Accepted refereed manuscript of:
Jones EV, Macintosh D, Stead S & Gray T (2017) How effective are MPAs in conserving crab stocks? A comparison of fisheries and conservation objectives in three coastal MPAs in Thailand. *Ocean and Coastal Management*, 149, pp. 186-197.

DOI: https://doi.org/10.1016/j.ocecoaman.2017.09.012

How effective are MPAs in conserving crab stocks? A comparison of fisheries and conservation objectives in three coastal MPAs in Thailand

Abstract

Mangrove forest ecosystems support aquatic species important to tropical fishing communities, but habitat degradation and over-fishing have caused many coastal fishery stocks to decline. Marine Protected Areas (MPAs) are widely promoted as a management option to reverse this situation. Using swimming crabs as indicator species, this paper explores the ecological effectiveness of two community-led MPAs and one co-managed MPA in Ranong and Phang-nga Provinces in southern Thailand. Comparisons were made of two fisheries objectives: catch per unit effort (CPUE); and size frequency distribution of *Portunus* spp. and *Scylla olivacea*; and one conservation objective: catch composition on benthic aquatic species, between each of the three managed areas and three associated control sites to look for effects of management. Eight replicates of each survey were undertaken in each site: four in the wet season, from May to July 2011; and four in the dry season, from February to March 2012.

Two of the MPAs, one a no-take zone and one a gear limitation zone, both managed by local communities, showed a significant increase in CPUE of target species compared with their controls to the benefit of local fishers. There was, however, little evidence of management impact on the composition of benthic aquatic species compared to the controls, so community-led management did not increase biodiversity. The third MPA, a seasonal no-take zone, co-managed by local communities and local government, showed no significant effect on either CPUE of target species or composition of benthic aquatic species when compared to its control. However, on size frequency distribution, a higher abundance of all size classes of *Portunus pelagicus* was observed in all MPAs compared to their control sites. To conclude, the two community-led MPAs benefitted fishers but had no effect on marine biodiversity, while the co-managed MPA did not benefit either fishers or marine biodiversity. However, all three MPAs showed increased crab abundance in each size class.

Key words: Mangroves; *Portunus*; *Scylla;* marine protected area; co-management; community management; Thailand

1. Introduction

The ecological value of mangrove forest ecosystems is now widely recognised (Duke et al., 2007; Polidoro et al., 2010), as is the direct contribution they make to the livelihoods of millions of small-scale fishers (Sathirithai & Barbier, 2001; Sudtongkong & Webb, 2008). Mangrove-associated benthic invertebrates, such as shrimps, crabs and molluscs, are estimated to contribute income exceeding $4 billion (USD) worldwide, per year from wild caught stocks, and wild caught seed stocked for aquaculture (Ellison, 2008). In Thailand's coastal areas, crustaceans are one of the most important fishery groups, especially crab species in the family Portunidae.

Two species, *Portunus pelagicus* and *Portunus sanguinolentus* (both swimming crabs) dominate inshore waters, while *Scylla olivacea* (mud crab) is mangrove-associated. *P. pelagicus* and *P. sanguinolentu*s are open-water marine species found throughout the Indo-West Pacific oceans (Stephenson & Campbell, 1959). They exhibit fast growth rates and high fecundity, according to research conducted in cooler waters (Williams, 1982; Edgar, 1990; Weng, 1992; Johnston et al., 2011) but an understanding of their response to fishing pressure is less well known due to their wide-ranging lifecycles and limited research on the fecundity and growth of these animals in tropical waters (Sukumaran & Neelakantan, 1997). *Scylla olivacea,* also found throughout the Indo-West Pacific, has a life cycle restricted largely to estuarine habitats, especially mangroves (Hill, 1950; Le Vay, 2001), excluding the spawning phase when females migrate offshore (Hyland et al., 1984). More intensively studied, this species' response to fishing pressure is relatively well understood (Overton et al., 1997; Le Vay, 2001; Moser et al., 2005; Walton et al., 2006). The ecological sustainability of all these crab fisheries is causing concern, and a recent study in the Gulf of Thailand presents evidence of severe overfishing (Kunsook et al., 2014) for *Portunus* spp. However, current estimates of overfishing are based on a decline in the landed size of crabs rather than regional stock assessments and an understanding of catch rates in the fisheries.

In Thailand, mangroves are governed by the state and, on paper, are under strict protection in no-take Marine Protected Areas (MPAs). However, in practice, local fishing communities widely access and utilise these areas, and in response to declining resources, some communities have established their own MPAs to conserve and enhance their local fishing resources (Sudtongkong & Webb, 2008). MPAs are a spatial management tool advocated as a 'simple yet elegant solution' (Jentoft et al., 2007) for ecosystem protection and fisheries management (Lauck, T et al., 1998; Roberts, 2009; Roberts et al., 2005) and the international community has set global targets for 10% of the marine environment to be protected within MPAs by 2020 (CBD, 2010). The ecological effectiveness of MPAs to meet the twin objectives of fisheries enhancement and biodiversity protection is, however, a matter of controversy, with critics claiming that there is limited empirical evidence that supports their widespread adoption over other forms of marine management (Hilborn, 2004; Kearney et al., 2013; Caveen et al., 2013; Caveen et al., 2012; Willis et al., 2003). While this 10% target is expected to be implemented by the *state* [national] primarily to protect marine biodiversity, *local* [community] efforts do contribute, but tend to prioritise objectives specific to livelihoods, such as the management of commercially important fish species. It is believed that involving local communities in the management of MPAs can potentially increase their effectiveness through increased compliance with the MPA regulations (Pimbert & Pretty, 1995; Charles & Wilson, 2009; Christie & Pollnac, 2011), though there is limited empirical evidence to validate this belief (Agardy et al., 2011).

In this paper, we examine the management effectiveness of two community-managed MPAs and one co-managed MPA by quantifying these reserves' effects on indicators of crab productivity i.e. catch per unit effort and size composition, of three commercial species: *Portunus pelagicus; P. sanguinolentus* and *Scylla olivacea.* The composition of the other associated aquatic fauna was also studied to determine whether these types of MPA management can provide wider biodiversity conservation benefits.

2. Methodology

2.1 Study sites

The study area spanned the Andaman Sea provinces of Phang-nga and Ranong on the southwest coast of Thailand, extending along 60 km of coastline from the Kapoe Estuary (9º 35´N, 98º 30´E) to the southern point of Koh Phra Thong Island (9º 02´N, 98º 20´E). Within this study area, two estuarine MPAs, with associated control sites, hereafter called 'estuarine', and one inshore marine MPA, with an associated control site, hereafter called 'marine', were repeatedly surveyed over two seasons.

Site 1 – Tha Yang MPA. An estuarine MPA, protecting secondary mangrove forests along the main canal. This area is a community-managed, permanent no-take zone (NTZ) which was established in 2007 to protect the commercially important *Portunus* population. The MPA covers 1.2 km² of mangroves and canals with a depth range between 2 to 5 metres. I It was established by the community and has been under community protection for four years. Reports from stakeholders suggest that there is no fishing pressure in this MPA due to high compliance with the regulations and support from both surrounding fishing villages. No fishing activity was observed during the site surveys.

Site 2 – Thung Nam Dam MPA. An inshore marine MPA of sea grass beds and coastal beach forests which front a large mangrove system. This area is under co-management between local communities and the Department of Marine and Coastal Resources (DMCR), with international NGO support and was established after consultation with the community in 2006. It is a seasonal no-take zone, 3 km² in size with depth ranges from 4 to 9 metres, established to protect sea grass habitat, associated flagship species (dugongs and sea turtles) and *Portunus* species. The seasonal closure is from 1st Feb - 1st May and all fishing gears are restricted under a voluntary scheme.

Reports from stakeholders suggest that there has been only a small reduction in fishing pressure due to low compliance with the MPA regulations because of conflict on how the MPA was established. However, no fishing activity was observed during the site surveys.

Site 3 - Bang Ban Tip MPA. An estuarine MPA, protecting secondary fringing and interior mangroves. This area is a community-managed, gear limitation zone (GLZ), which has a ban on box traps and gillnets (only mullet nets are allowed), and landing restrictions on under-sized crabs (<100 mm carapace width). This GLZ was established in 2009 by the community to recover and protect the commercially important *Scylla olivacea* and *Portunus* species. The area consists of 3 km^2 of mangroves and canals, with depths ranging from 1 to 5 metres; it has been under community protection for three years. Reports from stakeholders suggest high compliance with the MPA regulations, enforced by the community and local fisheries officers, with intermittent poaching by fishers from outside the village who oppose the MPA. No fishing activity was observed during the site surveys.

For each MPA, comparable fishing grounds or control sites were identified for comparative analyses because there were no pre-management baseline data for any of the MPAs. Control sites were selected by prioritising depth, habitat and distance from the MPAs, the latter being no closer than 2 km to minimise bias through spill-over, but no further than 3.5 km to reduce area effect on species composition. Figure 1 presents these three MPAs and their three associated controls.

