Modelling the effect of the seasonal fishing moratorium on the Pearl River Estuary using ecosystem simulation

Abstract

The coastal ecosystem of the Pearl River Estuary (PRE) has been overfished and has received a high level of combined pollution in recent decades. Fisheries' stock assessments have shown a declining population and have led to the implementation of a number of management measures, including a fishing moratorium. This study evaluated the effect of a fishing moratorium on the sustainability of PRE fisheries through an ecosystem approach. Two Ecopath models of the PRE coastal ecosystem in 1998 and 2008 were applied to obtain snapshots of the ecosystem in different periods. An dynamic simulation of the period from 1998 to 2008 was developed using Ecosim based on the assumption that the seasonal moratorium was never applied to the PRE fisheries from 1999 onward, which resulted in the predicted ecosystem of 2008* (the so-called 2008* ecosystem). Then, the attribute indices of the 2008* ecosystem were compared with those of the actual 2008 ecosystem to investigate the effect of the fishing moratorium. Finally, a series of 100 years dynamics simulations was examined according to five scenarios based on the 1998 Ecopath model to explore better strategies for the fishing moratorium. The results show that the 2008* ecosystem was not supposed to feature a seasonal moratorium, as the system in 1999 was more deteriorated, immature and fragile than the actual ecosystem in 2008. The seasonal fishing moratorium did benefit ecosystem protection, although its effect on ecosystem recovery was limited. A comparative analysis of different scenarios indicates that most functional groups will decrease without executing a fishing moratorium (S1). The prolonged moratorium (S2) seemed to be slightly more beneficial to stocks recovery than S0, during which fishing operations were carried out following the present fishing moratorium policy. However, banning all fishing operations during the moratorium season (S3) has little effect on the recovery of fishing stocks from overexploitation. Moreover, reducing the fishing effort by 50% (S4) led to the largest increase in both fish stocks (28.0%) and total landings (43%).

Keywords: Ecopath with Ecosim; seasonal fishing moratorium; the PRE coastal ecosystem; Comparative analysis; fisheries strategies exploration

1. INTRODUCTION

It has been generally believed that the estuaries of large rivers and their adjoining coastal waters are typical marine ecosystems. The Pearl River is the second largest river in China in terms of flow rate. It is also the largest river that discharges into the north of the South China Sea (SCS). Currently, the coastal region of the Pearl River Estuary (PRE) is a significantly and quickly developing economic zone. Because of rapid economic development, the PRE region has experienced overfishing and pollution over the past three decades. The high population density and rapid development of industry and agriculture have resulted in severe stress on the aquatic environment. A great amount of waste, excessive reclamation, overfishing and frequent oil spills, among other factors, has greatly affected the water-related environmental quality of the PRE. Deteriorating environmental conditions have increasingly exerted strong effects on the estuarine ecosystem (Huang et al., 1997; Ke et al., 2007; Li and Huang, 2008; Liang et al., 2005). Moreover, the PRE coastal ecosystem has sustained high stress due to fisheries since the 1980s, which have been proposed as the first major human disturbance to coastal areas (Jackson et al., 2001).

Fishing fleets began to be privatized and investment in fisheries have increased since the economic reform initiated at the end of 1978, which resulted in a large increase in the number of fishing boats and improvement in fishing technology (Jia et al., 2005). Consequently, the landings of different fishing gear in the PRE experienced a substantial increase since 1979 and reached peak values in 1998 (Figure 1). The total number of landings in 1998 was nearly five times as high as in 1979. Figure 1 shows that trawling contributed to most catches among the fishing gear from 1979 through 2012, whereas the catch rates of trawlers in the PRE coastal sea dropped by more than 70% from 1986 to 1998 (Lu and Ye, 2001). Trawling had severely damaged groundfish stocks. The loss of groundfish stocks has been compensated for by a large increase in shellfish. Purse seining has likewise become problematic in the PRE. Purse seiners catch most juvenile species and fully exploited ones, strongly contributing to the overfishing of mostly fully exploited species. With anthropogenic activities exerting an increasingly strong effect on the estuarine ecosystem, it appears that the ecosystem has experienced large changes since 1979, switching from a large-size and high-value demersal fish-dominated ecosystem to an ecosystem dominated by small-size and low-value pelagic species (Duan et al., 2009a; Duan et al., 2009b; Jia et al., 2005).

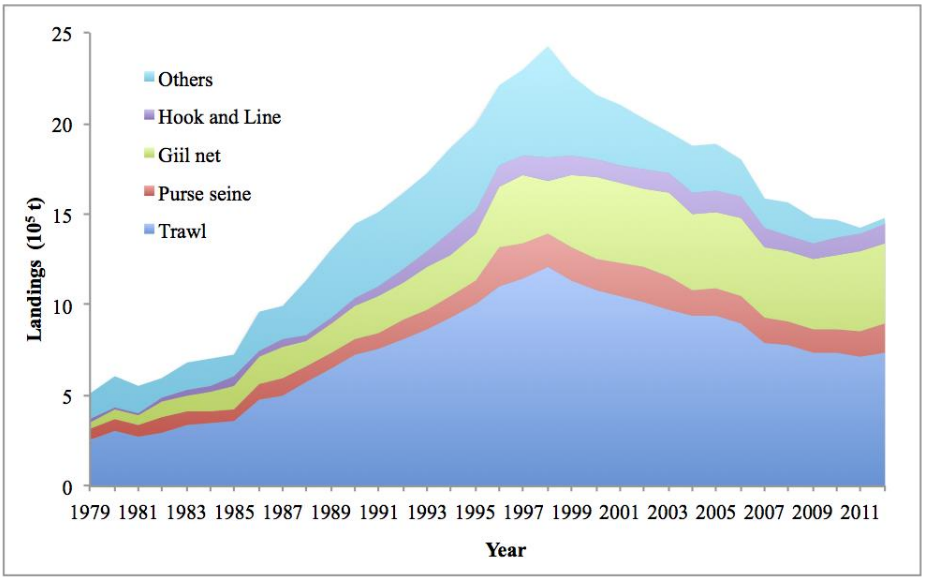

Fig. 1 Total number of landings of the coast of the PRE for different gear from 1979 to 2012

