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The Bio-economic Effects of Artificial Reefs: Mixed Evidence from Shandong, China

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Abstract

Artificial reefs are used to protect coastal habitats and rebuild fisheries. This engineering approach to fisheries management has gained popularity in many coastal areas, including China. In Shandong province alone, over USD 50 million were invested in artificial reefs during 2005–2013. Have artificial reefs achieved their biological and economic objectives? We compared reef and control sites in terms of catch and value per unit effort and average body length across species, based on surveys carried out during 2012–2013. We found that in aggregate, with all fish and invertebrates combined, artificial reefs did not improve the overall catches or revenues. Instead, seasonal fluctuations were prominent. However, when we allow for species-specific differences and focus on the common fish species, we find that an artificial reef can increase the catch and value per unit effort on average by approximately 40% compared to the control sites. The difference between these contrasting results occurs because some of the dominant species that comprise the bulk of the catches did not benefit from the reef, while many of the less dominant ones did so. This underlines the importance of being specific about what is meant by "benefiting fisheries" when evaluating artificial reefs, as well as when the objectives of reef projects are formulated in the first place. The positive effects of artificial reefs can be caused by the reefs themselves and by their influence on fishing patterns. Our study was not designed to separate these effects but we suggest that in Shandong, restrictions on fishing access may have been as important as the presence of the reef itself.

Keywords: Fisheries management, artificial reefs, linear mixed-effects model, CPUE, VPUE, socio-economic analysis

1 Introduction

Artificial reefs (AR), engineering structures deployed on the sea floor, have been regarded as a useful tool to manage fishing activities, enhance the productivity of fish stocks, and mitigate habitat deterioration (Baine, 2001; Bortone et al., 2011). The use of ARs varies strongly by country, with their purpose ranging from supporting recreational fishing and restricting the entry to marine protected areas to restoration and sustaining coastal fisheries. Japan has been one of the pioneers in fishing reef technology, aided by generous subsidy programs: during 1976–1987, Japanese government invested nearly \$100 million annually to construct a total of 1.4 million m^3 of ARs (Grove et al., 1989). Over the years, ARs have been spreading to many parts of the world, including Southeast Asia (Islam et al., 2014), the Persian Gulf (Feary et al., 2011), North America (Thanner et al., 2006), Australia (Branden et al., 1994), and Europe (Santos and Monteiro, 1997; Jensen, 2002). The development of modern ARs in China dates back to late 1970s, and has undergone an experimenting phase during 1979–1987 (Shen and Heino, 2014) and a formal deployment phase since 2001 (Yang et al., 2005). Despite the increasing popularity of AR programs worldwide, uncertainty remains regarding whether ARs achieve the intended fisheries enhancement or other objectives.

There is a long-standing debate of whether the biological effects of ARs emerge through 'attraction' where fish from surrounding areas are concentrated near a reef, without net increase in abundance, or 'production' where ARs increase fish abundance by providing new habitats (Pickering and Whitmarsh, 1997; Powers et al., 2003; Brickhill et al., 2005). To date, scientists appear to embrace the attraction hypothesis (Lindberg, 1997; Feary et al., 2011; Tessier et al., 2014), although several empirical studies have backed the production hypothesis too (e.g., Cresson et al., 2014; Lowry et al., 2014). Osenberg et al. (2002) argued that attraction and production should be treated as end-points on a continuum; where a particular system lies along the continuum will depend on reef design and species characteristics. Indeed, existing evidence suggests that fish recruitment, aggregation, and diversity are strongly influenced by physical attributes of the reef such as structural complexity (Spieler et al., 2001), reef size, orientation and depth (Pickering and Whitmarsh, 1997), by local environmental factors such as sedimentation load and water circulation (Perkol-Finkel et al., 2006; Wang et al., 2016), and by ecological processes such as predation and competition (Leitao et al., 2008)

Compared to the progress with biological evaluations of ARs, economic evaluations of ARs have only started to emerge relatively lately. Economic evaluations include socio-economic impact and efficiency assessments (Milon et al., 2000). Polovina and Sakai (1989) examined production change of two fisheries in Japan and found that *Octopus* catches were increased by 4% per 1000 m^3 of artificial reef deployed, but that the catches of flatfishes did not increase. Whitmarsh et al. (2008) showed that in southern Portugal the fishing revenue from AR sites is 1.7 times of that from the control sites. On the contrary, Islam et al. (2014) did not find benefits provided by concrete-based AR structures to the drift net users in Terengganu, Malaysia. Some studies have found that ARs can bolster local economy through ecotourism (Leeworthy et al., 2006; Kirkbride-Smith et al., 2013), but their ability to reduce pressure on the surrounding natural reefs may be limited (Oliveira et al., 2015). While authors may report positive or economic negative outcomes, many of them warn against the ‘double-edged sword’ effect of AR programs. As Milon (1989) put it, an AR that is effective in aggregating fish may jeopardize the overall economic performance of a fishery if access to the resource is not controlled.

