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

Al-Maslamani, Ibrahim; David, Smyth; Giraldes, Bruno; Chatting , Mark; Al-Mohannadi , Mohammed; Le Vay, Lewis

Journal of Sea Research

DOI:
10.1016/j.seares.2018.07.006

Published: 01/10/2018

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

Al-Maslamani Ibrahim [a]; Smyth David [b*]; Giraldes Bruno [a]; Chatting Mark [a];

Al-Mohannadi Mohammed [c]; Le Vay Lewis [d]

[a] Environmental Studies Center, Qatar University, Doha, Qatar P.O. Box 2713

[b*] School of Natural and Built Environment, Queen's University Belfast, Northern Ireland, UK

[c] Centre for Applied Marine Sciences, Bangor University, Menai Bridge, Wales, UK

[d] Department of Fisheries, Ministry of Municipality and Environment, State of Qatar

Corresponding author: Smyth. D[b*] School of Natural and Built Environment, Queen's University Belfast, Stranmillis Road. BT9 5AG (D.M.Smyth@qub.ac.uk : smythdm3@gmail.com)

Running Page Head: "Oyster decline in traditional fishing grounds"

ABSTRACT

The territorial waters of Qatar once supported dense assemblages of the pearl oyster *Pinctada radiata.* The oysters settled on a patchy network of limestone platforms *(hairãt)* and provided a suite of ecosystem services to the surrounding marine environment. Commercially important fish species are associated with *hairãt* and as a result, industrial fishing with traps focused on these areas. This study has shown that heavily-fished areas are presently in a state which can be considered non-favorable to conservation while areas closed to fishing are recovering. It is probable that an increase in fishing activity using traditional Gargoor traps and grapple retrieval are responsible for the current ecological status of the *hairãt.* The intensity in trap fishing appears to be having a detrimental effect on species such as corals, sea grasses and oysters. The decline in the standing stock of oysters is dramatic with an estimated reduction ratio of 580:1 between 2002 and 2016. As fishing damage appears to be a significant contributor to these losses, measures such as spatial protection of productive shallow offshore habitats and restriction on fishing effort are urgently required to address the decline. Strategic oyster stock enhancement through the re-seeding of selected areas could boost the recovery of damaged *hairãt* as *P. radiata* ecosystem services return.

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

INTRODUCTION

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

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

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

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

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

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

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

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

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

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

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

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

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

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

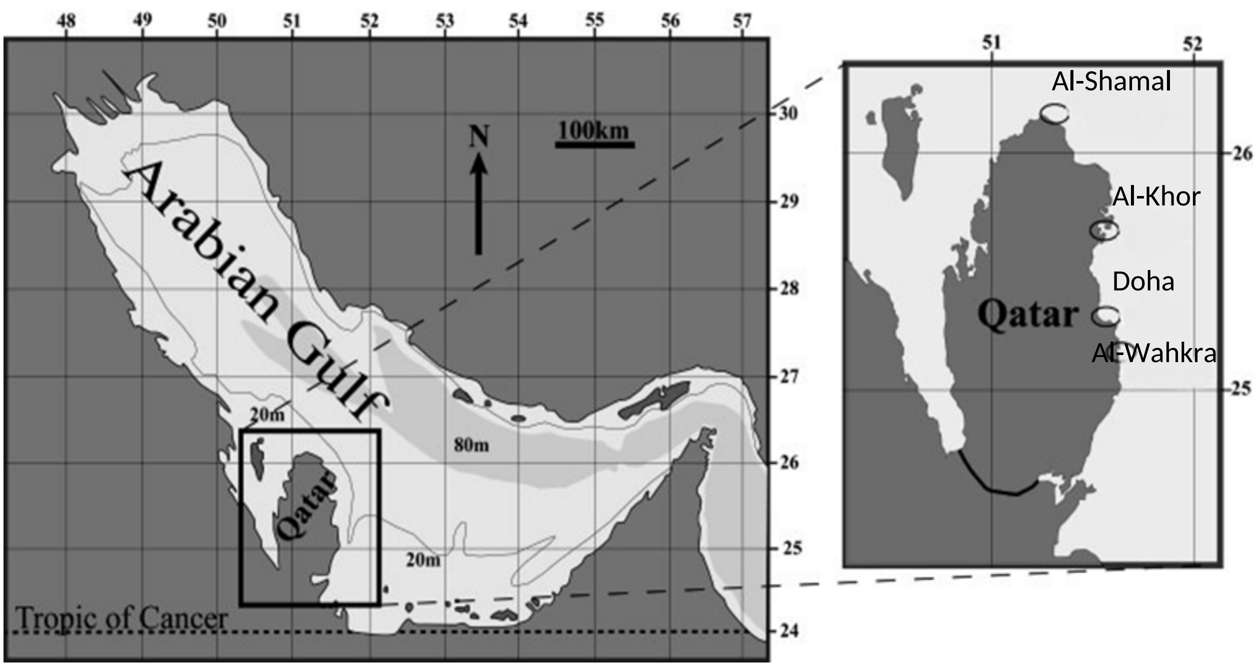

Fig 1. Qatar and its main fishing ports within the Arabian / Persian Gulf

MATERIALS AND METHODS

Study area

Qatar is situated on the west coast of the Arabian Gulf on the peninsula bordering Saudi Arabia and the United Arab Emirates at 25°30′N and 51°15′E (Fig 1). It has a total coastline of 563 km with a hydrodynamic regime typified by a south easterly surface current. Sea water surface temperatures range from 18.7 to 35.0 °C. Where depth is < 15 m, temperatures remain relatively constant between15 to 20 °C throughout the year. Salinities above the thermocline fluctuate between 35.5 to 44.5 ppt depending on season (Kampi and Sadrinasab, 2006).

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

Fishing intensity also influenced survey site location and was provided by the QFD in the form of days at sea and landings records for the Gargoor fleets in proximity to an associated port (Fig 1). The majority of sites were located within designated open fishing zones (Fig 2). Only one northerly site was accessible during the survey, as *hairãt* in this region are located either on or straddling a contentious maritime border with Bahrain (Fig 2). The areas with *hairãt* closed to fishing were under the jurisdiction of the Qatar Coastguard. Coordinates identifying these sites were plotted and presented in Fig 2 and are referred to as 'protected areas / potential fishing grounds'. This terminology has been used to emphasise that although marked as closed on nautical charts the sites could be potentially open for fishing in the future. Also the sites are not strictly policed and illegal fishing activity cannot be completely ruled out.

The location and areal extent of *hairãt* (4740.15 x 10^6 m^2) was predicted from depth-habitat relationships (Walton et al., 2017; Egerton et al., 2018) and calculated using Arc GIS© software and bathymetry data from the NOAA National Center for Environmental Information. All mapping was to a resolution of one minute. The total available fishing area on *hairãt* was calculated using results as per Arc GIS© pixel allocations in m^2. Coverage within restricted sites was subtracted from the total accessible area to provide the amount of available open fishing grounds within each zone (Table 1) presented in Fig 2.

