Ecosystem-based fisheries management in the Pearl River Delta: Applying a computable general equilibrium model

Ying Wang, Zhejiang Sci-tech University, China
Jianfeng Hu, Zhejiang Sci-tech University, China
Haoran Pan, Beijing Normal University, China
Pierre Failler, University of Portsmouth, UK

Abstract

A comprehensive social-economic-ecological framework is applied to evaluate the input and output control policies for fishery management in the Pearl River Delta of China. The paper proposes a computable general equilibrium model that can evaluate changes in the economic and social indicators. This model is connected to an Ecopath with Ecosim model that can simulate the dynamics of an ecosystem. The integrated model is constructed to investigate how different scenarios of fishing effort and catch management reflecting varying levels of input and output in four fishery management simulations result in different states of the socio-economic and ecosystem structure. Four alterative fishing management scenarios are defined, namely: (1) fishing intensity and policies maintain the status quo; (2) fishing vessel reduction is implemented as an input control policy; (3) fishermen are required to transfer to others jobs as an input control policy; and (4) total landing reduction is implemented as an output control policy. The modelling results show that the output control policy has the most positive effect on ecosystem restoration and can increase overall social welfare. The fishermen switching policy leads to the most positive increase in economic indicators, whereas the policy of fishing vessels reduction has a positive effect on ecosystem restoration, but also a considerable, negative impact on the social and economic dimensions. To achieve sustainable utilisation of marine fishery resources, China’s fisheries management policies should be developed to enhance total output control and ecosystem reconstruction.

Keywords

Input and output control policy; Ecosystem-based fisheries management; Computable general equilibrium; Ecopath with Ecosim; Pearl River Delta

1. Introduction

Over the last three decades, many important marine fish populations have been severely exploited. This has resulted in enormous losses for economies that are dependent upon local coastal fishery. Fishery management experts have recognised that, in order to effectively address the problem of overfishing, multidisciplinary analysis of complicated ecosystem dynamics and the socio-economic drivers shaping the interaction between humans and the environment must be developed [1]. Changes in fishing activities have implications for the local fisheries supply sectors, as well as processing, distribution, servicing, and other supply-chain sectors. Furthermore, variations of marine ecosystem and fishery policy implementation impact the industrial sectors and the social structure of fishing communities differently [2].

Thus, an integrated approach that links the ecosystem model and the socio-economic model using fisheries economics is needed to understand both the ecosystem and the socio-economic systems [3]. The ecosystem can be modelled as a food web of interacting species. The economic system has similar characteristics to the ecosystem and can also be represented by a complex network of interacting components. On this basis, ecosystem-based fishing management (EBFM) has become a promising research method [4, 5]. However, analyses involving this model are scarce because it requires complex interdisciplinary research [6-8].

Jin et al. (2003) [9] and Byron et al. (2015) [10] seek sustainable fishery management policies by connecting the regional economic input-output model with a linear ecological model of coastal marine ecosystems. Kaplan and Leonard (2012) combine an Atlantis ecosystem model with the input-output model to predict the ecological and economic impacts of fishery management [11]. Wang et al. (2016) design an ecological–economic–social integrated framework to evaluate the effectiveness of different policies in the Pearl River Estuary of China. In particular, a linear social accounting matrix (SAM) model is used in conjunction with the ecosystem model built by the Ecopath with Ecosim (EwE) software [12].

Although the linear economic-ecological models mentioned above can handle numerous elements (such as species production sectors, households, etc.), the approach cannot capture the nonlinear interactions between socio-economic systems and ecosystems. Since the prices are fixed in a linear system, substitution of goods and services is not allowed [13]. Thus, the optimal fishing intensity for maximum social welfare cannot be obtained using the linear model. However, the nonlinear dynamic general equilibrium model can describe utility maximisation for consumers and profit maximisation for producers. As the three earliest applications of a general equilibrium model in marine EBFM, Finnoff and Tschirhart (2003, 2008) [3, 14] and Hussain and Tschirhart (2013) [15] adopt a dynamic general equilibrium model of Alaskan fisheries to estimate the welfare variation associated with setting different allowable fishing quotas. The impacts of the fish stock rebuilding policy [16] and reducing subsidy policy [17] are analysed in a general equilibrium model with heterogeneous fishing vessels. There are a few studies where the dynamic Stochastic General Equilibrium model is employed to solve the uncertainties in fisheries and natural resource management [18-20]. In particular, the Computable General Equilibrium (CGE) model provides an analytical framework to assess the impact of fishery policies on regional economies and social welfare. Carvalho et al. (2011) evaluate the effects of fishery subsidy adjustment policies on regional economic variables of the Azores by constructing a dynamic CGE model [21]. The impacts on regional fisheries from supply-side and demand-side changes [22, 23] and multiregional economic impacts [24] are evaluated using the CGE model for Alaskan fisheries.

However, those regional economic models did not capture the interaction relationships between the socio-economic system and the ecosystem. Only one study, Jin et al. (2012), has coupled the CGE model with an end-to-end model of a marine food web to measure the effects of implementing EBFM in New England [2]. That study proposed that future research should connect the CGE model with top-down ecosystem models or Ecopath with Ecosim (EwE) models to conduct simulations of policy implementation effects.

Although the EBFM approach has frequently been utilised for fishery policy analysis, to the best of our knowledge, no study has focused on the integration of an ecosystem model covering the full trophic spectrum species into a nonlinear regional economic system model. A multidisciplinary framework is established in this study for assessing the multiple dimensions of the impacts on the ecosystem and socio-economic systems. The integration of the two system models is achieved by coupling the CGE model and the EwE model to simulate the impact of a fishery input and output control policy on economic output, labour income, capital income, and household welfare in the Pearl River Delta (PRD) in China. In addition, the response of coastal fishery resources and ecosystem structures to policy adjustments have been included in the simulation.

2. Methods

The public resource characteristics of marine fisheries pose various challenges to management. The PRD is formed by the alluvial accumulation of the Pearl River and its tributaries, which eventually flow into the South China Sea. Meanwhile, the Pearl River Estuary is rich in nutrients and biological species, and is therefore one of the most famous fishing grounds in China [25]. However, with the rapid population growth in the PRD and the increasing consumption of food, the pressure on fishery resources has increased [12]. Consequently, the PRD is facing a double crisis of economic rents dissipation and resource depletion, which is a quintessential example of non-economic growth [26]. The Chinese government has played a key role in promoting the sustainable exploitation of marine fishery in the PRD by implementing multiple fishery management policies, including a range of input controls, output controls, technical control instruments, and other economic measures [27-29]. This study analyses this situation using an integrated framework that links a CGE model to an EwE model to understand the trade-off of input and output control policy effects in the PRD in terms of economic, social, and ecological aspects.

