayat & Al-Ansi 2008)	Standing Stock ($\times 10^6$) 2016
North	1719	51,570	85
East	3020	105,700	181
South	closed zone	11	0.44
	Total	157,281	267

409 * Survey sites in modelled regions included; North- Bio 9, East- Bio 7, Bio 5, Bio 3, Bio 1, ED 1-6 and South- SD 1-7.*

410

411 Fishing intensity score

412 It was necessary to determine a combined value for available fishing area and total fishing coverage in
 413 order to calculate an “Intensity Score” (*IS*) as the Al-Khor, Doha and Al-Wakra fleets shared the eastern
 414 fishing grounds. Total fishing coverage was $509 \times 10^6 \text{ m}^2$ divided by (*TAj*) of $3015 \times 10^6 \text{ m}^2$ with a
 415 score of (*IS*) 0.169 indicating high intensity. The northern port of Al-Shamal had a total fishing coverage
 416 of $0.054 \text{ m}^2(\times 10^6)$ divided by a (*TaAj*) of $1720 \text{ m}^2(\times 10^6)$ which produced an (*IS*) $< 3.2 \times 10^5$ which was
 417 considered low intensity.

418 DISCUSSION

419 The present survey of the ecological status of *hairāt* and their associated standing stock of *Pinctada*
 420 *radiata* within Qatar’s territorial waters is the first to be undertaken in the region. The results are
 421 emphatic; analysis of 1,800 digital images of the seabed revealed evidence of old fishing damage at all
 422 survey sites, with the most impacted sites exhibiting recent damage in $>50\%$ of the sampled areas. In
 423 these high intensity areas, fishing pressure is extreme; it was not uncommon to have over 10 boats
 424 fishing more than 6000 traps on relatively small *hairāt*. The indicators of damage (encrusted shell
 425 fragments, rubble, broken shell, broken coral and dead coral) are all recognised as signs of damage

426 associated with seabed drag from bottom-tending gear (Calderwood et al., 2015) and the clarity of
427 separation between survey sites which previously displayed an oyster and coral dominant biotope <15
428 years ago suggests benthic disturbance has taken place recently and therefore since the 1992 trawl
429 ban (Carter, 2005; Al-Khayat and Al-Ansi, 2008; Smyth et al., 2016b).

430 Trap fisheries which experience high intensity repetitive fishing within a localised zone can undergo
431 detrimental changes to the targeted resource and its associated environment (Cury et al., 2003). The
432 effects of continuous deployment and retrieval of traps in other fisheries has resulted in similar findings
433 to those presented in this research, including in the Foveaux Straits New Zealand, Georges Bank
434 Maine USA, Quebec Canada, (McQuinn et al., 1988; Watling and Norse, 1998; Cranfield et al., 2003;
435 Shester and Micheli, 2011; Coll et al., 2012). The cumulative damage on trap-fished grounds often
436 results in a habitat altering reduction in ecosystem functionality (Fogarty, 2013). If the practice
437 continues unchecked, the continuous degradation can lead to the affected benthos being considered
438 unfavourable to conservation or beyond restoration (Eno et al., 2001; Kleisner et al., 2013). Thrush et
439 al., (2001) showed that bottom-tending gears which remove and smooth habitat structure can
440 significantly decrease biodiversity and lead to a scraped barren featureless seabed, a habitat
441 description which was prevalent within a recent biotope survey of Qatari offshore sites (Smyth et al.,
442 2016b).

443 Although Gargoor fishing is considered a static low impact method of fishing, our study has shown that
444 a very significant benthic surface area exposed to seabed drag during retrieval. Sites where dredging or
445 trawling was once common and evidence exists of historical benthic habitat damage are usually typified
446 by an accompanying low biodiversity index (Walting and Norse, 1998) and indeed this was the scenario
447 described by Smyth et al., (2016b). The loss of biodiversity and non-commercially relevant fauna as a
448 result of seabed drag is often overlooked by many fishery managers as it has no obvious economic
449 value. However, the often instantaneous removal of supposedly insignificant species can have a
450 damaging cascade effect which will eventually contribute to the decline of a valuable fishery (Eno et al.,

451 2001; Yoshikawa and Asoh, 2004). The majority of non-market species are integral components in
452 maintaining the existence of commercial species firstly as part of a trophic chain and secondly by
453 providing structural habitat (Cranfield et al., 2003). The importance of maintaining a complex benthic
454 topography in the region was established during recent hydroacoustic surveys within fished *hairāt*
455 which showed that the highest densities of high value commercial fish species were associated with the
456 most complex topographical habitats, whereas sites with a featureless profile had low-density
457 assemblages of low value fish (Egerton et al., 2018).