Both consumer surplus and producer surplus approaches can be used in assessing the economic performance of ARs (Milon et al., 2000). The consumer surplus approach is typically applied in cost-benefit analysis of demand for diving sites, demand for recreational fishing sites, and preference for marine habitat preservation, while the producer surplus is often used in measuring fishermen’s profit change. Because the primary objective of ARs in Shandong is fisheries enhancement (Yang, 2016), a producer surplus approach is more suitable in our case. Building upon Milon (1989; 2000), Whitmarsh et al. (2008) applied value per unit effort (VPUE), defined as catch per unit effort (CPUE) times the unit price of catch, to analyze producer surplus and profits due to ARs. Although VPUE only captures partial direct-use values of ARs (Whitmarsh et al., 2008), it has the advantage of being simple and objective, because price data reflect market information revealing people’s true preferences, and CPUE data are based on biological surveys. Moreover, the motivating effect of VPUE in fishermen’s targeting decisions is well documented (e.g., Marchal et al., 2007; Bastardie et al., 2013). By contrast, methods based on interviews or questionnaires (Polak and Shashar, 2013; Islam et al., 2014), often used in consumer surplus studies, may be susceptible to the ‘cheap talk’ problem (Farrell and Rabin, 1996): the extent of the true information that is revealed might be limited when communication is direct and costless.

Our study is set out to assess the catch and income generating potential of three artificial reefs in Shandong, China. The impact of these reefs on fish biodiversity has already been presented by Wang et al. (2016); here, we focus on their fisheries impacts.

Specifically, we hypothesize that ARs would result in greater CPUE, VPUE, and average species size, compared to the adjacent control sites. Differing from Whitmarsh et al.'s work, we account for species-specific effects while measuring the economic impacts. Very little is known about the performance of the artificial reefs deployed in China, especially outside the country. The main contributions of our paper are twofold: (a) documenting empirical experiences with Chinese artificial reefs to fill the existing knowledge gap, and (b) investigating the effect of artificial reef on fishery production and fishing revenue while taking species-specific differences into consideration.[add few words on importance of this aspect.]

2 The artificial reef development in Shandong

The large-scale deployment of artificial reefs (AR) in China started around 2001. Shandong, situated in the east coast of China, is a forefront province in the AR development. The deployment of AR program is closely linked to the development of sea ranching where artificial reefs are placed in the sea and hatchery-produced fish fries are released there, allowing the fry to grow in the wild. Learning from the experience of neighbouring countries (Grove et al., 1989), Chinese government considers sea ranching as an important tool to revive the marine-based economy as its coastal fisheries are being depleted (Yang, 2016). The depletion of coastal fisheries resources in China started to occur in early 1980s (Zhong and Power, 1997) and has continued (Cao et al., 2017); e.g., by 2002 over half of the economically important species in the East China Sea were severely depleted or being depleted (Ling et al., 2006). In the case of the Yellow and Bohai Seas, the major fishing grounds for the Shandong-based fisheries, a long-term ecosystem survey showed that the overall catch rate declined from 420 kg/h to 8 kg/h during 1959–2008 (Jin et al., 2013).

While the definition of a sea ranch varies in China, the following characteristics are typical: (1) the primary goal is to boost fisheries production; (2) property rights and sea boundary are clearly defined; (3) the recruitment of young fish relies on fry produced elsewhere; (4) the use of artificial reefs to simulate natural habitats that allow young fish to grow naturally, with or without a limited degree of externally provided feed (Yang, 2016). The construction of ARs constitutes a key component in the sea ranching program. In 2005, Shandong provincial government initiated a 10-year Fisheries Resources Restoration (FRR) program that greatly facilitated the deployment of ARs in the province. As of October 2013, the government had invested a total of RMB 300 million ($\sim$ USD 50 mill.) in ARs. This has led to the construction of 170 artificial reef projects, with a total volume of 10 million m^3 and occupying 15,000 ha sea floor along the coast (Shandong Provincial Department of Ocean and Fisheries, 2014).

The operational model of AR in Shandong typically is a public-private partnership model with three key stakeholders involved: the government, an expert panel, and a company. The government initiates a sea ranching program and provides initial funding. Experts provide technical assistance and advice during the deployment phase, especially with respect to fry cultivation and AR design. A selected company signs a long-term lease contract with the government and is guaranteed an exclusive use of a reef area. The main source of income for AR companies is generated from cultivation of high-value bottom-living species such as sea cucumber and abalone. Some AR companies also run recreational fisheries, but the income is minor. Companies would require recreational fishers to follow a number of rules, including maximum catch per boat (e.g., 20 kg) and a ban on juvenile fish. Of the three sites in our study, one (Rongcheng) is without recreational fisheries.

3 Material and methods

Sampling design

We carried out surveys near three islands in Shandong, namely Lidao, Xiaoshidao, and Qiansandao, located in the cities of Rongcheng, Weihai, and Rizhao, respectively, during September 2012–August 2013 (Fig. 1). The ARs in these three sites were all deployed during 2005–2010 through the government-subsidized Fisheries Resources Restoration Program. The sites are managed by three different reef companies. All survey sites (including reef sites and controls sites) are located within the reef area where the reef companies have exclusive access rights.

The site characteristics and reef material are given in Table 1 and Appendix A. The reef in Rizhao is twice as large as those in the other sites, and its material also differs from the others because of the greater bottom depth. Each AR site has a control site at a distance of about 800 meters. The choice of control site follows the principle that the environmental factors of a reef site and its control site should be similar (Zhang et al., 2006). Different from the other two sites, the control site for Rongcheng has a natural reef. In order to capture seasonality, the surveys were scheduled for different months, namely September and December in 2012, and January, May and August in 2013.

We applied standardized trammelnets in our sampling. Gillnets are often used in fisheries surveys, also when studying reefs (Kasim et al., 2013; Whitmarsh et al., 2008). Compared to simple gillnets, trammelnets effectively capture a broader size range of fish (Salvanes, 1991). Trammelnets are also commonly used by artisanal fishermen active in the adjacent areas, in part because engine-powered bottom trawling is banned in nearshore waters. Other gears in use are gillnets, handlines, and traps. Thus, trammelnet sampling provides a measure of fish density in a way that is relevant for assessing socio-economic impacts of ARs.