The PRD is a typical estuarine ecosystem with obvious marginal effects and abundant biological species. This is illustrated by the complex interactions between species in the food web. Accordingly, understanding the relationship between these functional groups is integral to establishing an ecological EwE model and to designing policies to guide the sustainable use of resources. The species in the EwE model are described in terms of the relationship between predators and prey, and some of them are commercially exploited fish. The economic value of these fish constitutes the total value added of the marine fishing sector of the economic system. The human socio-economic system is described by the CGE model which is constructed based on the SAM that provides detailed information for economic data and structure. The CGE model provides a typical multi-sector and multi-factor analytical framework of the PRD economy in which all product and factor markets are cleared following the Walrasian general equilibrium theory. Finally, the socio-economic system and ecosystem are linked to produce an integrated framework for fisheries management. The detailed interactions between the human socio-economic system and the coastal ecosystem are showed in **Figure 1**.

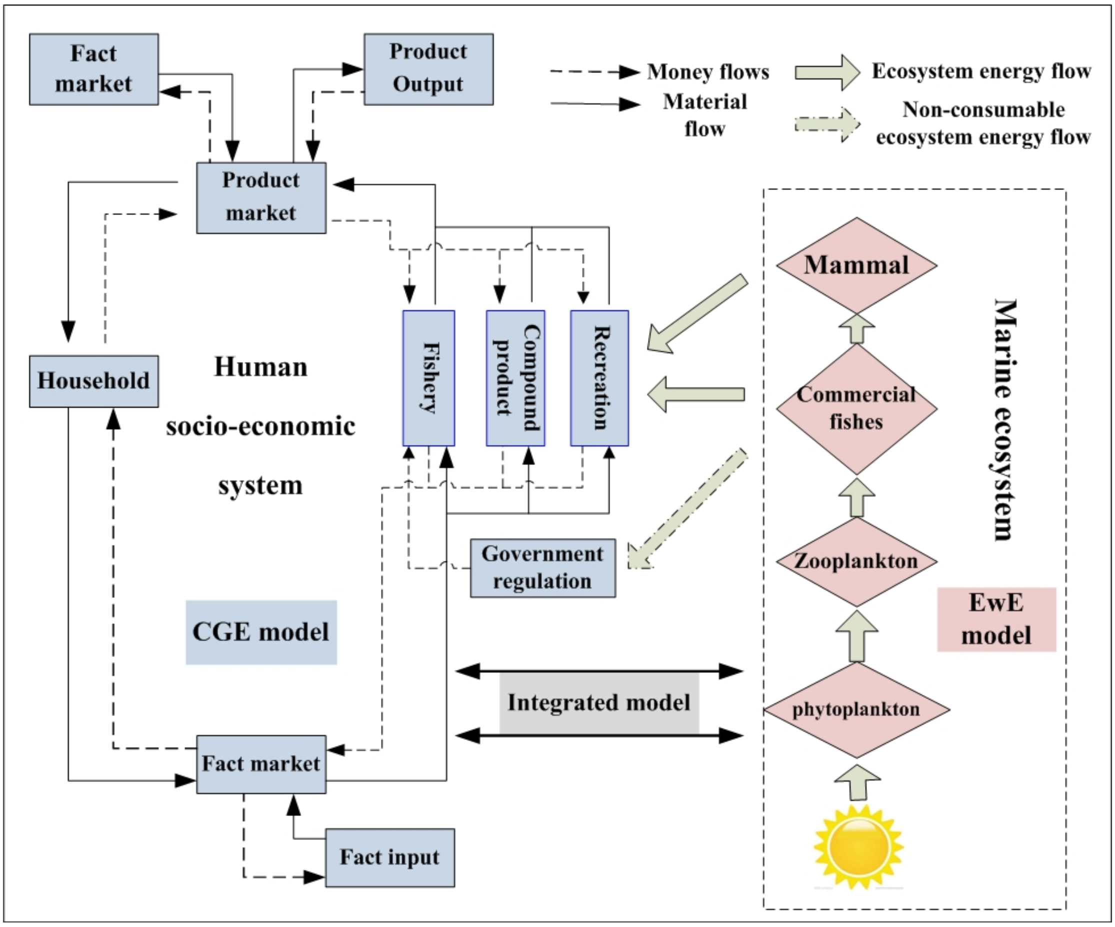

Figure 1. Basic components of an integrated model

2.1 The CGE model of the PRD fishing industry

A foundational model for the PRD economy is provided by the multi-sectoral CGE model, which can be calibrated using the SAM database for 2015. The definition of accounts in the CGE model is derived from SAM, which is based on the input-output table of 135 industries released by the Guangdong Provincial Bureau of Statistics. The detailed data of SAM accounts are from the input and output extension table of Guangdong province in 2015, the Guangdong finance yearbook for 2016, and the Guangdong statistical yearbook for 2016, and so on, while the data of fishery-related production and institutional accounts are from the China fishery statistical yearbook for 2016.

For the research purpose of this paper, the structure of the SAM for the PRD economy is aggregated into six sectors, including marine fish-harvesting, other seafood industry (freshwater fish-harvesting, marine aquaculture, freshwater aquaculture), aquatic product processing, fishery manufacturing (including fishing machinery and tools manufacturing, fish feed and medicines, and the fishery construction industry), fishery servicing, and non-fishery industry (other agriculture, industry, and services sectors). The details of the SAM are presented in **Appendix I**. Values in CNY (¥) are converted into USD ($), based on the fixed average exchange rate for 2015 of $1 = ¥6.2284.

The CGE model is established and solved through the general algebraic modelling system [30] and incorporates the behaviour equations of four economic agents: enterprises, households, government, and the rest of the world. The following briefly describes the model structure and the behaviour of each economic agent. Due to the large number of equations, parameters, and variables, the CGE model is not shown here, but it is available upon request.

2.1.1 Enterprises

According to the SAM, industries are aggregated into six sectors. Hence, commodities are also aggregated into six commodities groups, which are produced by the corresponding industry sectors. All of the economic agents in the PRD are assumed to be rational: producers seek to maximise profits and consumers pursue utility maximisation. The economic production in PRD has a two-layered nested structure. In the first level, constant elasticity of substitution (CES) functions describe the relationship between composite factors and aggregate intermediate inputs. In the second level, the aggregate intermediate input is expressed as a Leontief function of disaggregated intermediate input, whereas the optimal level of value-add is achieved by applying labour and capital according to the CES function.