Three fishing teams surveyed each MPA and the associated controls on the same day and the same tide. Each pair of sites were sampled with two sets of gears commonly deployed by local fishers: 1) 100 mm mesh bottom-water gill nets used to target *Portunus* spp. and; 2) 25 mm mesh box traps used to target *Scylla* and *Portunus* spp. (see Appendix A for gear dimensions). For crab nets, each fishing team deployed two nets 400-600 metres apart per site. For *Portunus* box traps each site was sampled with four lines of gear and dropped 200-300 metres apart. Box traps targeting *Scylla* sp. were buoyed and clustered in groups of seven in four sample areas ~200 metres apart in the small canals in the mangroves. After the gears were retrieved, each gear was lifted, labelled and either processed on the boat (traps) or wrapped for clearing back on shore (nets). Each animal caught was identified, sexed (if crab), measured (mm) and weighed (g) before being returned to the fishers.

To standardise fishing gear and mirror local fishing practices, the same gear set-up used by local fisheries was employed in the survey design and local fishers assisted with their deployment. To standardise fishing duration, all teams followed a deployment schedule which aimed to submerge the gill nets and *Portunus* traps for ~22 hours and the *Scylla* box traps for ~3.5 hours. These reflected the deployment times of the local fishers, so the results are comparable with other fisheries data. All surveys were conducted mid-way between neap and spring tides, and systematic sampling was applied - i.e. a random starting point in a pre-defined area was selected and then the gears were placed at regular intervals from that point to avoid variations or significant changes in depth contours and habitat. Eight replicates of each survey were completed, four in the wet season, from May to July 2011; and four in the dry season, from February to March 2012.

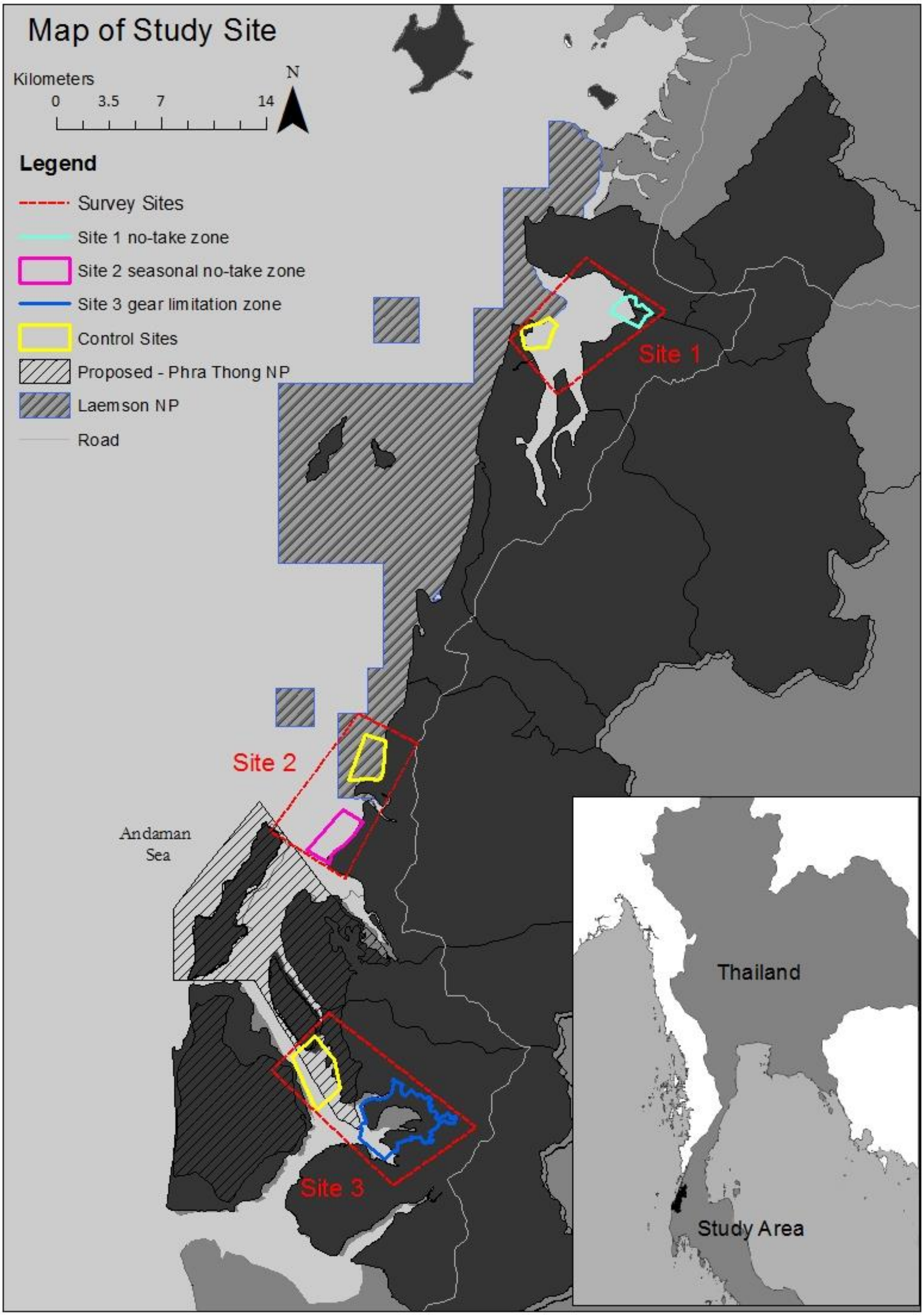

Figure 1: Maps of study sites

2.2 Data Analyses

Multivariate statistical analyses were carried out using PRIMER v6 (PREIMER-E, Plymouth) to assess differences in benthic aquatic community assemblages. Multi-Dimensional Scaling (MDS), using a

Bray-Curtis similarity measure was applied to plot community assemblages on square-root transformed data, as both dominant and rare species were deemed important to the analyses. MDS plots were conducted on net data only, as this was the only gear that sampled the fuller range of benthic species; the trapping methods used bait and therefore biased results toward carnivorous/scavenger species. Plots were produced which allowed comparison across all survey areas and between the MPAs and their control sites. Permutational analysis of variance (PERMANOVA; Anderson 2001) using a pair-wise Bray-Curtis similarity measure was used to test for significant difference between all areas and between each type of MPA and its associated control site. The analyses were run on factors assigned to benthic aquatic community composition, which allowed identification of which factors differed significantly from each other - e.g. type of habitat or form of protection. Due to a small sample size when divided into factors, PERMANOVA's Monte Carlo tests, which use chi-square variables combined with eigenvalues to construct the asymptotic permutation distribution, were used to conduct post hoc pair-wise *t*-tests to compare similarities in benthic assemblages between sites which were: 1) protected or unprotected and; 2) under MPA management. This was conducted for both seasons independently and then repeated on the combined data to look for differences between seasons.

Catch Per Unit Effort (CPUE) was calculated by unit of net and units of traps for all gears and Mann-Whitney U tests were used to compare differences in catch rates between the MPAs and their associated control as the data did not fulfil the requirements of parametric normality on the biomass and abundance variables. Median catch rates and range were quantified by unit, which was 'per panel' = 80 metres for gillnets and 'per trap' for box traps. Size frequency distribution were derived from both gillnets and *Portunus* box traps to determine whether there was an effect of management on the size distribution of the two dominant *Portunus* species.

3. Results

The results are divided into three sections: 1) an analysis of benthic species caught in gill nets in protected (MPAs) and non-protected (control) sites to assess benthic aquatic community composition for contributing towards conservation objectives; 2) an analysis of CPUE from both traps and gill nets between the MPAs and their associated controls to assess whether they contribute to local people's fisheries objectives and; 3) an analysis of the size distribution of the target species in the MPAs and their associated control areas to assess the size of target species for meeting local people's fisheries objectives.

3.1 Conservation Objectives

Comparison of Benthic Assemblages: The total number of animals caught over the survey period was 4,735: 2,764 in the bottom-water gill nets and 1,968 in the box traps . Dominant groups of animals included Decapoda and Stomatopoda (crayfish, lobster, crab, prawn and shrimp); Perciformes and Tetraodontiformes (bony fish); Pleuronectiformes (flatfish); Neogastropoda (marine snails) Anguillifomes (eels); Aulopiformes (lizardfish); Carcharhiniformes, Myloibatiformes and Torpedinifrormes (sharks and rays); and Siluriformes (catfish). Ordination plots using MDS were created from all surveys to compare benthic species assemblages caught in gill nets (2,764) in the three study areas over both seasons (Fig. 2). They show that the benthic aquatic species composition found in the estuarine and marine sites are dissimilar, while aquatic species composition found in the same habitats are clustered with high levels of similarity. Similarity is measured by the distance between two points on the plot - those that are closest together are most similar and cluster at a higher percentage; while those that are furthest apart are the most dissimilar and cluster at a much lower percentage. Distinctions were found between the two seasons in the marine sites, with high levels of clustering in the dry (open circles) and wet season date points (filled circles). The aquatic species composition in the estuarine sites was interspersed between seasons (squares), which indicates that estuarine aquatic species composition is quite stable throughout the year, whereas marine aquatic species composition shows some seasonal variation.

