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Evaluating impacts of pulse fishing on the effectiveness of seasonal closure

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Abstract

Seasonal fishing closures are often used in fisheries management to conserve overfished stocks. As one of the unintended consequences, fishermen often contend for maximizing catches immediately after reopening fisheries. The resultant large catch landings in a short time period (i.e., pulse fishing) may undermine the benefit of closure. We implemented an end-to-end model OSMOSE-JZB (Object-oriented Simulator of Marine ecOSystem Exploitation OSMOSE) modelling ecosystem in the Jiaozhou Bay located in China to evaluate the impact of pulse fishing on the effectiveness of seasonal closure at levels of fish community, population, and individual. Our study demonstrated that the three-month closure was successful in conserving fish stocks. There were small variations on ecological indicators (i.e., total biomass of the community, mean trophic level of the community, mean trophic level of the catch, and Shannon-Wiener biodiversity index) when pulse fishing occurred. Pulse fishing seemed not to result in a great shift in community structure. Compared to other species, the biomass of two large predatory fishes were more susceptible to pulse fishing. Pulse fishing could change the pressure of predators to fish stocks via food webs, especially for young individuals. Our simulations indicate that we can improve the effectiveness of seasonal closure by managing pulse fishing. Although the results derived in this study may be specific to the target ecosystem, the general approach is applicable to other ecosystems when evaluating fishing impacts.

Key words: OSMOSE, pulse fishing, seasonal closure, Jiaozhou Bay

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1 Introduction

Overfishing is a great threat to sustainable utilization of fisheries resources (Coll et al., 2013). As one of the most advocated management tools, seasonal closures can bring a reduction of fishing effort conducive to recovering fish stocks, and aim to benefit both ecosystems and fisheries socio-economics (Clarke et al., 2015; Samy-Kamal et al., 2015). Unrestrictive fishing activities can undermine the benefits from the fishing closure (Murawski et al., 2005). The large catch landing in a short time period can cause irreversible harm to fish stocks (Purcell et al., 2013) and incidental bycatch has a serious impact on food webs and ecological structure (Fulton et al., 2011). In practice, fishermen may change their harvest strategies in order to compensate the loss of landings resulted from the fishing moratorium (Wang et al., 2015). As the recognized importance of fishermen's behaviors in determining the effectiveness of fisheries management (van Putten et al., 2012), it is necessary to evaluate the impacts of temporal variations in fishing effort after seasonal fishing closure.

There is increasing attention to small-scale fisheries because of their contribution to food supply and economic income (Weeratunge et al., 2014). The fishermen's decisions, such as timing for entering or exiting fishing ground, are closely related to fish stock state (Kiyama and Yamazaki, 2018). Pulse fishing is a fishing strategy that lands large catches in a short time period and lets the stock grow in other periods (Da-Rocha et al., 2014). The summer moratorium of fishing has been implemented over twenty years across the coastal waters in China. Fishermen seek to maximize commercial profits resulting in an especial pulse fishing that substantial fishing effort concentrates immediately after the end of the seasonal closure. This management strategy seems not to prevent the depletion of natural resources resulting from high fishing effort. There are increasing reports on overexploited fisheries and disappeared fishing grounds (Jiang et al., 2009; Zhu, 2009). Many studies attribute this failure to high fishing effort during fishery openings (Wang, 2008; Shen and Heino, 2014). The effectiveness of the seasonal closure is closely tied to the seasonality of fishing effort during the harvest season (Cohen et al., 2013; Ichinokawa et al., 2015). Therefore, we hypothesize

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that whether it is possible to improve the effectiveness of seasonal closures by managing pulse fishing.

There is a worldwide consensus that ecosystem modelling approaches are a necessary complement to single-species approach by supporting fisheries management with the incorporation of ecosystem consideration (Hilborn, 2011). Ecosystem models can provide insights into variations of ecosystem under different management strategies, which can help optimize decision-making and avoid potentially irreversible harm on fish stocks (Fogarty, 2014). This study focuses on the small-scale fishery in the Jiaozhou Bay, China where has implemented the seasonal closure to conserve fish resources. Using an end-to-end model "OSMOSE-JZB" (Object-oriented Simulator of Marine ecOSystem Exploitation OSMOSE) previously developed for the Jiaozhou Bay (Xing et al., 2017), we simulate ecosystem dynamics from low trophic level (LTL) to high trophic level (HTL) in an individual-based way that helps us understand the change of different ecological components after pulse fishing arising.

There are four typical pulse fishing scenarios considered here based on different patterns of pulse fishing occurring in other coastal waters of China. The impact of pulse fishing on the effectiveness of seasonal closure were systematically evaluated at the levels of fish community, population, and individual. Besides, a scenario without seasonal closure was simulated used for quantifying the effectiveness of seasonal closure. Given the varying nature of fishing effort in realism ecosystem, an uncertainty analysis was conducted to examine the impacts of uncertain fishing effort on the model prediction and the robustness of our results. Our findings can improve our understanding on how pulse fishing influences the effectiveness of seasonal closure in a semi-closed bay ecosystem. This study can facilitate the development of management strategies for sustainable fisheries and the application of ecosystem modelling approach in tactical management.

2 Materials and methods

2.1 *Study area and data*

The Jiaozhou Bay is a semi-enclosed bay located on the south of Shandong Peninsula in China (Fig. 1). Its trophic structure has shifted greatly over the last two decades because of long-term heavy fishing activities (Han et al., 2015). The seasonal closure has been implemented over twenty years to improve the fisheries status. As there are few regulations constraining pulse fishing, it is essential to investigate whether pulse fishing impacts the effectiveness of the seasonal closure in the Jiaozhou Bay. An operational model OSMOSE-JZB was developed to simulate the ecosystem dynamics in the Jiaozhou Bay based on data collected from stratified random bottom trawl surveys in February, May, August, and November of 2011.

2.2 *The operational model: OSMOSE-JZB*

The OSMOSE-JZB consists of two main sub-models that describe two main biotic components: LTL groups and HTL groups. The spatio-temporal dynamics of LTL groups were simulated by hydrological model (the Finite Volume Coastal Ocean Model FVCOM; Chen et al., 2003) coupled with biogeochemical model (the North Pacific Ecosystem Model Used for Regional Oceanography NEMURO; Aita et al., 2007). The dynamics of trophic interactions and full life cycle for the HTL groups were described by OSMOSE model developed by Shin and Cury (2001). The model builds the food web, including five LTL groups and fourteen HTL groups, based on a one-way coupling approach that the distribution of LTL groups is just provided as a prey field for HTL groups. The detailed inputs, parameterization and calibration of the model can be found in Xing et al. (2017). We provide here a brief description of model structure and fishing configuration for each scenario. The detailed information of fourteen HTL groups are listed in Table 1.