The trammelnets we used are 28 meters long and 3 meters high with an outer stretched mesh size of 10 cm and an inner mesh of 4.2 cm. The hanging ratios were 0.56 and 0.44, respectively. To prevent potential damage, the bottom of the net was attached to a half-meter long rope fixed onto a rock. While placing a net, the floats were adjusted to keep the rope straight and to ensure a half-metre minimum distance between the net and the seabed. Each site (including reef sites and control sites) was sampled at least once per season. Because nets were occasionally lost to currents or stolen, effective sampling frequency differs by site (Table 1). The nets were soaked for 24 hours. The catch was brought to a lab for identification and measurement.

Data

We have chosen catch per unit effort (CPUE) and value per unit effort (VPUE) as our primary indicators to measure the bio-economic effect of an artificial reef. These measures complement each other because CPUE describes the biological state of the resources as well as direct use values in terms food production, whereas VPUE measures the use values in monetary terms. Species-specific CPUE is calculated in kg per standard unit of effort, here defined as one trammelnet soaked for 24 hours. VPUE is simply a product of CPUE (C) and price (P), either per species (i) or summed over all species caught in the same net at the same time, e.g.,VPUE $= \sum_{i=1}^{s} C_i * P_i$. Prices (Table 2) were collected separately from Chengyang aquatic products market, which is the largest seafood market in Qingdao, the largest coastal city in Shandong. The species-specific price is fixed in our study, reflecting the average market conditions in year 2014. The data are summarized in

Table 2. The species include both fish and invertebrates; the fish can be further separated into demersal and pelagic species.

In addition, we use species-specific mean body size as an additional measure of the biological state of the resources: if an AR results in reduced fishing mortality, then we expect mean size to increase.

Statistical analyses

We used two types of models in the analyses: species-aggregated models and species-disaggregated models. We applied mixed-effects log-linear models in both. The species-aggregated model analyzed both 20 'common fish' species (species that were present at both control and reef sites) and all species. The species-disaggregated model is only run for the common fish species. Focusing on the common species is justified because we do not want rare and poorly sampled species to obscure the effects of ARs. We focused on the fish because our sampling with trammelnets was more suited to catch fish rather than invertebrates.

We tested a number of model specifications (e.g., including different interactions). Models were selected using the Akaike Information Criterion (AIC) (the final models are listed in Table 4 and 5). The following two examples are provided for illustrative purposes:

Mixed-effects :

$$\log(\text{VPUE}) \sim \alpha_a + \text{reef} + \text{month} + (1 \mid \text{site}) + \mu_a \tag{1}$$

$$\log(\text{VPUE}) \sim \alpha_g + \text{reef} + \text{month} + \text{type} + (1 \mid \text{site}) + (1 \mid \text{species}) + \mu_g \tag{2}$$

The explanatory variables in the both models are categorical variables: 'reef' is coded as binary variable (with 0 for a control site and 1 for an artificial reef site), 'type' refers to fish type (demersal vs. pelagic), 'site' has three levels (Table 1), and 'month' has five levels (September, December, January, August and May), ordered according to the occurrence of sampling date.

In the mixed-effects models, we treat reef, month and fish type as fixed effects, but species and site as random effects, because we are interested in the specific effects of the artificial reef and five sampling months, but not in a specific species or sites. The models were estimated using the R package `lme4` (Bates et al., 2015).

4 Results

The overall catch characterization

There were 69 different species caught in our surveys, 42 of which were fish and 27 invertebrates. This reflects the highly mixed nature of the fisheries in Shandong, with many different species contributing to the catches (Table 3). Over half of the species (37) were caught at both reef and control sites (hereafter referred to as the 'common species'), 26 were reef-only species, and 6 species were caught only at the control sites. Of the 37 common species, 20 were fish and 17 invertebrates (Table 2).

The catches from a single trammelnet placed in water for 24 hours were generally low in terms of total weight, consisting of small-sized fish (Table 2). There is a prominent seasonal pattern in catch, with the highest CPUE achieved in autumn, followed by a strong decline towards the winter (Fig. 2). This is true for both the reef and control sites. Shandong has cold winters, and survey sites are relatively shallow (between 5–20 metres); as temperatures drop, fish tend to move to deeper waters. However, the mean body lengths appear highest in the winter (December–January; Fig. 2). Among the control sites, the highest CPUE is observed in Rongcheng. Unlike other control sites, the Rongcheng control site has natural reefs (Table 1).

The common species contribute 90% to the aggregate VPUE, whereas the share for the reef-only species is only 9%. On average, the price of the species caught only in reef sites is about 1.2 times the price of the common species and 2.7 times of the species only caught at control sites. This indicates that reef sites are capable of attracting more valuable species.

The reef effect

We aggregated the data over species per net (total biomass per net) and estimated the reef effect with a log-linear model. The results suggest that artificial reefs do not improve any of the three aggregate measures, CPUE, VPUE and size (Table 4).There are some seasonal patterns. The most dominant feature is that CPUE in the winter months (Dec. and Jan.) and May are 30%–60% less than that in September if we count all species (Model 1, Table 4). The effect becomes much weaker when only common fish species are measured (Model 2, Table 4); instead, we found body length of the common fish species in winter and spring months are about 30% greater than that in September.