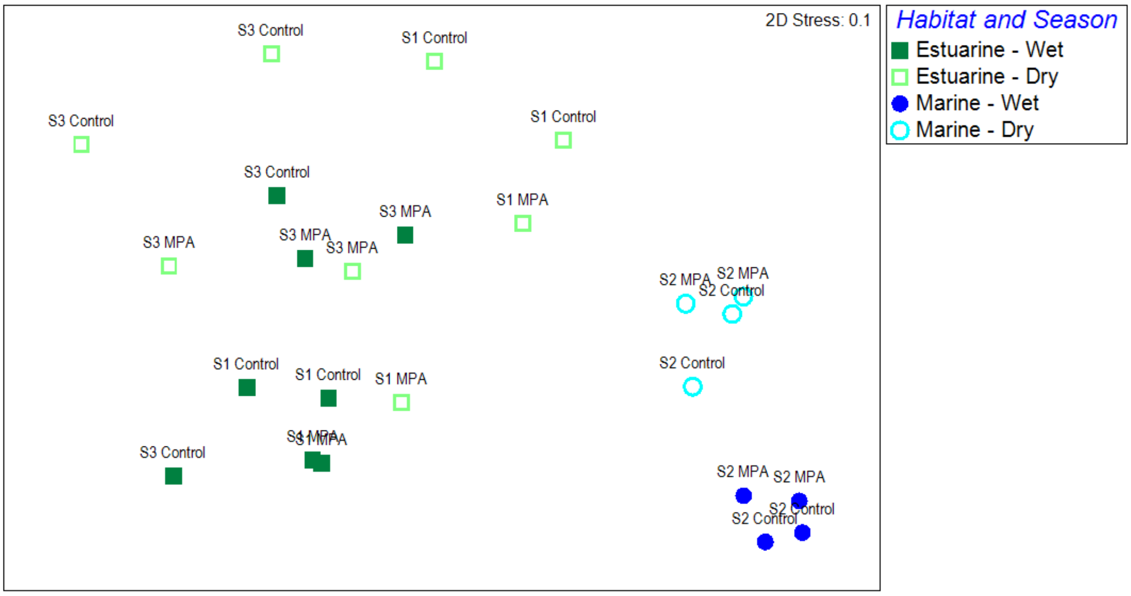

Figure 2: Two-dimensional non-metric Multi-Dimensional Scaling (MDS) plots of square-root transformed species abundance caught using gillnets during both seasons. Grouped by site (S) and displayed by MPA and control. Open circles = marine sites (site 2) during the dry season; closed circles = marine sites (site 2) during the wet season; open square = estuarine sites (sites 1 and 3) during the dry season; closed square = estuarine sites (sites 1 and 3) during the wet season.

Separate ordination plots for each season show dissimilarity between habitats which is much more pronounced in the wet season (Fig. 3a) than in the dry (Fig. 3b). Replications at each MPA and their associated controls show clustering with high levels of similarity, which would imply that site effect is having more impact than management on the composition of benthic aquatic species. In other words, the benthic species assemblages similarities are driven by environmental parameters and habitat, not because of reduced fishing pressure due to management intervention. This finding appears stronger in the wet season (Fig. 3a), where almost all surveys in each MPA are clustered with their associated controls, but dissimilar from the other sites, excluding the control site for the gear limitation zone (S3). The plot from the dry season (Fig. 3b) shows much more similarity between the marine and estuarine sites, and less clustering between the MPAs and the controls at each site.

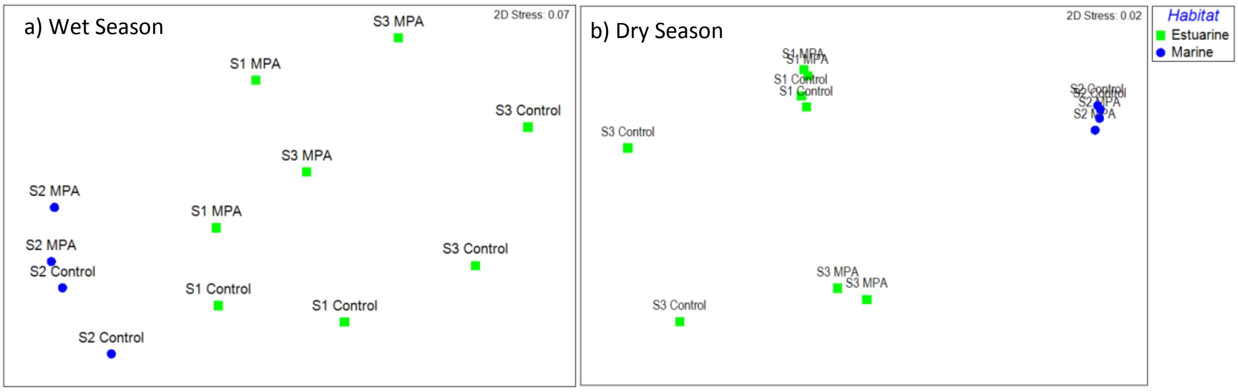

Figure 3: Two-dimensional non-metric Multi-Dimensional Scaling (MDS) plot of square-root transformed species abundance caught by gillnets during the wet season (a) and dry season (b) grouped by site (1 to 3) and whether it is from the MPA and control area.

PERMANOVA main tests (*F*) for the wet season revealed significant differences between habitats (*P* <0.001) but not between sites under MPAs and no protection (Table 1). Pair-wise *t*-tests (*t*) were performed on habitat which, again, found significant differences between estuarine and marine sites (*P* = 0.002), and post-hoc Monte Carlo tests resulted in significant difference being attributed to habitat as all significant p (MC) values were between the MPA and the control sites in the opposite habitats rather than between the MPAs and their comparable control site within the same habitat.

PERMANOVA main tests for the dry season also revealed significant differences between habitats, (F = 7.4057, *P* = 0.004) but, again, not between MPA and the areas with no protection. Pair-wise *t*-test showed similar patterns as the previous season with significant differences between estuarine and marine sites (t = 2.7213, *P* = 0.003), and Monte Carlo tests once again showed differences between survey areas in different habitats rather than from the MPAs and the controls in the same habitat (Table 1).

Table 1: Result of PERMANOVA main tests (F) and post hoc pairwise t-tests (t) on square-root transformed abundance data for crab assemblages sampled by bottom water gillnets. Results are based on Bray-Curtis similarity measures and main P-values were obtained using 999 permutations. Post hoc P-values were obtained using Monte Carlo test as sample size was insufficient and too few unique permutations existed on all samples.

Wet Season	df	MS	F	*P*	perms
Habitat	1	11308	16.266	**0.001**	858
Protection	1	690.45	0.437	0.746	861
Groups	df	*t*	*P*	p (MC)	perms

Estuarine + Marine	12	4.0331	**0.002**		854
Site 1 + Control	4	1.1375		0.332	15
Site 3 + Control	4	1.3563		0.196	15
Site 2 + Control	4	0.8099		0.515	15
Dry Season	df	MS	F	*P*	perms
Habitat	1	6214.8	7.4057	**0.004**	427
Protection	1	1770.3	1.3791	0.239	417
Groups	df	*t*	*P*	p (MC)	perms
Estuarine + Marine	10	2.7213	**0.003**		436
Site 1 + Control	2	1.3228		0.204	15
Site 3 + Control	4	1.2003		0.278	15
Site 2 + Control	2	0.9764		0.448	3

Bold indicates significance at p <0.05

To conclude, the ordination plots indicate that aquatic community composition is similar in the MPAs and their associated control sites. This suggests that the survey areas are suitable for comparison of CPUE and the size distribution of target crab species because they support similar species composition, so any differences in abundance of such species could be due to MPA management measures. Statistical tests did not reveal any significant difference between each MPA and its associated control site for benthic aquatic species composition, but species composition differed significantly in relation to habitat type. Thus, it appears that MPA management is not having any conservation impact in relation to benthic aquatic biodiversity.

3.2 Fisheries Objectives

Fisheries objectives are related to target species protection, including vulnerable life stages. This section analyses CPUE and size composition of the three target crab species (*P. pelagicus, P. sanginolentus* and *S. olivacea*), plus three other species, two *Charybdis* and a third *Portunus* species that are caught and retained as valuable by-catch.

Gill net: CPUE of target crab species for site 1, the no-take MPA, and site 3, the gear limitation MPA (both community-led MPAs), were higher in the MPAs than in their associated controls for target biomass and target abundance (Fig. 4c and d). For all aquatic species (Fig. 4a and b) this was also the case, apart from in site 3 in the wet season when the total biomass was higher in the control compared with the MPA.

Mann-Whitney U test between the MPA and its control area revealed that site 1 was the only site with significant differences between the MPA and its associated control for all four indicators in the wet season - biomass all species (Z = 2.117, U = 22, *P* = 0.034), abundance all species (Z = 2.080, U = 22.5, *P* = 0.037), biomass target species (Z = 2.268, U = 20, *P* = 0.023) and abundance target species (Z = 2.587, U = 16, *P* = 0.010) (Table 2). At this site, there was almost double the biomass per unit in the MPA, and CPUE was almost twice the abundance of all species and target species in the MPA compared to the control site (Fig. 4). There was also a significant difference found in the dry season between the site 1 MPA and the control, but only for two of the indicators, total biomass all species

(Z = 1.995, U = 13, P = 0.046) and abundance all species (Z = 2.261, U = 10.5, P = 0.024) again at double the amounts being extracted from the MPA compared with the control. Whilst not significant, higher values were recorded for biomass target species and abundance target species in both estuarine MPAs compared with their control areas (Fig. 4c and d).