2.1.2 Households

Households are divided into fishery households and non-fishery households, and each household group receives a share of labour income, capital income from the factor market, and other transfers from the government and enterprises. Household expenditure consists of household consumption and personal income tax. Household consumption is derived by maximising a Stone-Geary utility function by allocating different goods and services according to the linear expenditure system. Households pay personal income tax to the government at a fixed percentage of their net income. Household savings are proportional to disposable income after tax, and the propensity to save is exogenous.

2.1.3 Government

Taxation is the source of fiscal revenue for the government, such as direct taxation on employment income and capital income, indirect taxation on products and production, and transfer payments from other regions and the central government. Total government expenditures are a function of public consumption, subsidies on products and production, and transfer payments to households and enterprises. Government saving is positive when government revenue exceeds government expenditure; otherwise, savings are negative, and the government runs a deficit.

2.1.4 Rest of the world

On the export side, enterprises allocate their output, such as seafood, between the local market under investigation and the market outside the modelled region. Export activities are modelled based on enterprises' decisions, which maximise their total revenues from the sales of their commodity to local markets and the rest of the world according to the constant elasticity of transformation function.

Thus, commodities are imported from outside the region, in addition to being produced locally. Import activities are modelled based on household decisions, which minimise the costs of composite goods consumption by applying the CES Armington trade aggregation function. This function specifies how households choose between domestically produced commodities and imported commodities.

2.1.5 Market equilibrium

The CGE model is balanced (that is, in all markets, supply equals demand) when all commodity and factor markets can find the equilibrium solution of quantities and prices. All of the factor markets and the commodity markets follow the condition of market clearing.

2.1.6 Welfare evaluation

Welfare economics evaluates the social welfare change based on two indices: equivalent variation and compensating variation (CV). The CV is applied to evaluate social welfare, which reflects the change in household utility assuming that prices change after policy simulations.

$$CV = e(P1, u(QH1)) - e(P1, u(QH0)) \qquad \text{(Eq. 1)}$$

where *e* (P, *u*) is the expenditure function in microeconomics, P1 is the price of the commodity after the policy simulation, U(QH0) is the household utility levels in the status quo, and U(QH1) is the household utility levels after policy simulation.

2.2 Ecosystem model

In our study, the coastal ecosystem of the PRD is constructed using the EwE software, which has been extensively applied for simulating the food web of aquatic ecosystems [31]. The methodology of Ecopath and Ecosim modelling is detailed in the EwE instruction manual [32, 33]. The Ecopath is a static modelling approach that seeks to achieve a balance of mass and focuses on energy flows between trophic levels in the ecosystem. The mass balance equation for each functional group is based on several components and can be expressed as:

$$B_i \times (P/B)_i \times EE_i = Y_i + \sum_{j=1}^{n} B_j \times (Q/B)_j \times DC_{ij} + E_i + BA_i \qquad \text{(Eq. 2)}$$

where B_i and B_j are the biomass of functional group *i and j*, and $(P/B)_i$ is the ratio of production to biomass and determines the total mortality of *i*. EE_i is the ecotrophic efficiency, which represents the proportion of production utilised by the ecosystem. Y_i is the fishing mortality. $(Q/B)_j$ is the ratio of food consumption to the biomass of predator *j*. DC_{ij} is a matrix that represents the proportion of group *i* in the diet of predator *j*. E_i is the net migration rate of *i,* and BA_i is the biomass accumulation rate of *i*.

A 2008 Ecopath model has been introduced in previous studies (whereby the biomass data were derived from a 2008 survey of fishery resources in the South China Sea) [34]. In this model, all of the species of the PRD ecosystem are categorised into 24 functional groups with 10 commercially exploited fish species, including *Trachurus japonicus*, *Decapterus maruadsi*, Trichiurus haumela, Saurida., *Psenopsis anomala*, *Upeneus bensasi*, *Nemipterus virgatus*, *Priacanthus macracanthus*, *Priacanthus tayenus*, and cephalopods. The data and parameters estimation for the Ecopath ecosystem models are summarised in **Appendix II**.

Ecosim is a dynamic model based on a static Ecopath model, which describes the impact of the changes of fishing intensity on the ecosystem over time. The differential equation describing the prediction of biomass time series data in Ecosim is as follows:

$$dB_i / dt = g_i \sum_{j=1}^{n} Q_{ji} - \sum_{j=1}^{n} Q_{ij} + I_i - (M_i + F_i + e_i) B_i \qquad \text{(Eq. 3)}$$

where dB_i/dt represents the growth rate of biomass B_i for the functional group *i* over time period *t*. g_i is net growth efficiency. Q_{ij} is the biomass consumption rate of predator *j* to prey *i*. I_i and e_i are the immigration and emigration rate, respectively. M_i is the non-predatory natural mortality rate, and F_i is the fishing mortality rate.

As one of the key parameters to describe the strength of the relationship between predator and prey, the vulnerability parameter can be calculated using the actual catch data and fishing effort (power of fishing vessels and fisherman labour) time series data from 1995 to 2015. Based on these inputs, the Ecosim model can dynamically predict the biomass, catch, and other ecosystem indicators from 1995 to 2015 according to alternative fishing strategies.

2.3 Linking the economic model with the ecosystem model and policy scenarios

A key question in managing public resources is whether to control the input (capital, labour and other intermediate input, etc.) or the actual output level. Chinese fishery authorities have implemented a series of input control policies (fisheries license system, “double control” system of total number and total engine power of marine fishing vessels, fishermen exit and relocation system) and output control policies (“zero growth” and “negative growth” policy of marine fishing catches) to strengthen the protection of marine ecosystems in the PRD [35]. According to the fishing policy implemented in the PRD, we designed four scenarios to simulate the policy effect in terms of economic, social, and ecological aspects.

SCN0 or current situation: the baseline, this scenario depicts the effects of fishing intensity and policies under current trends.

SCN1 or fishing vessel reduction policy (input control): To address the rapid growth in the number of fishing vessels, Guangdong Ocean and Fisheries Bureau proposed a reduction in the number of marine fishing vessels, a plan which could be implemented on an annual basis. To illustrate the effect of this policy, we assume a 10% reduction in the fishery manufacturing products as an intermediate input of marine fishing in the CGE model. Accordingly, the fishing effort of fishing vessel input is reduced by 10% in the EwE model for 2015.