With Concern about overfishing, China put forward a zero-growth policy on fresh and seawater natural catches in 1999 and imposed a seasonal fishing moratorium. Beginning in 1999, the fishing moratorium was implemented from 1 June to 1 August every year in the northern South China Sea (SCS) (north of 12°N). During the fishing moratorium season, all fishing operations, excluding the use of gillnets, fishing cages and hooks and lines, are banned to conserve fisheries' resources and promote the sustainable development of the fishing industry. The approach is considered a concrete and effective measure for managing fishing effort and is expected to be beneficial to restoring the fisheries' resources. According to some researches (Dai et al., 2001; Liu et al., 2008b; Research group of summer fishing moratorium effect of Guangdong province in the South China Sea, 2014; Xiao, 2005), there was a considerable increase in fish catches in the two months immediately following the fishing moratorium, and the fishing moratorium was considered to be effective in protecting fisheries' resources and improving production. Until recently, the policy of fishing moratorium had been carried out for more than a decade. The effect of the fishing moratorium is still a matter of academic discussion. To data, there have been many studies on the effect of the seasonal moratorium (Chen, 2003; Hou et al., 2009; Huang, 2002; Liu and Chen, 2001; Wu, 2008; Yang and Zhou, 2013). These studies have been mostly extended qualitatively by comparing of community construction and the constitution of catches before and after the seasonal moratorium (Jiang et al., 2009; Robert, 1998; Schrank, 2005; Shi et al., 2008). Additionally, the advantages and disadvantages cancel out. Studies (Cheung and Pitcher, 2006; Pitcher et al., 2002) using ecosystem simulation models have suggested that the effects of the moratorium would be small given the sustained high fishing effort in the region. The published empirical studies that have evaluated the effect of the moratorium on exploited populations or ecosystem dynamics in the PRE are lacking. No detailed quantitative analysis has been conducted to data based on ecosystem simulation.

This study aimed to synthesize related information and time series data for dynamic simulation in Ecosim to quantify the changes of the coastal ecosystem under the moratorium as well as the possible effect of fishing on overall performance of the entire ecosystem. Moreover, fishing moratorium scenarios were simulated in Ecosim to explore better fishing strategies for the future.

2. METHODS AND MATERIALS

2.1 The study area

The Pearl River Estuary, located in the southern Chinese province of Guangdong, represents the typical coastal ecosystem of China. The coastal ecosystem of the PRE examined in our study, which extends from 112°30′E to 115°30′E, 21°00′N to 23°00′N, is a typical ecosystem of China’s coastal sea with an area of 72 600 km^2 (Fig. 2). The area covers the shelf from the coast to depths of approximately 100 m and it has the characteristics of delta coastal waters driven by salinity gradients arising from the combined effects of watershed and the open sea, with a NE-SW orientation (Xue et al., 2001).

The Pearl River Estuary is subjected to the effects of three water sources: the discharge of the Pearl River oceanic waters from the South China Sea, and coastal waters from the South China Coastal Current (Yin et al., 2004). The resultant nutrient-enriched waters provide large biological productivity and sustain the most important commercial fisheries (Li et al., 2000; Wang and Lin, 2006). The coastal fish production is mainly dominated by *Trichiurus lepturus* (largehead hairtail), *Nemipterus virgatus* (golden threadfin bream), sharks, *Decapterus maruadsi* (Japanese scad), *Sardinella aurita* (round sardinella), *Trachurus japonicus* (jack mackerel), *Siganus puellus* (masked spinefoot), *Argyrosomus argentatus* (silver croaker), *Saurida tumbil* (greater lizardfish), *Upeneus bensasi* (Bensasi goatfish), *Psenopsis anomala* (melon seed), and *Thamnaconus hypargyreus* (lesser-spotted leatherjacket), which are mainly exploited by trawling, purse-seining, long lining and gill netting (Mai et al., 2007). The PRE coastal ecosystem also plays a role as a natural refuge and nursery area for hundreds of species, including some local and endangered species such as D. *maruadsi* (Japanese scad), *S. aurita* (round sardinella), and *Larimichthys crocea* (Croceine croaker). The entire system shows diverse productivity, strong fishing activity, and complex food web relationships (Zhang, 2004). The dramatic expansion of fishing fleets, accompanied with mechanization and other technological advancements, have resulted in over-exploitation of near-shore and, more recently, offshore fisheries resources (Cheung and Sadovy, 2004; Shindo, 1973). These existing fishing fleets are highly capable of promoting already fully exploited fish stocks to an even greater overfished state (Ju, 2012; Lin and Zou, 2014; Liu et al., 2008a). Although the fish stocks appear to be able to recover from fishing pressure (Myers and Worm, 2005), the recovery rate depends on the productivity of the stocks and the level of depletion (Safina et al., 2005). Many marine fishes species have collapsed due to overexploitation, and it has been reported that many stocks have shown little or even no sign of recovery after up to 15 years, which suggests that fishes would be depleted to a level at which their recovery may be impaired (Hutchings, 2000; Hutchings and Reynolds, 2004). Therefore, over-exploitation in the PRE has aroused serious concern regarding fishery management and biodiversity conservation.

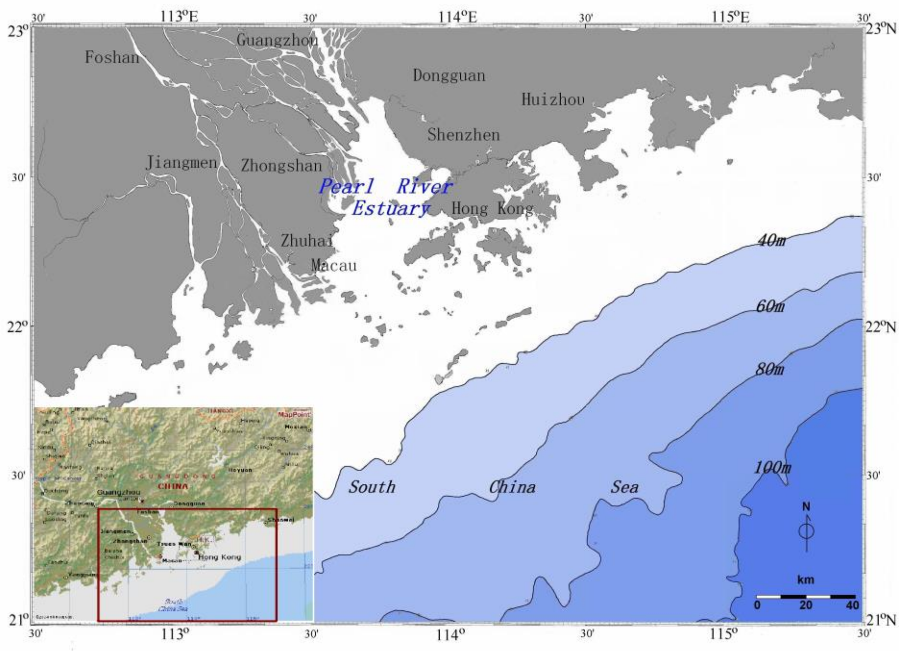

Fig.2 Map of the Pearl River Estuary (PRE) coastal ecosystem

2.2 The dynamic modeling approach

Ecopath with Ecosim (EwE) is ecosystem-based analysis software designed for the straightforward construction, parameterization and analysis of mass-balance trophic models of aquatic and terrestrial ecosystems (Christensen et al., 2005). Based on an approach proposed by Polovina (Polovina, 1984a, b) and further developed by Christensen and Pauly (Christensen and Pauly, 1992), Ecopath relies on straightforward mass-balance constraints to define trophic fluxes between functional groups. To date, the software has been optimized for direct use in fishery management as well as to address environmental questions through the inclusion of a temporal dynamic model (Ecosim) and a spatial dynamic model (Ecospace) (Christensen et al., 2005). The prominent advantage of this approach lies in its suitability in the application of a broad field of theories that are useful for ecosystem studies, which include thermodynamic concepts, information theory, trophic level description and network analysis (Müller, 1997). The approach has been used to analyse different aspects of the resulting food web network (Sandberg, 2007; Vidal and Pauly, 2004; Villanueva et al., 2008). Moreover, mass-balanced models enable comparisons between different ecosystems and between different periods of the same ecosystem (Díaz López et al., 2008; Neira et al., 2004; Panikkar and Khan, 2008; Shannon et al., 2003). Furthermore, the approach is often used to explore fisheries' management policy options under Ecosim (Araujo et al., 2008; Chen et al., 2009; Tsehaye and Nagelkerke, 2008; Viet Anh et al., 2014). In this study, dynamic simulations were performed in Ecosim to investigate the effect of the fishing moratorium on the PRE coastal ecosystem as well as to explore possible fishing strategies.