OSMOSE is a multi-species individual-based model assum-

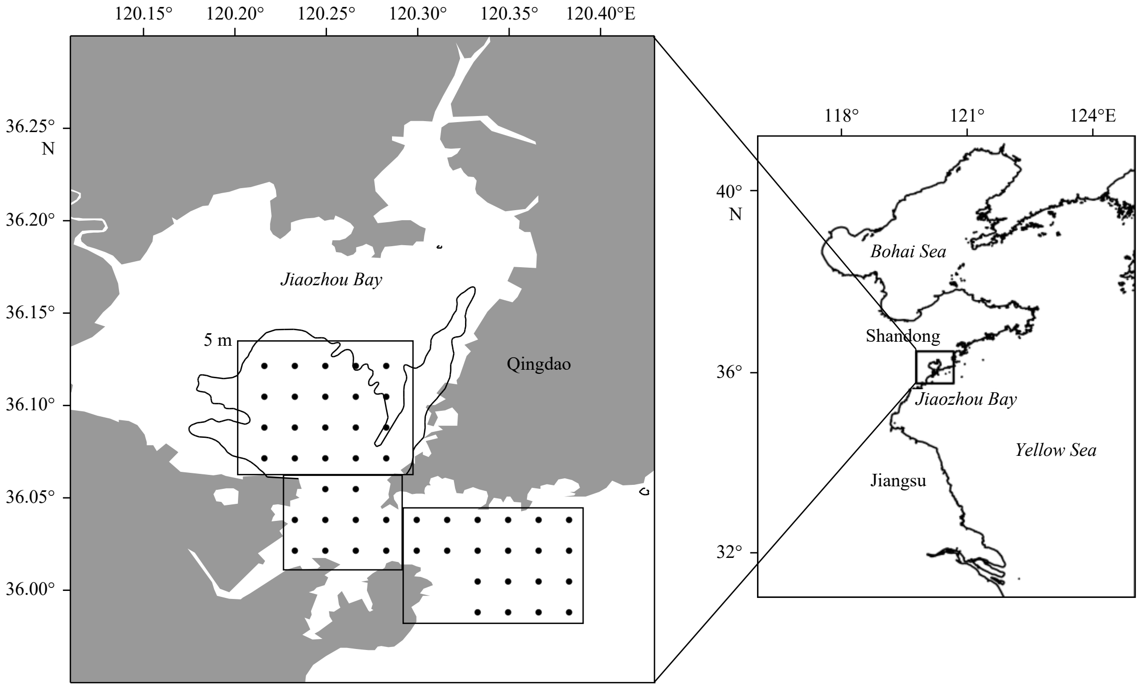

Fig. 1. Location and sampling design of survey in the Jiaozhou Bay, China. Stratified random bottom trawl surveys were conducted in February, May, August, and November of 2011.

Table 1. The fourteen high trophic level (HTL) groups modelled explicitly in the OSMOSE-JZB model

Type	HTL groups		Time steps for moving out of simulated domain	F_{annual}	
				(F0–F4)	(F5)
1	SP0:	*Oratosquilla oratoria* (Japanese mantis shrimp)	null	0.420	0.560
	SP1:	*Palaemon gravieri* (Chinese ditch prawn), *Parapenaeopsis tenella* (Smoothshell shrimp), *Alpheus japonicus* (Japanese snapping shrimp)	null	0.365	0.486
	SP5*:	*Octopus* sp. (Octopus)	null	0.350	0.467
	SP6:	*Sebastes schlegelii* (Korean rockfish)	null	0.482	0.643
	SP8:	*Amblychaeturichthys hexanema* (Pinkgray gody)	null	0.229	0.306
	SP11:	*Johnius belangerii* (Belanger's croaker)	null	0.411	0.548
	SP13:	*Cynoglossus joyneri* (Red tonguesole)	null	0.365	0.486
2	SP2*:	*Charybdis bimaculata* (Two-spot swimming crab)	0–5	0.314	0.470
	SP4*:	*Loligo* sp. (Squid)	0–5	0.419	0.628
	SP9*:	*Thryssa kammalensis* (Kammal thryssa)	0–5	0.467	0.700
3	SP3*:	*Charybdis japonica* (Japanese swimming crab)	0–11	0.246	0.369
	SP12*:	*Trachypenaeus curvirostris* (Southern rough shrimp)	0–11	0.426	0.638
4	SP7*:	*Pholis fangi* (Gunnel)	12–17	0.426	0.480
5	SP10*:	*Liparis tanakae* (Tanaka's snailfish)	0–5, 12–17	0.488	0.586

Note: The common name of each HTL group showed in the bracket. There were 24 time steps in a simulated year numbered from zero to 23. F_{annual} was annual fishing mortality summing up fishing mortality rate at each time step. The migratory species marked with *, and others were sedentary species that spent all life in the Jiaozhou Bay. In contrast to other migratory species, *Octopus* sp. migrated into or outside the study area at a given age, and could be available to fishing all year round similar to sedentary species.

ing that predation is opportunistic based on fish size and spatial co-occurrence. The fish school is a basic unit in the model, and the model assumes that the process of predation is opportunistic and size-based (Shin and Cury, 2004). There are six crucial processes in each time step (half a month), including: (1) the spatial distribution and random-walk movement mimicking the dispersal of fish (Halouani et al., 2016); (2) opportunistic predation based on spatial co-occurrence and size adequacy between prey and predators (Travers et al., 2009); (3) various mortalities (e.g., predation mortality, fishing mortality, starvation mortality, additional natural mortality) applied simultaneously based on the stochastic algorithm method making all mortalities stochastic and competitive that avoids the bias of mortality outcome (Grüss et al., 2016b); (4) growth calculated using von Bertalanffy growth model and weight-length relationship when individuals ingest enough food; (5) species-specific reproduction depending on sex ratio, spawning biomass, relative fecundity, and the percentage of eggs produced at each time step; and (6) migratory species migrating into or outside the simulated area (Xing et al., 2017).

The food web built by the OSMOSE-JZB covers most of components in the Jiaozhou Bay, and is proved that this model can capture ecosystem dynamics there (Xing et al., 2017). The summer moratorium of fishing from June 1 to September 1 is included in the model, and fishing effort is assumed to be evenly applied to the fisheries during the fishing season. In other words, fishing mortality rate of each HTL group set in the model is uniform in fishing season. Individuals are harvested when they reach the size of recruitment into fisheries, which is set at 5 cm for all HTL groups.

2.3 *Simulation scenario*

This study focuses on the evaluation of impacts of potential pulse fishing on the ecosystem of the Jiaozhou Bay where fishing is prohibited from June to August each year. The fishing mortality rate (i.e., M_{fishing}) was used to indicate impacts of fishing effort on fish individuals. We assumed the fishing effort was directly proportional to fishing mortality rate, and changed the M_{fishing} at each time step to imitate different scenarios. According to relevant studies on potential pulse fishing situations in China (Chen, 2007; Zhu, 2009; Shen and Heino, 2014), there were six scenarios (i.e., F0–F5 below) considered here:

(1) F0: The configuration of fishing mortality rate is the same to the initial OSMOSE-JZB model built by Xing et al. (2017) (Fig. 2a).