SCN2 or fishermen transfer to other jobs policy (input control): To help local fishermen transform their role in the fishing industry, subsidising aquaculture gear and offering training are means of supporting China's strategic shift from fishery as a quantity-expansion industry to an efficiency-motivated industry [36, 37]. We model this in the CGE model by assuming a 10% transfer of fishermen to other sectors in 2015, while the total supply of labour is kept constant. Accordingly, there is a 10% reduction of marine harvesting fishermen in the EwE model for 2015.

SCN3 or total landing reduction policy (output control): As output control is almost totally absent in the PRD, comparing the policy effects between output and input control policy could be insightful. We assume that total fishery landings are reduced in 2015 and model this by a 10% reduction in the value-add of marine harvesting in the CGE model and a corresponding 10% reduction of the total landings of commercially harvested fish species in the EwE model.

In this study, the CGE model and the EwE models are calibrated using economic and ecological data, respectively. The model is constructed to calculate the equilibrium quantities at a given baseline set of prices (P0) in the current state. To simulate the changes of economic, ecological, and social dimensions under different policy scenarios, the equilibrium solution is obtained by re-running the model after changing the variables.

3. Results of the policy simulation

3.1 Economic impacts

The values of major macroeconomic variables for the four fishery policy simulations in 2015 are listed in **Table 1**. In the baseline scenario (**SCN0**), all prices of commodities and factors are assumed to be 1.0 and the welfare change is assumed to be 0. **SCN1** describes the 10% reduction of marine fishing vessels policy scenario. **SCN2** describes the policy of a 10% transfer of marine fishermen to other sectors. **SCN3** describes the 10% reduction of total catch of commercially harvested fish species in the PRD policy scenario. In this study, the percentage changes with respect to the baseline macroeconomic indicators are used to describe the impacts of fishery policies on economic indicators.

Table 1. Values for key microeconomic variables and fishing policy examined under four scenarios in 2015

Economic indicator	Sectors	SCN0（million $）	SCN1（%）	SCN2（%）	SCN3（%）
Industry output	Marine fish-producing	2175	-3.677	-3.540	-6.366
	Other seafood industries	16245	0.029	0.256	0.064
	Aquatic product processing	3512	-0.904	-0.498	-0.296
	Fishery manufacturing	2285	-8.428	-8.163	-0.068
	Fishery servicing	16480	0.022	0.112	0.008
	Total fishery industry	**40696**	**-0.727**	**-0.543**	**-0.384**
	Other agriculture	53727	-0.055	-0.020	0.005

	Other manufacturing	2621500	-0.459	-0.433	-0.004
	Other services	989210	0.019	1.462	-0.043
	Total non-fishery industry	**3664437**	**-0.327**	**0.072**	**-0.014**
	Total	**3705133**	**-0.331**	**0.065**	**-0.018**
Labour income	Marine fish-producing	827	4.074	-10.000	4.055
	Other seafood industries	6430	0.016	-0.152	-0.049
	Aquatic product processing	659	-0.228	-0.092	0.459
	Fishery manufacturing	427	0.033	-0.438	0.082
	Fishery servicing	3783	0.009	-0.287	0.022
	Total fishery industry	**12126**	**0.278**	**-0.873**	**0.286**
	Other agriculture	42969	-0.005	-0.094	0.010
	Other manufacturing	235050	-0.026	-0.408	0.026
	Other services	284250	0.007	0.359	-0.035
	Total non-fishery industry	**562269**	**-0.008**	**0.004**	**-0.006**
	Total	**574396**	**-0.001**	**-0.015**	**-0.000**
Capital income	Marine fish-producing	15	4.078	1.553	4.058
	Other seafood industries	115	0.023	1.627	-0.044
	Aquatic product processing	835	-0.222	1.463	0.462
	Fishery manufacturing	492	0.038	1.114	0.086
	Fishery servicing	67	0.016	2.208	0.028
	Total fishery industry	**1525**	**-0.068**	**1.397**	**0.318**
	Other agriculture	769	0.001	1.686	0.014
	Other manufacturing	202120	-0.020	1.143	0.030
	Other services	226390	0.013	2.871	-0.031
	Total non-fishery industry	**429279**	**-0.002**	**2.055**	**-0.002**
	Total	**430804**	**-0.003**	**2.053**	**-0.001**

SCN0: Column 3 in Table 1 presents the baseline values of six fishery-related sectors—marine fish-harvesting, other seafood industry, aquatic product processing, fishery manufacturing, fishery servicing, and non-fishery industry—for the key variables, including total output, labour income, and capital income.

SCN1: Column 4 in Table 1 indicates that when a 10% reduction in fishing vessels occurs, total output in the marine fishery harvesting sector, aquatic product processing sector, and fishery manufacturing sector decreases by 3.677%, 0.904%, and 8.428 %, respectively. By contrast, the total output in the other seafood industry and fishery servicing sector increases by 0.029% and 0.022%, respectively, because the output of other aquatic products are alternatives to marine fishing products. Accordingly, the increase in the total output of fishery servicing, including recreational fishery, is due to the increase in consumer demand. Additionally, the reduced intermediate inputs of fishery manufacturing leads to an increase in demand for fishing vessel crews and capital. Thus, labour income and capital income of the marine fish-producing sector increase by 4.078% and 4.074%, respectively, and the price of labour and capital both increase by 0.200%. The impacts on prices of related seafood are mixed. Reduced supplies lead to a 5.100% increase in marine fish products' prices, while the aggregate price of other alternative seafood falls by 0.200%.

Total output in the aggregated non-fishery industry (agricultural products and industrial products) also shows less significant decreases in percentage terms than the seafood sectors because the demand for commodities supplied by the non-fishery industry declined. The manufacturing sector is most heavily affected with a decline of 0.459% due to the fishing vessel reduction policy, which led to a reduction in fishery-related manufacturing. On the whole, total output, labour income, and capital income in the PRD economy decrease by 0.331%, 0.001%, and 0.001%, respectively.

SCN2: Column 5 in Table 1 presents the results for scenario 2, which details the impact of a 10% transfer of marine harvesting fishermen to other sectors. Output in the marine fishery harvesting, aquatic product processing, and fishery manufacturing sectors fall by 3.540%, 0.498%, and 8.163%, respectively. However, output in the other seafood industry and fishery servicing sectors increase by 0.256% and 0.112%, respectively, because of the labour transfer to alternative production industries. The labour income decreases in virtually all fishery-related sectors, especially the marine fish-producing sector (-10%), although the price of labour increases by 1.2%. By contrast, capital income exhibits an increase from 1% to 2.5% because the demand for capital increases. The prices of marine harvesting species, aquatic processed products, fishery-related manufacturing products, and fishery-related services increase by 0.9%, 0.7%, 0.6%, and 0.9%, respectively.