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

Region	Total Hairãt area m^{-2} ($x10^6$)	Closed Fishing area m^{-2} ($x10^6$)	Total Fishing area m^{-2} ($x10^6$)
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**

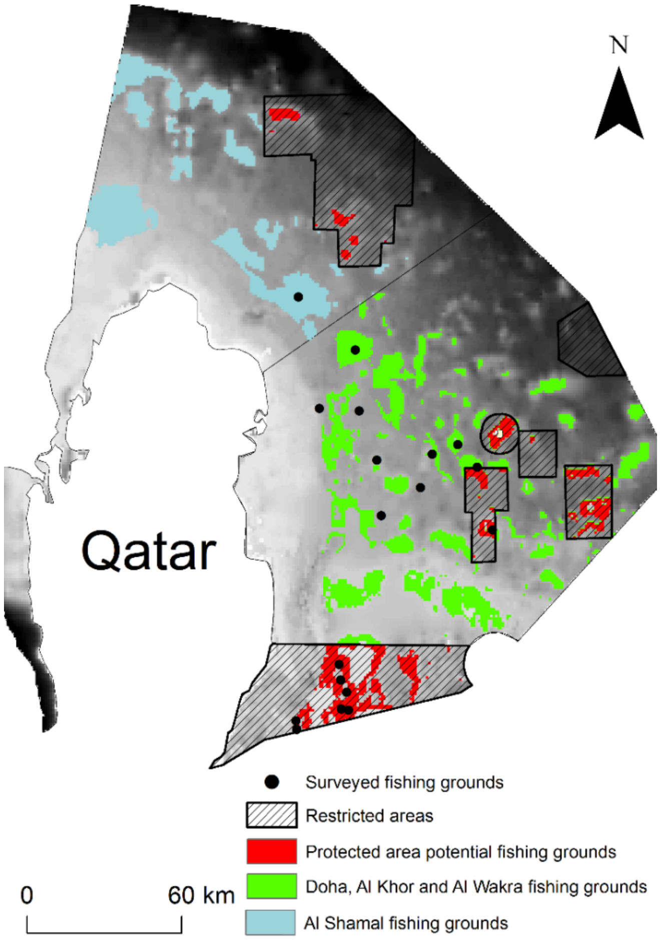

Fig 2. Historically renowned *hairãt* oyster sites at 8-25m depth as per Carter (2005), showing survey sites and restricted and protected area potential fishing grounds. Grey scale background represents bathymetry.

Methodology

A team of scientific divers was used to collect quantitative and qualitative data from the 18 sites. Divers carried out video and digital still surveys along 4x100 m transect lines separated by a 10 m gap as outlined in Smyth et al., (2009) and Giraldes et al., (2015). On commencement and completion of the transect travel a diver deployed surface marker buoys from which longitudinal and latitudinal co-ordinates were recorded using a Garmin ® GPS plotter. Using underwater cameras divers recorded substratum type, environmental damage, dominant key species and *P. radiata* density within a 0.25 m^2

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

Seafloor imagery

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

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

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

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

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

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

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

Analysis of fishing intensity

Fishing activity in the region was considered intense (Fig 3 A-G). *Hairãt* were visited on a regular basis by > 300 vessels > 15 m long each fishing 400 - 600 (225 cm) Gargoors. The *hairãt* closer to shore had the additional pressure of 350 smaller vessels < 15m which fished 50 to 150 traps (Al-Abdulrazzak et al., 2015; Pauly and Zeller, 2016). The QFD provided statistical data in relation to the number of licensed vessels per port, annual active fishing days and number of licensed Gargoor per vessel for 2014 and 2015 (Fig 3 G). This information was used to produce a fishing intensity score based on a similar formula used by MacDonald et al., (1996).