Ecosim allows for dynamic simulation at the ecosystem level, with key initial parameters inherited from the base Ecopath model. It estimates changes in biomass among functional groups in an ecosystem as functions of abundance among other functional groups and time-varying harvest rates, taking into account predator-prey interactions and foraging behaviors (Pauly et al., 2000). The basics of Ecosim consist of biomass dynamics expressed through a series of coupled differential equations. As it is a well documented and widely used model, it will not be expatiated here. The theory and method are detailed in the Ecopath with Ecosim user guide (Christensen et al., 2005).

2.3 The effect of fishing moratorium and fisheries policy exploration

Dynamic simulations were executed based on the improved Ecopath models, developed using EwE 6.0 in previous studies (Duan et al., 2009a; Duan et al., 2009b; Wang et al., 2012), in which the fisheries were divided into five types of gear (i.e. trawl, purse net, hook and line, gill net and others) for fishing moratorium simulation. The conceptual diagram of the PRE coastal ecosystem is shown is Fig.3. The ecosystem was divided into 24 functional groups in both Ecopath model: 1. *Sousa chinensis* 2.Sharks 3.*Trachurus japonicus* 4.*Decapterus maruadsi* 5.*Trichiurus haumela* 6.*Saurida.* 7.*Psenopsis anomala* 8.Upeneus *bensasi* 9.*Nemipterus virgatus* 10.*Priacanthus macracanthus* 11.*Priacanthus tayenus* 12.Other pelagics 13.Other demersals 14.Other zoobenthos 15.Benthic crustaceans 16.Polychaetes 17.Mollusks 18.Echinoderms 19.Cephalopods 20.Jellyfish 21.Zooplankton 22.Phytoplankton 23.Benthic producers 24.Detritus. Fig. 3 also describes the major flows of biomass and trophic interaction among functional groups in the PRE ecosystem. And the size of each round box is proportion to the biomass it represents.

The basic parameters of the Ecopath model for the actual 1998 and 2008 ecosystem were derived from stock assessment reports, the published literature, or government statistics. Input data and estimated parameters (bold) for the actual 1998 and 2008 ecosystem models are summarized in Table 1. And the databases of these two models are available at EcoBase model repository (http://sirs.agrocampus-ouest.fr/EcoBase/).

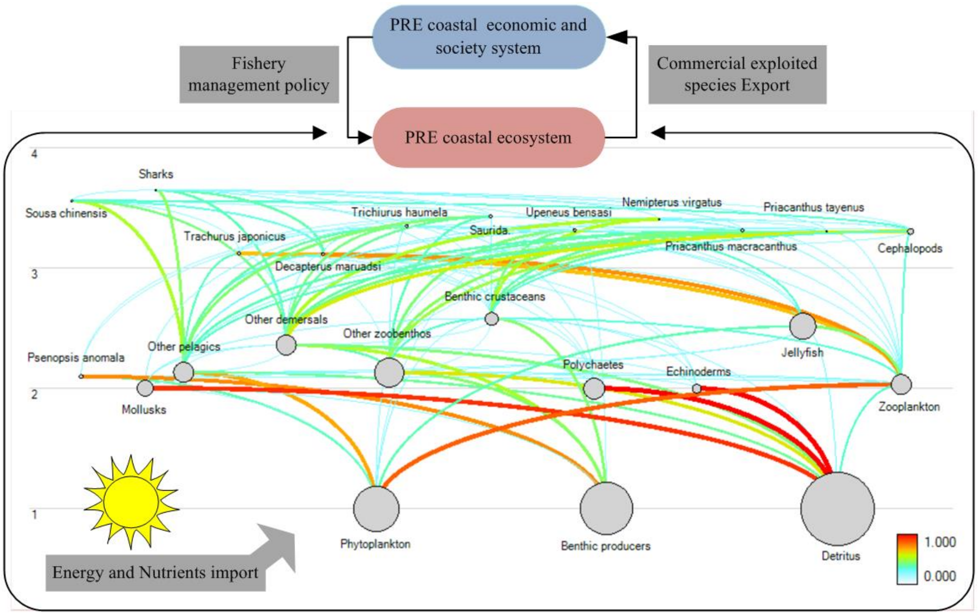

Figure 3 Conceptual diagram of the PRE coastal ecosystem

Table 1 Input and estimated parameters (bold) for Ecopath models of the PRE ecosystem in 1998 and 2008

