Environment, Development and Sustainability
https://doi.org/10.1007/s10668-021-01410-3

Modeling the impacts of fishing regulations in a tropical Indian estuary using Ecopath with Ecosim approach

G. B. Sreekanth[1] · S. K. Chakraborty[2] · A. K. Jaiswar[2] · P. U. Zacharia[3] · K. S. Mohamed[3]

Received: 8 August 2020 / Accepted: 5 April 2021

Abstract

In this study, we measured the impacts of an effective fishing regulation on the sustainability of fisheries in Zuari estuary, a tropical estuary situated along western coast of India through an Ecosim approach. Ecosystem indicators for 2016 and 2031 (for each Ecosim scenario) were measured to compare and contrast the decadal changes in the status of the ecosystem between these two periods. Four different hypothetical fishing patterns were simulated to explore the best suited management scenario. The ecosystem indices of 2031 ecosystem were compared with that obtained for 2016 to evaluate the possible effects of fishing regulations. The functional groups showed a decline in their biomass when no fishing regulations are implemented (S1). The direct fishing effort reductions of all the fleets (S4) and ban/reduction of indiscriminate fishing fleets (S2-immediate ban and S3-gradual reduction) showed a more or less similar trend for recovery of fish stocks through diverse fisheries policies. A complete ban of indiscriminate fishing seems to slightly more advantageous than the direct reductions in the fishing effort for all the fleets in terms of stock recovery (130%), Q statistic (1.15), Shannon diversity (1.43), mean trophic level of ecosystem (2.98), mean trophic level of the catch (2.91) and fish catch in the gillnet fleet (200%). The simulations have also suggested that a complete control for mechanized fishing fleets will be the best possible management strategy for the recovery of fish stocks in the ecosystem.

Keywords Zuari estuary · Ecopath with Ecosim · Fishing regulation · Indiscriminate fishing · Simulation

---

✉ G. B. Sreekanth
gbsree@gmail.com; gb.sreekanth@icar.gov.in

[1] ICAR-Central Coastal Agricultural Research Institute, Old Goa, Goa 403402, India

[2] ICAR-Central Institute of Fisheries Education, Mumbai, Maharashtra 400061, India

[3] ICAR-Central Marine Fisheries Research Institute, Kochi, Kerala 682018, India

1 Introduction

Estuaries are one of the highly productive aquatic ecosystems in the world (Costanza et al., 1997; Qasim & Sen Gupta, 1981; Whitfield, 1999), and they provide valuable support to human population, such as coastal protection, water purification, fisheries resources, carbon sinks, tourism opportunities and recreational services (Beck et al., 2001; Lira et al., 2018). Estuaries along the west coast of India represent tropical monsoonal estuaries, which are under the influence of southwest monsoon (Ansari et al., 1995; Shetye et al., 2007). Zuari estuary is a highly productive, macro-tidal estuary, and could be considered as a characteristic tropical monsoonal estuary. This estuary receives freshwater influx through the "southwest monsoon" from Zuari River and marine influx from the Arabian Sea (Ansari et al., 1995).

Traditional gillnet fishery (70 vessels) in the estuary provides employment to about 500 traditional fishermen (Sreekanth et al., 2020). However, the indiscriminate operation of trawlers and mini-purse seiners creates overexploitation of fisheries resources. Recently, the banks of the estuary have been transformed into a rapidly developing economic zone, and thus, the ecosystem region has experienced indiscriminate overfishing and pollution from industries, and agriculture over the last two decades (Shetye, 2011; Shetye et al., 2007; Shirodkar et al., 2012). These anthropogenic stressors along with devastation of mangrove habitats have reduced the water quality and has impaired the ecosystem balance (Kessarkar et al., 2015; Shirodkar et al., 2012). Ultimately, these stressors have resulted in depletion of the fish populations, and thereby the small-scale fisheries, which will have direct impacts on the livelihood of the traditional fishermen (Ansari et al., 1995; Sreekanth, Lekshmi, et al., 2017; Sreekanth, Manju Lekshmi, et al., 2017).

Globally, the fishing fleets have been intensified following the motorization and mechanization of fishing crafts and have overexploited the inshore fisheries resources especially in small-scale fisheries settings (Pauly, 1997). The fisheries resources are assumed as renewable resources and seem to be able to recoup from the overexploitation by fishing fleets (Pauly et al., 2000). However, the indiscriminate and intensified fishing could lead to depletion of fisheries resources because of the large differences in the ratio of harvest and rate of recruitment (Christensen, 2000; Pauly, 1997). In many parts of the world, many inshore fish species have depleted or even collapsed due to overexploitation, and it has been reported that many more species have shown no signs of recovery, which denotes that fish populations would be minimized to a level at which their recruitment will be negligible (Hutchings & Reynolds, 2004).

Fishing fleet in the estuary is not well monitored and the operation of mechanized fishing boats are not controlled. The fish catch from the estuary began to increase since 2000 and reached the highest values in 2012, and further, the catch decreased from 2012 to 2017 (Fig. 1). Indescriminate fishing (Trawling and mini-purse seining) contributes to fish catch, which is unreported from the estuary. Globally, trawling has severely affected the demersal fish stocks and shrimps, and similarly, purse seines overexploited the pelagic species and juvenile stocks (Pauly et al., 1998). With the increasing anthropogenic impacts in the Zuari estuary, it seems that the ecosystem has undergone paradigm shift in since 2000, shifting in terms of its trophic organization since 2000, from an ecosystem with a top-down control (large size and high value predatory species controlled) ecosystem to the one with a bottom-up control (small pelagic species dominated) (Ansari et al., 2003; Sreekanth et al., 2016, 2018).

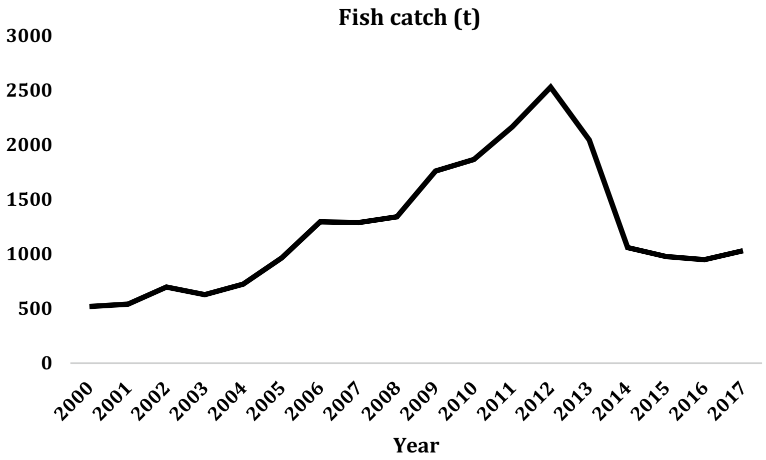

Fig. 1 The total fish catch (tonnes) from Zuari estuary from 2000 to 2017 (Personal communications with fishermen and experts, Directorate of Fisheries, 2018)

In response to the concerns about overfishing and continuous demand from traditional fishermen of Zuari estuary, the fisheries department has put forward a fisheries patrolling system for the coast of Goa since 2017. This patrolling is implemented to ensure that, excluding the traditional gillnet fishery, there are no other fishing operations are carried out in the estuary. Therefore, the indiscriminate operation of trawlers and mini-purse seiners are expected to reduce soon. This approach is considered to be a concrete and effective strategy for managing the fishing effort and is anticipated to restore the fisheries resources in the due course. Studies worldwide have reported that the reduction of fishing effort in coastal ecosystems yielded increase in fish catches as well as fish stocks for majority of the ecosystem (Liu et al., 2008; Manickchand-Heileman et al., 2004; Colléter et al., 2013). Therefore, the policy of reducing fishing effort is found to be a better solution for sustainable fisheries in coastal ecosystems (Xiao, 2005). However, being a multi-species and multi-fleet fisheries, the implementation of a complete fishing regulation would be a time-consuming process, and therefore, the effect of fishing effort reduction is still a critical subject for further studies (Mohamed et al., 2008). The effect of seasonal fishing regulations on fish stocks and fish catches has been investigated by several researchers (Hou et al., 2009; Robert, 1998; Schrank, 2005; Wang et al., 2015; Yang & Zhou, 2013).