(2) F1: Fishing mortality rate peaked at a maximum at the beginning of the fishing season. Afterwards it dropped abruptly, and then remained at a steady level until the end of the fishing season (Fig. 2b).

(3) F2: Fishing mortality rate reached a maximum, at a lower level than in F1, during the start of the fishing season, and then declined to reach a low stable level until the end of the fishing season (Fig. 2c).

(4) F3: Fishing mortality rate went up first with a peak in late autumn, and then dropped in subsequent time. Similarly to F1 and F2, fishing mortality rate remained at a low level from the start of the next year to the end of the fishing season (Fig. 2d).

(5) F4: Fishing mortality rate peaked twice, at the start and at the end of the fishing season (Fig. 2e).

(6) F5: The season closure was revoked. The additional fishing effort was imposed to HTL groups in June, July, and August. The fishing mortality rate followed a uniform distribution (Fig. 2f).

The first scenario (i.e., F0) was considered as the reference/base state representing the ecosystem of the Jiaozhou Bay without pulse fishing. The subsequent three scenarios (i.e., F1–F3) simulated pulse fishing starting early in the fishing season with three different trends. Scenario F4 simulated pulse fishing occurring before the beginning and after the end of seasonal closure. The last scenario (i.e., F5) simulated the situation that removed seasonal closure, and aimed to quantify the effectiveness of seasonal closure by comparing with the scenario F0.

Some species included in the model might migrate out of the study area at a given time in a year as part of the life history (Xing et al., 2017). Migratory species were assumed to be not harvested during the seasonal closure or periods outside simulated area. Subsequently, HTL groups were divided into five types on the basis of time they stayed in the simulated domain (Table 1). The detailed temporal distribution of fishing mortality rates and annual fishing mortality rate (i.e., F_{annual}) in each scenario were shown in Fig. 2 and Table 1, respectively. Our study only investig-

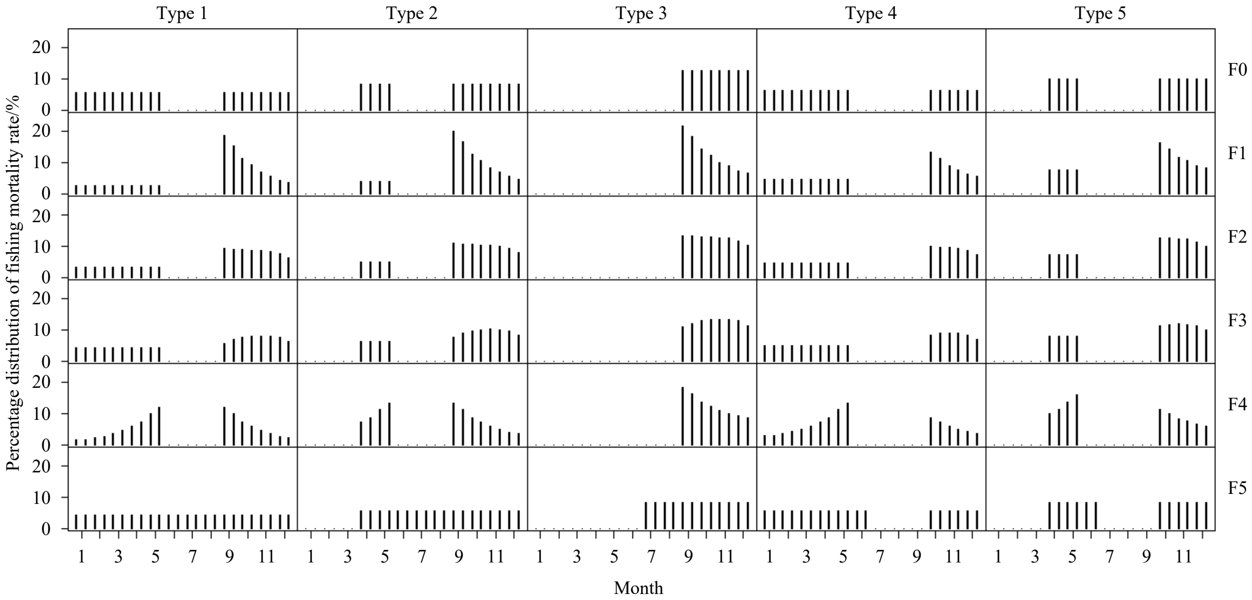

Fig. 2. Percentage distribution of fishing mortality rate for fourteen HTL groups that were divided into five types set explicitly in six scenarios. The sum of percentages for each HTL group at each time step was equal to 100% in a simulated year (24 time steps). The fishing mortality rate at each time step calculated dependent upon corresponding percentage ratio and annual fishing mortality rate. A seasonal closure implemented from June to August in scenarios F0–F4, and revoked in scenario F5.

ates the influence caused by pulse fishing rather than the magnitude of fishing effort. The annual fishing mortality rates of each HTL group in four pulse fishing scenarios (i.e., F1–F4) were kept the same to scenario F0. In scenario F5, fish populations were assumed to confront with an identical fishing pressure in June, July, and August as other period of fishing. The fishing mortality rate of HTL groups were evenly assigned to each time step when they stayed in the Jiaozhou Bay. Thus, annual fishing mortality rates in scenario F5 were higher than those in other scenarios.

Each scenario had a burn-in period of 80 years to ensure that the model was in a stable state with the same input configuration to the initial OSMOSE-JZB model, and then ran the model for another 40 years with configuration set in each scenario. Given the stochasticity incorporated in the model (Grüss et al., 2016a; Fu et al., 2017), each scenario was ran 100 times, and the outputs were averaged over the 100 simulation runs. The OSMOSE version used here is v3.2 coded in JAVA, and available online (http://www.osmose-model.org/).

2.4 *Uncertainty analysis*

As a result of observation and/or management implementation errors, we made an analysis on the consequences of uncertainty in fishing effort for the first five fishing scenarios (i.e., F0–F4). The new temporal distribution of fishing mortalities for all species was constructed for each scenario with a 10% level of uncertainty assigned to their values. The errors of fishing mortality rate in each time step were randomly generated by a Monte Carlo method and followed a uniform distribution with bounds related to their level of uncertainty (Han et al., 2017). Therefore, the sum of fishing mortality rate at each time step (i.e., annual fishing mortality rate) of each HTL group could vary between 0.9 and 1.1 times the initial value. There were 100 new constructed temporal distributions of fishing mortalities generated for all species in each scenario. Each of the above simulations was repeated for 12 times considering the stochasticity of the model, meaning that there were 1 200 simulations ran for each fishing scenario.