458 The magnitude of decline in oyster stocks within Qatar's territorial waters is particularly alarming, as in
459 2002 the average densities of *P. radiata* were $> 50\text{m}^2$ (Al-Khyat and Al-Ansi, 2008) while in 2016 they
460 were $< 0.18\text{m}^2$. The Gunderson model (iii) revealed a 580:1 decrease in *P. radiata* stocks over a 14
461 year period. This scale of stock decline has implications beyond the loss of the oyster resource. The
462 associated removal of reef habitat and connected prey resources for economically important fishery
463 species, water column filtration, sediment stabilisation and benthic pelagic coupling can have profound
464 effects on overall ecosystem health (Peterson et al., 2003; Smyth et al., 2016b). Once an oyster reef
465 matrix is removed from a *hairāt* it will rapidly become a barren flat limestone platform which will be
466 vulnerable to the effects of sedimentation (Pilskaln et al., 1998; Calderwood et al., 2015) which can
467 interfere with larval settlement. Newell (1988) showed that in the case of *Crassostrea virginica* in
468 Chesapeake Bay, if an area is targeted continuously by fishermen the impact of seabed drag can not
469 only remove existing mature oysters but also impede the attachment of larvae. The early settlement
470 pediveliger stage and juveniles of *P. radiata* (Fig 4 A) would be particularly vulnerable as their byssal
471 threads are not sufficiently hardened (Gosling, 2003). Although the physical removal of oysters by
472 dragged static gear causes an immediate impact, it is the secondary effect of sediment re-suspension
473 during subsequent drag which influences long term recruitment (Smyth et al., 2016a). The persistent
474 turbulence and re-settlement of particulates covers and smoothers previously clean substrates and
475 hinders future successful attachments of larvae (Lenihan and Peterson, 1998; Vasconcelos et al.,

476 2011). This is of particular importance to oyster populations as they naturally settle in areas with
477 reduced tidal velocities and any re-suspension and settlement of particulate matter tends to be long-
478 term (Kennedy and Roberts, 2006). This may explain the lack of any significant epibiont recovery within
479 the heavily fished *hairât*.

480 Our findings indicate the potential for recovery following exclusion of fishing activity, at least in terms of
481 the density of vulnerable epibionts when comparing fished and non-fished zones. Similar recovery has
482 been recorded in rehabilitation zones which were once subjected to intense fishing activity such as the
483 Georges Bank Gulf of Maine, Scandola Nature Reserve Corsica, and Columbretes Island Marine
484 Reserve Spain (Francour, 1991; Gell and Roberts 2003; Goñi et al., 2010). Oyster reef restoration
485 programs can offer habitat managers an additional means of returning damaged or low biodiversity
486 indices sites into biologically functional species-rich environments (Coen and Luckenbach, 2000;
487 Peterson et al., 2003; Coll et al., 2012). The restoration of oyster reef assemblages offers considerable
488 benefits beyond their immediate boundaries and commercial fisheries. Sharma et al., (2015) revealed
489 the positive spill-over effects to seagrass bed coverage prior to and post restoration of a 65 m stretch of
490 oyster reef in the Northern Gulf of Mexico. It was noted that seagrass coverage 100 m beyond the
491 oyster assemblage increased exponentially over a 5 year period. An amelioration of hydrographic
492 conditions and an improvement in water quality post reef establishment was recorded, emphasising
493 that the loss of an oyster reef may jeopardize nearshore habitats as well those in its immediate vicinity.

494 The contribution of oysters in many global fisheries has now been recognised by the FAO with oysters
495 now more economically valuable in regards to the ecosystem services they provide than they are as an
496 independent commodity (Thrush et al., 2001; Bremec et al., 2008). Laing et al., (2006) undertook a
497 Cost Benefit Analysis (CBA) in relation to a feasibility study for the restoration of the European oyster
498 *O. edulis*. The CBA showed that the non-marketable benefits provided high value (e.g. biodiversity,
499 environmental services) even if the oysters themselves were economically non-viable.