Ecosystem models have been successfully applied in coastal ecosystems to describe ecosystem functioning, food web dynamics and time dynamic and spatially explicit simulations to detect variations in ecosystem indicators and fishery (Araujo et al., 2008; Chen et al., 2009; Manickchand-Heileman et al., 2004; Mohamed et al., 2008; Pitcher et al., 2000; Tsehaye & Nagelkerke, 2008; Wang et al., 2015). Ecosystem models can also be used to reveal otherwise unknown system properties and to emphasize the need to improve knowledge about specific parts of the system. Ecopath with Ecosim (EwE) software is specifically designed for construction, parameterization and analysis of mass-balance trophic models of various ecosystems including marine, coastal, freshwater and terrestrial ecosystems. The Ecosim module in the Ecopath with Ecosim (EwE) software is a time dynamic ecosystem modeling tool for addressing the issues of how ecosystems are likely to respond to changes in fishing patterns (Christensen et al., 2005). There is a large-scale application of Ecosim with dynamic simulations in aquatic ecosystem research for exploring policy options for fisheries management (Araujo et al., 2008; Chen et al., 2009; Manickchand-Heileman et al., 2004; Mohamed et al., 2008; Pitcher et al., 2000; Tsehaye & Nagelkerke, 2008; Wang et al., 2015). Many research reports indicate that the Ecosim can be used for optimizing fishing scenarios for fisheries management and to highlight the dynamics of fishing communities to project sustainable fisheries in terms of economic, social and ecological contexts (Pitcher et al., 2000; Manickchand-Heileman et al., 2004; Araujo et al., 2008; Chen et al., 2009; Mohamed et al., 2008; Tsehaye & Nagelkerke, 2008; Wang et al., 2012).

Therefore, in this study, the base Ecopath model for Zuari estuary, which was constructed in 2016 (Sreekanth et al., 2018), was used to simulate various scenarios of fishing regulations in the estuary using the Ecosim module. We identified four hypothetical fishing scenarios that could be applied in the estuarine system. In the first scenario, the indiscriminate fishing practices would be continued and no fishing regulations. Second scenario considered an immediate ban on the indiscriminate fishing fleets from 2017 onwards. A gradual ban on the indiscriminate fleets represented the third scenario. All fishing fleets were reduced to half in the fourth scenario. The objective of the current study is thus to explore the utility of Ecosim to measure the relative changes in biomass of functional groups, ecosystem indicators (Kempton's Q statistic, Shannon Index, mean trophic level of the ecosystem, mean trophic level of the fish catch) and fish catch in gillnet fleet under different fishing scenario simulations at two time periods–2016 and 2031.

2 Material and method

The Zuari estuary (39.9 km^2) is a tropical estuary situated along the western coast of India (Fig. 2) characterized by high rainfall (3000 mm year^{-1}), high temperature (32–35 °C), longer photoperiod (11.9 h day^{-1}) and long flushing period (4–5 months) (Shetye et al., 2007). The ecosystem of Zuari estuary defined in this study covers the 10 km stretch upstream from the mouth, which is approximately 5 km wide and 5–6 m deep. The estuary is highly productive, in which the freshwater is well mixed with marine water by tides and winds (George et al., 2013; Qasim, 1973). The annual freshwater runoff into the estuary is 9 km^3, and this discharge makes the estuary highly productive and dynamic (Qasim, 1973). The currents are largely influenced by tides (semi-diurnal) with a mean range of 1.3 m (Ansari et al., 1995; Qasim, 1973). These conditions are favorable to high biological productivity, and thus, the diversity of fish assemblages is very high in the ecosystem (Ansari et al., 1995; Sreekanth et al., 2018). The operation of indiscriminate fishing practices such as trawl and mini-purse seines is often reported from the estuary, which is detrimental to the ecosystem and ecosystem functioning (website references W1, W2, W3, W4). Trawling and mini-purse seining contributed to majority of the fish catch (unreported) from the estuary since 2010 to 2017. The fishing fleet of the estuary includes small-scale gillnet fishery (70 vessels), occasional mini-purse seine (a seine net operated outside the bay generally that measures 200 m in length) fishery and trawl fishery. The major fish species harvested from the estuary are mackerel, sardine, white sardine, mullet, whitebait, moustached anchovy, silverbelly, carangid, croaker, catfish, crab and shrimp (Sreekanth et al., 2018). The Zuari estuary also functions as a foraging and nursery habitat for many of the vulnerable and near threatened (designated by IUCN) aquatic species such as *Glaucostegus granulatus*, *Epinephelus coioides* and *E. diacanthus.*

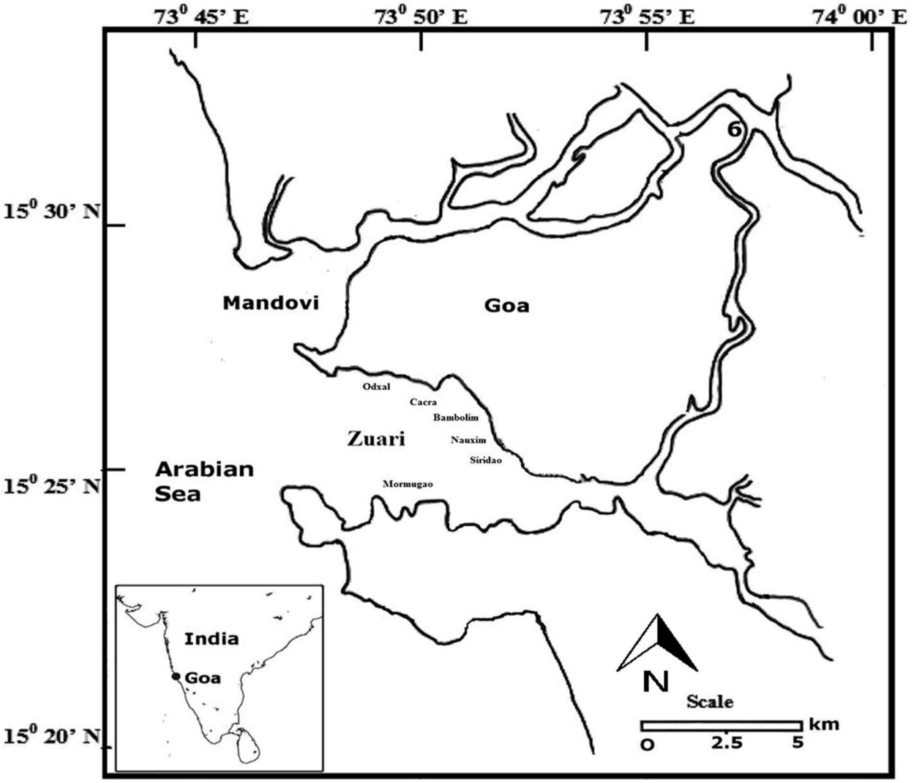

Fig. 2 The map representing the study area, Zuari estuary, central west coast of India

2.1 Ecosim model

Ecopath with Ecosim (EwE) software is an efficient tool designed for the construction, parameterization and analysis of trophic mass-balance models for various terrestrial and aquatic ecosystems structure (Christensen et al., 2005). In this model, the functional groups are considered to represent primary producers (phytoplankton and benthic algae), consumers (zooplankton, crabs, shrimps, fishes, dolphins and birds) and non-living component (detritus). The Ecosim model is widely used to explore fisheries policy simulations under the dynamic module, Ecosim (Araujo et al., 2008; Chen et al., 2009; Wang et al., 2015). The Ecosim model uses the ecosystem parameters derived from the base Ecopath model, and this dynamic model measures the alterations in biomass of ecological groups under time dynamic catch rates, prey–predator relationships and trophic dynamics (Christensen et al., 2005).