While the species-aggregated CPUE, VPUE and size did not show clear reef effects, species-disaggregated analyses for the 20 common fish species yield different results: the CPUE and VPUE of the reef sites are 40% higher compared to the control sites (Model a & b in Table 5). Body size shows similar tendency, but result is not significant (Model c in Table 5). Here species is treated as a random effect, such that the results can be interpreting as applying for an 'average' fish species; rare and abundant species get similar weight, in contrast to the aggregate models that are dominated by the most abundant species. Between-species variation was particularly important for VPUE, with the variance of the estimated random effect exceeding that of the residuals (Table 5).

The seasonal effects are retained in the disaggregated model: the VPUE and CPUE in December and August were 50%–60% lower and the body length in August was about 10% smaller compared to the reference month (September). Because of differences in mobility, and the closer association of demersal species with bottom structures compared to pelagic species, we expected to find a stronger reef effect for demersal species. Contrary to our expectation, we could not find any difference (i.e., non-significant reef $\times$ demersal/pelagic interaction). However, the VPUE was 90% lower for the pelagic species (Model a in Table 5. This difference is mainly caused by a lower price for pelagic fish, because CPUE and body size show no significant difference between fish types (Model b & c in Table 5).

5 Discussion

We have studied the biological and economic effects of artificial reefs in terms of CPUE, body length and VPUE in Shandong, China. The results are mixed: Whether species are benefiting from an artificial reef depends on the model we use and the species type (fish versus invertebrates). In the aggregated models, where catches are aggregated across all species caught in the same net, artificial reef did not increase the total CPUE nor VPUE. By contrast, the species-disaggregated models for the 20 common fish species showed a positive result: the VPUE and CPUE of fish species in the reef sites are 40% higher.

The divergence between the two analyses is caused by two factors. First, our sampling with trammelnets is more suited for capturing fish than invertebrates. Second, and more importantly, the results suggest that species that dominate the total catch benefit less (if at all) from the deployment of artificial reefs than the average fish species do. In essence, the species-aggregated model gives weight to species in proportion to their dominance in catch, while the disaggregated model measures the mean relative effect across individual fish species. Importantly, the two most abundant species caught in control sites, *Konosirus punctatus* and *Sebastes schlegelii*, accounting for respectively 19% and 12% of the site- and month-averaged total catch, had lower CPUE at the reef sites compared to the control sites (Table 3). In particular, the large negative effect (-80%) of the reef for the dominant pelagic *K. punctatus* is masquerading the positive effect for many other species. Nevertheless, the results of the aggregated models do not qualitatively change if we remove this species from the analyses (not shown).

The average species-level improvement on VPUE and CPUE resulting from the deployment of artificial reefs was about 40% in our study. The result is in line with other studies. For example, Whitmarsh et al. (2008) found that VPUE of the reef sites in southern Portugal was 70% higher.

High CPUE indicates that the density of fish is higher at artificial reef sites than at control sites. Unfortunately, we cannot distinguish whether this is due to the aggregation effect of an artificial reef, or because of different fishing pattern between reef and control sites allows the fish biomass at reef sites to partially recover. Nevertheless, the larger mean body size at the reef sites compared the control sites suggests lower mortality at the reefs, and that the positive reef effects at least partially stem from local recovery (Appendices B). Yet, at a finer level, the patterns are again highly varied, with some species being bigger at the reefs while others are not.[****]

Evidently, the exact magnitude of reef gains depends on a number of factors such as reef age, gear type, which prices are used (e.g., ex-vessel prices or market prices) and reef objective design. Kasim et al. (2013) found that hand-lining gave a higher VPUE than gillnets. Moreover, the aggregation effect of reefs may vary with their age. Researchers have found that resource abundance around the reef area tends to increase fast in the initial years, before an equilibrium level is reached (Bohnsack and Sutherland, 1985; Wang et al., 2008). Existing evidence suggests that the magnitude of improvement is often moderate, and that costly reef projects may sometimes be hard to justify economically (Sutton and Bushnell, 2007). A thorough evaluation on the costs and benefits of artificial reefs is thus critical prior to their wide deployment. Our study serves as a step-stone to pinpoint how this can be done. However, as mentioned previously, there are also other stakeholders involved in AR program. A full economic performance analysis shall also consider the producer surplus of companies and consumer surplus of other relevant stakeholders.

Our study involves some important limitations. Firstly, the price data in this study are species-specific averages. Because fish in the reef catches were on aware bigger, and because bigger fish often enjoy higher prices (Zimmermann and Heino, 2013), our use of average prices has likely underestimated the effect of reefs on the revenue. Secondly, we have no data on fishing activities in the surveyed reef and control sites, which prevents us from disentangling the physical and policy effects of reefs.

The AR programs in Shandong are operated under public-private partnerships (PPP). PPP helps to attract private funding into public projects, but public and private interests are not always in agreement. The contracted AR companies in Shandong have exclusive access rights to the area. This prevents artisanal fishers from entering the study sites, but it does not prevent the companies from fishing themselves. Unfortunately, we cannot access these data because companies are not required to report their catches. However, fishing intensity in our study area is likely moderate for two reasons: first, AR companies primarily rely on income from sea cucumber and abalone cultivation, the value of reef fish is too low to attract major interest; second, for the AR companies running recreational fisheries as an addition, they have incentive to protect the fish from over-exploitation for the sake of their business. Nevertheless, a lesson from Shandong is that the choice of operation model of an AR program shall match with its objective. If the main goal is to restore biological and ecological functions, mechanisms to avoid companies from abusing resources is critical in the design of a PPP model. As Wilson et al. (2002) pointed out, the artificial reefs are just one of many solutions to restore fisheries, and combining artificial reefs with instruments to reduce fishing intensity such as 'no-take zones' is essential. Islam et al. (2014) emphasized that non-restricted harvesting is the reason that ARs failed to bring economic benefits to artisanal fishermen in Malaysia.