Group name/parameters	1998							2008						
	ETL	**B**	**P/B**	**Q/B**	**EE**	**P/Q**	**Y**	**ETL**	**B**	**P/B**	**Q/B**	**EE**	**P/Q**	**Y**
1.Sousa chinensis	**3.60**	0.007	0.11	14.00	**0.02**	**0.01**	-	**3.56**	0.002	0.11	14.00	**0.05**	**0.01**	-
2.Sharks	**3.85**	0.002	0.40	6.83	**0.34**	**0.06**	-	**3.65**	0.004	0.40	6.83	**0.36**	**0.06**	-
3.Trachurus japonicus	**3.12**	0.012	2.15	10.47	**0.97**	**0.21**	0.01	**3.12**	0.012	2.15	10.47	**0.96**	**0.21**	0.02
4.Decapterus maruadsi	**3.11**	0.016	1.87	11.08	**0.92**	**0.17**	0.02	**3.11**	0.018	1.87	11.08	**0.90**	**0.17**	0.01
5.Trichiurus haumela	**3.36**	0.025	1.21	6.21	**0.95**	**0.19**	0.02	**3.35**	0.022	1.21	6.21	**0.94**	**0.19**	0.02
6.Saurida.	**3.39**	0.062	1.42	7.99	**0.93**	**0.18**	0.05	**3.43**	0.012	1.42	7.99	**0.99**	**0.18**	0.01
7.Psenopsis anomala	**2.10**	0.056	3.62	31.01	**0.94**	**0.12**	0.13	**2.10**	0.031	3.62	31.01	**0.99**	**0.12**	0.11
8.Upeneus bensasi	**3.37**	0.019	1.01	11.35	**0.94**	**0.09**	0.01	**3.31**	0.011	1.01	11.35	**0.92**	**0.09**	0.01
9.Nemipterus virgatus	**3.43**	0.013	2.07	7.23	**0.93**	**0.29**	0.02	**3.40**	0.007	2.07	7.23	**0.95**	**0.29**	0.01
10.Priacanthus macracanthus	**3.33**	0.009	1.97	9.69	**0.94**	**0.20**	0.01	**3.31**	0.013	1.97	9.69	**0.95**	**0.20**	0.02
11.Priacanthus tayenus	**3.32**	0.012	3.72	12.42	**0.98**	**0.30**	0.03	**3.30**	0.007	3.72	12.42	**0.92**	**0.30**	0.02
12.Other pelagics	**2.13**	0.740	3.72	12.70	**0.85**	**0.29**	1.57	**2.13**	0.670	3.72	12.70	**0.43**	**0.29**	0.50
13.Other demersals	**2.39**	0.679	3.47	12.89	**0.49**	**0.27**	0.04	**2.36**	0.660	3.47	12.89	**0.68**	**0.27**	0.77
14.Other zoobenthos	**2.15**	1.690	6.55	26.21	**0.47**	0.25	-	**2.13**	2.030	6.55	26.21	**0.42**	0.25	-
15.Benthic crustaceans	**2.60**	0.560	6.52	26.09	**0.71**	0.25	0.72	**2.58**	0.260	6.52	26.09	**0.45**	0.25	0.04
16.Polychaetes	**2.00**	0.800	4.93	19.71	**0.88**	0.25	-	**2.00**	0.740	4.93	19.71	**0.74**	0.25	-
17.Mollusks	**2.00**	0.700	4.80	19.20	**0.79**	0.25	0.38	**2.00**	0.414	4.80	19.20	**0.80**	0.25	0.23
18.Echinoderms	**2.00**	0.240	8.63	34.52	**0.95**	0.25	-	**2.00**	0.125	8.63	34.52	**0.74**	0.25	-
19.Cephalopods	**3.34**	0.099	3.10	11.97	**0.81**	**0.26**	0.18	**3.30**	0.074	3.10	11.97	**0.96**	**0.26**	0.20
20.Jellyfish	**2.52**	1.530	5.00	25.00	**0.05**	**0.20**	0.03	**2.52**	1.350	5.00	25.00	**0.05**	**0.20**	0.05
21.Zooplankton	**2.03**	0.666	32.00	192.00	**0.90**	**0.17**	0.11	**2.03**	0.690	32.00	192.00	**0.78**	**0.17**	0.11

22.Phytoplankton	**1.00**	7.59	71.50	-	**0.22**	-	-	**1.00**	10.715	71.50	-	**0.16**	-	-
23.Benthic producers	**1.00**	17.40	11.89	-	**0.11**	-	0.02	**1.00**	22.000	11.89	-	**0.09**	-	0.02
24.Detritus	**1.00**	200	-	-	**0.12**	-	-	**1.00**	200.000	-	-	**0.08**	-	-

ETL-effective trophic level, B—biomass (t· km-^{2}), P/B—production to biomass ratio (year^{-1}), Q/B—consumption to biomass ratio, (year^{-1}), P/Q—production to consumption ratio(year^{-1}), EE — ecotrophic efficiency, Y-total fishery catch rate (t· km^{2}year^{-1}).

The model, as implemented in Ecosim, argues that "top-down vs. bottom-up" control is in fact a continuum, where low *v* implies bottom-up and high *v* implies top-down control. Additionally, the simulation results are very sensitive to the values of *v*. In this study, the *v* values were attained through time series data fitting. Generally, the more time series data used and the better fittings obtained, the more reliable and reasonable are the *v* values that we can obtain. Therefore, the simulation-based predictions will be more credible. Due to the difficulty associated with data accessibility, time series data regarding catches and CPUE data for seven functional groups from 1991 to 2008 were used to perform a dynamic simulation in Ecosim under the condition of actually increasing fishing effort until the best fit was obtained (see Fig.4). Data pertaining to catches, CPUE and fishing effort were all obtained from the local fisheries' statistics yearbooks. The total sum of square error (SS) was decreased from 24.25 to 18.31 after altering the vulnerability (V) factor.

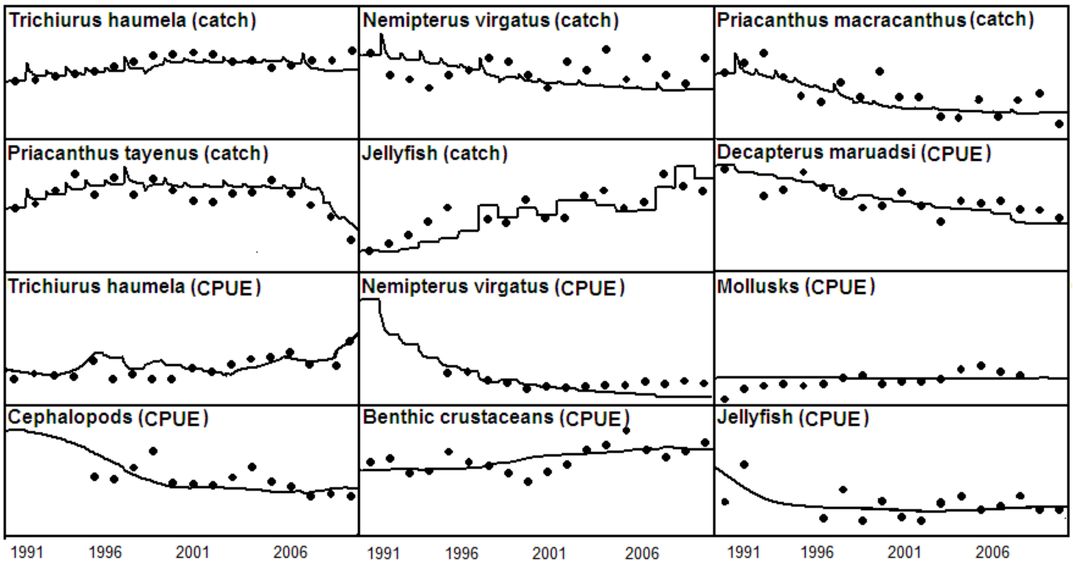

Fig.4 Best fits obtained for 12 sets of times series data regarding catch and relative abundance for seven groups via dynamic simulations using Ecosim

A dynamic simulation of the period from 1998 to 2008 using Ecosim was then performed based on the 1998 Ecopath model based on the assumption that the seasonal moratorium had never been applied to the PRE fisheries from 1999 onward, in which the *v* values were obtained from the best time series fitting indicated above. Through dynamic simulation, the predicted parameters for 2008 were used to construct another Ecopath model, the 2008* model. Then, the cumulative effects of fishing and the attributes of the ecosystem represented by the 2008* model were compared with those of the actual ecosystem with the fishing moratorium executed. Finally, simulations were developed for a period of 100 years under five scenarios based on the 2008 Ecopath model: S0: continuation the present fishing moratorium, S1: no fishing moratorium executed, S2: extension of the duration of the moratorium (i.e., 1 June to 1 September), S3: listing all fishing gear as banned based on the original fishing moratorium policy, and S4: no fishing moratorium executed, with the fishing effort of all fishing gear reduced by 50%. All of the scenario simulations were implemented, provided that the fishing effort was sustained and not prolonged.