The time dynamic simulation was performed based on the Ecopath model developed in the year 2016 for Zuari estuary using EwE version 6.4 in previous studies (Fig. 3, Sreekanth et al., 2020), in which the fishing fleet were divided into three types (trawl, purse seine and gillnet) for simulations. The trophic network flow diagram of Zuari estuary food web is presented in Fig. 3. There were 22 functional groups in the 2016 Ecopath model: birds (BD), dolphins (DO), large pelagics (LP), rays and skates (RS), cephalopods (CEP),

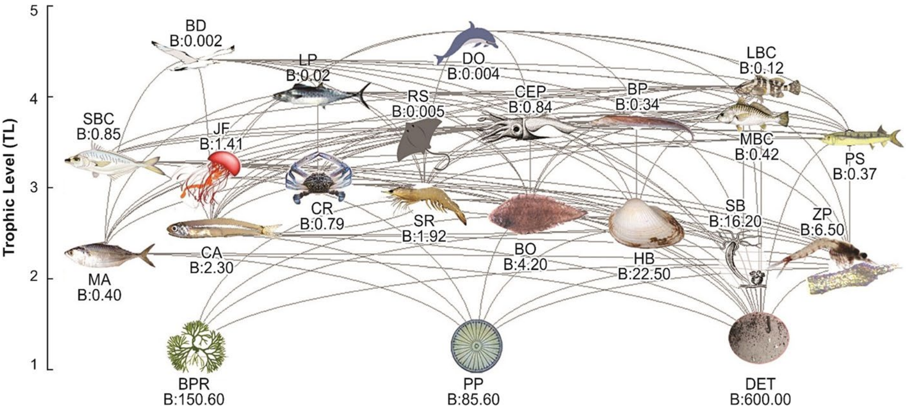

Fig. 3 Trophic flow diagram of the Zuari estuarine food web (BD-birds, DO-dolphins, LP-large pelagics, LBC-large benthic carnivores, MBC-medium benthic carnivores, BP-benthopelagics, CEP-cephalopods, RS-rays and skates, PS-piscivores, SBC-small benthic carnivores, JF-jellyfish, CR-crabs, SR-shrimps, BO-benthic omnivores, HB-heterotrophic benthos, SB-sessile benthos, MA-mackerel, CA-clupeids and anchovies, ZP-zooplankton, BPR-benthic producers, PP-phytoplankton, DET-detritus), 'B' represents the biomass (t km^{-2} year^{-1}) for each functional group

benthopelagics (BP), large benthic carnivores (LBC), medium benthic carnivores (MBC), piscivores (PS), small benthic carnivores (SBC), mackerel (MA), clupeids and anchovies (CA), crabs (CR), shrimps (SR), benthic omnivores (BO), heterotrophic benthos (HB), sessile benthos (SB), jellyfish (JF), zooplankton (ZP), benthic producers (BPR), phytoplankton (PP) and detritus (DET). The network flow diagram also reports the biomass flows and trophic fluxes across the groups in the Zuari estuary. The validity of the model was successfully evaluated through the PREBAL routine, which recognizes the discrepancies of the input data (Heymans et al., 2016). The model showed reasonably good estimate of pedigree (0.76). The input data and estimated parameters for the 2016 Ecopath model are described in Table 1.

2.2 Fishing policy simulations

We used the temporal dynamic module *Ecosim* (Walters et al., 2000) to perform fishing scenarios of decrease or increase in fishing effort. Ecosim simulations are sensitive to the 'vulnerability' settings, which incorporates density dependency and reflects how far a group is from carrying capacity (Christensen et al., 2005). As a prerequisite to incorporate prey predator interactions in Ecosim, a vulnerability setting is required for all predator–prey interactions, which defines the status of prey groups from predators (whether vulnerable or protected) (Christensen et al., 2005). The vulnerability of each functional group was set to be corresponding to its trophic level in the Ecopath model. Ecosim assumed default values defined for tropical ecosystems were used for all other parameter settings (base proportion of nutrients: 1.000; minimum foraging time: 0.1, maximum relative feeding time: 2.00, feeding time adjustment rate: 0.5, density dependent catchability: 1.000). Ecosim calculates corresponding changes in relative biomass of ecological groups when the fishing effort of the fleet is altered. Ecosim also provides dynamic simulations of various scenarios in which the fishing mortality rates are changed and generates plausible model representation of what might occur when the

Table 1 Estimates of parameters of the Zuari 2016 Ecopath model (Sreekanth et al., 2018)

Ecological group	TL	B	P/B	Q/B	EE	P/Q
DO	4.66	0.004	0.07	16.22	0.000	0.004
BD	4.34	0.002	0.08	58.02	0.000	0.001
LBC	4.04	0.118	4.50	12.20	0.953	0.369
LP	3.97	0.02	2.40	8.20	0.884	0.293
MBC	3.75	0.42	3.90	10.80	0.991	0.361
BP	3.64	0.34	3.10	9.60	0.993	0.323
RS	3.57	0.005	1.70	7.30	0.706	0.233
CEP	3.56	0.84	4.20	7.90	0.995	0.532
PS	3.46	0.37	2.30	9.30	0.969	0.247
SBC	3.19	0.85	5.20	20.53	0.997	0.253
JF	3.02	1.41	4.86	28.50	0.904	0.171
CR	2.94	0.79	6.70	20.50	0.990	0.327
SR	2.77	1.92	6.80	24.20	0.988	0.281
BO	2.58	4.20	3.20	11.30	0.900	0.283
HB	2.43	22.50	3.40	16.70	0.482	0.204
CA	2.37	2.30	7.30	26.30	0.960	0.278
MA	2.16	0.40	6.80	20.20	0.573	0.337
SB	2.04	16.20	9.80	45.00	0.872	0.218
ZP	2	6.50	25.50	240.00	0.758	0.106
BPR	1	150.60	12.80	0.00	0.068	
PP	1	85.60	96.20	0.00	0.152	
DET	1	600.00			0.118	

TL: trophic level, B: biomass (t km^{-2} year^{-1}), P/B: production/biomass (year^{-1}), Q/B: consumption/biomass (year^{-1}), EE: ecotrophic efficiency, P/Q: production/consumption or gross efficiency of food conversion (year^{-1}), BD-birds, DO-dolphins, LP-large pelagics, LBC-large benthic carnivores, MBC-medium benthic carnivores, BP-benthopelagics, CEP-cephalopods, RS-rays and skates, PS-piscivores, SBC-small benthic carnivores, JF-jellyfish, CR-crabs, SR-shrimps, BO-benthic omnivores, HB-heterotrophic benthos, SB-sessile benthos, MA-mackerel, CA-clupeids and anchovies, ZP-zooplankton, BPR-benthic producers, PP-phytoplankton, DET-detritus

fishing effort on ecological groups is altered. A dynamic simulation of the period from 2016 to 2031 using Ecosim was performed on the 2016 Ecopath model based on the assumption that fishing regulations will be implemented from 2017 onwards. Four simulations were carried out for a period of 15 years (Fig. 4) under four different scenarios on the 2016 Ecopath model:

1. Scenario 1 (S1): No regulation on fishing fleets (Gillnet fleet increase by 50%, Trawl and Purse seine fleet increase by 100% at the end of 2031),
2. Scenario 2 (S2): Immediate ban on indiscriminate fishing fleets (Ban from 2017 onwards), Gillnet fleet gradually increase by 50% at the end of 2031.
3. Scenario 3 (S3): Gradual ban on trawl and purse seine fishing fleets (reduction of the trawl and purse seine fleets to 50% in 2017 and then gradual reduction by 10% during 2018 and 2019, 5% reduction till 2025 and complete ban then onwards) and gradual increase in Gillnet fleet by 50% at the end of 2031.

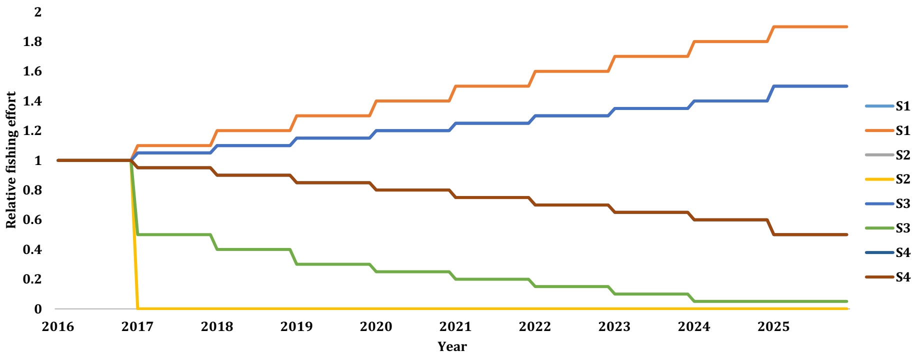

Fig. 4 Performed simulations of changing fleet fishing effort: S1, S2, S3 and S4. GN: Gillnet fleet, TRPS: Trawl and Purse seine fleet

4. Scenario 4 (S4): Reduction of all fishing fleets by 50% at a rate of 5% year till 2026 and then complete ban on the trawl and purse seine fleets, keeping gillnet fleet at the same level of fishing (50% of the base level) from 2026 to 2031.

For each scenario, fishing effort on the fleet was changed rather than changing fishing mortality on individual groups. This is because of the existence of multispecies and multigear fisheries in the estuary. Therefore, the fishing is relatively non-selective, and it is not justifiable to adjust the fishing mortality on individual groups.