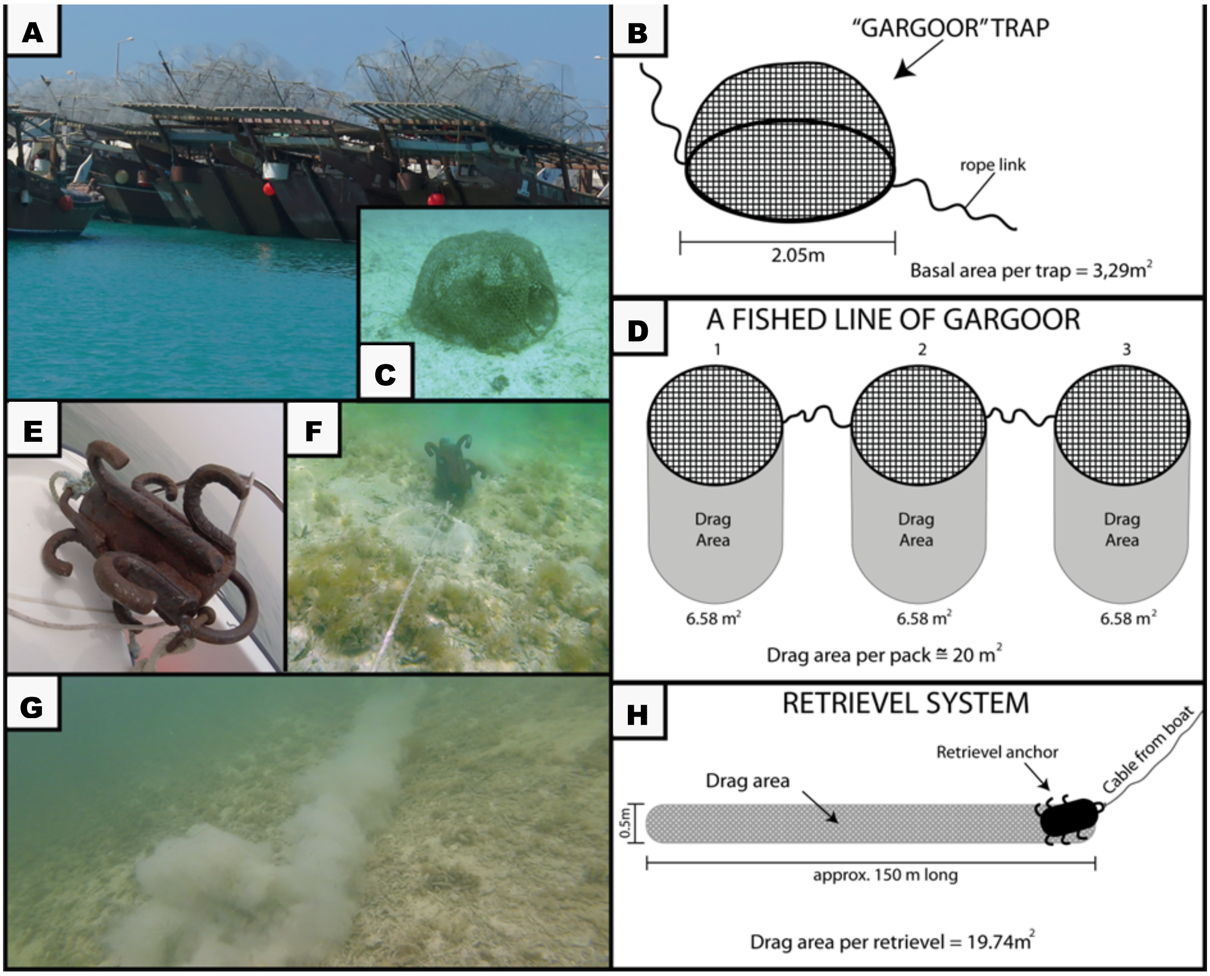

Fig 3. A) Dhow loaded with Gargoor; B-D) Gargoor *in-situ*, trap and line of drag area; E-G) *Manshal* / multi-grapple line retrieval; H) Multi-grapple drag area. (Drag area estimated from information provided by QFD officials in relation to length of trap line attachment rope and average retrieval rope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

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

(A) annual area fished per vessel

(Nv) number of traps fished per vessel

(D) trap diameter (Fig 3 B)

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

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

(Nd) number of trap collection days per annum.

Area of fished m^{-2}: $(Nd \times Nv \times G)/3 + (Nv \times Nd \times L \times D)$ *(i)*

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

$$\boldsymbol{F = \frac{AxV}{TAj}} \qquad (ii)$$

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

Pinctada radiata standing stock estimates

In order to determine the standing stock of oysters (Fig 4 A-F) since the first documented surveys in 1992 (Al- Madfa et al., 1998) and 2002 (Al-Khayat and Al-Ansi, 2008), a comparison of oyster density per m^2 within the replicated surveyed areas was undertaken using *P. radiata* observations gathered from the CPCe© data for each of the 100 m transects. Estimates were made of the current standing stocks of oysters within the fished and protected offshore zones using the following mathematical model adapted from Gunderson (1993):

$$\boldsymbol{P = \sum_{i=1}^{h} (Ri\,.a)C\,i}$$

(iii)

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

Ri = Area of region I in m^2.

a = Area sampled within a single sampling unit.

Ci = Mean no. of oysters observed per sample unit in region *i* based on

n = Samples.

h = Number of regions composing the survey.

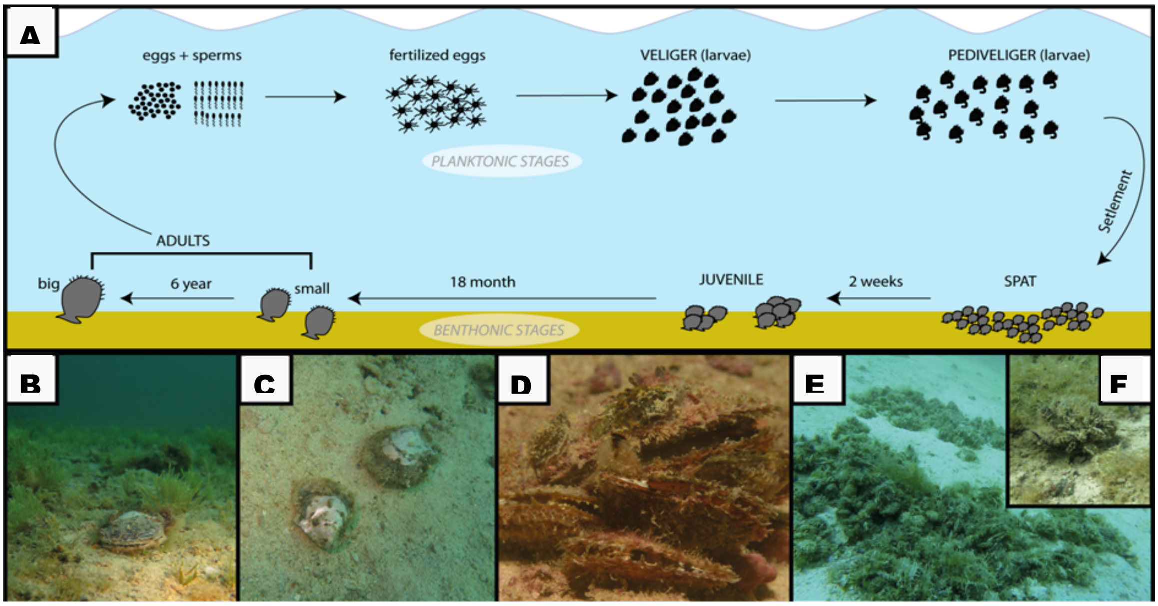

Fig 4. A) Life cycle of *Pinctada radiata*; B-D) Adult *P. radiata*; E-F) Juvenile *P. radiata*.

The total population resident in the entire survey area, ‘P’, was determined using an estimate of survey region in m^2. The surface area, ‘Ri’, for the regions was estimated using scaled images of the *hairãt* from Arc GIS© with a pixel value (924.33 m^2) per calculated area (Table 1). Value ‘a’ is a constant which refers to the area sampled within a single sampling unit (9 m). ‘Ci’ refers to the mean number of oysters observed per sampling unit in region ‘i’ based on ‘n’ samples.