6 Conclusions

We have investigated whether the deployment of artificial reef in Shandong improves fisheries catches and revenues. Our results are mixed: in aggregate, with all fish and invertebrates combined, artificial reefs did not improve the overall catches or revenues. When we allow for species-specific differences and focus on the common fish species, we found that an artificial reef can increase the catch and value per unit effort on average by approximately 40% compared to the control sites. The difference between these results occurs because some of the dominant species that comprise the bulk of the catches did not benefit from the reef, while many of the less dominant ones did so. This underlines the importance of being specific about what is meant by "benefiting fisheries" when evaluating artificial reefs and when the objectives of reef projects are formulated in the first place. Moreover, we emphasize that artificial reef projects alone are not sufficient to ensure the biological and economic goals. Restricting fishing access in the reef area is a key to achieve the biological goal of an AR program, an argument that has been reiterated also by other studies (Wilson et al., 2002).

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7 Figures

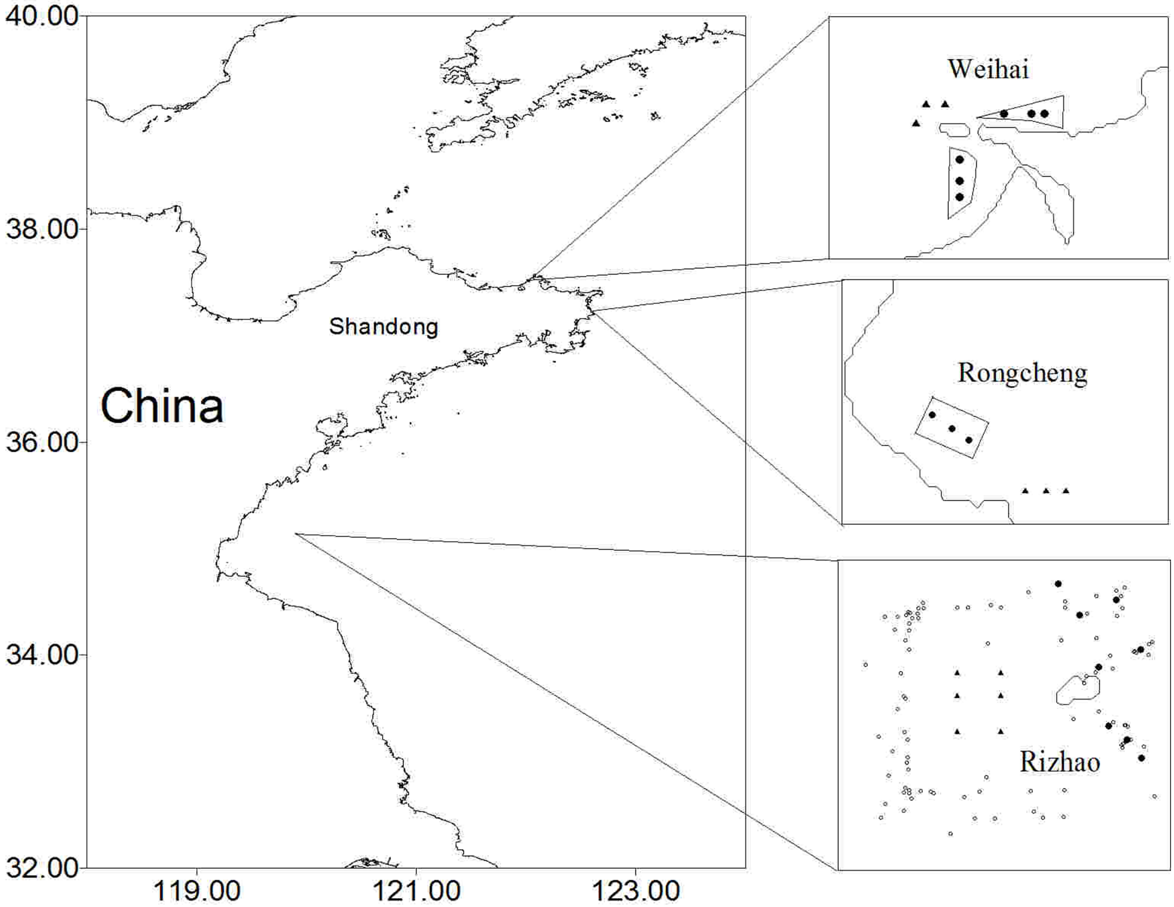


Figure 1: The location of the sampled artificial reef sites and their control sites along the Shandong coast. The filled circles stand for artificial reef sites and the filled triangles indicate the control sites. The hollow rectangle represents the deployed reef area in Weihai, Rongcheng and Rizhao.