Further, in each scenario, the trends in relative biomass estimates of benthic predatory fish (LBC and MBC), pelagic predatory fish (PS and LP), small benthic fish (BP and SBC), small pelagic fish (CA and MA), rays and skates, shrimps and crabs were compared to identify the best scenario among the simulations. To evaluate the ecosystem structure, mean trophic level (TL) of the catch, mean TL of the ecosystem, biomass of higher TL ($TL > 3.5$) and biomass of lower TL (2 to 3) were measured in each scenario. Besides, a set of ecosystem integrity indicators such as Kempton's index of biodiversity (Q) and Shannon diversity index (H) were used to assess the ecological diversity, biomass diversity and sustainability of fisheries, respectively, in each scenario simulation.

1. Kempton's index of biodiversity (Q): The index expresses the biomass species diversity of functional groups in an ecosystem. The 'Q' statistic is estimates as from the interquartile slope of the cumulative functional group abundance curve (Kempton & Taylor, 1978).

$$Q = S/\left[2\log\left(\frac{R2}{R1}\right)\right] \tag{1}$$

   where S is the total number of functional groups in the model; $R1$and $R2$ are the biomass values of the 25th and 75th percentiles in the cumulative abundance distribution. Kempton index increases with increase in biomass of high trophic level species and decreases with increased fishing impacts.
2. Shannon diversity index (H): The Shannon diversity index represents a combination of species richness and biomass. It quantifies the uncertainty in predicting the species identity of an individual when taken at random from a dataset. If there is only one species in the dataset, then Shannon entropy exactly equals zero (there is no uncertainty in

predicting the species). This index increases with increasing diversity and evenness in the dataset). The index is calculated by the following equation,

$$H = \sum_{i=1}^{s} P_i \ln P_i \tag{2}$$

where S is the total number of functional groups in the model; P_i is the proportional abundance of individual ecological group.

3 Results

3.1 Scenario 1 (S1): No regulation on fishing fleets

The results of the first scenario simulation, where no fishing regulations are enforced, suggest a decrease in biomass of majority of the functional groups (20 of the 22 groups, barring HB and ZP) to varying degrees in the 15 years' simulation period (Fig. 5a). In this simulation, catches of all the major fish groups had heavily reduced in 2031 compared to 2016. The most negatively affected groups were BO, SBC, MA, BP, CA, MBC, LP, CR and PS. The decline in percentage biomass for fishery groups ranged from 13% for shrimps to 98% for BO. The average increase in overall biomass was about 130% compared to the base value in 2016. The fishing scenario, S1, had negatively affected the functional groups, for which the relative biomass has declined drastically (> 50% decline) in 2031. Shrimps exhibited the lowest decrease in biomass in 2031 compared to their biomass in 2016. This pattern of biomass variability in shrimp (one of the important prey group in

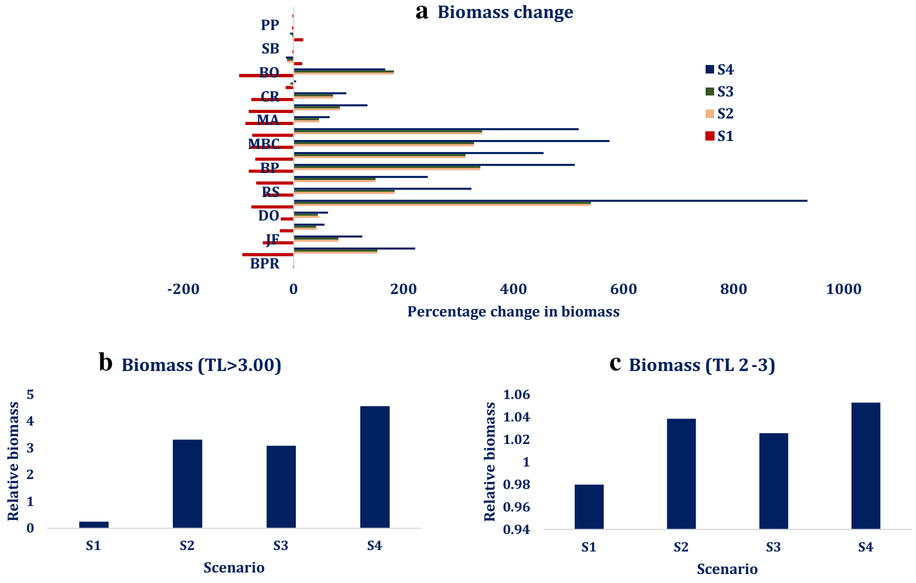

Fig. 5 The impacts (positive and negative) on percentage change in biomass of various functional groups (in 2031 compared to 2016) (**a**), relative change in high trophic level (TL > 3) (**b**) and low trophic level (TL 2–3) biomass (**c**) in various fishing scenarios

the ecosystem) suggested that the change in one species because of fishing would affect other species through the food chain. Therefore, the decrease in biomass at the high trophic level groups reduced predation pressure on the lower trophic levels. The results suggested that the exploited functional groups in the 2031 ecosystem were more responsive to the increase in fishing effort (S1), where no fishing regulation was implemented. The biomass of high trophic level groups declined to a very low level (0.24) (Fig. 5b) in 2031 in the scenario simulation. However, the lower trophic level groups showed only a 2% decline in their biomass (Fig. 5c). The mean trophic level of the ecosystem (2.80) and mean trophic level of the catch (2.75) had also yielded the lowest values in 2031 in this scenario (Fig. 6). The ecosystem diversity indices, Kempton's Q and Shannon diversity showed a decline over the years under this scenario. The Q statistic dropped from 1.25 in 2016 to 1.01 in 2031, whereas the Shannon index slightly dropped to 1.27 (2031) from 1.33 (2016). The benthic predators, pelagic predators, small benthic fish, small pelagic fish, rays and skates, and shrimps declined heavily in the first scenario, which is depicted in Fig. 7.

3.2 Scenario 2 (S2): Reducing the mechanized fishing effort and increasing gillnet fishing

Scenario two has considered a strict implementation of the ban on the trawl and purse-seine fleets in the estuary from 2017 onwards. In this scenario, a complete ban on trawl and purse seine fleets was enforced from 2017 onwards together with a 50% increase in the gillnet fleet in 2031. Eighteen of the 22 functional groups increased in their biomass in 2031 with the S2 fishing scenario (Fig. 5). The increase in percentage biomass for fishery groups ranged from 46% for MA to 538% for LP. The average increase in overall biomass was about 130% compared to the base value in 2016. Shrimps exhibited a slight decrease (about

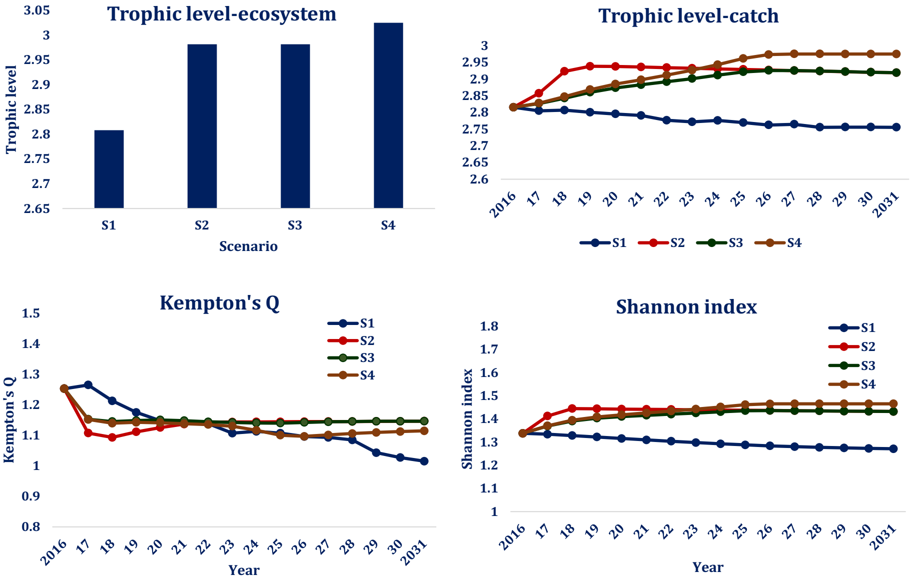

Fig. 6 The variation of ecosystem indicators (mean trophic level of the ecosystem, mean trophic level of the catch, Kempton's Q, and Shannon index) in various fishing scenario simulations