The gear limitation MPA (site 3) did not show any significant differences for the indicators between the MPA and the control in the wet season, but significant differences were observed in the following dry season for biomass target species (Z = 2.317, U = 8, P = 0.021) and abundance target species (Z = 2.034, U = 10.5, P = 0.042) with higher median values in the MPA (Table 2 and Fig. 4).

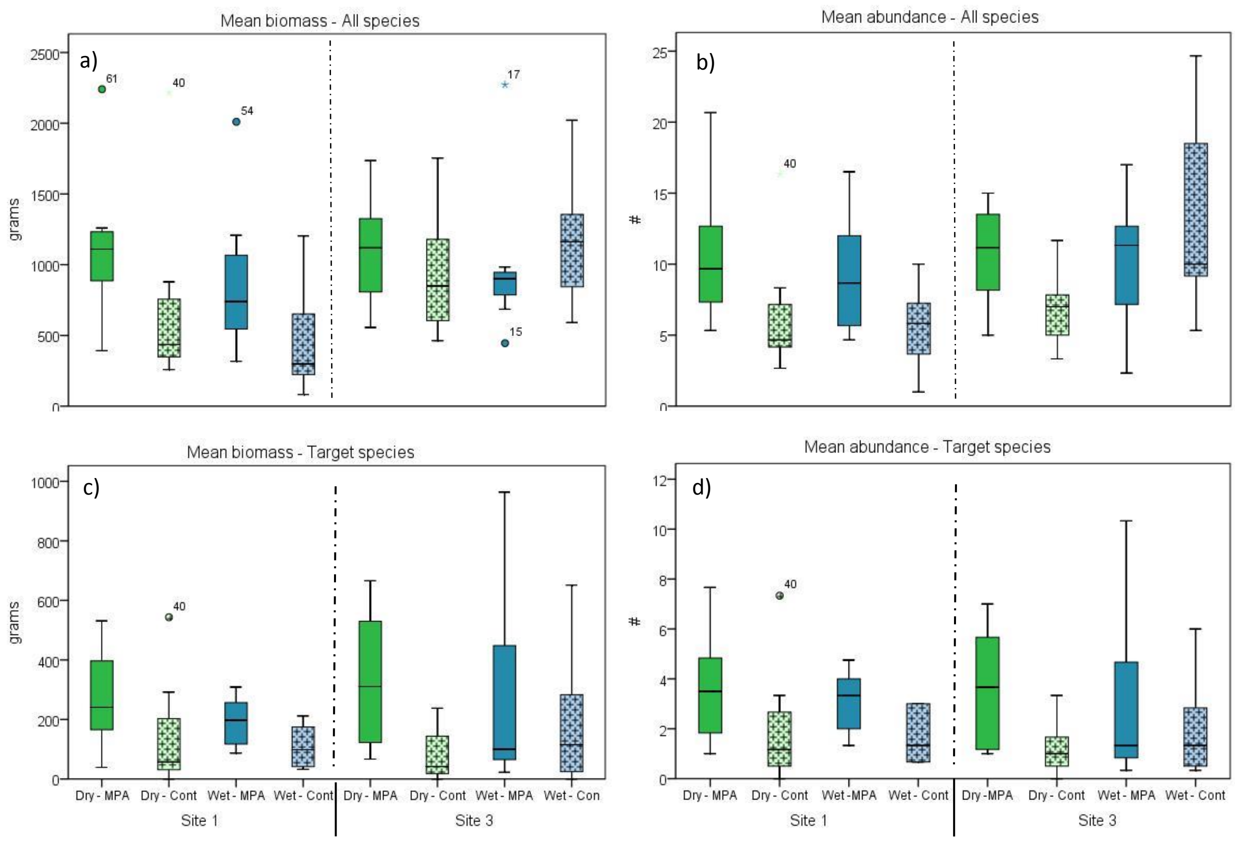

Figure 4: Box plots of gill net CPUE for Site 1 - the no-take MPA managed by the local community – and Site 3 the gear limitation MPA also managed by the local community. Green plots show CUPE during the dry season and blue plots show CUPE during the wet season. Solid plots are the MPAs, dotted plots are the controls.

Table 2: Median catch rates per unit effort with range from all three fishing gears. Unit effort is per panel (80m) for gill nets and per trap for *Portunus* traps and *Scylla* traps and weight is in grams (g). Significance was tested by Mann-Whitney U tests between the MPAs and control areas. * indicates significance at p< 0.05 ** at p< 0.01

	Indicators	Total biomass per panel (g)		Total abundance per panel (#)		Target biomass per panel (g)		Target abundance per panel (#)	
		Md	MW	Md	MW	Md	MW	Md	MW
Estuarine Site 1 – No-Take Zone									
Wet - MPA	Gill Nets	739.6 ± 1692		8.6 ± 11.8		197.4 ± 221.9		3.3 ± 3.4	
Wet - control	Gill Nets	299.1 ± 1121	*(22)* ***2.117****	5.8 ± 9.0	*(22.5)* ***2.080****	98.1 ± 179.3	*(20)* ***2.268****	1.3 ± 2.3	*(16)* ***2.587*****
Dry - MPA	Gill Nets	1109 ± 2848		9.6 ± 15.3		240.9 ± 492.3		3.5 ± 6.7	
Dry - control	Gill Nets	436.1 ± 1958	*(13)* ***1.995****	4.6 ± 13.6	*(10.5)* ***2.261****	57.4 ± 543.4	*(17) 1.575*	1.2 ± 7.3	*(14.5) 1.841*
Wet - MPA	Portunus Trap	37.3 ± 114.2		0.8 ± 2		16.8 ± 64.8		0.3 ± 1.3	
Wet - control	Portunus Trap	27 ± 88.7	*(166) .920*	0.6 ± 1.7	*(151)1.318*	14.8 ± 74.2	*(181) .514*	0.3 ± 1.1	*(192.5) .205*
Dry - MPA	Portunus Trap	46.6 ± 146.2		0.9 ± 2.2		32.3 ± 163.2		0.6 ± 2.5	
Dry - control	Portunus Trap	28.4 ± 64.3	*(65)* ***2.375*****	0.6 ± 1.4	*(52.5)* ***2.858*****	27.2 ± 40.9	*(82.5) 1.720*	0.3 ± 0.8	*(60)* ***2.577*****
Wet - MPA	Scylla Trap	53.7 ± 205.4		1.0 ± 4.3		0.0 ± 47.5		0.0 ± 0.3	
Wet - control	Scylla Trap	52.5 ± 132.3	*(173) .478*	1.3 ± 4	*(161.5) .914*	0.0 ± 37.9	*(174) .451*	0.0 ± 0.3	*(186.5) .114*
Dry - MPA	Scylla Trap	37.6 ± 124.8		1.0 ± 2.9		0.0 ± 20.3		0.0 ± 0.1	
Dry - control	Scylla Trap	48 ± 293.6	*(105) .867*	1.2 ± 5.7	*(112) .885*	0.0 ± 33.1	*(117.5) .397*	0.0 ± 0.1	*(112.5) 1.018*
Marine Site 2 – Seasonal No-Take Zone									
Wet - MPA	Gill Nets	1203 ± 2286		17.5 ± 97		173.3 ± 775		3.7 ± 12	
Wet - control	Gill Nets	1026 ± 3194	*(26) .630*	18.7 ± 78.7	*(27.5) .473*	194.8 ± 845.5	*(32) .001*	4.5 ± 10.7	*(28.5) .369*
Dry - MPA	Gill Nets	638.5 ± 1678		8.7 ± 17		232.5 ± 1128		2.8 ± 10.7	
Dry - control	Gill Nets	1130 ± 2321	*(24) .840*	8.3 ± 21.3	*(29) .316*	171 ± 542	*(27) .525*	2 ± 5	*(27) .527*
Wet - MPA	Portunus Trap	58 ± 82		1.4 ± 0.5		42.5 ± 73.5		0.9 ± 1.1	
Wet - control	Portunus Trap	73.6 ± 112.9	*(101) .309*	1.8 ± 0.8	*(85.5) 1.605*	52.6 ± 111.8	*(99) 1.093*	1.1 ± 2.5	*(76)* ***1.968****
Dry - MPA	Portunus Trap	46.5 ± 68.8		1.2 ± 0.5		37.5 ± 64.5		0.8 ± 1.6	
Dry - control	Portunus Trap	48.2 ± 115.4	*(124) .151*	1.2 ± 0.6	*(101) 1.018*	42.1 ± 105.9	*(110.5) .661*	0.7 ± 1.8	*(120) .302*
Wet - MPA	Scylla Trap	51.5 ± 299.2		1.1 ± 4.9		0.0 ± 45		0.0 ± 0.4	
Wet - control	Scylla Trap	43 ± 166.2	*(113) .565*	1.2 ± 3.3	*(109) .845*	0.0 ± 54.5	*(123) .189*	0.0 ± 0.3	*(100) 1.255*

Table 2 Continued.