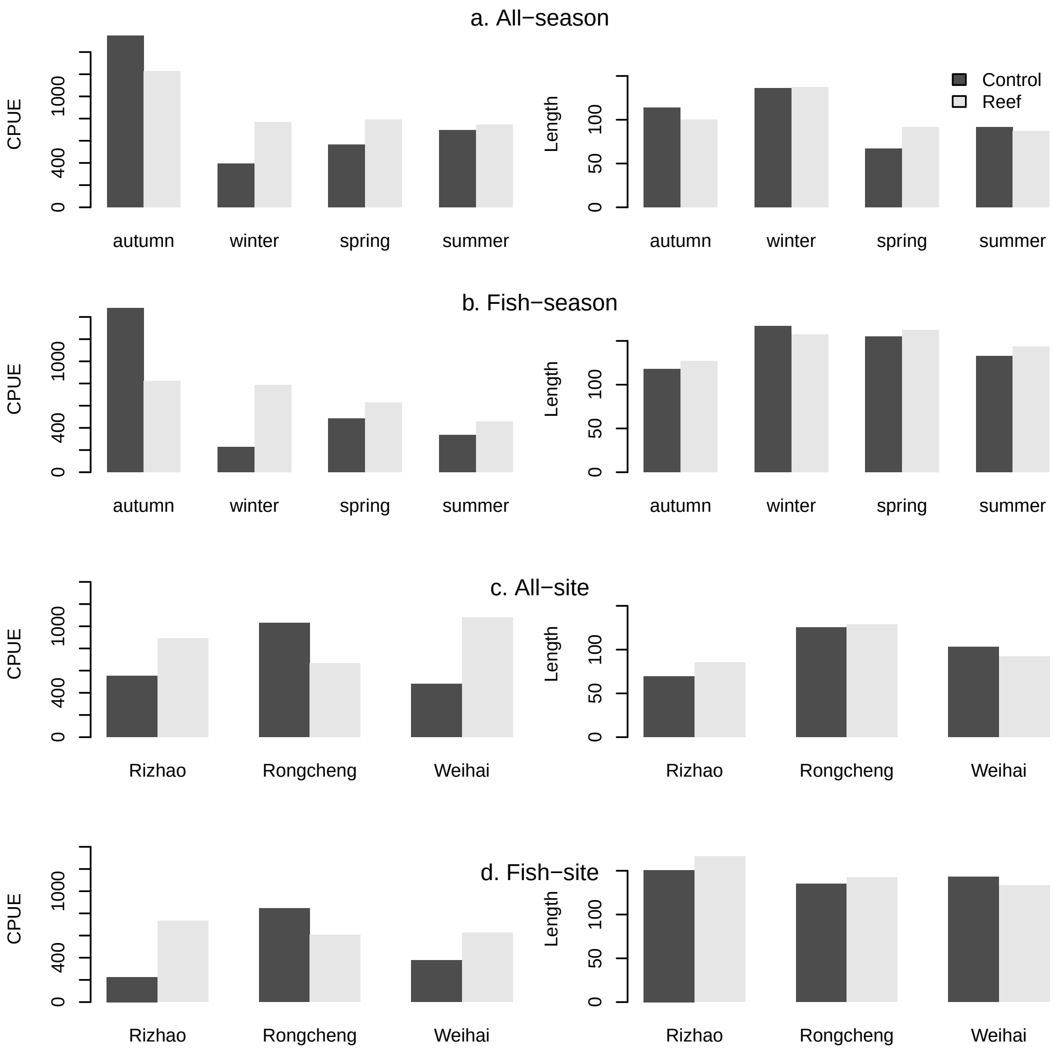


Figure 2: Season specific mean CPUE (g/net/day) and body length (mm) by species type and site. 'All' refers to both fish and invertebrates while 'Fish' means that only fish species are included. We refer September as 'autumn', December and January as 'winter', May as 'spring' and August as 'summer'.

8 Tables

Table 1: Sampling sites and their characteristics. Source: Zhang et al. (2006).

Site name	Site type	Sampling (times)	Specific location	Depth (m)	Reef area (hm^2)/ distance (m)	Bottom type	Reef material
Rizhao	Reef	24	Qiansandao	18–24	200	Fine sand & rocks	2,4,6,7
	Control	17	”	18–24	800	Fine sand & rocks	
Weihai	Reef	13	Xiaoshidao	5–15	97	Muddy sand	1,3,5
	Control	8	”	5–15	800	Muddy sand	
Rongcheng	Reef	19	Lidao	6–12	96	Hard substrates	1,3,5
	Control	9	”	6–12	800	Natural reef	

Notes: 1=natural rocks; 2=concrete pipes; 3= concrete A-shape blocks; 4=concrete cubic blocks; 5= clustered rocks; 6=layered concrete planks; 7=retired wooden shipwrecks. Appendix A shows examples of these structures.

Table 2: Summary statistics: Listed species are common species caught both at control and reef sites. Mean CPUE (g/net/d) is the species-specific total catch per net averaged over all surveyed stations and months. Mean length (mm) refers to the species-specific mean length per net averaged over all stations and months. 1 RMB ∼ 0.15 USD in 2012–2013. 'F' indicates fish (demersal or pelagic), 'NF' indicates non-fish (invertebrate). '+' = present but not measured.