RESULTS

Habitat Imagery analysis

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

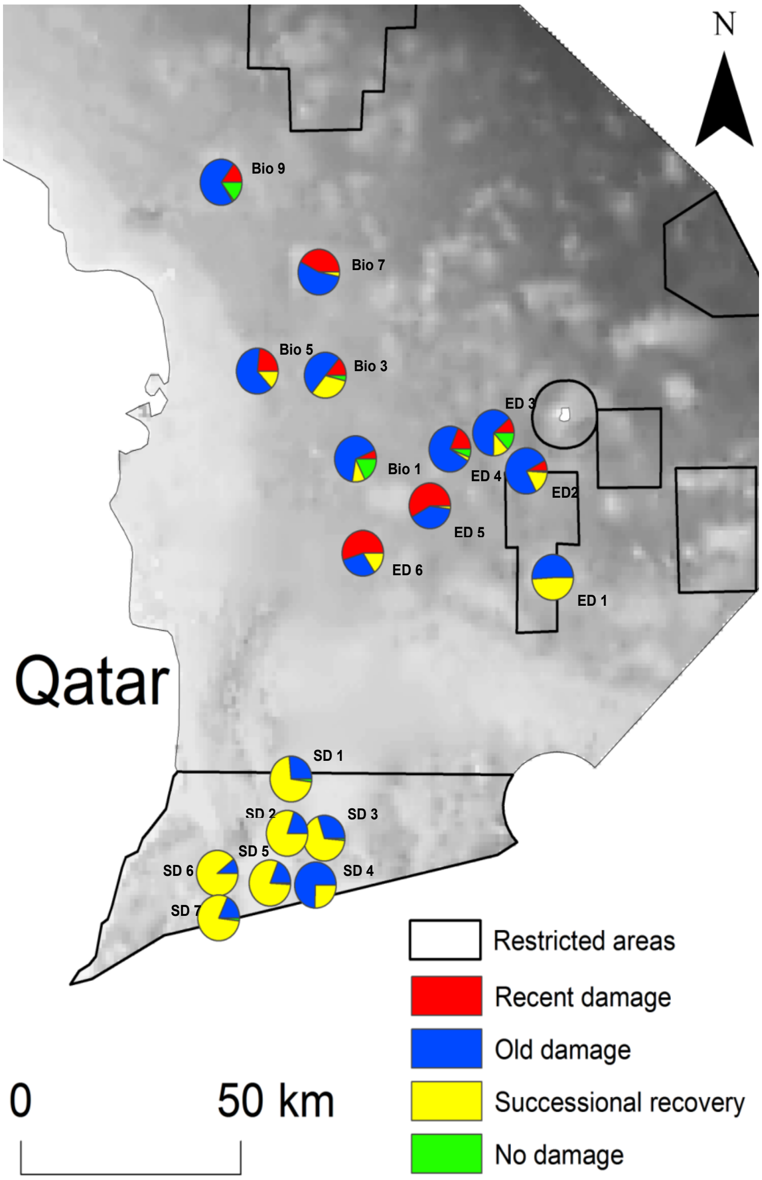

Fig 5. Fractionally proportioned habitat categories per surveyed site.

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

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

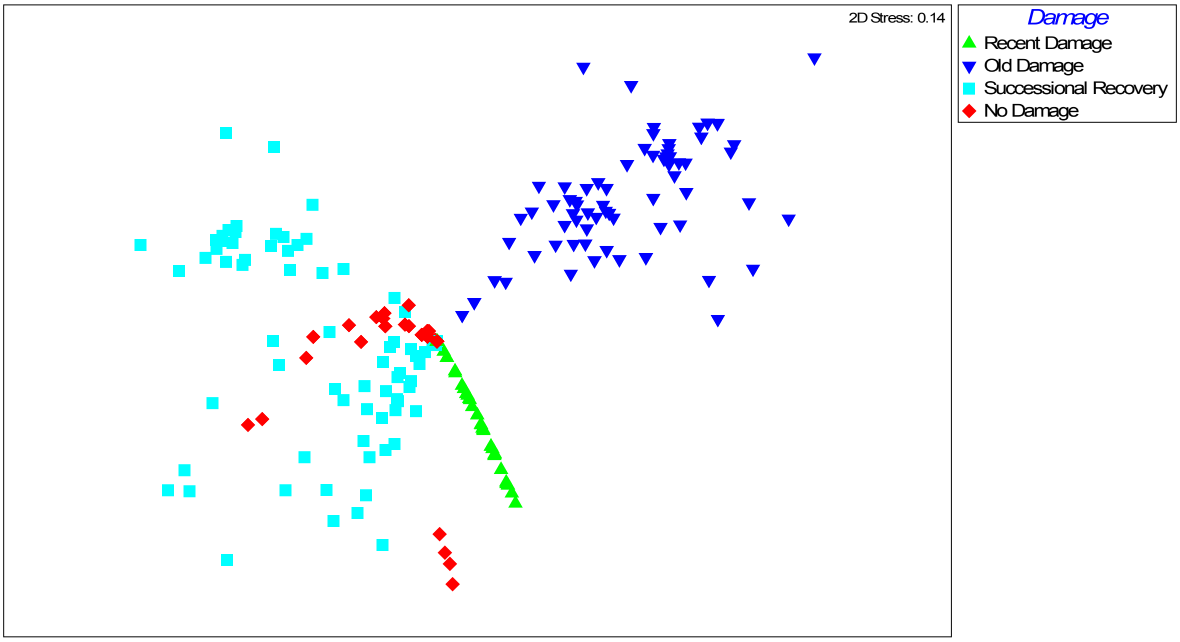

Fig 6. MDS of differences between CPCe observational categorisations indicative of epibionts.

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

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

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

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

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

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

The majority of observations were made in Trans 2 the first central transect that lay on the border of the protected zone suggesting most fishing activity was taking on the perimeter of protection. One totally protected site (SD 3) also had no live epbiota recorded and indeed the data confirmed that fishing incursions must have been taking place as Broken Coral, Broken Shell and Seabed Gouge / Scrape were all recorded as the highest ranking average abundances within factors (Table 3 c). A SIMPER average dissimilarity value of < 27.00% was revealed for the majority of sites showing a relatively constant percentage representation of individual constituent components within transects at each site.

Tables 3 (a-c)

3 a) *Site Bio. 1 Fished Zone*	ANOSIM *p*-value 0.0001			ANOSIM *R*-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
Pinctada radiata	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 *p*-value 0.0001			ANOSIM *R*-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 *p*-value 0.0001			ANOSIM *R*-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 *p*-value 0.0001			ANOSIM *R*-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%							

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 *Ellisella sp.*	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 *p*-value 0.0001		ANOSIM *R*-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 *p*-value 0.0001		ANOSIM *R*-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 %							

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 *p*-value 0.0001		ANOSIM *R*-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
Pinctada radiata	1.62	8.95	59.45	0	0	1.05	0.38
SIMPER Average Dissimilarity 18.08 %							

Site SD. 3 Protected Zone	ANOSIM *p*-value 0.005		ANOSIM *R*-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 *p*-value 0.0001		ANOSIM *R*-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 *p*-value 0.0001		ANOSIM *R*-value 0.86				
Green Algae	4.28	17.64	17.64	0.63	0	2.41	037
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
Pinctada radiata	2.73	11.26	52.94	0	0	1.67	0
SIMPER Average Dissimilarity 24.27 %							
Site SD. 6 Protected Zone	ANOSIM *p*-value 0.0001		ANOSIM *R*-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. *Pinctada*	2.6	15.8	59.02	0.3	0	1.5	0
Pinctada radiata	2.58	15.78	74.81	0	0	1.5	0
SIMPER Average Dissimilarity 16.41 %							
Site SD. 7 Protected Zone		ANOSIM *p*-value 0.0001		ANOSIM *R*-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 %							

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

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

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

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

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

PERMANOVA examining observational variations in Soft Epibiota between Fishing Intensity and Closed and Open fishing zones				
Source of variation	df	MS	Pseudo- F	*P*
Fishing Intensity	2	0.04	3.52	< 0.005
Closed and Open zones	1	0.06	5.33	< 0.005
Residual	12			
Total	17			