2.5 *Measurement of fisheries status*

The relative change of fisheries status were evaluated at levels of community, population, and individuals. The variations of fish community structure were described by four common ecological indictors relating to unexploited level (i.e., total biomass of the community), community trophic composition (i.e., mean trophic level of the community), capturing “fishing down marine food webs” (i.e., mean trophic level of the catch), and diversity (i.e., Shannon-Wiener biodiversity index) (Fulton et al., 2005; Shannon et al., 2009). Two population-based indicators (i.e., species biomass and mean body length) were used to measure the response of different fishes to different harvest strategies in terms of biomass and size structure of populations. At the level of individual, fishing mortality rate (i.e., M_{fishing}) stood for direct impact of fishing on individuals of two age groups, which were represented juveniles (below 1 year old) and adults (from 1 to 2 years old), respectively. The variations of predation mortality rate (i.e., $M_{\text{predation}}$) of individuals from two age groups showed how altered fishing activities changed interspecies interactions. The indicator M_{sum}, summing up relative predation and fishing mortality rates, was used to quantify the change of combined pressures from predators and fishing when pulse fishing arose (i.e., F1–F4) or seasonal closure was revoked (i.e., F5) (Table 2).

3 Results

3.1 *Relative changes at community level*

There were small differences of all ecological indicators between pulse fishing scenarios (i.e., F1–F4) and reference state (i.e., F0). Pulse fishing seemed not to result in a marked change of community structure (Fig. 3). The H' had an upward trend in scenario F1. In scenario F4, all ecological indicators increased when pulse fishing occurred. The H' was more susceptible to pulse fishing compared with other ecological indicators. In contrast to pulse fishing scenarios, all ecological indicators in scenario F5 declined after imposing additional fishing effort to fish

Table 2. Definitions of indicators that characterize fish community, population, and individual

Level	Indicators	Symbol	Definition	Sources
Community	total biomass of the community	Bio_{com}	$Bio_{\text{com}} = \sum_i Bio_i$, where Bio_i denotes the biomass of a given species i in the simulated area	
	mean trophic level of the community	mTL	$mTL = \sum_i TL_i \times \frac{Bio_i}{Bio_{\text{com}}}$, where Bio_i denotes the biomass of a given species i; Bio_{com} indicates the total biomass of the community; TL_i represents the mean trophic level of species i, and its computational method was the same as in Xing et al. (2017)	Travers et al. (2006), Xing et al. (2017)
	mean trophic level of the catch	mTL_{catch}	$mTL_{\text{catch}} = \sum_i TL_i \times \frac{Bio_{\text{catch},\,i}}{Bio_{\text{catch, com}}}$, where $Bio_{\text{catch},i}$ denotes the biomass of the given species i in catch; $Bio_{\text{catch, com}}$ indicates the total yield; TL_i represents the mean trophic level of a target species i, and its computational method was the same as in Xing et al. (2017)	Travers et al. (2006), Xing et al. (2017)
	Shannon-Wiener biodiversity index	H'	$H' = -\sum_i P_i \times \ln(P_i)$, where P_i denotes the proportion of the biomass of a given species i in the community	Travers et al. (2006)
Population	species biomass	Bio_i	The total biomass of a species i in the simulated area	
	mean body length	$\bar{L}$	$\bar{L} = \sum \frac{Lj}{N_i}$, where L_j is the body length (cm) of an individual fish j and N_i is the number of the species i in the simulated area	Shin et al. (2005)
Individual	predation mortality rate	$M_{\text{predation}}$	provided in outputs of the model	
	fishing mortality rate	M_{fishing}	provided in outputs of the model	
	relative change of two mortality rates	M_{sum}	the sum of relative fishing and predation mortality rates between pulse fishing scenarios and reference state	

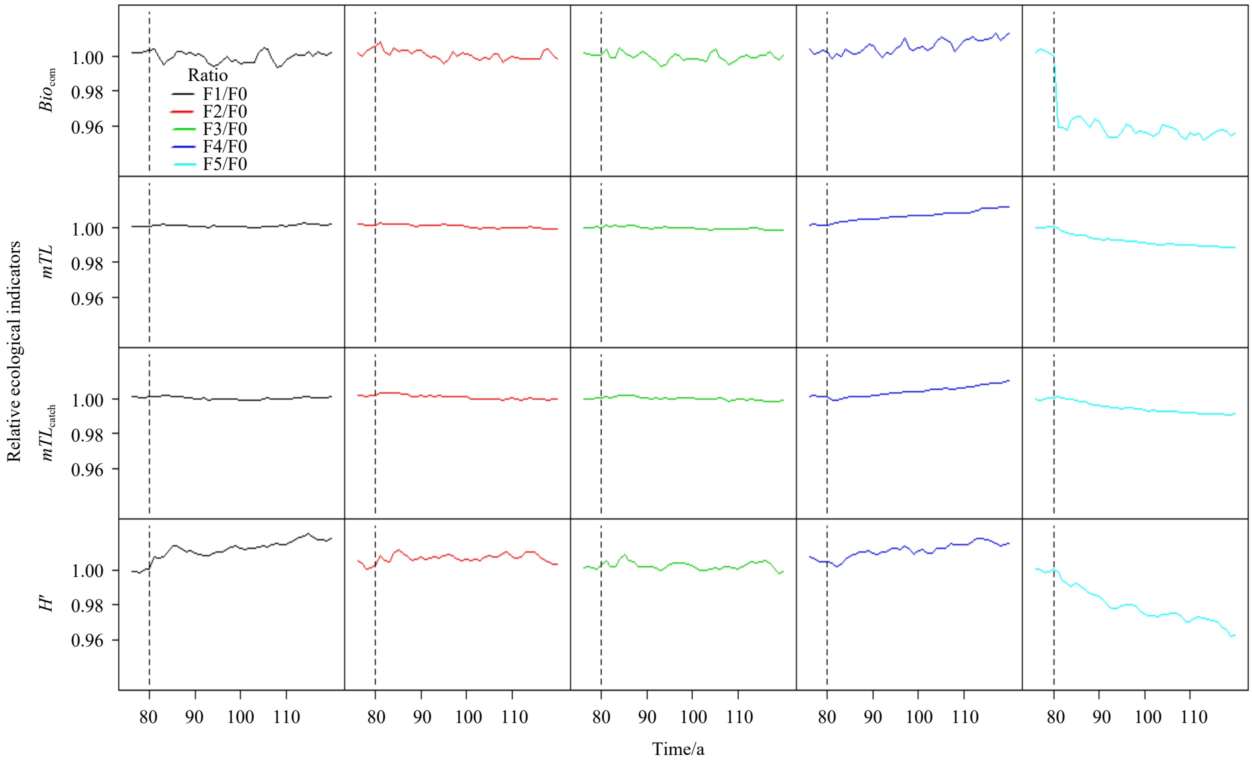

Fig. 3. Relative average annual changes in four ecological indicators, i.e., total biomass (Bio_{com}), mean trophic level of the community (mTL), mean trophic level of the catch (mTL_{catch}) and Shannon-Wiener biodiversity index (H'), of 14 HTL groups after the 75th year. The ratios of four indicators between last five fishing scenarios (i.e., F1–F5) and reference state (i.e., F0) were represented for relative changes. The predicted data are averages of 100 repeated simulations.

stocks in June, July, and August (Fig. 3). It implied that the three-month seasonal closure effectively protected fish community from further eroding by fishing activities.