Species name	Type	Price (RMB/kg)	Mean CPUE		Mean length	
			control	reef	control	reef
Agrammus sp.	F	50	171	306	137	139
Callionymus kitaharae	F	12	8	3	98	65
Thamnaconus septentrionalis	F	80	143	113	168	145
Chelidonichthys spinosus	F	60	41	118	134	230
Cleisthenes herzensteini	F	90	215	126	147	164
Engraulis japonicus	F	1	14	16	75	86
Hexagrammos otakii	F	50	161	709	166	181
Konosirus punctatus	F	32	1050	160	158	166
Lateolabrax maculatus	F	84	100	141	192	201
Paralichthys olivaceus	F	72	77	58	169	102
Pseudorhombus cinnamoneus	F	72	62	345	66	258
Pseudosciaena polyactis	F	40	38	109	135	148
Scomber japonicus	F	12	105	78	161	155
Sebastes hubbsi	F	92	34	71	95	95
Sebastes schlegelii	F	92	647	293	137	133
Sillago sihama	F	42	43	26	148	131
Sparus macrocephalus	F	130	84	330	138	154
Thryssa kammalensis	F	1	128	179	96	108
Verasper variegatus	F	50	48	192	126	138
Yongeichthys criniger	F	30	49	89	188	240
Alpheus heterocarpus	NF	50	+	4	+	18
Aphelasterias japonica	NF	1	24	32	46	45
Asterias amurensis	NF	1	48	49	62	49
Asterina pectinifera	NF	4	83	58	58	57
Charybdis bimaculata	NF	50	10	25	20	22
Charybdis japonica	NF	50	154	192	50	50
Dorippe japonica	NF	0.2	23	20	35	26
Glyptocidaris crenularis	NF	16	247	619	63	85
Hemicentrotus pulcherrimus	NF	190	62	32	46	+
Luidia quinaria	NF	1	152	51	66	59
Octopus ocellatus	NF	72	40	29	43	44
Oratosquilla oratoria	NF	40	52	59	29	50
Oregonia gracilis	NF	4	12	14	35	55
Parapanope euagora	NF	0.2	9	5	26	22
Parthenope validus	NF	0.2	71	87	30	32
Rapana venosa	NF	30	25	134	66	+
Trachypenaeus curvirostris	NF	132	12	14	28	29

Table 3: Top species ranked in terms of mean CPUE (g/net/day) averaged across all survey sites and months. Share= $\frac{\mathrm{CPUE}_i}{\sum_i \mathrm{CPUE}_i} * 100\%$

Rank	Species	CPUE	Share (%)	Species	CPUE	Share (%)
	Reef sites			*Control sites*		
1	Hexagrammos otakii	709	10.0	Konosirus punctatus	1050	19.3
2	Scomberomorus niphonius	698	9.8	Gadus	672	12.4
3	Glyptocidaris crenularis	619	8.7	Sebastes schlegelii	647	11.9
4	Pseudorhombus cinnamomeus	345	4.8	Glyptocidaris crenularis	247	4.6
5	Sparus macrocephalus	330	4.6	Cleisthenes herzensteini	215	4.0
6	Agrammus sp.	306	4.3	Atrina pectinata	206	3.8
7	Sebastes schlegelii	293	4.1	Agrammus sp.	171	3.1
8	Pseudopleuronectes yokohamae	280	3.9	Hexagrammos otakii	161	3.0
9	Saurida elongata	276	3.9	Charybdis japonica	154	2.8
10	Pholidae	224	3.2	Blennius yatabe	152	2.8
11	Charybdis japonica	192	2.7	Luidia quinaria?	152	2.8
12	Verasper variegatus	192	2.7	Cantherines septentrionalis	143	2.6
13	Thryssa kammalensis	179	2.5	Chaeturichthys stigmatias	142	2.6
14	Konosirus punctatus	160	2.2	Thryssa kammalensis	128	2.4
15	Argyrosomus argentatus	142	2.0	Scomber japonicus	105	1.9
	Subtotal		69.4%			80 %

Table 4: Estimating reef, time and site effects in the species-aggregated models. Parameters estimated on the logarithmic scale are additive; back-transformation to the natural scale gives multiplicative effects (column 'Multpl.'). Price=$\frac{VPUE}{CPUE}$. The displayed models are the ones with the lowest AIC score. Reference levels are Reef: control, month: September,and site: Rizhao; significance codes: '***' $p < 0.001$,'**' $p < 0.01$, '*' $p < 0.05$, '+' $p < 0.1$.

Dep. var.	Ind. var.	Fixed effects			Random effects	
		Estimate	t-value	Multipl.	Groups	Variance
1. All species						
(a) log(VPUE)	(Intercept)	10.46	18.54		Site	0.10
	Reef	0.15	0.42	1.17	Residual	2.77
	Dec	-0.79	-1.17	0.46	Obs.	90
	Jan	**-2.42****	-3.19	0.09		
	May	-0.79	-1.43	0.45		
	Aug	-0.17	-0.31	0.84		
(b) log(CPUE)	(Intercept)	6.66	20.44		Site	0.00
	Reef	0.14	0.62	1.15	Residual	1.04
	Dec	**-0.84***	-2.08	0.43	Obs.	90
	Jan	**-1.33****	-2.90	0.27		
	May	**-0.60**$^{+}$	-1.77	0.55		
	Aug	-0.28	-0.84	0.76		
(c) log(length)	(Intercept)	4.61	24.58		Site	0.07
	Reef	0.03	0.41	1.03	Residual	0.11
	Dec	0.15	1.07	1.16	Obs.	88
	Jan	**0.31**$^{+}$.	1.83	1.37		
	May	**-0.32****	-2.81	0.73		
	Aug	-0.17	-1.50	0.85		
2. Common fish						
(a) log(VPUE)	(Intercept)	10.09	20.15		Site	0.17
	Reef	0.17	0.54	1.19	Residual	1.64
	Dec	-0.62	-1.16	0.54	Obs.	73
	Jan	-0.27	-0.38	0.76		
	May	-0.06	-0.14	0.94		
	Aug	-0.57	-1.26	0.57		
(b) log(CPUE)	(Intercept)	6.09	14.14		Site	0.14
	Reef	0.33	1.23	1.39	Residual	1.16
	Dec	**-0.90**$^{+}$	-1.99	0.41	Obs.	73
	Jan	-0.16	-0.26	0.86		
	May	-0.35	-0.86	0.71		
	Aug	-0.62	-1.63	0.54		
(c) log(length)	(Intercept)	4.78	58.34		Site	0.01
	Reef	0.03	0.68	1.04	Residual	0.04
	Dec	**0.25****	2.95	1.29	Obs.	72
	Jan	**0.25***	2.22	1.29		
	May	**0.27****	3.42	1.31		
	Aug	0.10	1.33	1.10		