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

PERMANOVA examining observational variations in Hard Epibiota between Fishing Intensity and Closed and Open fishing zones				
Source of variation	df	MS	Pseudo- F	*P*
Fishing Intensity	2	0.01	2.4	< 0.01
Closed and Open zones	1	0.02	3.09	< 0.0005
Residual	12			
Total	17			

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

Oyster standing stock comparisons

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

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

Region	*hairãt area ($x\ 10^6\ m^2$) Fishing zone Area*	*Standing Stock ($x10^6$) 2002 (Al-Khayat & Al-Ansi 2008)*	*Standing Stock ($x10^6$) 2016*
North	1719	51,570	85
East	3020	105,700	181
South	closed zone	11	0.44
	Total	**157,281**	**267**

*** 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.***

Fishing intensity score

It was necessary to determine a combined value for available fishing area and total fishing coverage in order to calculate an "Intensity Score" *(IS)* as the Al-Khor, Doha and Al-Wakra fleets shared the eastern fishing grounds. Total fishing coverage was 509 $x10^6$ m^2 divided by (TAj) of 3015 $x10^6$ m^2 with a score of *(IS)* 0.169 indicating high intensity. The northern port of Al-Shamal had a total fishing coverage of 0.054 $m^{-2}(x10^6)$ divided by a $(TaAj)$ of 1720 $m^2(x10^6)$ which produced an *(IS)* < $3.2 \times 10^{\mathbf{5}}$ which was considered low intensity.

DISCUSSION

The present survey of the ecological status of *hairãt* and their associated standing stock of *Pinctada radiata* within Qatar's territorial waters is the first to be undertaken in the region. The results are emphatic; analysis of 1,800 digital images of the seabed revealed evidence of old fishing damage at all survey sites, with the most impacted sites exhibiting recent damage in >50% of the sampled areas. In these high intensity areas, fishing pressure is extreme; it was not uncommon to have over 10 boats fishing more than 6000 traps on relatively small *hairãt.* The indicators of damage (encrusted shell fragments, rubble, broken shell, broken coral and dead coral) are all recognised as signs of damage associated with seabed drag from bottom-tending gear (Calderwood et al., 2015) and the clarity of separation between survey sites which previously displayed an oyster and coral dominant biotope <15 years ago suggests benthic disturbance has taken place recently and therefore since the 1992 trawl ban (Carter, 2005; Al-Khayat and Al-Ansi, 2008; Smyth et al., 2016b).

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

Although Gargoor fishing is considered a static low impact method of fishing, our study has shown that a very significant benthic surface area exposed to seabed drag during retrieval. Sites where dredging or trawling was once common and evidence exists of historical benthic habitat damage are usually typified by an accompanying low biodiversity index (Walting and Norse, 1998) and indeed this was the scenario described by Smyth et al., (2016b). The loss of biodiversity and non-commercially relevant fauna as a result of seabed drag is often overlooked by many fishery managers as it has no obvious economic value. However, the often instantaneous removal of supposedly insignificant species can have a damaging cascade effect which will eventually contribute to the decline of a valuable fishery (Eno et al., 2001; Yoshikawa and Asoh, 2004). The majority of non-market species are integral components in maintaining the existence of commercial species firstly as part of a trophic chain and secondly by providing structural habitat (Cranfield et al., 2003). The importance of maintaining a complex benthic topography in the region was established during recent hydroacoustic surveys within fished *hairãt* which showed that the highest densities of high value commercial fish species were associated with the most complex topographical habitats, whereas sites with a featureless profile had low-density assemblages of low value fish (Egerton et al., 2018).

The magnitude of decline in oyster stocks within Qatar's territorial waters is particularly alarming, as in 2002 the average densities of *P. radiata* were > 50m^2 (Al-Khyat and Al-Ansi, 2008) while in 2016 they were < 0.18 m^2. The Gunderson model (*iii)* revealed a 580:1 decrease in *P. radiata* stocks over a 14 year period. This scale of stock decline has implications beyond the loss of the oyster resource. The associated removal of reef habitat and connected prey resources for economically important fishery species, water column filtration, sediment stabilisation and benthic pelagic coupling can have profound effects on overall ecosystem health (Peterson et al., 2003; Smyth et al., 2016b). Once an oyster reef matrix is removed from a *hairãt* it will rapidly become a barren flat limestone platform which will be vulnerable to the effects of sedimentation (Pilskaln et al., 1998; Calderwood et al., 2015) which can interfere with larval settlement. Newell (1988) showed that in the case of *Crassostrea virginica* in Chesapeake Bay, if an area is targeted continuously by fishermen the impact of seabed drag can not only remove existing mature oysters but also impede the attachment of larvae. The early settlement pediveliger stage and juveniles of *P. radiata* (Fig 4 A) would be particularly vulnerable as their byssal threads are not sufficiently hardened (Gosling, 2003). Although the physical removal of oysters by dragged static gear causes an immediate impact, it is the secondary effect of sediment re-suspension during subsequent drag which influences long term recruitment (Smyth et al., 2016a). The persistent turbulence and re-settlement of particulates covers and smoothers previously clean substrates and hinders future successful attachments of larvae (Lenihan and Peterson, 1998; Vasconcelos et al., 2011). This is of particular importance to oyster populations as they naturally settle in areas with reduced tidal velocities and any re-suspension and settlement of particulate matter tends to be long-term (Kennedy and Roberts, 2006). This may explain the lack of any significant epibiont recovery within the heavily fished *hairãt*.

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

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

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

One of the most valuable species which is targeted by Gargoor is the grouper *Epinephelus coloides,* which can reach > 38 US$ / Kg. QFD have reported an annual decrease in grouper size and landings since 2010 with a subsequent rise in market price, making the species even more lucrative to fishermen. Consequently the Gargoor fleets target the stocks evermore intensely thereby exposing their associated habitat to increased fishing disturbance. If this exploitation continues un-checked the possible outcome could be comparable to that which occurred with the Nassau grouper *Epinephelus striatus* in the trap fishery of the US Virgin Islands (Garrison et al., 1998). As stocks of grouper declined trapping increased and over a six year period during the 1970s, the spawning aggregation which consisted of tens of thousands of fish was wiped out (Olsen and La Place, 1979). Additional problems related to trap use were observed in areas of intense fishing. The small mesh sizes of the traps lead to reduced productivity through growth over-fishing a result of the premature removal of juvenile fish (Sary et al., 1997; Robichaud et al., 2000). Grandcourt et al., (2004) identified the potential problems of premature removal by Gargoor in the Arabian Gulf and proposed that the high incidence of juvenile catches should be addressed with a re-design of fishery gear. QFD are currently examining potential solutions to this issue with the development of large mesh panels and escape hatches. Qatar University is also playing an active role in addressing the use of the destructive *Manshal* grapple retrieval system and is currently in the advanced stages of developing a lightweight gliding trap collection device.