3.2 *Relative changes at population level*

The two large predatory fishes, *S. schlegelii* (SP6) and *J. belangerii* (SP11), were more sensitive to pulse fishing, and other species fluctuated around the reference predictions. The biomass of *S. schlegelii* declined in front three pulse fishing scenarios, but increased significantly in scenario F4. The biomass of *J. belangerii* increased except in scenario F3 for which the biomass was close to that in F0. Similar to predictions in pulse fishing scenarios, both of them showed a stronger response to additional fishing effort in June, July, and August, and declined to a lower level than other scenarios (Fig. 4).

The annual mean length of each HTL group varied less than species biomass, indicating that the modelled species size structure did not display a big shift. Mean body lengths of six species were sensitive to varying fishing effort: (1) For the first three pulse fishing scenarios (i.e., F1–F3), the mean body lengths of three species groups, i.e., small shrimp groups (SP1: *P. gravieri*, *P. tenella*, and *A. japonicus*), *J. belangerii*, and *C. joyneri* (SP13), were smaller than those for the reference state, however, the mean length of *S. schlegelii* was greater; (2) *O. oratoria* (SP0) showed a downward trend in scenarios F1 and F5; (3) in contrast to *S. schlegelii* and *J. belangerii*, the mean length of small shrimp groups, *Octopus* (SP5*), and *C. joyneri* increased in scenario F4; and (4) the proportion of small individuals for the small shrimp groups *A. hexanema* (SP8) in F5 increased contrary to *S. schlegelii* and *C. joyneri* (Fig. 5).

3.3 *Relative changes at individual level*

Pulse fishing could change the combined effects of the two mortalities and have greater impacts on the predation pressure exerted upon juveniles than upon adults. This suggests that fishing activities could have indirect impacts on juveniles via trophic interactions. There was a decline in the predation mortality of *S. schlegelii* (SP6) juveniles in July and August in F4 (Fig. 6), accompanied by an increased population biomass (Fig. 4). By contrast, the predation pressure on juveniles increased with subsequent decreased population biomass under other pulse fishing scenarios. This indicated that the survival of juveniles was crucial to fish repopulation and fishing activities were capable of influencing fish stock status during the closed fishing season as well. Although there were small variations on their fishing mortalities due to a low level of fishing pressure, the predation mortality of both age groups of *A. hexanema* (SP8) changed, implying pulse fishing indirectly influenced individuals through trophic interactions. *Johnius belangerii* (SP11) had a similar response to the fishing scenario F4. However, the biomass of *J. belangerii* increased in scenarios F1 and F2 (Fig. 4) with multilevel declines in predation mortality on juveniles.

In scenario F5, the pressure of predators to juveniles of three fishes decreased in most months after removing the three-month seasonal closure (Fig. 6), by contrast, there were small changes of predator pressures on adults (Fig. 7). The additional fishing effort in August had a bigger impacts on young *S. schlegelii* than other two fishes (Fig. 6). Adults of *S. schlegelii* and *J. belangerii* were underwent higher fishing pressures from June to August (Fig. 7).

3.4 *Uncertainty analysis*

In the uncertainty analysis, the biomass of *S. schlegelii* (SP6) and *J. belangerii* (SP11) had the same variation trends to simulated outcomes for the five scenarios (Fig. 8): (1) both fish species reached a stable biomass in scenario F0; (2) the biomass of *S. schlegelii* declined in the subsequent three scenarios (F1, F2 and F3), and increased in the last scenario (F4); and (3) the biomass of *J. belangerii* increased in scenarios F1 and F4, and had small variations in scenarios F2 and F3.

4 Discussion

Some closures are not effective on biological conservation due to cursory design and evaluation (Ichinokawa et al., 2015).

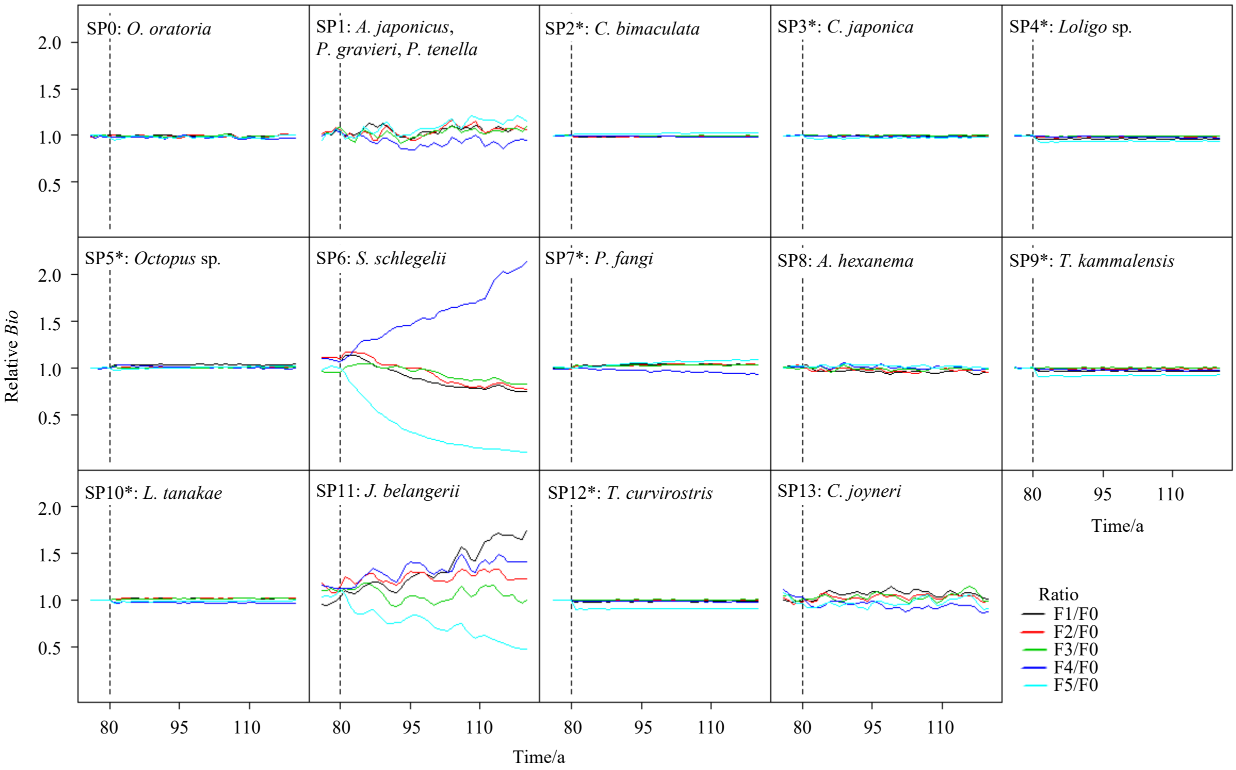

Fig. 4. Relative average annual changes in biomass of 14 HTL groups after the 75th year. The relative changes were represented by the ratios of species biomass between last five fishing scenarios (i.e., F1–F5) and reference state (i.e., F0). The predicted data are averages of 100 repeated simulations. * Migration species.