Table 5: Estimating reef, time and site effects in the species-disaggregated, mixed-effects models for the 20 common fish species. Estimates on the logarithmic scale are additive, back-transformation to the original scale gives multiplicative effects (i.e., Multipl.). Reference levels are reef: control, month: september, site: Rizhao. and fish type: demersal. 'Obs.'= number of observations. Only models with lowest AICs are displayed. Significance codes: '**' $p < 0.01$, '*' $p < 0.05$, '+' $p < 0.1$.

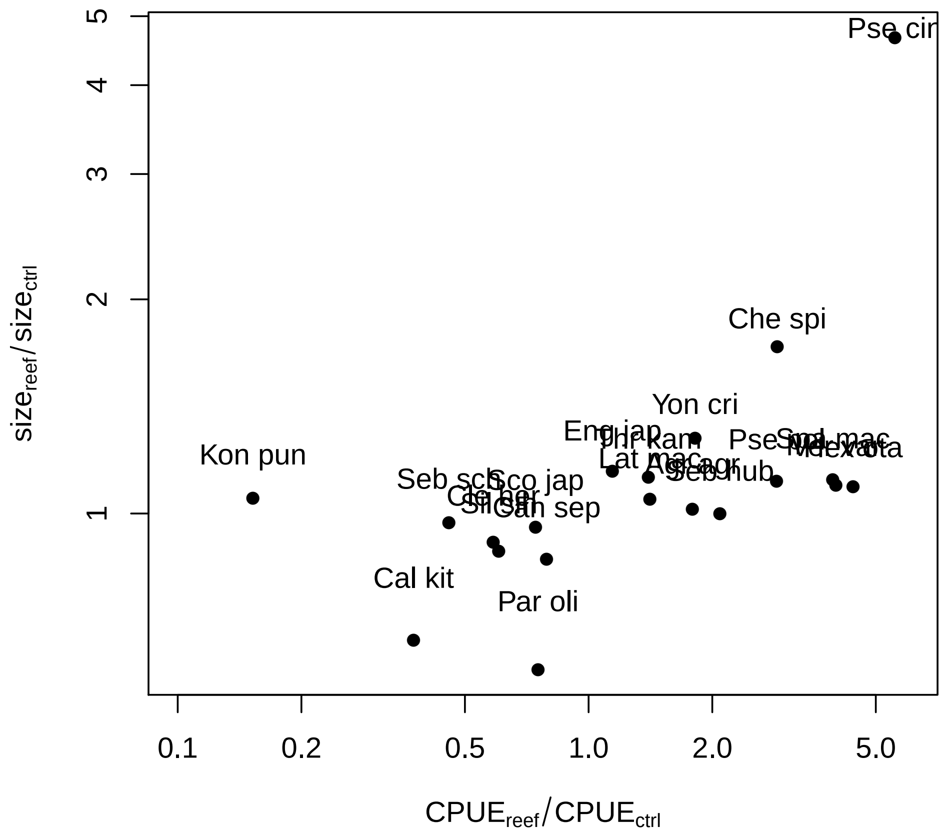
Dep. Var.	Fixed effects				Random effects	
	Ind. var.	Estimate	t-value	Multipl.	Random	Variance
(a) log(VPUE)	(Intercept)	8.74	17.64		Species	2.676
	Reef	**0.35***	2.09	1.42	Site	0.000
	Dec	**-0.71***	-2.19	0.49	Residual	1.118
	Jan	0.28	0.55	1.32	Obs.	189
	May	-0.43	-1.56	0.65		
	Aug	**-0.84****	-3.33	0.43		
	Pelagic	**-2.42***	-2.78	0.09		
(b) log(CPUE)	(Intercept)	4.74	15.67		Species	0.455
	Reef	**0.35***	2.07	1.42	Site	0.00
	Dec	**-0.69***	-2.27	0.50	Residual	1.125
	Jan	0.29	0.59	1.34	Obs.	189
	May	-0.42	-1.59	0.66		
	Aug	**-0.86*****	-3.59	0.42		
	Pelagic	-0.38	0.94	0.68		
(c) log(length)	(Intercept)	4.90	56.08		Species	0.048
	Reef	0.07	1.61	1.07	Site	0.010
	Dec	0.09	1.08	1.09	Residual	0.074
	Jan	0.10	0.81	1.11	Obs.	182
	May	0.02	0.31	1.02		
	Aug	**-0.12**$^{+}$	-1.94	0.88		
	Pelagic	-0.12	-0.93	0.89		

Appendices

A Artificial reef materials in Shandong

1- Natural rocks 	2-Clustered stones 
3-Concrete A-shaped blocks 	4-Concrete cubic blocks 
5-Concrete pipes 	6-Multiple layer concrete slabs 

B Correlation of CPUE and body length



Notes: Dots represent 20 fish species that are caught both in a reef and a control site. Texts denote the abbreviated species names listed in Table 2. Pearson's product-moment correlation is 0.597 (p=0.180) for 20 species, but becomes 0.48 (p=0.0336) after excluding outlier *Konosirus punctatus*.