500 Auster and Langton (1999) emphasised the importance of benthic environmental features when
501 describing distributions of demersal fish species at spatial and temporal scales (Brander, 2007; Perry et
502 al., 2010). Regional scale patterns of fish stock distribution and abundance can be affected by small-
503 scale variations which have been attributed to differing topographic benthic structures such as oyster
504 beds and rocky reefs (Cranfield et al., 2003; Schejter et al., 2008). In Qatar several valuable
505 commercial species are habitat dependant on the features and fauna associated with the *hairāt*. Fish
506 such as the groupers (Epinephelidae), rabbit fish (Siganidae), emperors (Lethrinidae), and snappers
507 (Carangidae) are all considered *hairāt* dependent in the region (Hartman and Abrahams, 2000;
508 Grandcourt, 2012; Al-Abdulrazzak et al., 2015). They have an intrinsic connectivity to the habitats
509 requirements and if deprived of the specific ecosystem components and the topographical nature
510 associated with their life stages they will cease to exist. Unfortunately this habitual association niche
511 means they are an easily located stationary catch (Smith et al., 2008). It is therefore quite probable that
512 the heavily fished *hairāt* are experiencing the combined effects of habitat destruction and stock
513 overexploitation.

514 One of the most valuable species which is targeted by Gargoor is the grouper *Epinephelus coloides*,
515 which can reach > 38 US\$ / Kg. QFD have reported an annual decrease in grouper size and landings
516 since 2010 with a subsequent rise in market price, making the species even more lucrative to
517 fishermen. Consequently the Gargoor fleets target the stocks evermore intensely thereby exposing
518 their associated habitat to increased fishing disturbance. If this exploitation continues un-checked the
519 possible outcome could be comparable to that which occurred with the Nassau grouper *Epinephelus*
520 *striatus* in the trap fishery of the US Virgin Islands (Garrison et al., 1998). As stocks of grouper declined
521 trapping increased and over a six year period during the 1970s, the spawning aggregation which
522 consisted of tens of thousands of fish was wiped out (Olsen and La Place, 1979). Additional problems
523 related to trap use were observed in areas of intense fishing. The small mesh sizes of the traps lead to
524 reduced productivity through growth over-fishing a result of the premature removal of juvenile fish (Sary

525 et al., 1997; Robichaud et al., 2000). Grandcourt et al., (2004) identified the potential problems of
526 premature removal by Gargoor in the Arabian Gulf and proposed that the high incidence of juvenile
527 catches should be addressed with a re-design of fishery gear. QFD are currently examining potential
528 solutions to this issue with the development of large mesh panels and escape hatches. Qatar University
529 is also playing an active role in addressing the use of the destructive *Manshal* grapple retrieval system
530 and is currently in the advanced stages of developing a lightweight gliding trap collection device.

531 In conclusion, evidence of fishing disturbance was discovered at the offshore *hairāts* accompanied by
532 an alarming decline in *P. radiata* standing stocks. The marine ecosystems of Qatar are currently
533 exposed to considerable anthropogenic and environmental stressors (Sheppard et al., 2010). It appears
534 that increased fishing activity is having a detrimental input and should be addressed. Concern over the
535 effects of fishing on ecosystem health has led several countries worldwide to closing parts of their
536 associated seas in an attempt to preserve fish production (Bradstock and Gordon, 1983; McClanahan
537 and Arthur, 2001). If a proportion of the now unproductive *hairāt* could be protected and re-seeded with
538 small translocated or cultured assemblages of *P. radiata* it is possible that the associated ecosystem
539 services could be restored. The subsequent benefits would not only be ecological but could also
540 augment and restore many of the economically valuable fishery stocks which are currently in decline.
541 This study has highlighted that habitat recovery is possible and underway within zones closed to
542 fishing. If additional strategically positioned protected plots were introduced within the southern
543 hydrodynamic corridor it could lead to the further augmentation of *P. radiata* via oyster larval dispersal
544 linkage. As oyster stocks increased the additional habitat enhancing services they would provide could
545 return the barren *hairāt* to the once historically renowned biogenic entities that they once were.

546

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