	Gear	Total biomass per panel (g) Md	MW	Total abundance per panel (#) Md	MW	Target biomass per panel (g) Md	MW	Target abundance per panel (#) Md	MW
Estuarine Site 3 – Gear Limitation Area									
Wet - MPA	Gill Nets	900.7 ± 1827.4		11.3 ± 14.7		100.1 ± 940.8		1.3 ± 9.9	
Wet - control	Gill Nets	1162 ± 1430	*(17) .958*	10 ± 19.3	*(19) .704*	115.3 ± 651.3	*(21) .448*	1.3 ± 5.6	*(19.5) .645*
Dry - MPA	Gill Nets	1119 ± 1179		11.2 ± 10		310.8 ± 599.2		3.6 ± 6.0	
Dry - control	Gill Nets	849.8 ± 1291	*(22) .694*	7 ± 8.3	*(12) 1.852*	42.2 ± 237.9	*(8)* ***2.317****	1.0 ± 3.3	*(10.5)* ***2.034****
Wet - MPA	Portunus Trap	38.1 ± 145.3		0.5 ± 1.5		25.4 ± 66.4		0.2 ± 0.8	
Wet - control	Portunus Trap	21.3 ± 65.9	*(90.5) .894*	0.4 ± 0.9	*(77.5) 1.444*	11.3 ± 44.8	*(76) 1.517*	0.1 ± 0.3	*(61)* ***2.169****
Dry - MPA	Portunus Trap	65.3 ± 112.9		1.0 ± 3.4		53.5 ± 94.4		0.7 ± 1.8	
Dry - control	Portunus Trap	40.9 ± 75.6	*(84) 1.658*	1.0 ± 2.3	*(120) .302*	26 ± 69.3	*(81) 1.777*	0.2 ± 0.5	*(60.5)* ***2.571*****
Wet - MPA	Scylla Trap	55.9 ± 240.7		1.2 ± 5.6		8.3 ± 79.1		0.1 ± 0.6	
Wet - control	Scylla Trap	77.4 ± 95.3	*(111) .641*	1.6 ± 4	*(64)* ***3.171*****	0.0 ± 0.0	*(75)* ***2.003****	0.0 ± 0.0	*(64)* ***3.182*****
Dry - MPA	Scylla Trap	121 ± 248		2.5 ± 4.7		12.1 ± 63.3		0.1 ± 0.6	
Dry - control	Scylla Trap	100.2 ± 162	*(94) 1.281*	3.2 ± 4.1	*(88) 1.635*	0.0 ± 39	*(73)* ***2.076****	0.0 ± 0.1	*(88) 1.706*

For the marine site, the seasonal no-take MPA (site 2), CPUE showed no significant differences between the MPA and associated control for any of the indicators measured, but the control area had the highest CPUE for total biomass (Table 2). Box plots of the marine sites displays higher biomass and abundance in the control site than the MPA in the wet season (Fig. 5) and mixed between the MPA and the control in the dry season.

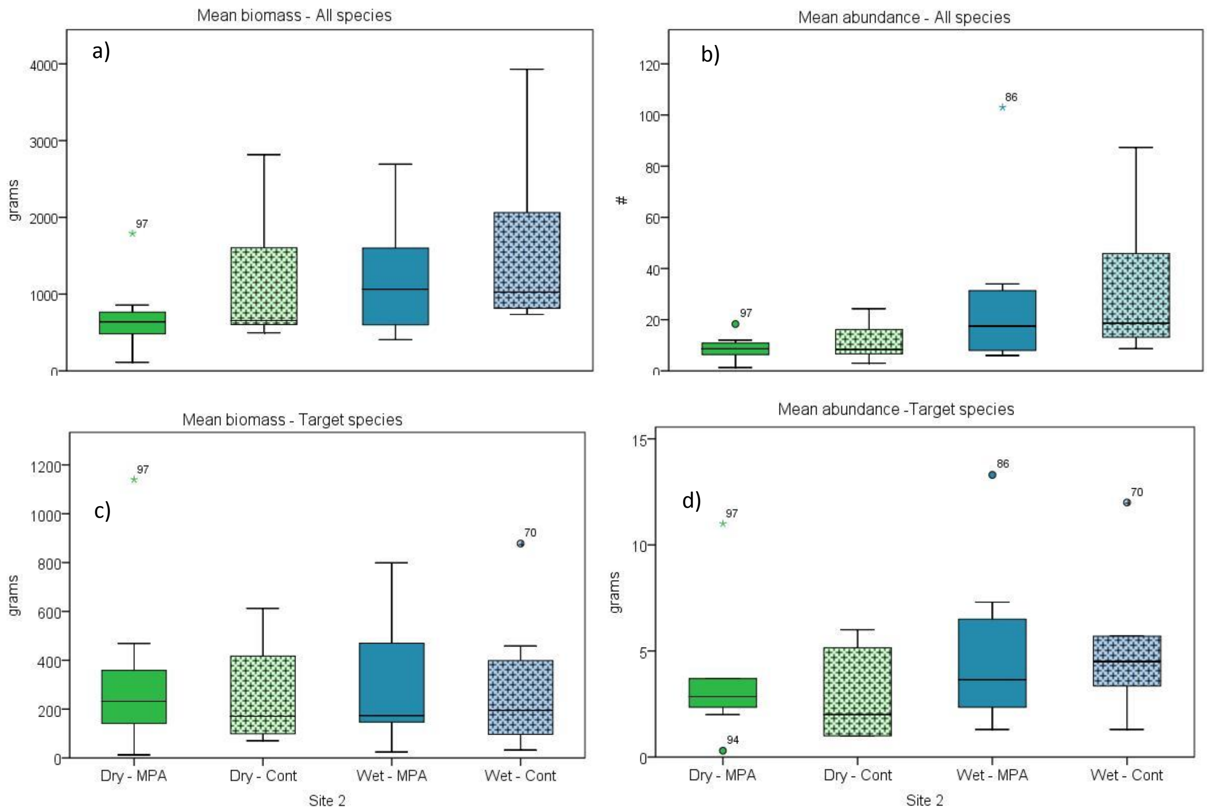

Figure 5: Box plots of gill net CPUE for Site 2 - the seasonal no-take MPA co-managed by the local community and local government. Green plots show CUPE during the dry season and blue plots show CUPE during the wet season. Solid plots are the MPAs, dotted plots are the controls.

Portunus Traps: CPUE - In the two estuarine sites (sites 1 and 3) there was no significant difference between the four indicators for the MPAs and their associated controls during the wet season (Table 2), apart from site 3, the gear limitation area, which had a significant difference between abundance of target species caught, with the higher number in the MPA ($Z = 2.169$, $U = 61$, $P = 0.030$). For the dry season, highly significant differences in CPUE were found at site 1 for three of the four indicators - biomass all species ($Z = 2.375$, $U = 65$, $p = 0.018$), abundance all species ($Z = 2.858$, $U = 52.5$, $P = 0.004$) and abundance target species ($Z = 2.577$, $U = 60$, $P = 0.010$), with a higher CPUE in the no-take MPA compared to the control (Table 2). For the marine site (site 2), one indicator showed a higher abundance of target animals ($Z = 1.968$, $U = 76$, $P = 0.049$) with the higher number found in the control site. This pattern was reflected in the rest of the indicators, with higher CPUE in the control site than the seasonal no-take MPA (Table 2).

***Scylla* Traps: CPUE** - The only site that showed a significant difference was the gear limitation area (site 3). For the wet season, three of the four indicators were significant, the first - abundance all species ($Z = 3.171$, $U = 64$, $p = 0.002$) - was found to have the highest proportion in the control site, in contrast to biomass of target species ($Z = 2.003$, $U = 75$, $p = 0.045$) and abundance of target animals ($Z = 3.182$, $U = 64$, $p = 0.001$) with the higher median found in the MPA (Table 2). This supports the management objective to recover and protect the *Scylla* stocks in this managed area. This pattern was repeated again in the dry season, but only one indicator - abundance of target animals ($Z = 2.076$, $U = 73$, $p = 0.038$), was significant. However, due to very low catch rates for *Scylla olivacea* (only 77 individuals were caught during these surveys) the data should be treated with caution in terms of any inferences drawn.

Size Distribution of Target Species: Size distribution plots of *P. pelagicus* in both estuarine sites show higher abundance of all sizes classes below 140 mm in the MPAs (solid line) compared to the control (dotted line). Catch rates of animals over 140 mm in width were low at all sites (Fig. 6).

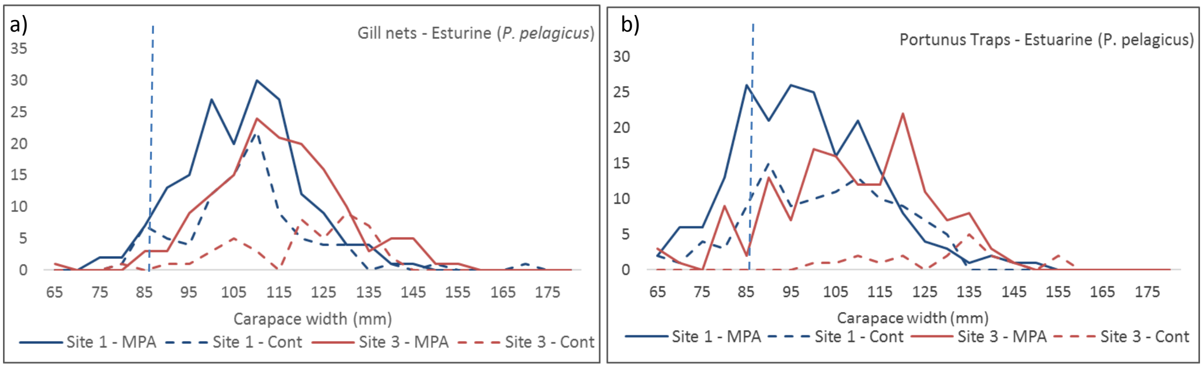

Figure 6: Size distribution of *Portunus pelagicus caught* from gillnets and traps in the estuarine sites. Dotted line represents minimum size at maturity for females (85-90 mm) and males (80-90 mm). Solid line = MPA, dotted line = control.