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

Acknowledgements

This research was supported by the Qatar National Research Foundation, National Priorities Research programme project NPRP-6-1680-4-027. The authors express their sincere thanks to Dr Mohammad Abdullah of the Qatar Department of Fisheries, Mr Reyniel Gasang, Ms. Aisha Al-Ashwel, Ms. Najat Al-Omary, Ms. Noora Alfardi and the crew of the RV Janan.

REFERENCES

Al-Abdulrazzak, D., Zeller, D., Belhabib, D., Tesfamichael, D., Pauly, D., 2015. Total marine fisheries catches in the Persian/Arabian Gulf from 1950 to 2010. Reg. Stud. Mar. Sci. *2*, 28-34.

Al-Ansari, A.A., Shams, A.J., Aldaggal, A.A., 1994. Trial production of pearl oyster of *Pinctada radiata* trochophore larvae for use as larval food. Minister of Commerce and Agriculture Directorate of Fisheries National Mariculture Center, Bahrain.

~~Al-Ansari., 2017. Fishing fleet manager. Pers. Comment~~

Al-Khayat, J.A., Al-Ansi, M.A., 2008. Ecological features of oyster beds distribution in Qatari waters, Arabian Gulf. Asian J. Sci. Res. 1, 544-561.

Al-Madfa, H., Abdel-Moati, M.A.R., Al-Gimaly, F.H., 1998. *Pinctada radiata* (Pearl Oyster): a bioindicator for metal pollution monitoring in the Qatari waters (Arabian Gulf). Bull. Environ. Contam. Toxicol. 60, 245-251.

Al-Mohammadi., 2017. QFD Qatar Fisheries Department. Pers. Comment.

Althaus, F., Williams, A., Schlacher, T.A., Kloser, R.J., Green, M.A., Barker, B.A., Schlacher-Hoenlinger, M.A., 2009. Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. Mar. Ecol. Prog. Ser. 397, 279-294.

Auster, P.J., Langton, R.W., 1999. The effects of fishing on fish habitat. Am. Fish. Soc. Symp. 22, 150-187.

Beck, M,, Brumbaugh, D.R., Airoldi, L., Carranza, A., Coen, L.D., Crawford, C., Defeo, O., Edgar, G.J., Hancock, B., Kay, M.C., Lenihan, H.S., Luckenback, M.W., Toropova, C.L., Zhang, G., Guo, X., 2011. Oyster reefs at risk and recommendations for conservation, restoration, and management. Bio. Science 61, 107–116.

Bento, R., Feary, D.A., Hoey, A.S., Burt, J.A., 2017. Settlement Patterns of Corals and other Benthos on Reefs with Divergent Environments and Disturbances Histories around the North eastern Arabian Peninsula. Front. Mar. Sci. 4, 305. doi: 10.3389/fmars.2017.00305

Botsford, L.W., Castilla, J.C., Peterson, C.H., 1997. The management of fisheries and marine ecosystems. Sci. 277, 509–515.

Bouma, T.J., Olenin, S., Reise, K., Ysebaert, T., 2009. Ecosystem engineering and biodiversity in coastal sediments: posing hypotheses. Helgol. Mar. Res. 63, 95-106.

Bradshaw, C., Veale, L.O., Hill, A.S., Brand, A.R., 2001. The effect of scallop dredging on Irish Sea benthos: experiments using a closed area. Springer Netherlands, In Coastal Shellfish—A Sustainable Resource pp 129-138.

Bradstock, M., Gordon, D.P., 1983. Coral-like bryozoan growths in Tasman Bay, and their protection to conserve commercial fish stocks. NZ. J. M Freshw. Res. 17,159-163.

Brander, K.M., 2007. Global fish production and climate change. Pro. Natl. Acad. Sci. 104, 709-714.

Bremec, C., Escolar, M., Schejter, L., Genzano, G., 2008. Primary settlement substrate of scallop, Zygochlamys patagonica (King and Broderip, 1832) (Mollusca: Pectinidae) in fishing grounds in the Argentine Sea. J. Shellfish. Res. 27, 273-280.

Burdett, A.L., (Ed.) 1995. Records of the Persian Gulf Pearl Fisheries. Archive Editions, pp 1857-1962.

Calderwood, J., O'Connor, N.E., Roberts, D., 2015. Effects of baited crab pots on cultivated mussel (Mytilus edulis) survival rates. ICES J. Mar. Sci. 72, 1802-1810.

Carleton, J.H., Done, T.J., 1995. Quantitative video sampling of coral reef benthos: large-scale application. Coral Reefs 14, 35-46

Carter, R., 2005. The history and prehistory of pearling in the Persian Gulf. J. Econ. Soc. Hist. Orient. 48, 139-209.

Clarke, K.R., Warwick, R.M., 1994. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. National Environment Research Council, Swindon UK.

Coen, L.D., Luckenbach, M.W., 2000. Developing success criteria and goals for evaluating oyster reef restoration: ecological functioning or resource exploitation? Ecol. Engin. 15, 323–343.

Coll, M., Piroddi, C., Albouy, C., Ben Rais Lasram, F., Cheung, W.W., Christensen, V., Palomares, M.L., 2012. The Mediterranean Sea under siege: spatial overlap between marine biodiversity, cumulative threats and marine reserves. Glob. Ecol. Biogeogr. 21, 465-480.

Connor, D. W., Allen, J. H., Golding, N., Howell, K. L., Lieberknecht, L. M., Northen, K. O., Reker, J. B., 2004. The Marine Habitat Classification for Britain and Ireland Version 04.05- Sublittoral Sediment Section. *Peterborough: Joint Nature Conservation Committee (JNCC)*.

Cranfield, H.J., Manighetti, B., Michael, K.P., Hill, A., 2003. Effects of oyster dredging on the distribution of bryozoan biogenic reefs and associated sediments in Foveaux Strait, southern New Zealand. Cont. Shelf. Res. 23, 1337-1357.

Cury, P., Shannon, L., Shin, Y.J., 2003. The functioning of marine ecosystems: a fisheries perspective. Responsible fisheries in the marine ecosystem, Cambridge University Press, pp 103-123.

De Young, C., (Ed.) 2006. Review of the state of world marine capture fisheries management: Indian Ocean (No. 488) FAO.

Egerton, J., Al-Ansi, M., Abdallah, M., Walton, M., Hayes, J., Le Vay, L., 2018. Hydroacustics to examine fish association with shallow offshore habitats in the Arabian Gulf. Fish. Res. 199, 127-136.

Eno, N.C., MacDonald, D.S., Kinnear, J.A., Amos, S.C., Chapman, C.J., Clark, R.A., Munro, C., 2001. Effects of crustacean traps on benthic fauna. ICES J. Mar. Sci. 58, 11-20.

Fogarty, M.J., 2013. The art of ecosystem-based fishery management. Can. J. Fish. Aqua. Sci. 71, 479-490.

Francour, P., 1991. The effect of protection level on a coastal fish community at Scandola Corsica. Rev. Ecol. Terre. Vie. 46, 65-81.