Total output in the aggregated non-fishery industries increases by less than 0.1% as a result of demand reduction with respect to the commodities produced by the non-fishery sectors. Output, labour income, and capital income in the aggregated non-fishery industries increase by 0.072%, 0.004%, and 0.024%, respectively. The service sectors are most affected, with total output, labour income, and capital income of the service sector all increasing. This effect is driven by the increase in labour input and capital return. On the whole, total output and capital income increase by 0.065% and 2.05%, while labour income decreases by 0.015%.

SCN3: Column 6 in Table 1 presents the results for the output control policy simulation, whereby the total catch of the marine fishery industry is reduced by 10%. Output in the marine fishery harvesting and aquatic product processing sectors decrease by 6.366% and 0.296%, respectively. However, the labour income in these two sectors increase by 4.055% and 0.459%, respectively, while the capital income in these two sectors increase by 4.058% and 0.462%, respectively. Although the total catch reduction policy results in a reduction in total output, the increase in catch prices brings about an increase in factor returns. The labour and capital income in the other fishery-related and non-fishery industries exhibit a slight increase because the output control policy promotes a transfer of surplus labour and capital to other industries. The price of seafood from marine harvesting increases by 10.8% and the price for processed seafood increases slightly (0.2%) due to the reduction in the total catch from marine fishery. This is because, according to the elasticity of supply and demand, consumers' purchasing intention turns to available substitutes, thus increasing demand for relatively cheaper substitutes, which leads to an increase in the price of substitutes.

It is also notable that the impact of the output reduction policy on total output is much less in the aggregated non-fishery sectors (-0.014 %) than for the combined fishery sector (-0.384 %). For instance, total output in the manufacturing and service sectors fall by 0.004% and 0.043%, respectively, but output increases by 0.005% in other agriculture sectors. With the supply of fishing vessel crews transferring to other fishing relevant sectors, the price of labour is decreased by 0.300% (although this is not shown in the table). Output, labour income, and capital income in the aggregated non-fishery sectors decrease by 0.014%, 0.006%, and 0.002%, respectively. The service sectors are most affected whereby total output, labour income, and capital income of the service sector decrease. This is caused by the reduction in the seafood supply. In general, total output in the PRD economy decreases slightly (-0.018%), while the labour income and capital income both decrease less than 0.001%.

3.2 Social impact

The utility and welfare change of households represent an estimation of the social impact of fishery policies.

SCN1: The total utility of fishery households falls by 0.48%, and the welfare cost is 4.5 million dollars for fishermen. Additionally, the non-fishery household income category also experiences a welfare loss (86.82 million dollars). The fishing vessels reduction policy causes the total utility of society to decline by 0.040% and the welfare cost is 91.18 million dollars.

SCN2: The total utility of fishery and non-fishery households both decrease in Scenario 2 (**Table 2**). In percentage terms, the decrease is largest for the fishery household classes (-2.00%). Fishery households also experience the largest losses in welfare, as measured by the Hicksian CV (-19 million dollars). In contrast to the previous scenario, in this case, since the transfer of fishermen to other sectors increases total outputs for the service industry in the economy, this leads to a positive welfare change for non-fishery sectors (0.76%) and all household categories (0.75%). This result occurs because the production factors of labour which are transferred to non-fishery production sectors are outweighed by the reduction in the output of commodities.

SCN3: Note that, unlike the other two scenarios, the income and welfare of fishery households increase in scenario 3 (**Table 2**). The utility of the fishery household class increases 0.95% when total marine fish-harvesting is cut by 10% and the welfare gain for fishermen is 9 million dollars based on the Hicksian CV. Total utility of non-fishery households increases by 0.08%, and the welfare gain for the non-fishery household's category is 173 million dollars.

Table 2. The utility and welfare change for two aggregated households' categories

Households	Utility change (%)				Welfare change (million dollars)		
	SCN0	SCN1	SCN2	SCN3	SCN1	SCN2	SCN3
Fishery households	379	-0.476	-2.002	0.947	-4.50	-19	9
Non-fishery households	107401	-0.038	0.760	0.076	-86.82	1734	173
All households	**379**	-0.040	0.751	0.079	**-91.18**	**1703**	**182**

3.3 Ecological impact

Based on the current exploitation intensity (SCN0), stocks of commercially exploited fish species are expected to decrease during the 20-year (1995–2015) simulation period, except benthos such as zoobenthos, benthic crustaceans, and echinoderms (**Figure 2**). The total biomass of the 10 commercially exploited species exhibits a population reduction of nearly 31.5% compared to their original of biomass levels in 1995 (baseline) for SCN0.

Compared with the current exploitation intensity (SCN0), total annual production of the 10 commercially exploited species for SCN1, SCN2, and SCN3 increases by 18.6%, 25.7%, and 36.8%, respectively. The results indicate that the total biomass is higher in all three hypothetical scenarios (SCN1, SCN2, SCN3) than in SCN0, whereby the most significant improvement is recorded when the policy of a 10% reduction in the total catch of commercially exploited species (SCN3) is implemented. Specifically, in SCN3, the populations of all of the commercially exploited species are significantly higher than under the original fishing intensity scenario (SCN0), such as the *Trachurus japonicus*, *Decapterus maruadsi*, and other pelagic species are almost restored to the 1995 baseline population level. The biomass levels of functional groups in lower trophic levels, such as zoobenthos, echinoderms, etc., show decreasing population trends. This can be explained by the increase of predator biomass in higher trophic levels. Comparing SCN1 to the status quo scenario (SCN0), the abundance of commercially exploited species increases under the fishing vessels reduction scenario (SCN1). The trend toward population recovery is even more significant in the fishermen transfer scenario (SCN2) than in the fishing vessels reduction policy scenario (SCN1) (**Figure 2**). In sum, these simulations (SCN1, SCN2, and SCN3) suggest that the restoration of predators will reduce the biomass of benthos, such as zoobenthos, benthic crustaceans, and echinoderms in the lower trophic levels. (The trophic levels of 24 species are calculated using the EwE model and presented in **Appendix II**).