Plots of *P. pelagicus* in the marine sites (Fig. 7) had higher abundance in all size classes up to 135 mm carapace width for both gears in the MPAs. Two size groups appear for *P. pelagicus* in traps from site 2 (Fig. 7b), possibly indicating two moult periods; however caution is required due to the small sample sizes involved.

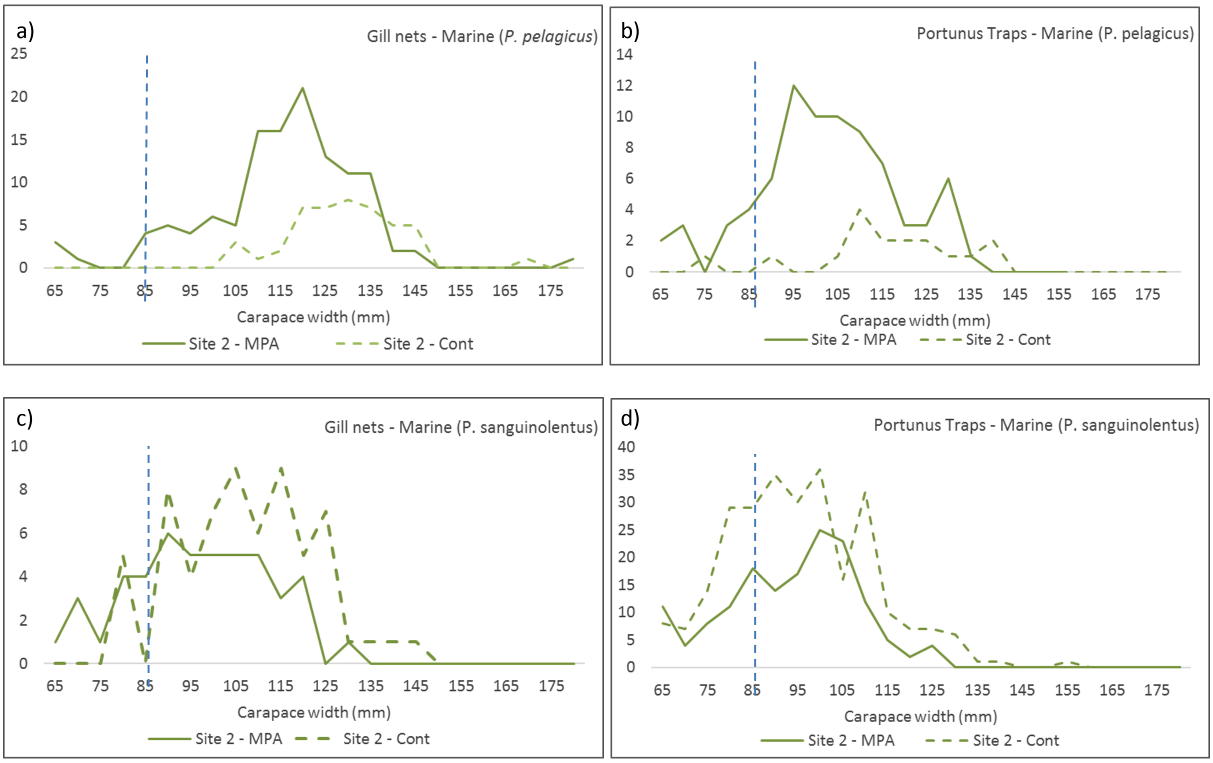

Figure 7: Size distribution of *Portunus pelagicus* (a, b) and *Portunus sanguinolentus* (c, d) from gillnets (a, c) and traps (b, d). Vertical dotted line represents minimum size at maturity for female and male is 85-90mm and 80-90mm respectively. Solid line = MPA, dotted line = control.

For *P. sanguinolentus* (Fig. 7c, d) which was absent in the estuarine sites, potentially due to its lower intolerance to low salinity levels, higher abundance and larger sized animals were recorded in the control site for both gillnets and box traps. These species were more abundant in the traps compared with abundance level recorded for those caught using gill nets.

Mann-Whitney U tests found significant differences for *P. pelagicus* ($Z = 2.089$, $U = 33161$, $p = 0.037$) with the smaller median size of animals caught in site 1 (*Md* - MPA 99.5 mm, control 103 mm) and site 3 ($Z = 4.859$, $U = 5968$, $p > 0.001$) in the MPAs (*Md* - MPA 110.2 mm, control 121.1 mm) compared to their controls. Both sites were in the estuarine areas. This was also the case for *P. pelagicus* in site 2 ($Z = 5.639$, $U = 5786$, $p > 0.001$) in the inshore marine area, with smaller median size for animals in the MPA compared to the control (*Md* - MPA 110.3 mm, control 122 mm). No significant differences were found for *P. sanguinolentus* ($Z = 1.904$, $U = 30178$, $p = 0.057$) with similar median sizes of animals in the control and the MPA (*Md* - MPA 93.8 mm, control 94.9 mm).

4. Discussion

This section discusses how far the three different MPA management regimes met (1) fisheries objectives; and (2) other conservation objectives.

4.1 Meeting Fisheries Objectives

The CPUE findings support the hypothesis that community-led MPAs can have a significant positive effect on target species (*Portunus* spp.) abundance. The CPUE from the community-led no-take MPA (site 1) showed higher medians for all indicators in the MPA compared with the control in both seasons. Of these, the differences for the gill nets surveys were significant for six of the eight indicators, and for *Portunus* traps surveys three indicators were significant. The second community-led MPA - the gear limitation MPA (site 3) - showed higher medians on almost all CPUE variables from both gill nets and *Portunus* trap surveys in the MPA compared with the control, but only four were significant. In a number of cases the target species biomass and abundance were double in this community-managed MPA compared with the control except for gill nets in the wet season. This result passes the criteria proposed by Willis *et al.* (2003) for only inferring an effect if there is > 100% increase in values between MPAs and control sites.

In the third case - the seasonal no-take MPA (site 2) - which was co-managed, no significant differences were found, and therefore the MPA is not considered to be having any effect on target species abundance. In this site, only one indicator out of the eight showed a significant result with higher median abundance indicated in the control site. For almost all variables, the control site had higher median CPUE than the MPA in the wet season. This altered for the gill net surveys during the dry season, which coincides with the closed period, when target species variables showed higher medians in the MPA, but this was not the case for the *Portunus* trap surveys and no variable was significant.

Five factors could explain this poor performance by the seasonal no-take co-managed MPA (site 2). First, internal conflict occurred in the communities involved in the MPA management over the use of different fishing gear (Jones, 2014), and this could have exacerbated non-compliance. Second, a reported lack of consultation with displaced fishers when the MPA was established could have led to a backlash against MPA regulations by local people. This was not the case in the two community-managed MPAs, which both enjoyed wide support within their respective communities. Third, there was no legal framework to exclude fishing gear from the site area. Fourth, the seasonal nature of this MPA, with limited time for species recovery, may be ineffective: found seasonal non-take zones are less effective than permanent protection. Fifth, natural variability in productivity in this highly energetic marine environment could have resulted in higher abundance and biomass in the control area. These marine sites were more dissimilar than the estuarine sites in the ordinate plots (Fig. 3) suggesting that aquatic communities in site 2 and its control were different and the two survey areas supported different compositions of animals. This highlights the challenges of determining confounding factors in site selection for MPA assessments (Willis *et al*, 2003) and the difficulty of isolating effects of management.   Whilst we recognise these challenges, the MDS plots do demonstrated that aquatic community composition was similar between the first and third MPAs and their respective control sites, which suggests that   it is sound to infer that the higher abundance observed in the community-led MPAs (site 1 and 3) is due to management. This conclusion is reinforced by higher abundance being observed in these two estuarine MPAs over the two seasons.

We would therefore argue that the community-led estuarine MPAs in sites 1 and 3 are showing positive effects for target species in the MPA, and that the findings should be used as an indication of the value of community-led MPAs. However, a lack of a suitable baseline before management was implemented, and the difficulty of achieving appropriate sampling to get sufficient power in the analysis, lead us to be cautious in interpreting these results.

A higher abundance of all size classes of *P. pelagicus* was observed in all three MPAs compared with their respective controls. In the two estuarine sites, the gear limitation MPA (site 3) had larger individuals than the no-take MPA (site 1). This is most likely due to the southern sites (site 3) being located in the vicinity of a large canal with a deep gully (~8 metres) which could have influenced the area by bringing in larger individuals, whilst the northern site (site 1) was more sheltered and at a constant depth. In the inshore seasonal no-take MPA (site 2), *P. pelagicus* had a higher abundance across all size classes but the opposite was observed for *P. sanginolentus*, where higher abundance was recorded in the control. This is most likely due to habitat, with the presence of seagrass beds in the MPA and harder rocky habitat in and around the control area. *P. sanginolentus* is known to favour rocker substrate (Lee & Hsu, 2003), therefore habitat type is most likely the key driver in size distribution at the site rather than management. Locality is an important factor in this study, because a key contributor to the success of community-managed MPAs is likely to be the locality of these MPAs, which are closer to the villages and therefore easy to monitor and enforce and support compliance with the rules, which appeared to be lacking in the co-managed MPA.