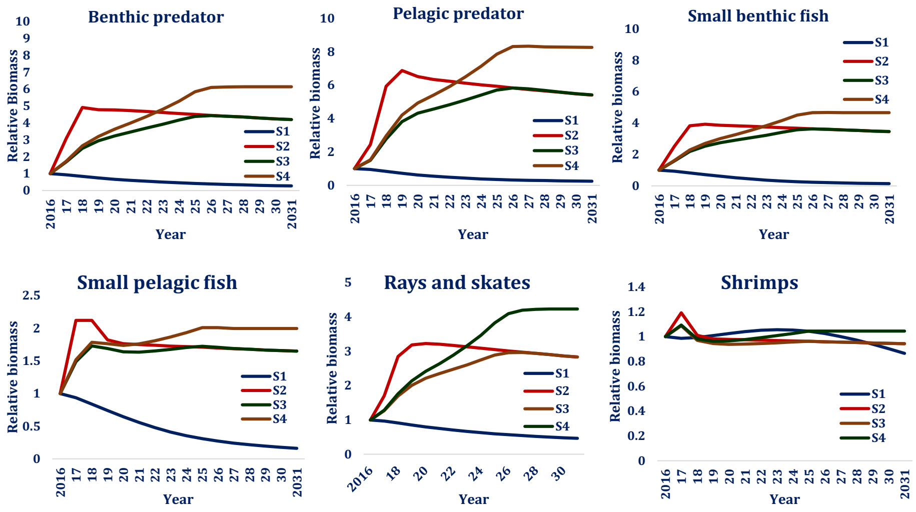

Fig. 7 Simulation results of relative biomass for benthic predators, pelagic predators, small benthic and pelagic fish, rays and skates and shrimps under various fishing scenarios

6%) in biomass in 2031 compared to the base year. Due to the decrease in the fishing effort in this scenario, a drastic hike in biomass was expected for the former group. However, the increase in biomass for major predator groups has exerted predation pressure on shrimps (one of the major prey group), and subsequently caused a decline in the biomass of the latter group. In this scenario, the overall fishing effort has decreased drastically in the estuary with a complete ban on the two major fleets. As expected, most of the fish groups benefitted in this scenario with a projected biomass increase of about 130%, especially for the top predators. The relative biomass of high trophic level groups escalated to a very high level (3.3) (Fig. 5) in 2031 compared to 2016. However, the lower trophic level groups showed only a 2% increase in their biomass. The mean trophic level of the ecosystem (2.98) and the mean trophic level of the catch (2.91) had also yielded showed an increment in 2031 (Fig. 6). The Shannon diversity showed an increase over the years under this scenario. The index has increased from 1.33 in 2016 to 1.43 in 2031. The Q statistic dropped initially from 1.25 to 1.10, and however, it has improved over the years to a value of 1.15. The biomass of benthic predators, pelagic predators, small benthic fish, small pelagic fish, rays and skates exhibited a drastic increase in the initial years of the simulation and showed a gradual decrease towards the end of the simulation period. However, the relative biomass was very high for all these groups (ranging from 1.6 for benthic predators to 5.4 for pelagic predators) in 2031 compared to 2016. Shrimps declined slightly in relative biomass (0.94 compared to 2016) at the end of simulation.

3.3 Scenario 3 (S3): Gradual removal of indiscriminate fleets and gradual increase in Gillnet

The scenario S3 was characterized by a gradual removal of the mechanized fleets and a subsequent increase in the gillnet fleet. In this scenario, a gradual ban on the trawl and purse seine fleets was enforced 2017 onwards together and which was extended as a complete ban from 2026 onwards. On the other hand, a gradual increase in the gillnet fleet to the tune of 50% is also imposed. Eighteen of the 22 functional groups increased in their biomass in 2031 compared to 2016 (Fig. 5). The increase in percentage biomass for fishery groups ranged from 46% for MA to 540% for LP. The average increase in overall biomass was about 120% compared to the base value in 2016. Shrimps again exhibited a marginal decrease (about 6%) in biomass in 2031 compared to the base year. However, the increase in biomass for major predator groups exerted predation pressure on shrimps (one of the major prey group), thereby causing a decline in their biomass. All the fish groups were all benefitted in this scenario with an increase in their biomass through the simulation period. The highest increase in relative biomass was observed for LP, PS, BP, MBC, RS, BO, SBC and CEP. The relative biomass of high trophic level groups remained at a higher level (3.01) (Fig. 5) in 2031 compared to 2016. However, the lower trophic level groups showed only a 2% increase in their biomass similar to the second scenario. The mean trophic level of the ecosystem (2.98) and the mean trophic level of the catch (2.91) had also showed an increment in 2031 (Fig. 6). The Shannon diversity index showed an increase over the years following this scenario. The index has increased from 1.33 in 2016 to 1.43 in 2031. The Q statistic dropped initially from 1.25 to 1.10, and however, it has improved over the years to a value of 1.15. The biomass of benthic predators, pelagic predators, small benthic fish, small pelagic fish, rays and skates exhibited a drastic increase in the initial years of the simulation and showed a gradual decrease towards the end of the simulation period. However, the relative biomass was very high for all these groups (ranging from 1.6 for benthic predators to 5.4 for pelagic predators) in 2031 compared to 2016. Relative biomass of shrimps declined slightly (0.94 compared to 2016) at the end of the simulation.

3.4 Scenario 4 (S4): Uniform reduction in fishing fleets followed by a complete ban on mechanized fleets

In the fourth scenario S4, 20 functional groups increased in their biomass values for 2031 (Fig. 4). The increase in percentage biomass for fishery groups ranged from 4% for shrimps to 930% for LP. The average increase in overall biomass was about 200% compared to the base value in 2016. All the exploited groups showed an increase in biomass in 2031 compared to 2016 following this scenario. The lowest increase was again for shrimps, which is due to the higher levels of predation rates on this group. The highest increase in relative biomass was observed for LP, MBC, PS, BP, LBC, RS, CEP, SBC, BO and CA. The relative biomass of high trophic level groups showed the highest value in this scenario (4.5) (Fig. 5) in 2031 compared to 2016. The lower trophic level groups showed a slight (5%) increase in their biomass. In 2031, the mean trophic level of the ecosystem (3.02) and mean trophic level of the catch (2.97) had shown the highest increment in this scenario compared to 2016 (Fig. 6). The Shannon diversity index has increased from 1.33 in 2016 to 1.46 in 2031. The Q statistic has dropped from 1.25 in 2016 to 1.11 in 2031. The biomass of benthic predators, pelagic predators, small benthic fish, small pelagic fish, rays and skates exhibited a continuous increase from 2016 to 2031. The highest increase in relative biomass for these groups was observed in S4. The relative biomass was very high for all these groups (ranging from 1.9 for small pelagic fish to 8.3 for pelagic predators) in 2031 compared to 2016. The relative biomass of shrimps has improved slightly (1.04 compared to 2016) at the end of the simulation.

3.5 Patterns in the fish catch under various scenario simulations

A comparison of catches by gear types in the first scenario S1 indicates that for most of the groups, the relative catch declined in 2031 compared to 2016. In this 15-year simulation, the increasing fishing effort of the mechanized fleets and gillnet fleet caused a reduction in the fish catch for most of the groups (Fig. 8). In trawl fleet, the average relative catch had declined to 0.56, whereas the catches of shrimps had showed an increasing trend (1.72). The catches of BO, SBC, MA, BP, CA, MBC, LP, CR and PS in 2031 have declined to half of their catch value recorded in 2016 (Fig. 8). Similarly, in the purse-seine fleet, the average relative catch has declined to 0.53 in 2031. The catches of BO, SBC, MA, BP, CA, MBC, LP, CR and PS in 2031 have declined to half of their catch value when compared to 2016 (Fig. 8). In gillet fleet, the catches of BO, SBC, MA, BP, CA, MBC, LP, CR, PS, LBC and CEP reduced to half of their catch in 2016 (Fig. 8). The improvement in catch was only observed for shrimps in all the fleets, which might be a reflection of the increase in biomass followed by the removal of the predator species at the escalated fishing efforts.