3. RESULTS

3.1 Comparative analysis of the attributes of ecosystem maturity

There are many attributes indices related to the ecosystem's maturity obtained from the Ecopath model (see Table 2), in agreement with the theories of Odum and Christensen regarding the developmental stages that an ecosystem undergoes (Christensen, 1995; Odum, 1969). In this study, the predicted parameters that originated from the hypothetic ecosystem were input the software to construct a new Ecopath model (i.e. 2008* model) for the comparative analysis of system properties. The summary statistics of the 2008* model and the two actual models for 1998 and 2008 are shown in Table 2.

Ecopath model allows the quantification of the input-output relationships and the comparison of the habitats by means of the Total System Throughput (TST), which is the sum of the energy flows and represents the size of the ecosystem in terms of flows (Ulanowicz, 1986). The total system throughput estimated for the actual 2008 ecosystem (2311.86 t km^{-2} $year^{-1}$) and hypothetic 2008* ecosystem (2314.64 t km^{-2} $year^{-1}$) was relatively high compared to the value obtained for the 1998 ecosystem (1773.18 t km^{-2} $year^{-1}$), indicating that more energy flowed through the ecosystem in 2008 and 2008* compared to that in 1998. In addition, other flows through the studied ecosystem, such as the sum of all production, total net primary production and net system production, were also larger in the 2008* ecosystem (1083.45, 1027.61 and 905.05, respectively) than in the 2008 (1082.41, 1026.41 and 903.69, respectively) or 1998 (808.45, 749.48 and 621.00, respectively) ecosystems. Thus, the system flows, which increase with the amount of material flowing through the ecosystem, were higher in the hypothetical 2008* ecosystem.

The mean trophic level of the catch and gross efficiency are two indicators that represent the trophic level of fisheries' harvesting species. The mean trophic level of the catch in the 2008 model (2.40) is slightly higher than the levels measured for the 2008* (2.38) and 1998 model (2.34), demonstrating that the fisheries in 2008 were mainly targeting at top predators. This finding can be explained by different fishing fleet structure; the gillnet catch in 2008 (25.0%) contributed more than that in 1998 (12.2%), causing the mean trophic level of the catch to be high. The gross catch efficiencies in 2008 and 2008* were the same (0.002), but two times lower than the catch efficiency in the 1998 model (0.004). The gross efficiency of the catch was low when fisheries targeted top trophic level species and high when fisheries harvest low trophic levels (Viet Anh et al., 2014). The lower gross efficiencies in 2008 and 2008* indicated that harvesting species occupied high trophic levels.

The highest parameter values of the 2008* model were observed to be similar to those of the 2008 model. The connectance index (CI) and the system omnivory index (SOI) are two indices reflecting the complexity of the interrelationships within an ecosystem; the indices are also used to describe the maturity of an ecosystem and are expected to be higher in a mature system (Odum and Barrett, 1971). The 2008* model had almost same CI, SOI and overhead values (proposed as a possible measure of ecosystem stability) as the 2008 model. The value of overhead can be used as a measure of ecosystem stability (Rutledge et al., 1976), and the total system overhead and ascendency relative to capacity are mutually exclusive. The maturity shows a strong negative correlation with relative ascendency and thus a strong positive correlation with system overhead (Christensen, 1995). The reduction in the relative overhead from 1998 (64.6 %) to 2008 (60%) and 2008* (60%) indicates that the system was likely more susceptible to stress-induced changes even under the fishing effort reduction policy. Moreover, total biomass/total throughput represents the unit of biomass necessary to maintain the unit of flow, which is expected to increase along with ecosystem maturity (Christensen et al., 2005; Odum, 1969). However, no significant differences in total biomass/total throughput were found between the 1998, 2008 and 2008* ecosystems (0.02).

Furthermore, the ecosystem represented by the 2008 model appears to be slightly more mature than that represented by the 2008* model according to some indices, such as the ratio between primary production and total system respiration (PP/R), Finn's cycling index (FCI) and predatory cycling index

(PCI). PP/R is considered an important descriptor of system maturity (Odum and Barrett, 1971); it is expected to approach to 1.0 as an ecosystem develops toward the “mature” stage, to be greater than 1 in the early developmental stage of a system and to be less than 1 when an ecosystem receives large inputs of organic matter from the outside (e.g., one suffering from organic pollution). The PP/R value in the 2008 model is 8.36 (Table 2), slightly lower than that in 2008* (8.38) but much higher than that in 1998 (5.83) and 1981 (2.86), as shown in a previous study (Duan et al., 2009b). This discrepancy suggests that the ecosystem in the actual 2008 model was more mature than that in 2008* when the fishing moratorium policy was implemented from 1999 onward. However, most of the attributes of ecosystem maturity and stability indicated that the PRE coastal ecosystem in 1998 was in a relatively mature condition compared with that in 2008, which suggest that ecosystem was still in recession starting in 1998. The primary production/biomass ratio is considered another important descriptor of thermodynamic order and system maturity (Odum and Barrett, 1971). The PP/B values were 25.72 and 25.78 in the 2008 and 2008* model, respectively, greater than that the value in 1998 (22.76), which suggests that a lower level of biomass accumulation compared with the primary production in 2008 and 2008* and the hypothetic 2008* ecosystem without the fishing moratorium policy induced a more immature character than that observed for the actual 2008 ecosystem.

The cycling flow is quantified by means of the Finn Cycling Index (FCI) and Predatory Cycling Index (PCI), which measure the importance of cycling as a Total System Throughput fraction, FCI with and PCI without taking into account the detritus flows (Christensen et al., 2005). The FCI of the 2008 ecosystem was 2.23, slightly higher than that in 2008* (2.22) but much lower than that in 1998 (2.72). The FCI has been suggested to be correlated with system maturity, resilience and stability (Christensen et al., 2005; Vasconcellos et al., 1997). A high FCI is a feature of a mature ecosystem. The higher FCI value in 2008 model indicates that the ecosystem under fishing moratorium from 1999 onward was more complex and appeared more mature than the ecosystem without fishing moratorium. Moreover, ECOPATH provides an estimation of Finn's mean path length, which measures the average path length of the recycled flow. Finn's mean path length is a maturity index, because the cycling is lengthy and slow in mature and complex ecosystems and short and rapid in perturbed ecosystems (Christensen, 1995; Odum, 1969). Finn's mean path length was higher in the 1998 ecosystem (2.32) than in the 2008 (2.22) and 2008* (2.22) ecosystem, which indicates that the energetic flows are more articulated, the cycles are longer, and the recycling of organic matter is slower in the trophic network of 1998.