Garrison, V.H., Rogers, C.S., Beets, J., 1998. Overfishing and in situ observations of fish traps in St. John, US Virgin Islands. Rev. Biol. Trop. 46, 41-59.

Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends. Ecol. Evol. 18, 448-455.

Giraldes, B.W., Silva, A.Z., Corrêa, F.M., Smyth, D.M., 2015. Artisanal fishing of spiny lobsters with gillnets — A significant anthropic impact on tropical reef ecosystem. Glob. Ecol. Conserv. 4, 572-580.

Goni, R., Hilborn, R., Díaz, D., Mallol, S., Adlerstein, S., 2010. Net contribution of spillover from a marine reserve to fishery catches. Mar. Ecol. Prog. Ser. 40, 233-243.

Gosling, E., 2003. Reproduction, settlement and recruitment. Bivalve molluscs: biology, ecology and culture. Fishing news books Blackwell Oxford pp 131-168.

Grandcourt, E.M., Al-Abdessalaam, T.Z., Francis, F., Al-Shamsi, A.T., 2004. Biology and stock assessment of the Sparids, Acanthopagrus bifasciatus and Argyrops spinifer (Forsskål, 1775) in the Southern Arabian Gulf. Fish. Res. 69, 7-20.

Grandcourt, E., 2012. Reef fish and fisheries in the Gulf. In Coral Reefs of the Gulf Springer Netherlands p 127-161

Gunderson, D.R., 1993. Surveys of fisheries resources. John Wiley & Sons UK.

Hall, S.J., 2002. The continental shelf benthic ecosystem: current status, agents for change and future prospects. Environ. Conserv. 29, 350-374.

Harding, J., Mann, R., 2003. Influence of habitat on diet and distribution of striped bass (*Morone saxatilis*) in a temperate estuary. Bull. Mar. Sci. 72, 3.

Hartman, E.J., Abrahams, M.V., 2000. Sensory compensation and the detection of predators: the interaction between chemical and visual information. Proc. R. Soc. B. 267, 571-575.

Hinz, H., Prieto, V., Kaiser, M.J., 2009. Trawl disturbance on benthic communities: chronic effects and experimental predictions. Ecol. Appl. 19, 761-773.

Hirst, N.E., Clark, L., Sanderson, W.G., 2012. The distribution of selected MPA search features and Priority Marine Features of the NE coast of Svcotland. Scottish Natural Heritage Commisioned Report 500. Edinbrugh: Scottish Natural Heritage p.132

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., et al., 2001. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Sci. 293, 629-637.

Jennings, S., Kaiser, M.J., 1998. The effects of fishing on marine ecosystems. Adv. Mar. Biol. 34, 201-352.

Kaiser, M.J., Collie, J.S., Hall, S.J., Jennings, S., Poiner, I.R., 2002. Modification of marine habitats by trawling activities: prognosis and solutions. Fish. Fish. 3, 114-136.

Kampi, J., Sadrinasab, M., 2006. Circulation of the Persian Gulf: A numerical study. Ocean Sci. 2, 27-41.

Kennedy, R.J., Roberts, D., 2006. Commercial oyster stocks as a potential source of larvae in the regeneration of *Ostrea edulis* in Strangford Lough, Northern Ireland. J. Mar. Biol. Ass. UK. 86, 153-159.

Kleisner, K., Zeller, D., Froese, R., Pauly, D., 2013. Using global catch data for inferences on the world’s marine fisheries. Fish. Fish. 14, 293-311.

Kohler, K.E., Gill, S.M., 2006. Coral Point Count with Excel extensions (CPCe): A Visual Basic program for the determination of coral and substrate coverage using random point count methodology. Comput. Geosci. 32, 1259-1269.

Laing, I., Walker, P., Areal, F., 2006. Return of the native - is European oyster (*Ostrea edulis*) stock restoration in the UK feasible? Aquat. Liv. Res. 19, 283–287.

Lenihan, H.S., Peterson, C.H., 1998. How habitat degradation through fishery disturbance enhances impacts of hypoxia on oyster reefs. Ecol. Appl. 8,128-140.

Linehan, J.E., Gregory, R.S., Schneider, D.C., 2001. Predation risk of age-0 cod (*Gadus morhua*) relative to depth and substrate in coastal waters. J. Exp. Mar. Biol. Ecol. 263, 25-44.

Lokrantz, J., Nyström, M., Norström, A.V., Folke, C., Cinner, J.E., 2009. Impacts of artisanal fishing on key functional groups and the potential vulnerability of coral reefs. Environ. Conserv. 36, 327-337.

Lorimer, J.G., 1915. Gazetteer of the Persian Gulf Oman and Central Arabia. Calcutta.

Luckenbach, M.W., Coen, L.D., Ross, P.G., Stephen, J.A., 2005. "Oyster Reef Habitat Restoration: Relationships between Oyster Abundance and Community Development Based on Two Studies in Virginia and South Carolina." J. Coastal. Res. 34, 64-78.

MacDonald, D.S., Little, M., Eno, M.C., Hiscock, K., 1996. Disturbance of benthic species by fishing activities: a sensitivity index. Aquatic. Conserv: Mar. Freshw. Ecosyst. 6, 257-268.

McClanahan, T.R., Arthur, R., 2001. The effect of marine reserves and habitat on populations of East African coral reef fishes. Ecol. Appl. 11, 559-569.

McQuinn, I.H., Gendron, L., Himmelman, J.H., 1988. Area of attraction and effective area fished by a whelk (Buccinum undatum) trap under variable conditions. Can. J. Fish. Aquat. Sci. 45, 254-260.

Mohammed, S.Z., Yassien, M.H., 2003. Population parameters of the pearl oyster *Pinctada radiata* (Leach) in Qatari Waters, Arabian Gulf. Turk. J. Zoo. 27, 339-343.

Neumann, H., Kröncke, I., 2011. The effect of temperature variability on ecological functioning of epifauna in the German Bight. Mar. Ecolo. 32, 49–57 doi:10.1111/j.1439-0485.2010.00420.x

Newell, R.I., 1988. Ecological changes in Chesapeake Bay: are they the result of overharvesting the American oyster, *Crassostrea virginica*. Understanding the estuary. Adv. Chesap. B. Res. 129, 536-546.

Olsen, D.A., La Place, J.A., 1979. A study of a Virgin Islands grouper fishery based on a breeding aggregation. Proc. Gulf. Carb. Fish. Inst. 32, 130-144.

Parker, S.J., Penney, A.J., Clark, M.R., 2009. Detection criteria for managing trawl impacts on vulnerable marine ecosystems in high seas fisheries of the South Pacific Ocean. Mar. Ecol. Prog. Ser. 397, 309-317.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., Zeller, D., 2002. Towards sustainability in world fisheries. Nature. 418, 689-695.