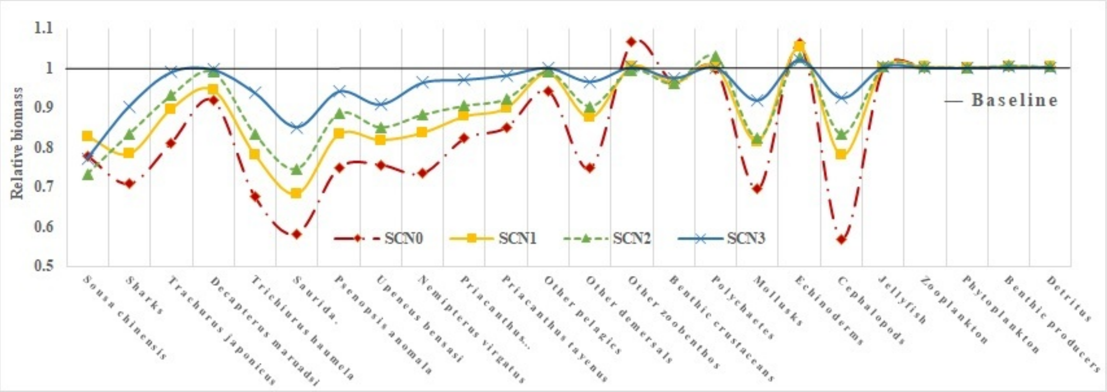

Figure 2. The biomass dynamics of 24 functional groups under four policy scenarios

The state of the entire ecosystem cannot be described by a single indicator. Three ecosystem indictors are selected to estimate the state of the PRD ecosystem: (1) ecosystem maturity, evaluated by the mean trophic level of fishery catch (MTLC) index; (2) ecosystem sustainability, evaluated by the fishing-in-balance (FiB) index; and (3) biological diversity, evaluated by Kempton's Q index. The theory and method of the three ecosystem indicators are detailed in Shannon et al. (2009) [38]. Compared with the baseline level (1995), the ecosystem indicators of the PRD in 2015 under the different scenarios are presented in **Table 3**.

Table 3. The ecosystem indicators change during four policy scenarios in 2015

	MTLC	FiB	Kempton's Q
Baseline (1995)	2.410	-0.021	4.390
SCN0	2.360	-0.025	4.257
SCN1	2.379	-0.005	4.461
SCN2	2.386	-0.010	4.548
SCN3	2.388	-0.003	4.650

Although the MTLC index is affected by fishing methods (such as fishing gear) [39], it is still recognised as an important ecological indicator for assessing the fishing pressure on ecosystems [40, 41]. Systems with lower values of MTLC are considered to be under more pressure. Although the fishing intensity has been assumed to be reduced in SCN1, SCN2, and SCN3, the MTLC index was still lower than the baseline year of 1995, which shows that fishing pressure still exists in the PRD ecosystem. The MTLC indicator declined most significantly in SCN0, which suggests that the current fishing intensity exerts the most pressure on the ecosystem.

FiB=0 when an increase/decrease in trophic level has a corresponding decrease/increase in the mass of fish caught. The FiB will decline if the increase in catches cannot compensate for a reduced trophic level [42]. Under the current scenario (SCN0), the FiB index declines in comparison to the baseline level from 1995, which proves that the fish resources in the PRD ecosystem are experiencing depletion. It is worth mentioning that smaller and lower trophic level species are gradually replacing larger and higher trophic level species. In addition, a positive trend for FiB is usually due to an increase in primary production or the geographical expansion and technological improvement of fishing [43]. The FiB index rises and approaches 0 under the total catch reduction policy (SCN3), indicating that the impact of fishing on the ecosystem can still result in the ecosystem carrying capacity being stable. The FiB index increases under the fishing vessels reduction policy (SCN1) and fishermen transfer policy (SCN2), indicating that the reduction in fishing intensity leads to an increase in primary production.

Kempton's Q index was proposed by Kempton and Taylor to represent the impact of different levels of fishing intensity on the biodiversity of the ecosystem [44, 45]. The observed decrease of the index in the status quo scenario (SCN0) is caused by the decline in biodiversity caused by overfishing. In all three hypothetical scenarios (SCN1, SCN2, SCN3), the Kempton's Q index increases and exceeds the diversity level in the base year (1995), possibly reflecting the fact that the decline in fishing intensity would lead to the restoration of the ecosystem and an increase in biodiversity.

Fishing activities cause ecosystems to be subjected to changing external environmental pressures, which can have various impacts on ecosystems and biological communities, including changes in ecosystem structure and a loss of biodiversity [42, 46, 47]. The fishing vessel reduction (SCN1), fishermen transfer (SCN2), and total catch reduction (SCN3) policies all have positive impacts on most ecological indicators, with the total catch reduction policy (SCN3) performing best among all of the ecological indicators assessed in this study. The effect of the fishermen transfer policy on ecosystem restoration is more obvious than that of the fishing vessel reduction policy.

4. Discussion

The fact that fishery resources are common goods leads to the problem of over-exploitation [48, 49]. In a fishing ground which is experiencing or threatened by overfishing, the fishery administration can attempt to deal with the problem through the use of either input or output controls [50, 51]. In theory, input controls are relatively simple to implement [52]. The government might establish a limited access program or reduce the number of fishing vessels and fishermen. With respect to output controls, two forms of controls have been emphasised: (i) total allowable catch (TAC) and quota management systems [53] and (ii) the individual transferable quota (ITQ) system [54, 55]. Currently, fishery management in the PRD relies on a range of input control policies which limit fishing capacity, including a fishing license system and "double control" system, as well as the fishermen exit and relocation system [27]. However, policies which focus on output control are almost absent, although a national total catch limit objective has been defined [28].

In this study, the socio-economic and ecological dimensions are considered to compare the effects of input and output control policies in the PRD. Specifically, a CGE model that can assess changes in the socio-economic system is integrated with an EwE model that can simulate the dynamics of the PRD ecosystem. In four simulations, we show how scenarios reflecting varying levels of input and output control policies result in different states of ecosystem structure and socio-economic benefits.

The economic impact of each simulation is different as a result of changing the intermediate input and final output. For instance, the output control policy has the greatest negative impact on the output of the marine fish-harvesting sector, but due to the substitution effect of other aquatic products, the output change of fishery-related industries is the smallest among the three hypothetical policy scenarios. At the same time, higher prices for marine harvesting products lead to higher returns on labour and capital investment. By contrast, the input control policy of reducing fishing vessels not only reduces the output of the marine fishing industry, but also causes a great negative impact on the fishery manufacturing industry. This leads to a decline in the total economic output. The input control policy of fishermen transfers not only has a less negative impact on fishery-related industries, but also increases the output, labour, and capital returns of non-fishery sectors. Therefore, the fishermen transfer policy is the best in terms of economic performance.