In scenario 2, due to the ban on the mechanized fleets, the relative catches of all the groups have increased and the total fish catch was doubled in gillnet fleet. The catches of LP, PS, BP, MBC, LBC, RS, BO, SBC, CEP, CA, CR and MA have doubled in 2031 compared to the base year (Fig. 8). As expected, there was an overall 50% decrease in the total catches especially for SR, BO, MA, JF, CR, CA, SBC, PS in 2031 compared to the base year 2016 because of the removal of the two fishing fleets. However, there was an improvement in total catches for LP, MBC, BP, RS and LBC. Again, shrimps displayed the lowest increase in the relative catch of gillnet fleet, due to the anticipated predatory pressure from high trophic levels in the reduced fishing effort scenario. In scenario 3, the relative catches of all the groups have improved in the gillnet fleet specifically for LP, PS, BP, MBC, LBC, RS, BO, SBC, CEP, CA, CR and MA (Fig. 8). Shrimps displayed the lowest increase in

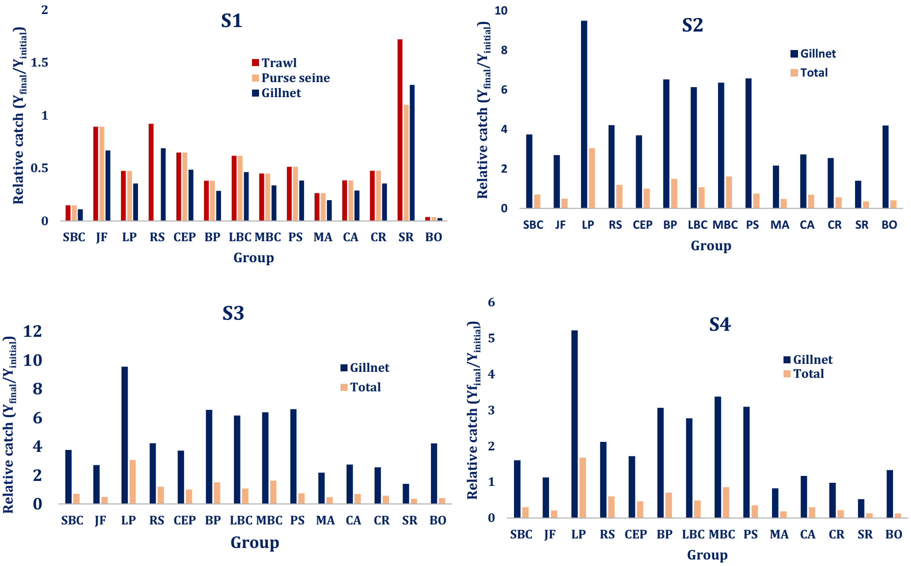

Fig. 8 The relative change in fleet wise catch and total catch for different fish groups in various fishing scenarios

relative catch of gillnet fleet, due to the anticipated predatory pressure from higher trophic levels. The total fish catch was reduced to half in 2031 compared to the base year, 2016.

In scenario 4, the relative catches of LP, MBC, PS, BP, LBC, RS, CEP, SBC, BO and CA have increased in the GN fleet (Fig. 8). The catches of mackerel and shrimps have reduced following the increase of predatory species in the ecosystem and reduction in the gillnet fleet. However, the total catch from gillnet fleet in 2031 was found near to the value in 2016. The total catch from all the fleets has nearly reduced by 80% compared to the base year. The relative catches for SR, MA, CR, CA, BO, SBC, CEP, RS, LBC, BP and PS have declined in total catch when compared to the base year. These reductions correspond to the effort reductions in all the fleets. The group LP has only showed an increase in catch in 2031 compared to the base year.

4 Discussion

The fishing regulations and decrease in the fishing effort for mechanized fleets achieved their targets of protecting the fish stocks and relative improvement in biomass for the fished ecological groups (Shannon et al., 2009; Tsehaye & Nagelkerke, 2008; Wang et al., 2015; Williams et al., 2006). In previous studies along the west coast of India, the seasonal fishing ban, which is enforced for two months during monsoon season, was found to yield better results for conservation of the fish stocks (Vivekanandan et al., 2010). Further, extension of the seasonal fishing ban for three months was suggested for the west coast of India (Mohamed et al., 2014). However, there is no seasonal ban implemented in the estuarine systems even as the estuaries provide spawning and nursery grounds for many species at different periods including monsoon, pre-monsoon and post-monsoon. The life history characteristics of multi-species, multi-gear harvests will be a critical factor influencing the efficacy of the fishing regulations for fisheries management (Wang et al., 2015). Therefore, enforcement of a seasonal fishing moratorium that has been successfully implemented in the Pearl River estuary (Wang et al., 2015) would not be practical for the tropical monsoonal estuaries. Thus, fishing effort regulations including the ban on indiscriminate fishing practices seem to be a better strategy for the Indian estuaries. The simulation results showed that both scenario2 and scenario3, wherein the removal of mechanized fishing fleets was implemented, yielded the best possible results in terms of recovery of fish stocks and increment in the catch for the gillnet fleet.

The efficacy of the fishing regulations was not tested along the estuaries along the Indian coast, and this study forms the first of that kind to simulate the impacts of fishing regulations on fish stock recovery and fish catch in a tropical monsoonal estuary. The study showed that the implementation of fishing regulation would yield restoration of majority of the fish stocks, which are already exploited in contrast to the recovery of some specific species in the seasonal fishing ban. Moreover, the total catch from the estuary has declined after the implementation of the regulation, which was primarily due to the ban/reduction in the indiscriminate fishing fleets in the estuary. At the same time, the catch rates in the gillnet fleet improved substantially following the fishing regulations. Some of the exploited groups were impacted when the fishing regulations were in place in scenario simulations. The variations were evident in the shrimp biomass when the fishing regulations were implemented. The former group was the main target species for the gillnet fishery and also served as the major prey resource for top predators (Sreekanth et al., 2020). Therefore, the fishing regulations influenced the groups in two ways: (1) increase in capture by gillnet fleet, (2) increase in predation by the predator groups. However, these groups represent highly resilient fish stocks, would be able to recover from these initial declines and can sustain in the ecosystem efficiently (Froese & Pauly, 2016).

There are several case studies documented around the world highlighted the effectiveness of fishing regulations to improve the status of fish stocks in coastal ecosystems. The fishing regulations appeared to be a better management solution than the periodic closures such as seasonal fishing ban (Wang et al., 2015). Seasonal fishing closures had resulted in larger fish harvest of the stocks immediately after the closure period (Hou et al., 2009; Mohamed et al., 2014). Moreover, these seasonal closures had also caused a gradual decline in the fish landings indicating that the seasonal ban only not sufficient to recover the fish stock under overexploited populations (Williams et al., 2006). Therefore, in tropical estuaries, other modes of fishing regulations could be a better option to recover the fish stocks and to yield sustainable fish harvest from the ecosystem (Ansari et al., 2003). In this study, scenario S2 and scenario S3 were identified as the better fishing policy strategies for the management of fish stocks in the estuary. However, it is suggested to follow the S2 scenario, as there will be an immediate ban on the indiscriminate fishing fleets, which will reduce the cost of fisheries management for the implementing agencies in terms of monitoring and surveillance.

Previous studies also suggested that the fishing regulations effectively improved the fishermen's cognizance of conservation of fish stocks and played an important role in the restoration of the resources (Wang et al., 2015). This positive impact was indicated by the increase in catch rates for fishing fleets in Pearl River estuary before and after implementation of the fishing regulations (Wang et al., 2015). Adebola and de Mutsert (2019) modeled the fishing effort in Nigerian coastal waters and suggested that closure of fishing up to 5 nm will benefit the reef fishes. By simulating the different fishing scenarios of Itapu reservoir for different kind of fishing fleets, Phillippsen et al., (2019) concluded that apart from the fishing effort, the introduction of species had negative impacts on the native species; further, he proposed reduction in the fishing effort for three species. Through simulation of trophic cascade on the West Florida Shelf, Chagaris et al (2015) inferred that the overexploitation of bait fishes could cause the reduction of large-sized fishes and the biomass would be improved for species whose diet consists of benthic-associated prey. Forrest et al (2015) in his studies have developed ecosystem models of the continental shelf and slope of New South Wales and concluded that an increase in the trawl fishery would subsequently decimate the long-lived elasmobranch and piscivorous fish species.