In previous work (Duan et al., 2009b), the PRE coastal ecosystem was compared by constructing Ecopath models for 1981 and 1998, which formed the foundation of the current study. The comparative analysis indicates that the coastal ecosystem of the PRE in 1981 was in a relatively mature condition compared with that in 1998. However, it was still at an intermediate-low developmental stage, even in 1981. In this study, the results of the comparative analysis implied that the ecosystem still deteriorated after ten years over-exploitation from 1998 to 2008. Although the fishing moratorium policy was effective in providing ecosystem protection, it protects the ecosystem from exacerbation and accomplished little with respect to ecosystem recovery.

Table 2 Summary of the indices for the PRE coastal ecosystem in 1998, 2008 and 2008*

Parameter（unit）	1998	2008	2008*
Ecosystem theory indices			
Total system throughput (t km^{-2} year^{-1})	1773.18	2311.86	2314.64
Sum of all production (t km^{-2} year^{-1})	808.45	1082.41	1083.45
Mean trophic level of the catch	2.34	2.40	2.38
Gross efficiency (catch/net p.p.)	0.004	0.002	0.002
Calculated total net primary production (t km^{-2} year^{-1})	749.48	1026.41	1027.61
Total primary production/total respiration (PP/R)	5.83	8.36	8.38
Net system production (t km^{-2} year^{-1})	621.00	903.69	905.05
Total primary production/total biomass (PP/B)	22.76	25.72	25.78
Total biomass/total throughput	0.02	0.02	0.02
Total biomass (excluding detritus) (t km^{-2})	32.93	39.90	39.87
Total catches (t km^{-2} year^{-1})	3.36	1.94	2.16
Connectance Index (CI)	0.28	0.28	0.28
System Omnivory Index (SOI)	0.13	0.12	0.12
Network flow indices			
Predatory cycling index (%of throughput without detritus) (PCI)	2.10	2.21	2.20
Finn's cycling index（% of total throughput）(FCI)	2.72	2.22	2.23
Finn's mean path length	2.32	2.22	2.22
Information indices			
Ascendency (% of capacity)	35.4	40.0	40.0
Overhead (% of capacity)	64.6	60.0	60.0

3.2 Cumulative impacts analysis

Figure 4 shows the cumulative impacts of fishing in both the actual 2008 PRE coastal ecosystem and the hypothetic 2008*model, which indicated that the changes in the susceptibility of the functional groups to human exploitation with effort increased by 10%. In Figure 5, the positive effects are shown above the base line and negative impacts are shown below. The effects of fishing on the specific species were compared between the actual 2008 ecosystem and the hypothetic 2008* ecosystem without the seasonal fishing moratorium. The results of cumulative impacts analysis indicated that the biomass of most of the functional groups in the hypothetic 2008* ecosystem was more sensitive than that of the functional groups in the ecosystem with the seasonal moratorium. Approximately 17 of 24 functional groups decreased both in the 2008 and 2008* ecosystem, when effort increased by 10%. In addition to *Priacanthus macracanthus*, all of the groups showed a lower degree of sensitivity to the 10% increase in fishing effort in the actual 2008 ecosystem, such as *Sousa chinensis* (-0.1841, -0.3498*, it presents *Sousa chinensis* decreased by 18.41% in the actual 2008 ecosystem and 34.98% in the hypothetic 2008* ecosystem, the same below), sharks (-0.04473, -0.09485*), *Trachurus japonicus* (-0.6105, -0.6858*), *Trichiurus haumela* (-0.4985, -0.7218*), *Saurida* (-0.6794, -0.7529*), *Psenopsis anomala* (-0.5789, -0.7382*), *Upeneus bensasi* (-0.4471, -0.7347*), *Nemipterus virgatus* (-0.37, -0.6858*), *Priacanthus tayenus* (-0.5044, -0.6143*), other pelagic (-0.0939, -0.1683*), other demersals (-0.04993, -0.1317*), *Polychaetes* (-0.0043, -0.006299*) , mollusks (-0.1269, -0.1405*), echinoderms (-0.04704, -0.05477*), Cephalopods (-0.7509, -0.7529*), zooplankton (-0.01247, -0.02845*). The result suggest that functional groups in the actual 2008 ecosystem presented a lower susceptibility to the increase in fishing effort than did the hypothetical 2008* ecosystem, indicating that the ecosystem under the seasonal moratorium was more resilient to anthropic disturbance. In other words, the seasonal fishing moratorium appeared to be effective in protecting the ecosystem from external disturbance.

Additionally, the biomass of several small and low-trophic-level functional groups, such as *Decapterus maruadsi, benthic crustaceans, Jellyfish, phytoplankton, other zoobenthos and detritus,* increased in the actual 2008 ecosystem, as observed for the 2008* ecosystem. This increase affected the interactions throughout the entire food web, and suggests that the change in one species due to fishing will affect other species through food chain transfer. Therefore, the decrease in biomass of the large and high-trophic-level functional groups reduces predation pressure on small and low-trophic-level functional groups.

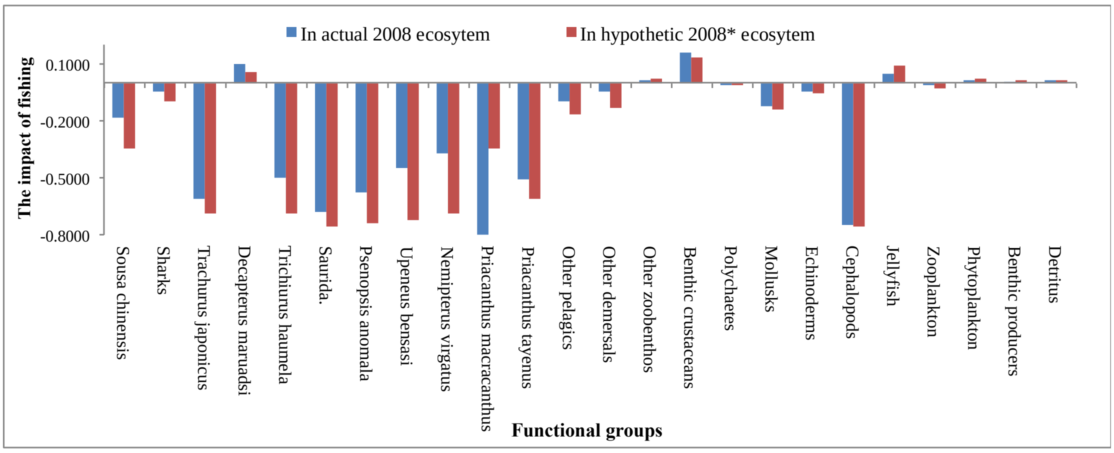

Fig.5 Cumulative effects of fishing gear on functional groups with effort increased by 10%

3.3 Policy simulation

The positive effect of the fishing moratorium on the ecosystem was confirmed, as indicated above. However, the aforementioned simulation was performed for a period lasting no more than 20 years. It was suggested that the duration of the fishing closure be prolonged. Moreover, it is generally considered that gillnets have placed pressure on fisheries and therefore should be banned during the moratorium season. Therefore, five scenarios were described in the methods section, were simulation in Ecosim based on the Ecopath model for 1998 to explore better fishing strategies as well as the long-time effect of the fishing moratorium policy on fish stocks. All simulations were executed under the condition of sustained fishing effort.