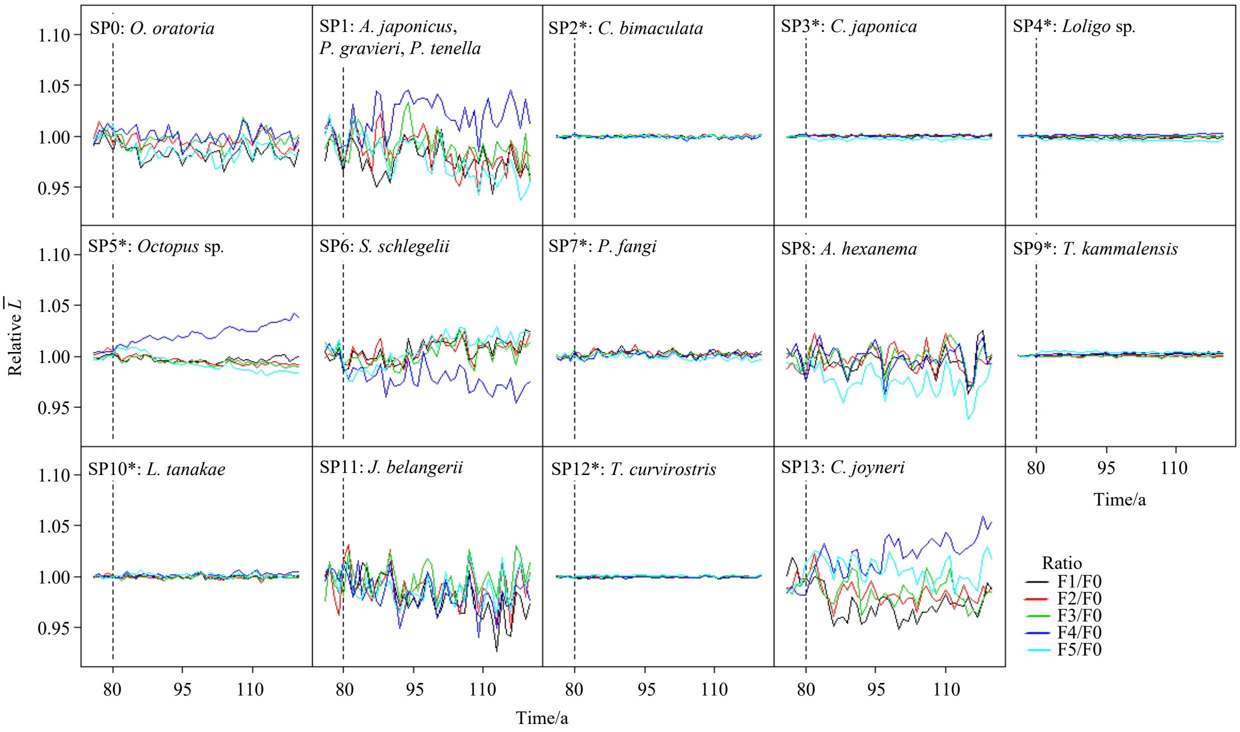

Fig. 5. Relative average annual changes in mean body length ($\bar{L}$) of 14 HTL groups after the 75th year. The relative changes were represented by the ratios of species-based mean body length between last five fishing scenarios (i.e., F1–F5) and reference state (i.e., F0). The predicted data are averages of 100 repeated simulations. * Migration species.

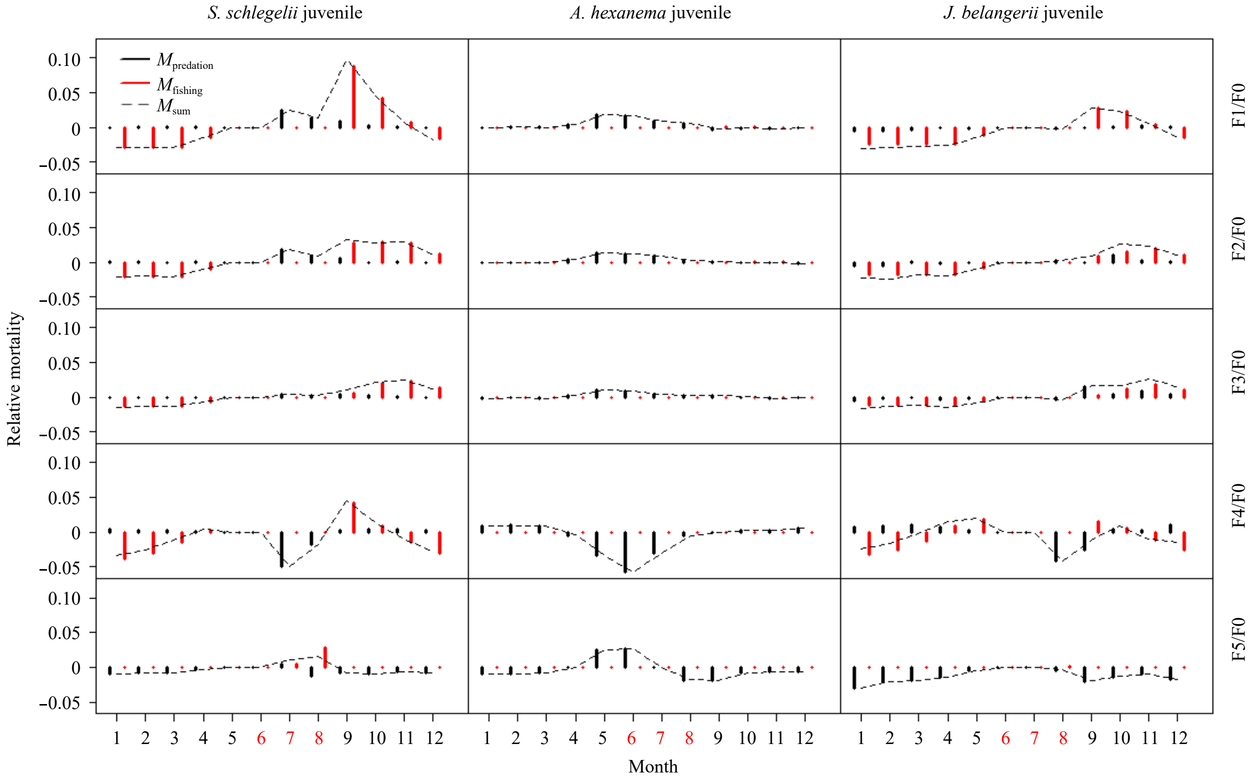

Fig. 6. Relative monthly mean changes of predation ($M_{\text{predation}}$, black bars) and fishing mortality (M_{fishing}, red bars) rates of young *S. schlegelii* (SP6), *A. hexanema* (SP8), and *J. belangerii* (SP11). The relative changes were the difference between predicted mortality rates in first five scenarios (i.e., F1–F5) and reference state (i.e., F0). The indicator M_{sum} (black broken line) was the sum of relative changes of $M_{\text{predation}}$ and M_{fishing}. The predicted data were averaged over 100 repeated simulations and over the last 40 years of the simulations. A seasonal closure implemented from June to August marked in red in scenario F0–F4, and revoked in scenario F5.