Regarding the social aspect, the output control policy increases total social welfare, while the fishermen transfer policy can increase the welfare of non-fishery households, but it has a great negative impact on the total welfare of fishery communities. The fishing vessels reduction policy also leads to a decline in overall social welfare.

In terms of ecology, our model results demonstrate that each of the policy changes positively affects the ecosystem. The output control policy has the most obvious effect on the protection of the PRD ecosystem, and the fishermen transfer policy has a more significant effect than the fishing vessel reduction policy.

Since 1987, China has proposed a total input control policy called the “double control” system, which is the control of both the total number of engine-powered fishing vessels and their total engine power [35]. Because the reduction in power is difficult to measure by economic indicators, this study simulates the policy impact of reduced input in the fishing vessels manufacturing industry. The simulation results show that the input control policy of a 10% fishing vessel reduction will reduce the output (-0.727%) and capital income (-0.068%) but increase the labour income in fishery sectors (0.278%). The input control policy did contribute to the recovery of fishery resources of 18.6% in terms of fish biomass, but it exemplified the least improvement in the status of the ecosystem. It also leads to a decrease in overall social welfare (-0.040%).

It is also important to help fishermen switch to other industries because of the loss of jobs they face from simply reducing the number of fishing vessels. To effectively maintain the social stability of the fishing community and promote the adjustment of the industrial structure of marine fisheries, since 2020, the Ministry of Agriculture has adopted a fishery subsidy for fishermen exiting and relocating to encourage the decommissioning of fishing vessels and relocation of fishermen [56]. The simulation indicates that the input control policy of a 10% fishermen transfer to other industries reduces the output of the fishery sector (-0.543%) and the welfare of fishermen (-2.000%), but it has a positive impact on output (0.072%), labour income (0.004%), and capital income (2.055%) of non-fishery sectors and the welfare of non-fishery residents (0.76%). It can increase the biomass of commercial exploited fish species in the ecosystem considerably. As a result, the maturity of the ecosystem, and especially the mean trophic level of fish caught, shows a larger improvement than under the fishing vessels reduction policy. Therefore, the fishermen transfer policy can not only effectively improve the total output and welfare of society, but also promote the restoration of the ecosystem.

One of the major trends in the management of marine fisheries in China is to implement a total catch control system [27]. However, since the estuary is characterised by an abundance of fish species and a large number of fishing vessels [25], it is challenging to assign vessels to a particular catch quota allocation system in the PRD. In 2018, catch quota pilot projects were introduced in five provinces in China [57]. Guangdong, as one of pilot provinces, has selected cowrie shell for catch quota management, and one relatively mature area in the Pearl River will be chosen as a pilot site by 2020 [57], which is considered a practical action to promote the implementation of output control policy in the PRD. This study finds that the output control policy of a 10% total catch reduction has the greatest negative impact on the total output of the marine fish-harvesting sector (-6.366%, **Table 1**) but smallest negative impact (-0.384%, **Table 4**) on the total output of fishery-related sectors. This is because other seafood industries and recreational fishery can compensate for the loss of marine fishery production. Furthermore, as a consequence of the increase in the return of factor inputs, labour income and capital income increase

0.286% and 0.318%, respectively. The output control policy also improves overall social welfare, which indicates that the effect of declining commodity prices outweighs a diminution in household income. The biomass of commercially harvested species is most effectively maintained under this policy scenario as compared to 1995 levels, which is evidenced by the ecosystem indicators. Actually, many fisheries management cases [55, 58, 59] indicate that an output control policy should be implemented in conjunction with an input control policy to achieve EBFM targets [60], because output control has been less successful with non-selective fishing methods and multispecies fisheries [61].

In summary, the output control policy has the best effect on ecosystem restoration and can increase overall social welfare. The fishermen transfer policy has the most obvious economic benefit, while the policy of fishing vessels reduction has an effect in terms of ecosystem restoration, but it has a greater negative impact on the social and economic dimensions.

The results of all simulations are summarised in **Table 4**.

Table 4. Summary of simulation results in the economic, ecological, and social dimensions

Dimension	indicators		SCN1	SCN2	SCN3
Economic	Total output (%)	Fishery sectors	-0.727	-0.543	-0.384
		Non-fishery sectors	-0.327	0.072	-0.014
	Employment (%)	Fishery sectors	0.278	-0.873	0.286
		Non-fishery sectors	-0.008	0.004	-0.006
	Capital income (%)	Fishery sectors	-0.068	1.397	0.318
		Non-fishery sectors	-0.002	2.055	-0.002
Social	Utility change (%)	Fishery households	-0.476	-2.002	0.947
		Non-fishery households	-0.038	0.760	0.076
	Welfare change (million dollars)	Fishery households	-4.50	-19	9
		Non-fishery households	-86.82	1734	173
Ecological	Relative total biomass changes of MES (%)		18.6	25.7	36.8
	Maturity – MTLC		2.379	2.386	2.388
	Sustainability – FiB index		-0.005	-0.010	-0.003
	Biomass diversity –Kempton's Q index		4.461	4.548	4.650

Note: MES - 10 commercial exploited species, MTLC - mean trophic level of fishery catch, FiB - fishing-in-balance index.

5. Conclusions

In light of the increasing conflicts between the social economy and the ecological environment in coastal waters, it is vital to identify a comprehensive approach to assessing the effectiveness of fishery policy. This research provides a new perspective using integrated CGE and EwE models to demonstrate the effects of input and output control policies on the coastal ecosystem, the regional economy, and social welfare, thus overcoming the limitations of traditional fixed price models in evaluating the socio-economic impacts of fishery policies.

The four simulations in this study consider the original fishing intensity, the reduction in fishing vessels, the transfer of fisherman to other industries, and the reduction in the total catch of commercially harvested fish species. The modelling results show that the output control policy not only has the greatest positive impact on ecosystem recovery, it could also improve the welfare of fishermen and non-fishermen households. However, its negative impact on the output of the marine fish producing industry is the greatest of all four scenarios. Input control policies have a less direct impact on the ecosystem and on welfare than the output control policy. The fishermen transfer policy has a positive impact on non-fishery sector employment and total economic output. The fishing vessels reduction policy is beneficial to the restoration of the ecosystem but has a negative effect on the economy and overall social welfare. Therefore, the policy of output control is recommended for fisheries with a serious fishery resource depletion issue, and an input control policy that combines fishermen switching industries and fishing boat cutbacks is recommended for fisheries with less serious resource depletion. To realise sustainable utilisation of marine fishery resources, China's fisheries management policies should be developed to enhance total output control and ecosystem reconstruction.