These scenarios were as follow: S0, continuation of the present fishing moratorium; S1, no fishing moratorium; S2, extension of the duration of the moratorium; S3, listing all fishing gear as banned; and S4, No fishing moratorium +fishing effort reduced by 50% for all gear.

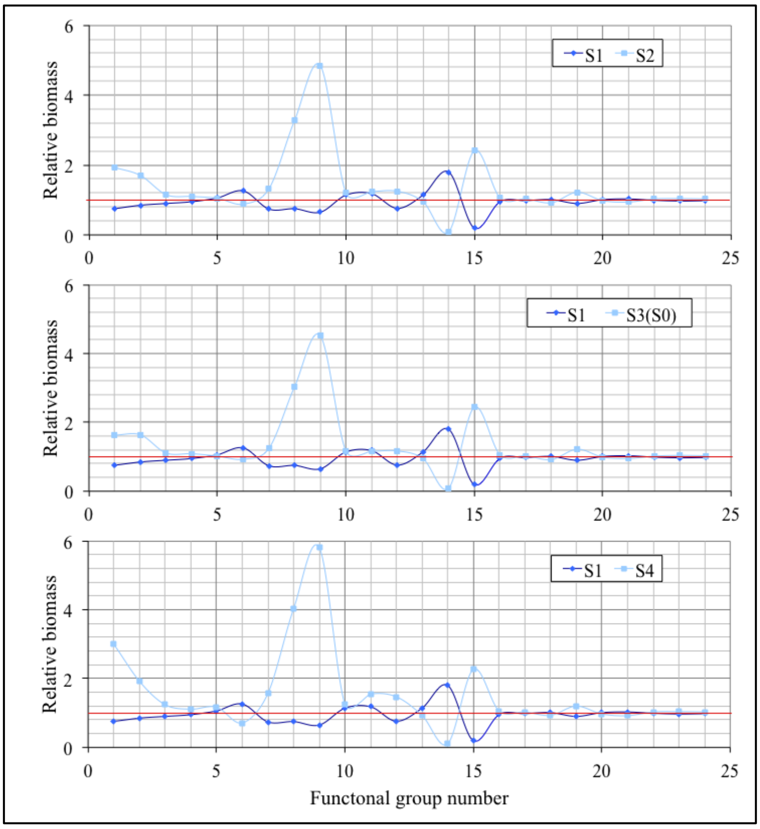

Fig.6 Responses of functional groups to various fishing scenarios

(The red line represents a baseline with the relative biomass equaling to 1. The functional numbers refer to the "methods and materials" section)

In Figure 6, all of the scenarios except for S1 surprisingly presented quite similar variation trends with respect to species abundance. Moreover, the variation trend of the species biomass was opposite of that observed in the other four simulated scenarios. As shown in Figure 6, the relative biomass values of most functional groups were below the baseline in S1, indicating that the stocks of most functional groups would shrink without the fishing moratorium, even if the fishing effort did not increase. In the other four scenarios, most of the functional groups appeared to have more or less recovery from over-exploitation. This was the case for most fish species (*Trachurus japonicus*, *Decapterus maruadsi*, *Trichiurus haumela*, *Psenopsis anomala*, *Upeneus bensasi*, *Nemipterus virgatus*, *Priacanthus macracanthus*, *Priacanthus tayenus*, and the other pelagic) and the predators at the top of the food chain (*Sousa chinensis* and Sharks). The benthic crustaceans and cephalopods also presented distinct increases in abundance. The prolonged moratorium duration (S2) seemed to be slightly more beneficial to stocks recovery than S0, during which fishing operations were carried out following the present fishing moratorium policy. However, banning all fishing operations during the moratorium season, which was the case in the S3 simulation, resulted in a nearly identical system output to that with S0. In the S4 simulation, there was a more prominent increase in abundance for most fish species than that observed in S0, S2 and S3. All of the above mentioned results indicate that the recovery of fishing stocks from over-exploitation exhibit similar tendencies either under the condition of intermittent control of fishing effort, such as the implementation of a fishing moratorium (as in S0, S2 and S3), or by reducing fishing effort in a straightforward manner (as in S4). Of course, it was assumed that there was no increase in fishing effort throughout the entire period. Furthermore, controlling fishing effort will not reduce the number of landings as assumed. The results indicate that the predicted number of landings in 100 years would rise to various extents in S0, S2, S3 and S4, in which fishing effort was controlled in different ways (Table 3). There were more prominent changes in the number of landings of all gears in S4 than in other scenarios. In contrast, the number of landings was predicted to decline after 100 years in S1, although no control over fishing effort was exerted. In Table 3, the landings of the trawl and the other gear appear to change. The landings changes of gill nets and hooks and lines were not abrupt. This type of difference in the landing changes among different gear may be related to the trend in the species stocks shown in Figure 6 through trophic interaction and fishing.

Comparative analysis of different scenarios indicated that the largest increase (28.0%) in fish stocks could be obtained in S4. Moreover, an increase (43%) in the total number of landings was observed. No distinct differences between S0 and S3 were observed.

Table 3 The landing changes ($Landing_{end}/Landing_{start}$) of different fishing gears in simulations

Gear	**S1**	**S2**	**S3 (S0)**	**S4**
Others	0.62	1.58	1.55	1.65
Trawl	0.75	1.43	1.40	1.48
Purse net	0.80	1.22	1.15	1.40
Gill net	0.92	1.17	1.13	1.29
Hook and line	0.92	1.17	1.13	1.28
Total	0.79	1.34	1.31	1.43

4. DISCUSSION

4.1 The advantage of the fishing moratorium

It is generally believed that the fishing moratorium achieved its purpose of protecting young fish aggregations, as the moratorium period coincided with the growth period of most of young fishes of various species. Based on the preliminary assessment for the first summer fishing moratorium in the South China Sea in 1999 (Liu and Chen, 2000) , during the first two month of opening closure, the fisheries' output increased by 30% from one year earlier. Fifteen years after the fishing moratorium was launched, a report on the summer fishing moratorium effect of Guangdong Province in the South China Sea in 2012 (Research group of summer fishing moratorium effect of Guangdong province in the South China Sea, 2014) suggested that the summer fishing moratorium effectively enhanced fishermen's awareness of protecting resources and the environment and played a very active role in fishery resource protection and restoration for the 15th consecutive year. For instance, the catch rate of the surveillance pair trawler in Gongguan (a city in the PRE) is 307.3 kg/h, more than double the catch rate in 1998 (150 kg/h), before the moratorium was executed.