The gear limitation MPA (site 3) had a significantly higher CPUE for both biomass and abundance of *Scylla olivacea* in the MPA compared to the control. Whilst this result meets the management objectives of the community-managed MPA, overall catch rates of *S. olivacea* were low throughout both seasons, which is an indication that fishing pressure is influencing abundance. In total, 712 traps were deployed - with an in-water time of over 282 hours, but only 77 mud crabs were caught. Most of these individuals came from the gear limitation MPA, which was managed under rules designed by the community to protect and enhance this species. Other theories on why the catch rate could have been so low include reports that the 'catchability' of *Scylla* increased with higher temperatures and at time of darkness i.e. dawn and dusk, due to the mud crabs' increased mobility at these times (Miller, 1990). The surveys were conducted early to mid-morning which may have affected catch rates negatively. However, darkness did not affect catch rates in surveys on South African mud crabs (Robertson, 1989). Another possible explanation is that the sites were located more seaward, and potentially are less favoured by *S. olivacea,* which prefers denser mangrove habitat (Macintosh et al., 1991). Predation is an additional factor for consideration, as *S. olivacea* is targeted by crab-eating monkeys (Macaca spp). However, the most likely reason is overfishing. Catches in these surveys were dominated by *Thalamita* spp., which inhabits similar habitat to *S. olivacea,* competes for resources (Walton et al., 2007) and is also predated on by Macaca spp. The high abundance of *Thalamita* in the surveys suggests the habitats and conditions sampled are suitable for *S. olivacea*, but their numbers have declined because of sustained overfishing.

This size distribution data of all species from the MPAs suggest that the two community-managed MPA (sites 1 and 3) are supporting recruits, i.e. small, under-target sized crabs, and potentially contributing to the fisheries by reducing the rates of growth overfishing of juvenile species. “One of the better ways to replenish stocks and increase yield is to protect stocks from growth overfishing… Designating nursery areas as reserves can protect juvenile fish from by-catch if the species are relatively sedentary during juvenile stage” (National Research Council Working Group, 2001). Reducing fishing mortality in undersized animals in protected grounds can encourage spill-over in these animals, which is likely due to the life cycle of *P. pelagicus* to move in the shallow marine areas for spawning and then into deeper water when adults (Chande & Mgaya, 2003; Xiao, 2004). As these animals are not of direct interest to the crab fishery at this size, the costs to fishers from being displaced from these areas is low and potentially more acceptable. Currently the two community-managed MPAs tested in this survey are the only two in the region where the protection of undersized animals is reported to be functioning and enforced.

For recruitment overfishing, i.e. where adult spawning stocks are depleted to a level where the reproductive capacity can no longer replenish itself, there are no current estimates for *P. pelagicus* or *P. sanguinolentus* nor is there any spatial protection as no MPAs are fully functioning in offshore waters (sub-littoral shallow waters) which the spawning stocks inhabit (Chande & Mgaya, 2003). This leads to questions on the scale of current protection and what is needed as these animals at this size are of direct interest to the fishery. These questions are beyond the scope of this article, but the *Portunus* fishery is believed to be overfished and in need of wider management (Boutson et al., 2009).

4.2 Meeting Other Conservation Objectives

Results from the MDS analyses indicate there is no evidence to suggest that management regimes or measures are having any effect on the community assemblages of benthic species, at least in terms of reducing their exploitation by bottom water gill nets. Significant differences were found between the estuarine and marine habitats, but not between management regimes. This was found for both seasons. This would suggest that area effect is shaping benthic aquatic species assemblage, which is not being altered by any form of protection. This is not surprising as the vast majority of small-scale gears employed in the area are not known to have significantly negative impacts on habitat. Noticeable shifts in aquatic species composition (spatial distribution of species diversity and relative abundance) are more often than not associated with habitat loss (National Research Council 2001; Sainsbury et al 1997 in Bianchi, 2000) and whilst effects can be felt from functional group removal, such as herbivorous groups in the Caribbean in the 1980s (Hughes, 1994) and extreme fishing pressure, e.g. sustained trawling or push nets, generally shifts in aquatic community assemblages are more likely through habitat and environmental change rather than through the targeting , or cessation of targeting, of certain species (Skilleter & Warren, 2000; Wilson et al., 2008; Seytre & Francour, 2013).

Understanding the effects of fishing on aquatic species community composition is less developed despite the long history of fisheries management because of weak theoretical foundations and fluctuations in short-term assemblage, so long-term monitoring is required but often lacking (Seytre & Francour, 2013). This is especially the case for highly energetic environments such as mangroves and estuaries which may require more tailored monitoring to take account of the biophysical characteristics that distinguish these areas from each other, and therefore may need different management measures and regimes to achieve conservation objectives. So, whilst the data did not reveal any differences for conservation by management, the survey period undertaken in this study was short, so it is difficult to infer with certainty that local community management, and equally co-management, is having no effect on benthic aquatic species composition and highlights the challenges of measuring the ecological effectiveness of MPAs.

5. Conclusions

There are three main findings of this study. First, MPAs managed by local communities contribute to fisheries objectives more effectively than do co-management regimes. The reason for this finding appears to be that the two community-led MPAs (sites 1 and 3) were both established and embraced by their respective communities with considerable enthusiasm, and have enjoyed high compliance rates with their regulations. By contrast, the co-management MPA (site 2) was established by the government after a controversial consultation process, and has experienced significant levels of non-compliance with its regulations, not least because of disputes over gears used. Second, there is no evidence to indicate that either community managed or co-managed MPAs contribute to marine conservation more broadly, though this could be due to the length of the survey period, rather than because they are ineffective. Third, all three MPAs studied represent noteworthy efforts in coastal management, in that higher abundance of all size classes of *P. pelagicus* was observed in all three MPAs compared with their respective controls.. The first finding confirms the hypotheses that community-led management can positively impact on crab populations and that the inhabitants of fishing villages have the capacity to implement such protection through community-led MPAs. This result supports the need for more bottom-up management which empowers local community-led management, assuming community member are supportive and compliance can be achieved.

However, there are three caveats to this conclusion. First, the data obtained in this study constitute only “a snapshot” of aquatic community composition over two seasons, and the results also suffer from the limitation that there are no pre-MPA baseline data. Whilst the significant increases in target species observed in the two community-led MPAs (sites 1 and 3) are encouraging, signifying good community capacity and a positive role local fishers can have in managing their own marine resources, future research should focus on longer-term monitoring and stock assessments in these and other community-managed MPAs. Second, these MPAs are small in area and few in number in Thailand and other Southeast Asian countries, and while small MPAs can empower resource users and improve fishery conditions for them, they do so only on a very localised scale. Larger MPAs, MPA networks, and international research collaboration on significant geographical scales are required to ensure the health of the wide-ranging commercial fishery stocks which support fishing communities throughout the coastal regions of the Indian and West Pacific Oceans. Third, a shift towards bottom-up governance needs approval from national and local governments, so community management cannot entirely replace the state’s stewardship of natural resources, but it can improve its implementation.

Acknowledgements

This work was undertaken as part of a joint funded UK Natural Environment Research Council (NERC) and Economic Social Research Council (ESRC) PhD and for their support we offer thanks. Special thanks go to the Department of National Park and the National Research Council of Thailand (NRCT) for allowing this work to take place. We would also like to thank the Mangroves for the Future initiative, and IUCN Asia Regional Office in Bangkok, for their in-country support.

Appendices

Appendix A: Dimensions and deployment method of each gear

Gear type	Volum	Size of gear	Gear set up	Deployme	Bait
Bottom-water gill nets	2 x 240m	3 panels of 80m x 1.5m nylon mesh. 100mm (k-k) mesh with a hanging ratio (E) 0.5	Net intertwined with plastic floats and lead weights and surface buoyed and weighted at each end with rocks	400-600m apart	none
Box traps for *Portunus*	4 x 10 traps	300mm x 600mm x 250mm with entrance slips at each end	Line of 10 baited traps tied 7m apart and buoyed at each end	200-300m part	*Sardinella* spp.
Box traps for *Scylla*	5 x 7 traps		Baited and separately ropes to a buoy	In groups of 5	*Sardinella* spp.

Reference

Agardy, T., di Sciara, G.N. & Christie, P. (2011) Mind the gap: Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, 35 (2), pp.226–232.

Bianchi, G. (2000) Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science*, 57 (3), pp.558–571. Available from: <http://icesjms.oxfordjournals.org/cgi/doi/10.1006/jmsc.2000.0727> [Accessed 8 July 2010].

Boutson, A., Mahasawasde, C., Mahasawasde, S., Tunkijjanukij, S. & Arimoto, T. (2009) Use of escape vents to improve size and species selectivity of collapsible pot for blue swimming crab Portunus pelagicus in Thailand. *Fisheries Science*, 75, pp.25–33.

Caveen, A.J., Gray, T.S., Stead, S.M. & Polunin, N.V.C. (2013) MPA policy: What lies behind the science? *Marine Policy*, 37, pp.3–10.

Caveen, A.J., Sweeting, C.J., Willis, T.J. & Polunin, N.V.C. (2012) Are the scientific foundations of temperate marine reserves too warm and hard? *Environmental Conservation*, 39 (03), pp.199–203.