Frequent revisions and modifications in fishing policies are required to conserve fishery resources and biodiversity in the tropical monsoonal estuaries (Ansari et al., 2003; Sreekanth et al., 2018). The fisheries management authorities along the west coast of India have understood that the fishing pressure is high and diverse management actions are required to conserve the fish stocks. These management actions include limiting the fishing fleet, prohibiting the destructive fishing practices, seasonal fishing ban, regulations on the use of mesh size for fishing gears, and demarcation of areas for fishing operations, etc. However, being a multi-species and multi-gear fisheries setting, the implementation of these measures attracts a lot of criticism and also will result in intra-sectoral conflicts between mechanized and traditional fishermen. There are no concrete efforts to delineate the positive and negative impacts of the fisheries management actions on the fish stocks. In this context, this study put forward an important step to examine the effect of the fishing regulations as a policy measure and to explore different fisheries management and conservation strategies. To evaluate the effects of the fishing regulations on the fish community, changes in the structure and function of the ecosystem in the Zuari estuary were examined based on scenarios simulation over the period from 2016 to 2031. The ecosystem indicators (S1, S2, S3 and S4) for 2031 calculated based on the various fishing scenarios were applied to the estuarine ecosystem. The comparison of the ecosystem features for the 2016 and 2031 indicated that the fishing regulations were effective in providing an ecosystem recovery. The fishing regulations could reveal a positive impact in terms of the recovery of highly sensitive fish stocks (predatory groups).

Assessing the fishing policy options and management scenarios could generate practicable solutions by evaluating their comparative advantages and disadvantages and the simulation approach helps to make policy decisions on the fisheries management. The comparison of various fishing scenarios revealed that most of the functional groups would decline in their biomass when no fishing regulations are implemented (S1). The direct fishing effort reductions (S4) and ban/reduction of indiscriminate fishing fleets showed a more or less similar trend for recovery of fish stocks through diverse fisheries policies (S2 and S3). The ban of indiscriminate fishing fleets (S2 and S3) offers better stock recovery and fish catch in gillnet fleet when compared to direct reductions in the fishing effort for all the fleets (S4). Moreover, the reduction in the gillnet fleet (in S4) would be detrimental because the traditional fisher folk depend on the gillnet fishery for their livelihood (Sreekanth et al., 2020). The scenario S2 indicated a comparative advantage from S3, in terms of overall recovery of fish stocks (130%) and the largest increment in gillnet catch (200%). Moreover, in S2, an immediate ban on the indiscriminate fishing fleets will be enforced, which will definitely ease the cost of fisheries management (monitoring and surveillance) for the implementing agencies. The Ecosim is a time dynamic module for policy evaluation, and here, we used it to measure the impact of indiscriminate fishing operations on the estuarine fisheries and ecosystem. Consequently, there are chances for estimation errors in the parameters of the EwE model used. Therefore, we may not consider EwE as a tool for making quantitative predictions, but rather, as a tool for screening alternate policies to find out choices that are worthy of more detailed analysis and experimental field testing. Moreover, it is pertinent to note that the simulation results are mean responses of the ecosystem, not actual transitional dynamics (Christensen et al., 2005). However, the current simulations suggest that the impacts of indiscriminate fishing could be minimized through strict policy decisions. Besides, the results of the study also force us to rethink on the fisheries management approaches and resource conservation strategies for tropical interface ecosystems. Our model and simulations reveal that the fishing regulations can play a major positive role in the recovery of fish community estuarine systems. But in a real sense, the question will be how to effectively reduce the fishing effort/ban the indiscriminate fishing operations. Hence, to safeguard the fish stocks, a comprehensive set of fishing regulations on fishing effort (limited access, size limits, species bans, catch control and gear restrictions) could be formulated for tropical estuaries.

Acknowledgements The authors are grateful to Director, ICAR-CCARI, Director, ICAR-CIFE, Mumbai, and the staff of the Fisheries department and Goa State Biodiversity Board for their support and guidance in this study. We owe a great deal to Mr. Vishwajeet Prajapati, KVK, North Goa, and research scholars (Department of Fisheries and Goa University) and fishermen along the Zuari estuary for their kind co-operation, technical help with the fishing experiments, sample collections and digital aids, in particular, fishermen of Zuari estuary, and Biodiversity Management Committees who has provided assistance in site selection and sampling.

Authors contribution GBS, SKC and AKJ contributed to conceptualization, methodology and sampling; GBS, AKJ and PUZ contributed to data collection, writing–original draft preparation. GBS, KSM, PUZ and SKC contributed to software analysis, writing–reviewing and editing.

Funding The authors thankful to Indian Council of Agricultural Research, New Delhi, for funding this research work.

Declaration

Conflict of interest The authors declare that they have no known competing interests or conflicting interest.

References

Adebola, T., & de Mutsert, K. (2019). Spatial simulation of redistribution of fishing effort in Nigerian coastal waters using Ecospace. *Ecosphere, 10*(3), e02623.

Ansari, Z. A., Chatterji, A., Ingole, B. S., Sreepada, R. A., & Rivonkar, C. U. (1995). Community structure and seasonal variation of an inshore demersal fish community at Goa, West Coast of India. *Estuarine, Coastal and Shelf Science, 41*, 593–610.

Ansari, Z. A., Sreepada, R. A., Dalal, S. G., Ingole, B. S., & Chatterji, A. (2003). Environmental influences on the trawl catches in a bay estuarine system of Goa. *Estuarine, Coastal and Shelf Science, 56*, 503–515.

Araujo, J. N., Mackinson, S., Stanford, R. J., & Hart, P. J. B. (2008). Exploring fisheries strategies for the western English Channel using an ecosystem model. *Ecological Modelling, 10*(4), 465–477.

Beck, M. W., Heck, K., Able, K., Childers, D., & Egglestone, D. (2001). The identification, conservation and management of estuarine and marine nurseries for fish and invertebrates. *Biosciences, 51*, 633–641.

Chagaris, D. D., Mahmoudi, B., Walters, C. J., & Allen, M. S. (2015). Simulating the trophic impacts of fishery policy options on the West Florida Shelf using Ecopath with Ecosim. *Mar Coast Fish, 7*(1), 44–58.

Chen, Z., Xu, S., Qiu, Y., Lin, Z., & Jia, X. (2009). Modeling the effects of fishery management and marine protected areas on the Beibu Gulf using spatial ecosystem simulation. *Fisheries Research, 100*, 222–229.

Christensen, V., Walters, C. J., & Pauly, D. (2005). *Ecopath with Ecosim: A User's Guide*. . Fisheries Centre of the University of British Columbia.

Christensen, V. (2000). Indicators for marine ecosystems affected by fisheries. *Marine and Freshwater Research, 51*(5), 447–450.

Colléter, M., Valls, A., Guitton, J., Morissette, L., & Arreguín-Sánchez, F. (2013). *EcoBase: A repository solution to gather and communicate information from EwE models*. Vancouver, Canada: Fisheries Centre of the University of British Columbia.

Costanza, R., D'Arge, R., De Groot, R., Farber, S., & Grasso, M. (1997). The value of world's ecosystem services and natural capital. *Nature, 387*, 253–260.

Directorate of Fisheries (2018) The report on the marine and inland fish catch for Goa state (2017–2018)

Forrest, R. E., Savina, M., Fulton, E. A., & Pitcher, T. J. (2015). Do marine ecosystem models give consistent policy evaluations? A comparison of Atlantis and Ecosim. *Fisheries Research, 167*, 293–312.

Froese R, Pauly D (2016) Fishbase worldwide web electronic publication. http://www.fishbase.org.

George, G., Desai, D. V., Gaonkar, C. A., Aboobacker, V. M., Vethamony, P., & Anil, A. C. (2013). Barnacle larval transport in the Mandovi- Zuari estuarine system, central west coast of India. *Journal of Oceanography, 69*(4), 451–466.

Heymans, J. J., Coll, M., Link, J. S., Mackinson, S., Steenbeek, J., Walters, C., & Christensen, V. (2016). Best practice in Ecopath with Ecosim food-web models for ecosystem-based management. *Ecological Modelling, 331*, 173–184.

Hou, X. Q., Chen, C. L., Sun, S. L., & Lu, S. Y. (2009). Research on marine fisheries resource investigation in Shenzhen City during the summer fishing moratorium, 2007–2008 (Translation, in Chinese). *Ocean Developers Manage, 1*, 106–112.

Hutchings, J. A., & Reynolds, J. D. (2004). Marine fish population collapses: consequences for recovery and extinction risk. *BioSciences, 54*, 297–309.

Kempton, R. A., & Taylor, L. R. (1978). The Q-statistic and the diversity of floras. *Nature, 275*(5677), 252–253.