In previous study (Cohen and Foale, 2013; Foale and Manele, 2004; Russ and Alcala, 1998), the fishing moratorium was believed to be more suitable for short-lived and fast-growing species than for long-lived and slow-growing species. Because most fish species in the PRE spawn annually in the summer, the main contribution of the summer fishing moratorium was to protect the spawning of importantly commercial fish. Recently, the average quality of catch has risen annually in each fishing areas of the South China Sea (Wu, 2008; Yang and Zhou, 2013), such as the yields of mackerel and cephalopods, which rose by 50% and 80% every year; moreover, yellow croaker, mackerel and hairtail significantly increased after the seasonal fishing moratorium. The annual spawning season for the majority commercial exploited species in the South China Sea is April to June, and the following months, July to September, represent a critical stage in juvenile development (Shi et al., 2008).

Therefore, the summer fishing moratorium in the PRE, which runs from 1 June to 1 August, increased the survival rate of juveniles and thereby promoted the stock. As such, the life history characteristics of multi-species, multi-gear harvests will be a critical factor impacting the efficacy of the fishing moratorium for fisheries management.

4.2 The disadvantage of the fishing moratorium

Debates regarding the effect of the fishing moratorium have not ceased since the policy was implemented. Pervious studies (Jiang et al., 2009) have indicated that the summer moratorium was only beneficial to certain stocks, but it did not play a positive role in the restoration of fish community structure and functional groups. For instance, the gill net fishery has not been included in the binding targets of the fishing moratorium; however, species such as pomfret and mackerel were commercially valuable species and have been confronted with more intense fishing pressure than other species. Moreover, they were the main target species for gill net fishing, which was not banned by the moratorium (Shi et al., 2008). Therefore, they were under intense fishing pressure, even during the moratorium

Additionally, it was suggested that this fishing restriction has injected more energy into the ecology, economy and society. Some research (Cohen et al., 2013; Murawski et al., 2005) suggests that periodically harvested closures represent a minor reduction in fishing grounds when they were closed, but when opened, they provide communities with an opportunity to boost fish catches to meet elevated social and economic demands. For example, during the first month of opening closure in Shenzhen City (Hou et al., 2009) and Maoming City (Li, 2005) of Guangdong Province in the PRE, the benefits of periodically harvested closures were completely removed under high fishing pressure, and the fishery output rapidly dropped to the levels observed during the same period before the fishing moratorium. Similar to other areas in the world, overall declines in target species populations occurred after fishing closure in Hawaii, indicating that the 1–2 year closure periods were too short for compensatory growth and reproduction (Williams et al., 2006).

Similar to many studies on the fishing moratorium in the East China Sea (Xu et al., 2003; Zhou, 2007) and South China Sea (Shi et al., 2008; Yang and Zhou, 2013), we found no limits on the volumes harvested during openings. High fishing pressures combined with a lack of restraint during harvests reduced the benefits gained from fishing moratorium strategies. Therefore, it is important to combine the fishing moratorium with other fishery management frameworks or, more generally, in contexts featuring mechanisms that limit or reduce fishing effort. Some examples of the currently employed fishing effort control include limited access, size limits, species bans, catch control and gear restrictions (Cohen and Foale, 2013).

5. CONCLUSION

Revisions of current management policy and tactics are needed to conserve fishery resources and biodiversity in the NSCS. Chinese fishery management authorities have recognized the current status of fishery resources and have initiated a range of fishery management policies. These policies include limiting new entry to fisheries and prohibiting the use of certain destructive fishing methods. However, the degree to which the regulations have been enforced has been questioned.

Two main reasons are usually advocated for use of seasonal fishing moratorium: reduction of effort and avoidance of disturbance during reproduction. Previous studies revealed that some target stocks have recovery gradually (Wu, 2008; Yang and Zhou, 2013). But how the status of the fish community has changed since the implementation of the moratorium was not clear. In this context, this study represents an important effort to investigate the effect of the fishing moratorium policy and to explore different management and conservation strategies using EwE to simulate the consequences of certain management measures on the ecosystem and evaluate changes in eutrophic conditions or address different aspects of fishing. To evaluate the effects of the summer fishing moratorium on the fish community, changes in the structure and function of the ecosystem in the PRE were examined based on scenarios simulation over the period from 1998 to 2008. A predicted ecosystem (2008*) was developed based on the hypothesis that summer closure was never applied to the PRE fisheries. The results of the comparisons of attribute indices between the actual 1998 ecosystem, the actual 2008 ecosystem and the hypothetical 2008* ecosystem indicated that the fishing moratorium policy was effective in providing ecosystem protection. However, it simple protects the ecosystem from exacerbation and accomplishes little with respect to ecosystem recovery. The PER ecosystem still face an increased risk of deterioration after ten years over exploitation. The cumulative impact analysis suggests that the ecosystem under summer closure shows more resilient to external anthropic disturbance. And the seasonal fishing moratorium had a beneficial effect on ecosystem protection and should be advocated and encouraged.

Evaluating alternative policy options and management scenarios can provide useful information on their relative advantages and disadvantages for authorities to make policy decisions. The comparative analysis of the different scenarios indicated that most functional groups would decrease without the execution of a fishing moratorium (S1). The recovery of fishing stocks from overexploitation will show a similar trend under the condition of intermittent control of fishing efforts through a variety of fishing moratorium policies (S0, S2, S3) as with reducing the fishing effort directly (S4). Furthermore, the prolonged moratorium (S2) seemed to be slightly more beneficial to fishing stock recovery than S0, during which the fishing operations were carried out following the present fishing moratorium policy. However, the result of banning all fishing operations during the moratorium season in the S3 simulation was not as expected: the system output with S3 was almost the same as with S0. Most importantly, reducing the fishing effort of all fishing gears by 50% (S4) led to the largest increase in both in the fish stocks (28.0%) and total landings (43%).

All of these factors indicated that the summer moratorium was only play a minor positive role in the restoration process of fish community structure and function in the PRE ecosystem. But in reality, the problem will be how to handle the inevitable fishing effort shifting from summer to other seasons, and annual fishing effort was not effectively cut down (Cohen et al., 2013; Jiang et al., 2009). Hence, to protect the health of the ecosystem, a comprehensive set of restrictions on fishing effort should be implemented in the fishery in the PRE coastal ecosystem.

ACKNOWLEDGEMENTS

This study is conducted under the project “An ecological computable general equilibrium (ECGE) model for marine fishery management: Linking ecosystem and socio-economic system” is supported by National Natural Science Foundation of China (Contract No.71303220), the project “Ecosystems, Societies, Consilience, Precautionary principle: Development of an assessment method of the societal costs for best fishing practices and efficient public policies” is funded by the EU Sixth Framework Program (Contract No.003711), and Science Foundation of Zhejiang Sci-Tech (ZSTU) (Grant No. 13092180-Y).

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