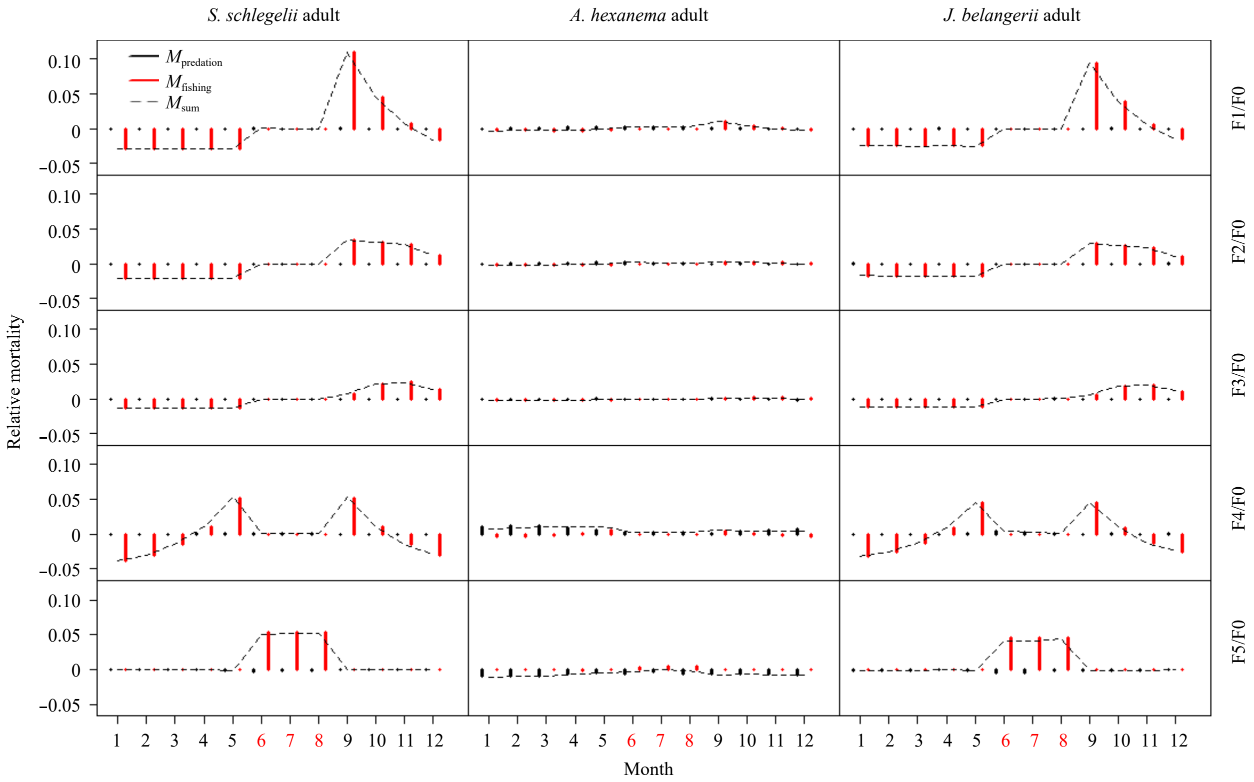

Fig. 7. Relative monthly mean changes of predation ($M_{\mathrm{predation}}$, black bars) and fishing mortality (M_{fishing}, red bars) rates of adults of *S. schlegelii* (SP6), *A. hexanema* (SP8), and *J. belangerii* (SP11). The relative changes were the difference between predicted mortality rates in first five scenarios (i.e., F1–F5) and reference state (i.e., F0). The indicator M_{sum} (black broken line) was the sum of relative changes of $M_{\mathrm{predation}}$ and M_{fishing}. The predicted data were averaged over 100 repeated simulations and over the last 40 years of the simulations. A seasonal closure implemented from June to August marked in red in scenario F0–F4, and revoked in scenario F5.

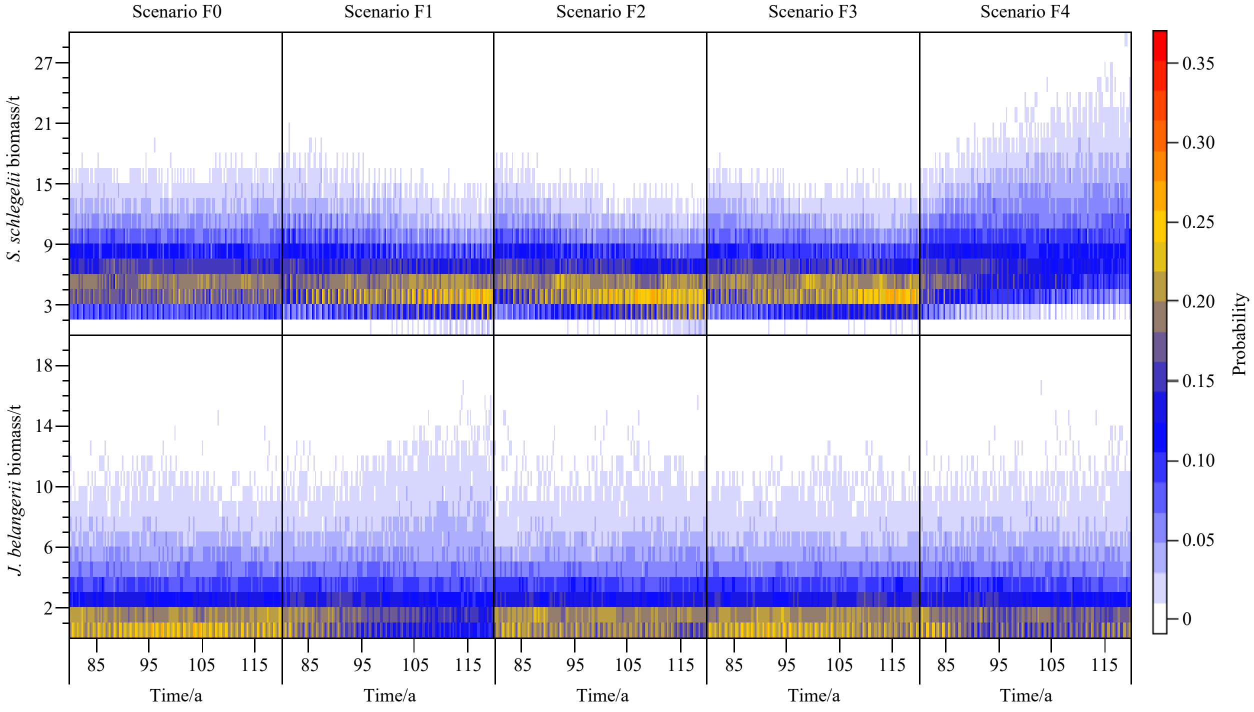

Fig. 8. Annual probability frequency distribution of *S. schlegelii* (SP6) and *J. belangerii* (SP11) population biomass under five fishing scenarios after the 80th year, with 1 200 Monte Carlo simulation runs and 10% level of uncertainty specified for fishing effort of all HTL groups at each time step.