CBD (2010) Aichi Biodiversity Targets 2011-2020 [Internet]. Available from: <http://www.cbd.int/sp/targets/>.

Chande, A.I. & Mgaya, Y.D. (2003) The Fishery of Portunus pelagicus and Species Diversity of Portunid Crabs along the Coast of Dar es Salaam, Tanzania. *Western Indian Ocean J. Mar. Sci.*, 2 (1), pp.75–84.

Charles, A. & Wilson, L. (2009) Human dimensions of Marine Protected Areas. *ICES Journal of Marine Science*, 66, pp.6–15.

Christie, P. & Pollnac, R.B. (2011) Socio-Economy - Social dynamics of scaling-up marine protected areas declaration and management. In: J. Claudet ed. *Marine Protected Areas: A Multidisciplinary Approach*. Cambridge, New York, Melbourne, Madrid, Cape Town, Singapore, Sao Paulo, Dekli, Toyko, Mexcio City, Cambridge University Press, pp.121–140.

Duke, N., Meynecke, J., Dittmann, S., Ellison, A., Anger, K., Berger, U., Cannicci, S., Diele, K., Ewel, K., Field, C., Koedam, N., Lee, S., Marchand, C., Nordhaus, I. & Dahdouh-Guebas, F. (2007) A World Without Mangroves? *Science*, 317, pp.41–43.

Edgar, G. (1990) Predator-prey interactions in seagrass beds. II. Distribution and diet of the blue manna crab Portunus pelagicus Linnaeus at Cliff Head, Western Australia. *Journal of Experimental Marine Biology and Ecology*, 139, pp.23–32.

Ellison, A.M. (2008) Managing mangroves with benthic biodiversity in mind: Moving beyond roving banditry. *Journal of Sea Research*, 59 (1-2), pp.2–15.

Hilborn, R. (2004) When can marine reserves improve fisheries management? *Ocean & Coastal Management*, 47, pp.197–205.

Hughes, T.P. (1994) Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science*, 265, pp.1547–1551.

Jentoft, S., Van Son, T.C. & Bjørkan, M. (2007) Marine Protected Areas: A Governance System Analysis. *Human Ecology*, 35, pp.611–622.

Johnston, D., Harris, D., Caputi, N. & Thomson, A. (2011) Decline of a blue swimmer crab (Portunus pelagicus) fishery in Western Australia—History, contributing factors and future management strategy. *Fisheries Research*, 109, pp.119–130.

Jones, E.V. (2014) Does Community Management still need the State? Comparative Analysis of Marine Protected Area Governance in Thailand. Newcastle University.

Kearney, R., Farebrother, G., Buxton, C.D. & Goodsell, P. (2013) How terrestrial management concepts have led to unrealistic expectations of marine protected areas. *Marine Policy*, 38, pp.304–311.

Kunsook, C., Gajaseni, N. & Paphavasit, N. (2014) A Stock Assessment of the Blue Swimming Crab Portunus pelagicus (Linnaeus, 1758) for Sustainable Management in Kung Krabaen Bay, Gulf of Thailand. *Tropical Life Science Research*, 25 (1), pp.41–59.

Lauck, T, Clark, C., Mangel, M. & Munro, G. (1998) Implementing the precautionary principle in fisheries management through marine reserves. *Ecological Applications*, 8 (1), pp.72–8.

Lee, H.-H. & Hsu, C.-C. (2003) Population Biology of the Swimming Crab Portunus sanguinolentus in the Waters off Northern Taiwan. *Journal of Crustacean Biology*, 23 (3), pp.691–699.

Macintosh, D.J., Aksornkoae, S., Vannucci, M., Field, M., Clough, C.D., Kjerfve, B., Paphavasit, N. & Wattayakorn, G. (1991) *Final report of the intergrated multidisciplinary survey and research programme of the Ranong mangrove ecosystem*. National Research Council of Thailand: Bangkok.

Miller, R.J. (1990) Effectiveness of crab and lobster traps. Canadian. *Journal of Fisheries and Aquatic Sciences*, 47, pp.1228–1251.

Moser, S., Macintosh, D.J., Laoprasert, S. & Tongdee, N. (2005) Population ecology of the mud crab Scylla olivacea: a study in the Ranong mangrove ecosystem, Thailand, with emphasis on juvenile recruitment and mortality. *Fisheries Research*, 71, pp.27–41.

National Research Council Working Group (2001) *Marine Protected Areas: Tools for Sustainable Ocean Ecosystems*. Washington DC, National Academy Press.

Overton, J.L., Macintosh, D.J. & Thorpe, R.S. (1997) Multivariate analysis of the mud crab Scylla serrata (Brachyura: Portunidae) from four locations in Southeast Asia. *Marine Biology*, 128, pp.55–62.

Pimbert, M.P. & Pretty, J. (1995) *Parks, People and Professionals: Putting `Participation' into Protected Area Management*. UNRISD, Geneva.

Polidoro, B. a, Carpenter, K.E., Collins, L., Duke, N.C., Ellison, A.M., Ellison, J.C., Farnsworth, E.J., Fernando, E.S., Kathiresan, K., Koedam, N.E., Livingstone, S.R., Miyagi, T., Moore, G.E., Ngoc Nam, V., Ong, J.E., Primavera, J.H., Salmo, S.G., Sanciangco, J.C., Sukardjo, S., Wang, Y. & Yong, J.W.H. (2010) The loss of species: mangrove extinction risk and geographic areas of global concern. *PloS one*, 5 (4), p.e10095.

Roberts, C.M. (2009) Effects of Marine Reserves on Adjacent Fisheries. *Science*, 1920 (2001).

Roberts, C.M., Hawkins, J.P. & Gell, F.R. (2005) The role of marine reserves in achieving sustainable fisheries. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 360 (1453), pp.123–32.

Robertson, W.D. (1989) Factors affecting catches of the crab Scylla serrata. (Forskâl) (Decapoda: Portunidae) in baited traps: soak time, time of day and accessibility of the bait. *Estuarine, Coastal and Shelf Science*, 29, pp.161–170.

Sathirithai, S. & Barbier, E.B. (2001) Valuing Mangrove Conservation in Southern Thailand. *Contemporary Economic Policy*, 19 (2), pp.109–122.

Seytre, C. & Francour, P. (2013) A long-term survey of Posidonia oceanica fish assemblages in a Mediterranean Marine Protected Area: emphasis on stability and no-take area effectiveness. *Marine And Freshwater Research*, 65 (3), pp.244–254.

Skilleter, G.. & Warren, S. (2000) Effects of habitat modification in mangroves on the structure of mollusc and crab assemblages. *Journal of Experimental Marine Biology and Ecology*, 244, pp.107–129.

Stephenson, W. & Campbell, B. (1959) The Australia portunids (Crustacea: Portunidae) III: The genera Portunus. *Asustralian Journal of Marine and Freshwater Research*, 10, pp.84–124.

Sudtongkong, C. & Webb, E.L. (2008) Outcomes of State- vs. Community-Based Mangrove Management in Southern Thailand. *Ecology and Society*, 13 (2), pp.27–50.

Sukumaran, K.K. & Neelakantan, B. (1997) Food and feeding of Portunus (Portunus) snaguinolenus (Herbst) and Portunus (Portunus) pelagicus (Linnaeus) (Brachyuara; Portunuidae) along Karnataka coast. *Indian Journal of Marine Sciences*, 26, pp.35–38.

Le Vay, L. (2001) Ecology and Management of Mud Crab Scylla spp. *Asian Fisheries Science*, 14, pp.101–111.

Walton, M., Levay, L., Lebata, J., Binas, J. & Primavera, J. (2007) Assessment of the effectiveness of mangrove rehabilitation using exploited and non-exploited indicator species. *Biological Conservation*, 138, pp.180–188.

Walton, M., Levay, L., Lebata, J., Binas, J. & Primavera, J. (2006) Seasonal abundance, distribution and recruitment of mud crabs (Scylla spp.) in replanted mangroves. *Estuarine, Coastal and Shelf Science*, 66, pp.493–500.

Weng, H.T. (1992) The sand crab (Portunus pelagicus (Linnaeus)) populations of two different environments in Queensland. *Fisheries Research*, 13, pp.407–422.

Williams, M.J. (1982) Natural food and feeding in the commercial sand crab Portunus pelagicus in Moreton Bay, Queensland. *Journal of Experimental Marine Biology and Ecology*, 59, pp.165–176.

Willis, T.J., Millar, R.B., Babcock, R.C. & Tolimieri, N. (2003) Burdens of evidence and the benefits of marine reserves: putting Descartes before des horse? *Environmental Conservation*, 30 (2), pp.97–103.

Wilson, S.K., Fisher, R., Pratchett, M.S., Graham, N. a. J., Dulvy, N.K., Turner, R. a., Cakacaka, A., Polunin, N.V.C. & Rushton, S.P. (2008) Exploitation and habitat degradation as agents of change within coral reef fish communities. *Global Change Biology*, 14 (12), pp.2796–2809.

Xiao, Y. (2004) Sex ratio, and probability of sexual maturity of females at size, of the blue swimmer crab, Portunus pelagicus Linneaus, off southern Australia. *Fisheries Research*, 68, pp.271–282.