Kessarkar, P. M., Suja, S., Sudheesh, V., Srivastava, S., & Rao, V. P. (2015). Iron ore pollution in Mandovi and Zuari estuarine sediments and its fate after mining ban. *Environmental Monitoring and Assessment, 187*(9), 572.

Lira, A., Angelini, R., Le Loc'h, F., Ménard, F., Lacerda, C., Frédou, T., & Frédou, F. L. (2018). Trophic flow structure of a neotropical estuary in northeastern Brazil and the comparison of ecosystem model indicators of estuaries. *Journal of Marine Systems, 182*, 31–45.

Liu, Y., Hog, M. J., & Ye, Q. T. (2008). Comparison of swing net catch before and after closed fishing seasons in Minnan Sea Waters (in Chinese). *Journal Fujian Fisheries, 4*, 60–63.

Manickchand-Heileman, S., Mendoza-Hill, J., Kong, A. L., & Arocha, F. (2004). A trophic model for exploring possible ecosystem impacts of fishing in the Gulf of Paria, between Venezuela and Trinidad. *Ecological Modelling, 172*(2), 307–322.

Mohamed, K. S., Puthra, P., Sathianandan, T., Baiju, M., Sairabanu, K., Lethy, K., Sahadevan, P., Nair, C., Lailabeevi, M., & Sivaprasad, P. S. (2014). *Report of the committee to evaluate fish wealth and impact of trawl ban along Kerala coast.* . Department of Fisheries.

Mohamed, K. S., Zacharia, P. U., Muthiah, C., Abdurahiman, K. P., & Nayak, T. H. (2008). *Trophic Modelling of the Arabian Sea Ecosystem off Karnataka and Simulation of Fishery Yields, Central Marine Fisheries Research Institute, Kerala, India, Technical Bulletin No. 51*

Pauly, D. (1997). Small-scale fisheries in the tropics: marginality, marginalisation and some implication for fisheries management. In: Pikitch et al., (Ed.), *Global trends: Fisheries Management, American Fisheries Society Symposium 20, Bethesda, Maryland*, pp. 40–49

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., & Torres, F. (1998). Fishing down marine food webs. *Science, 279*(5352), 860–863.

Pauly, D., Christensen, V., & Walters, C. (2000). Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. *ICES Journal of Marine Science, 57*, 697–706.

Phillippsen, J. S., Minte-Vera, C. V., Coll, M., & Angelini, R. (2019). Assessing fishing impacts in a tropical reservoir through an ecosystem modeling approach. *Reviews in Fish Biology and Fisheries, 29*(1), 125–146.

Pitcher, T. J., Watson, R., Haggan, N., Guénette, S., Kennish, R., Sumaila, U. R., Cook, D., Wilson, K., & Leung, A. (2000). Marine reserves and the restoration of fisheries and marine ecosystems in the South China Sea. *Bulletin of Marine Science, 66*(3), 543–566.

Qasim, S. Z., & Sen Gupta, R. (1981). Environmental Characteristics of the Mandovi-Zuari Estuarine System in Goa. *Estuarine, Coastal and Shelf Science, 13*, 557–578.

Qasim, S. Z. (1973). Productivity of backwaters and estuaries. *IBP Ecol Study, 3*, 143–154.

Robert, R. L. (1998). The biological characteristics of the Monterey bay squid catch and the effect of a two-day-per-week fishing closure. *CalCOFI Report, 39*, 204–211.

Schrank, W. E. (2005). The Newfoundland fishery: Ten years after the moratorium. *Marine Policy, 29*, 407–420.

Shannon, L. J., Coll, M., & Neira, S. (2009). Exploring the dynamics of ecological indicators using food web models fitted to time series of abundance and catch data. *Ecological Indicators, 9*(6), 1078–1095.

Shetye, S. R., Dileep Kumar, M., & Shankar, D. (2007). *Mandovi and Zuari estuaries.* . CSIR-National Institute of Oceanography.

Shetye, S. R. (2011). Indian estuaries: Dynamics, ecosystems, and threats. *National Academy Science Letters, 34*(7 & 8), 229–237.

Shirodkar, P. V., Deepthi, M., Vethamony, P., Mesquita, A. M., & Pradhan, U. K. (2012). Tide dependent seasonal changes in water quality and assimilative capacity of anthropogenically influenced Mormugao harbour water. *Indian Journal of Marine Sciences, 41*(4), 314–330.

Sreekanth, G. B., Chakraborty, S. K., Jaiswar, A. K., & Zacharia, P. U. (2018). A inventory on the coastal finfish and shellfish species of Zuari estuary, southwest coast of India. *Indian Journal of Geo-Marine Sciences*, *47*(5), 945–958.

Sreekanth, G. B., ManjuLekshmi, N., Chakraborty, S. K., Jaiswar, A. K., Zacharia, P. U., Singh, N. P., Vishnuradhan, R., & Pazhayamadom, D. G. (2016). Effect of monsoon on coastal fish diversity of Goa: an example from gill net fishery. *The Indian Journal of Fisheries, 63*(2), 8–18.

Sreekanth, G. B., Lekshmi, N. M., & Singh, N. P. (2017). Temporal Patterns in Fish Community Structure: Environmental Perturbations from a Well-Mixed Tropical Estuary. *Proceedings of the National Academy of Sciences, India, Section B: Biological Sciences, 87*(1), 135–145.

Sreekanth, G. B., ManjuLekshmi, N., Chakraborty, S. K., Jaiswar, A. K., & Singh, N. P. (2017). Seasonal fish species composition, catch rate and catch value in the small scale fishery of a tropical monsoon estuary along southwest coast of India. *Journal of Environmental Biology, 38*(1), 81–89.

Sreekanth, G. B., Chakraborty, S. K., Jaiswar, A. K., Zacharia, P. U., Mohamed, K. S., & Francour, P. (2020). Trophic network and food web characteristics in a small tropical monsoonal estuary: A comparison with other estuarine systems. *Indian Journal of Marine Science, 49*(5), 774–789.

Tsehaye, I., & Nagelkerke, L. A. J. (2008). Exploring optimal fishing scenarios for the multispecies artisanal fisheries of Eritrea using a trophic model. *Ecological Modelling, 212*(3–4), 319–333.

Vivekanandan, E., Narayanakumar, R., Najmudeen, T. M., Jayasankar, J., & Ramachandran, C. (2010). Marine fisheries policy brief-2; Seasonal fishing ban. *CMFRI Special Publication, 103*, 1–44.

Wang, Y., Duan, L., Li, S., Zeng, Z., & Failer, P. (2015). Modeling the effect of the seasonal fishing moratorium on the Pearl River Estuary using ecosystem simulation. *Ecological Modelling, 312*, 406–416.

Wang, Y., Li, S. Y., Duan, L. J., & Liu, Y. (2012). Fishery policy exploration in the Pearl River Estuary based on an Ecosim model. *Ecological Modelling, 230*, 34–43.

Walters, C., Pauly, D., Christensen, V., & Kitchell, J. F. (2000). Representing density dependent consequences of life history strategies in aquatic ecosystems: EcoSim II. *Ecosystems, 3*, 70–83.

Whitfield, A. K. (1999). Ichthyofaunal assemblages in estuaries: a South African case study. *Reviews in Fish Biology and Fisheries, 9*(2), 151–186.

Williams, I. D., Walsh, W. J., Miyasaka, A., & Friedlander, A. M. (2006). Effects of rotational closure on coral reef fishes in Waikiki-Diamond Head Fishery Management Area, Oahu Hawaii. *Marine Ecology Progress Series, 310*, 139–149.

Xiao, F. (2005). Analysis on result of closed summer season for Trawling at Minnanand Taiwan shallow bank fishing grounds (in Chinese). *Modern Fisheries Information, 20*, 23–25.

Yang, B. H., & Zhou, J. W. (2013). The effect valuation of 2012 summer fishing moratorium in the South China Seabase on the comparison among trawl, purse seine and gill net fishery (Translation, in Chinese). *Chin Fish, 3*, 73–75.

W1: https://goa365.tv/issues/E/illegal-trawler-fishing-in-river-zuari-a-law-order-problem-bmm-/09406.html

W2: https://www.navhindtimes.in/2018/02/28/goanews/trawlers-illegally-fishing-in-zuari/

W3: https://timesofindia.indiatimes.com/city/goa/trawler-trespasses-set-limits-in-zuari-seized/articleshow/70866404.cms

W4: https://timesofindia.indiatimes.com/city/goa/3-trawlers-caught-fishing-in-shallow-waters-off-siridao/articleshow/65546439.cms