In summary, the findings provide a decision-making framework for fisheries management departments to compare how to achieve expected management objectives through alternative management tools. The integration model provided in this study has three advantages over some other coupling models. (1) It considers non-linear interactions of economic agents. (2) It enables social welfare analysis. (3) It considers dynamic features of ecosystems. This study is conducted as an example application of an integrated ecological-economic system model. Future research can be focused on how to reflect the real world more closely by (1) setting the values of key elasticities in the CGE model and (2) using a coupling approach for two different types of models.

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Appendix I. The layout of the social accounting matrix (SAM) in the Pearl River Delta

Value ($ in million, $1=¥6.8	SEC1	SEC2	SEC3	SEC4	SEC5	SEC6	SEC7	SEC8	COM1	COM2	COM3	COM4	COM5	COM6	COM7	COM8	LAB	CAP	HH1	HH2	ENT	GOV	VATL	VATK	BUSTAX	TARIFF	INVSAV	ROW	TOTAL
SEC1									2175	0	0	0	0	0	0	0													2175
SEC2									0	16245	0	0	0	0	0	0													16245
SEC3									0	0	3512	0	0	0	0	0													3512
SEC4									0	0	0	2285	0	0	0	0													2285
SEC5									0	0	0	0	16480	0	0	0													16480
SEC6									0	0	0	0	0	53727	0	0													53727
SEC7									0	0	0	0	0	0	2621541	0													2621541
SEC8									0	0	0	0	0	0	0	989215													989215
COM1	0	0	145	0	0	0	1487	170											1	194		0					0	366	2364
COM2	0	0	1063	0	0	0	11134	1273											19	3355		0					0	820	17663
COM3	0	0	318	0	0	0	2515	1100											164	0		0					1	39	4136
COM4	37	267	0	64	8	118	1452	0											0	0		0					925	544	3416
COM5	58	423	0	0	562	825	1885	0											0	0		108					12598	38	16497
COM6	52	378	21	0	4142	2352	41630	9569											123	21933		0					-181	1893	81912
COM7	1089	7927	200	990	6627	5955	1761052	181967											2095	182114		304					435165	992641	3578125
COM8	97	706	234	118	1290	706	243040	251519											1377	245220		140438					39163	154786	1078696
LAB	827	6430	659	427	3783	42969	235048	284251																					574394
CAP	15	115	835	492	67	769	202120	226388																					430802
HH1																	656	33			3632	25							4346
HH2																	573739	65013			99612	35602							773966
ENT																	0	365756			0	79443							445199
GOV																			11	8180	21122	0	64785	55653	32978	53882	0	40304	276914
VATL	0	0	16	90	0	31	64648	0																					64785
VATK	0	0	20	103	0	1	55528	0																					55653
BUSTAX	0	0	0	0	0	0	0	32978																					32978
TARIFF	0	0	0	0	0	0	0	0	9	71	31	57	1	1409	47829	4474													53882
INVSAV																			556	312970	320833	20994						-167681	487672
ROW									180	1347	593	1075	16	26776	908755	85007													1023749
TOTAL	2175	16245	3512	2285	16480	53727	2621541	989215	2364	17663	4136	3416	16497	81912	3578125	1078696	574394	430802	4346	773966	445199	276914	64785	55653	32978	53882	487672	1023749	12712329

Note: SEC1-marine fish-producing, SEC2-other seafood industries, SEC3-aquatic product processing, SEC4-fishery manufacturing, SEC5-fishery servicing, SEC6-other agriculture, SEC7-other manufacturing, SEC8-other services, COM1-marine harvesting products, COM2-other fish-producing products, COM3-aquatic processing products, COM4-other fish-industry products, COM5-fish-servicing products, COM6-agricultural products, COM7-industrial products, COM8-services products, LAB-labour, CAP-capital, HH1-fishery household, HH2-non-fishery household, ENT-enterprise, GOV-government, VATL-labour value added tax, VATK-capital gains tax, BUSTAX-business tax, INVSAV-investing and saving, ROW-rest of the world.

Appendix II. Basic parameters of the Ecopath model in the Pearl River Delta (2008)

No.	Group name	Trophic level	B($t \cdot km^{-2}$)	P/B($year^{-1}$)	Q/B($year^{-1}$)	EE	P/Q
1	*Sousa chinensis*	3.56	0.002	0.112	14.000	0.053	0.008
2	Sharks	3.65	0.004	0.400	6.830	0.359	0.059
3	***Trachurus japonicus***	3.12	0.012	2.150	10.471	0.958	0.205
4	***Decapterus maruadsi***	3.11	0.018	1.870	11.079	0.900	0.169
5	***Trichiurus haumela***	3.35	0.022	1.210	6.210	0.944	0.195
6	***Saurida.***	3.43	0.012	1.420	7.993	0.989	0.178
7	***Psenopsis anomala***	2.10	0.031	3.620	31.010	0.988	0.117
8	***Upeneus bensasi***	3.31	0.011	1.013	11.352	0.915	0.089
9	***Nemipterus virgatus***	3.40	0.007	2.070	7.230	0.950	0.286
10	***Priacanthus macracanthus***	3.31	0.013	1.970	9.685	0.951	0.203
11	***Priacanthus tayenus***	3.30	0.007	3.720	12.420	0.923	0.300
12	Other pelagics	2.13	0.670	3.720	12.704	0.435	0.293
13	Other demersals	2.36	0.660	3.468	12.885	0.682	0.269
14	Other zoobenthos	2.13	2.030	6.553	26.212	0.415	0.250
15	Benthic crustaceans	2.58	0.260	6.522	26.088	0.455	0.250
16	Polychaetes	2.00	0.740	4.928	19.712	0.740	0.250
17	Molluscs	2.00	0.414	4.800	19.200	0.801	0.250
18	Echinoderms	2.00	0.125	8.629	34.516	0.742	0.250
19	**Cephalopods**	3.30	0.074	3.100	11.970	0.959	0.259
20	Jellyfish	2.52	1.350	5.000	25.000	0.055	0.200
21	Zooplankton	2.03	0.690	32.000	192.000	0.781	0.167
22	Phytoplankton	1.00	10.715	71.500	0.000	0.160	
23	Benthic producers	1.00	22.000	11.885	0.000	0.086	
24	Detritus	1.00	200.000			0.084	

Note: 1. The bold functional group is 10 commercial exploited species.

2. B—biomass, P/B—ratio of production to biomass, Q/B—ratio of consumption to biomass, P/Q—ratio of production to consumption, EE — ecotrophic efficiency

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