The large predatory fish are often exposed to a high fishing pressure, and the slow growth makes them need a long time to recover from the heavy fishing pressure (Foale and Manele, 2004; Myers and Worm, 2005). They are often at a disadvantage against small pelagic forage fish when the closed fishing season is short (Williams et al., 2006; Cohen and Foale, 2013). Our results showed that two large predatory fishes with small populations (*S. schlegelii* and *J. belangerii*) got more benefit from the temporal buffer for compensatory growth of adults provided by the summer moratorium of fishing in the Jiaozhou Bay compared to others (Fig. 4). The stocks of *S. schlegelii* and *J. belangerii* declined markedly after revoking the fishing closure (Fig. 4), even though the pressure of predators to their juveniles mitigated (Fig. 6). It implied that benefits from reduced predators did not compensate the loss of adults from fishing. Similar to other study (Clark et al., 2015), our findings suggest that the protection of matured individuals is important to improving productivities of fish resources.

Comparing to other species, large catches in a short time (i.e., pulse fishing) can markedly influence the effectiveness of seasonal closure in recovering two large predatory fishes (*S. schlegelii* and *J. belangeri*). But the responses of *S. schlegelii* and *J. belangeri* to pulse fishing were different due to the species-specific growth traits. For the first three pulse fishing scenarios (i.e., F1–F3), pulse fishing occurring at the beginning of fishing season aggravated the decline in the *S. schlegelii* population. Conversely, the large loss of *J. belangerii* in September and October seemed to be compensated by the growth of stock in other periods. This was mainly attributed to the intense fishing pressure at the beginning of fishing season had a bigger negative impact on the stock recruitment of *S. schlegelii* compared to *J. belangerii* (Fig. 6). As what was reported by Xing et al. (2017), the proportion of young *S. schlegelii* which were longer than the size at first catch was higher than that of *J. belangerii* in September and October due to their specific growth traits. *S. schlegelii* juveniles were more likely to harvest when pulse fishing occurred after the seasonal closure. This likewise resulted in a decrease in matured individuals that weakened the "top-down control" feedback (Vergnon et al., 2008).

For scenario F4, more juveniles of *S. schlegelii* and *J. belangerii* survived by means of an intense fishing pressure in May lowering the pressure of predators (Fig. 6). The subsequent fishing closure provided a respite that favored their further growth. The high fishing effort imposed to spawning stock seemed not to result in the decline of fish population. Our simulations demonstrated that the performance of seasonal closure could be improved by managing pulse fishing. The small fish *A. hexanema* seemed to have the resistibility to pulse fishing. This was mainly attributed to a low fishing pressure. There were small variations on the predation and fishing mortality of adults in the four pulse fishing scenarios (F1–F4) (Fig. 7). It suggests that the species, which are imposed to a high fishing intensity, have a priority to monitor when pulse fishing occurs.

Various indicators have been developed to characterize exploited ecosystems (Shin et al., 2010, 2012; Moffitt et al., 2016), but their responsiveness and performance should be well understood before advising fisheries management (Blanchard et al., 2010). This study provided some insights on the utilization of indicators in assessing the fisheries status. When the annual fishing pressures imposed to fish stocks were unchanged, the four ecological indicators (Total biomass, mean trophic level of community, mean trophic level of catch, and Shannon-Wiener index) were not sensitive to changes caused by pulse fishing. This was mainly attributed that pulse fishing had a small impact on dominant species populations. In addition to this, sensitive fishes, such as *S. schlegelii* and *J. belangerii*, might have contrary responses to pulse fishing. This resulted in an offset that influenced the performance of ecological indicators on investigating fishing impacts. None of size-based indicators can singly reflect the response of ecosystem to fishing activities (Shin et al., 2005). Compared to species biomass, the mean body length of fish population was difficult in capturing the change after pulse fishing occurring. This suggested that species biomass had a high priority to be included in the monitoring programs. Since it is indeed impossible to survey every fish stock as money and time are limited, this ecosystem modelling approach can help us define the priority of species in monitoring programs.

Most of ecosystem models are applied in strategic management (Zhang et al., 2015; Collie et al., 2016), and decision makings provided by these models are more easily influenced by varying degrees of uncertainty as a result of complex and hierarchical model structure (Link et al., 2012; Lassalle et al., 2014). As the limitation of biological information and survey data, the parameterization of OSMOSE-JZB model is simplified and fishing mortality rates are not calibrated with catch data (Xing et al., 2017). In order to further solidify our findings, we conducted an uncertainty analysis on fishing mortality rates in four pulse fishing scenarios. Although each scenario was assigned a 10% level of uncertainty on fishing mortality rate of each species, the biomass of *S. schlegelii* and *J. belangerii* had the same variation tendency to outcomes in scenarios without adding errors. The insufficient verification may result in imprecise parameter values and influence model accuracy. It is necessary to make a sensitivity analysis of the model in the future study. The food web was simplified in our model, especially by the removal of rare species and by the aggregation of various species into functional groups. Both aspects need to be considered when interpreting the model results (Ward et al., 2012; Xing et al., 2017). Fish could change their life history strategy in response to fishing activities (Jørgensen et al., 2007; Anderson et al., 2008). We assumed that these responses had no effect on the fixed relationships in the model, such as the ranges of predator-prey size ratio, the selectivity of fishing nets, and the potential variability in species life history traits. This will certainly increase uncertainty in modelling process.

Ecosystem-based fishery management (EBFM) is advocated to avoid irreversible ecological degradation and rebuild fishery resources (Patrick and Link, 2015; Eddy et al., 2017). Admittedly, the implementation of EBFM is still impeded in many developing countries due to inadequate data for running ecosystem models (Hamel and Bryant, 2017). Our simulations indicated that the efficiency of seasonal closure could be improved by managing seasonality of fishing effort. Some of our results may not be fully applicable to other fisheries ecosystems due to the specificity of the Jiaozhou Bay ecosystem and limited data used in modelling. Our ecosystem modelling approach can be used as a heuristic tool to investigate and quantify impacts of pulse fishing on the ecosystem.

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