Pauly, D., Zeller, D., 2016. Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. Nat. Commun. 7, 102.

Perry, R.I., Cury, P., Brander, K., Jennings, S., Möllmann, C., Planque, B., 2010. Sensitivity of marine systems to climate and fishing: concepts, issues and management responses. J. Mar. Syst. 79, 427-435.

Peterson, C.H., Grabowski, J.H., Powers, S.P., 2003. Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. Mar. Ecol. Prog. Ser. 264, 249–64.

Pilskaln, C.H., Churchill, J.H., Mayer, L.M., 1998. Resuspension of sediment by bottom trawling in the Gulf of Maine and potential geochemical consequences. Conserv. Biol. 12, 1223-1229.

Rice, J., 2006. Department of Fisheries and Oceans, Ottawa, ON (Canada); Canadian Science Advisory Secretariat, Ottawa, ON (Canada) Impacts of mobile bottom gears on seafloor habitats, species*, and communities: a review and synthesis of selected international reviews* (No. 2006/057). DFO, Ottawa, ON (Canada)

Rice, J., 2011. Managing fisheries well: delivering the promises of an ecosystem approach. Fish. 12, 209-231.

Robichaud, D., Hunte, W., Chapman, M.R., 2000. Factors affecting the catchability of reef fishes in Antillean fish traps. Bull. Mar. Sci. 67, 831-844.

Rodney, W.S., Paynter, K.T., 2006. Comparisons of macrofaunal assemblages on restored and non-restored oyster reefs in mesohaline regions of Chesapeake Bay in Maryland. J. Exp. Mar. Biol. Ecol. 335, 39–51.

Rothschild, B.J., Ault, J.S., Goulletquer, P., Heral, M., 1994. Decline of the Chesapeake Bay oyster population a century of habitat destruction and overfishing. Mar. Ecol. Prog. Ser. 3**,** 29-39.

Sanderson, W.G., Holt, R.H.F., Kay, L., Ramsay, K., Perrins, J., and others 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea. II. Epifauna recorded by divers and cameras. J. Mar. Biol. Ass. UK. 88, 143–149.

Sary, Z., Oxenford, H.A., Woodley, J.D., 1997. Effects of an increase in trap mesh size on an overexploited coral reef fishery at Discovery Bay, Jamaica. Mar. Ecol. Prog. Ser. 154, 107-120.

Schejter, L., Bremec, C.S., Hernández, D., 2008. Comparison between disturbed and undisturbed areas of the Patagonian scallop (Zygochlamys patagonica) fishing ground “Reclutas” in the Argentine Sea. J. Sea Res. 60,193-200.

Sharma, S., Goff, J., Moody, R.M., Byron, D., Heck, K.L., Powers, S.P., Ferraro, C., Cebrian, J., 2016. Do restored oyster reefs benefit seagrasses? An experimental study in the Northern Gulf of Mexico. Restor. Ecol. 24, 306–313 doi:10.1111/rec.12329

Sheppard ,C., Al-Husiani, M., Al-Jamali, F., Al-Yamani, F., Baldwin, R., Bishop, J., Jones, D.A and others (2010) The Gulf: a young sea in decline. Mar. Pollu. Bull. 60, 13-38.

Shester, G.G., Micheli, F., 2011. Conservation challenges for small-scale fisheries: Bycatch and habitat impacts of traps and gillnets. Biolo. Conserv. 144, 1673-1681.

Smith, M.D., Roheim, C.A., Crowder, L.B., Halpern, B.S., Turnipseed, M., Anderson, J.L., Asche, F., Selkoe, K.A and others 2010. Sustainability and global seafood. Sci. 327, 784–786.

Smyth, D., Roberts, D., Browne, L., 2009. Impacts of unregulated harvesting on a recovering stock of native oysters (*Ostrea edulis*). Mar. Pollu. Bull. 58, 916-922.

Smyth, D., Roberts, D., 2010. The European oyster (Ostrea edulis) and its epibiotic succession. Hydrobiologia. 655, 25–36.

Smyth, D., Al-Maslamani, I., Giraldes, B.W., Chatting, M., Al-Ansari, E., Le Vay, L., 2016a. Anthropogenic related variations in the epibiotic biodiversity and age structure of the “Pearl Oyster” *Pinctada radiata* within the eulittoral zone of Qatar. Reg. Stud. Mar. Sci. 5, 87-96.

Smyth, D.M., Al-Maslamani, I., Chatting, M., Giraldes, B.W., 2016b. Benthic surveys of the historic Pearl Oyster beds of Qatar reveal a dramatic ecological change. Mar. Pollu. Bull. DOI: 10.1016/j.marpolbul.2016.08.085

Soniat, T.M., Finelli, C.M., Ruiz, J.T., 2004. Vertical structure and predator refuge mediate oyster reef development and community dynamics. J. Exp. Mar. Biol. Ecol., 310, 163-182.

Stamatopoulos, C., Abdallah, M., 2016. Standardization of fishing effort in Qatar fisheries: methodology and case studies. J. Mar. Sci. Res. Dev. 5, 170. DOI: 10.4172/21559910.1000170.

Strain, E.M.A., Allcock, A.L., Goodwin, C.E., Maggs, C.A., Picton, B.E., Roberts, D., 2012. The long-term impacts of fisheries on epifaunal assemblage function and structure, in a special area of conservation J. Sea Res. 67, 58-68.

Thrush, S.F., Hewitt, J.E., Funnell, G.A., Cummings, V.J., Ellis, J., Schultz, D., Talley, D., Norkko, A., 2001. Fishing disturbance and marine biodiversity: the role of habitat structure in simple soft-sediment systems. Mar. Ecol. Prog. Ser. 223, 277-286. doi.org/10.3354/meps223277

Tolley, S.G., Volety, A.K., 2005. The role of oysters in habitat use of oyster reefs by resident fishes and decapod crustaceans. J. Shellfish res. 24, 1007–1012.

Vasconcelos, R.O., Carriço, R., Ramos, A., Modesto, T., Fonseca, P.J., Amorim, M.C.P., 2011. Vocal behaviour predicts reproductive success in a teleost fish. Behav. Ecol. 2, 375-383.

Watling, L., Norse, E.A., 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. Conserv. Biol. 12, 180-197. doi.org/10.1046/j.15231739.1998.0120061180.x.

Walton, M.E.M., Hayes, J., Al-Ansari, M., Abdallah, M., Al-Maslamani, I., Mohannadi, M., Le Vay. L and others 2017. Biodiversity of offshore benthic habitats supporting commercial fisheries in the Arabian Gulf. ICES J. Mar. Sci. 75, 178-189.

Wells, H.W., 1961. The fauna of oyster beds with special reference to the salinity factor. Ecol. Mono. 31, 239-266.

Yoshikawa, T., Asoh, K., 2004. Entanglement of monofilament fishing lines and coral death. Biol. Conserv. 117, 557-560.