ALEX SOUZA LIRA

APLICAÇÃO DE MODELOS TRÓFICOS EM UM ESTUÁRIO TROPICAL: UM ESTUDO DE CASO EM PERNAMBUCO

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Alex Souza Lira

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Dedicatória

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“If you have a dream shall protect”

Alex Souza Lira

Resumo

A abordagem Ecopath with Ecosim foi usada para descrever a estrutura e as interação da rede trófica e avaliar o efeito da pesca na rede trófica de um estuário tropical no Nordeste do Brasil (Estuário do Rio Sirinhaém – ESR). Para a construção do modelo Ecopath foram amostradas espécies de peixes e macroinvertebrados, entre 2013-2014, enquanto dados de produtividade primária e zooplâncton foram obtidos através da literatura. Os resultados mostraram que a transferência de energia na rede trófica do ESR foi baseada na produtividade primária (60,5%), com uma média de transferência de 6,79%, relativamente baixa quando comparada com o valor teórico de 10%, indicando uma baixa eficiência global de transferência, embora dentro dos valores reportados para vários modelos tróficos de estuários. O índice de espécie chaves e a análise de impacto trófico indicaram que o Camurim (*Centropomus* spp.), com nível trófico = 3,20, é um componente chave no sistema, podendo desencadear em um efeito cascata (top-down). O aumento da atividade de pesca causa uma redução da biomassa do Camurim, e consequentemente diminui a pressão de predação nas suas presas. Os indicadores ecológicos para a saúde do ecossistema indicam que ESR é um sistema em desenvolvimento, como a maioria dos estuários, demandando estratégias de manejo para a manutenção do estado de equilíbrio. Este estudo fornece as primeiras informações para descrever e avaliar o estado da rede trófica em um ecossistema estuarino tropical no nordeste do Brasil.

Palavras-chave: EwE, Rede Trófica, Indicadores Ecológicos, Espécies Chave, *Centropomus* spp.

Abstract

The ecosystem approach Ecopath with Ecosim was used to describe the food web structure, and to evaluate the fishing effects in a tropical estuary ecosystem in northeastern Brazil (Estuary of the Sirinhaém River – SRE). To build the Ecopath model we sampled fish species and macroinvertebrates between 2013-2014, while data of primary production and zooplankton were obtained from literature. Results showed that the energy transfer in the SRE food web was based mostly on grazing food chain (60.5%), with an average transfer efficiency of 6.79%, relatively low compared to the theoretical value of 10%, indicating low global transfer efficiency, although it is within the range reported in various estuarine trophic models. The keystone index together with the MTI (Mixed Trophic Impact) index indicated the snook (*Centropomus* spp.), with Trophic Level = 3.20, as a key component in the system, which could trigger a cascade effect (top-down). The increasing in the fishery activity causes a reduction in the Snook biomass, and consequently decrease the pressure of predation on their prey. The ecological indicators of ecosystem health indicated that the SRE is a system in development, as most estuaries, requiring management strategies to the maintenance for the equilibrium state.This study provides the first information to describe and evaluate the state of food web in a tropical estuarine ecosystem in Northeast Brazil.

Keywords: EwE, Food web, Ecological indicators, Keystone species, *Centropomus* spp

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1. Introdução

A partilha de recursos (GUEDES e ARAÚJO, 2008), preferência de habitat (WETHERBEE e CORTÉS, 2004), competição (SVANBÄCK e BOLNICK, 2007) e avaliação do fluxo energético (MODICA et al., 2015) são apenas algumas das temáticas obtidas com os estudos de ecologia trófica, sendo a avaliação holística das interações tróficas entre os diferentes organismos nestes ambientes é fundamental para a compreensão da dinâmica, produtividade e manutenção da sustentabilidade dos ecossistemas (YOUNG et al., 2015). Todas estas informações acerca da ecologia trófica são também consideradas de grande valia no desenvolvimento de estratégias de conservação de espécies e ecossistemas (SIMPFENDORFER et al., 2011).

Modelos ecossistêmicos são uma realidade no auxílio de mecanismos de gestão, devido à sua capacidade de inclusão de inúmeras variáveis e processos em diferentes escalas, associados a resultados quantitativos fáceis de interpretar (ROMAGNONI et al., 2015), facilitando tomadas de decisão pela gestão dos ecossistemas (FULTON et al., 2011; DICHMONT et al., 2013). O Ecopath com Ecosim (EwE) é um dos softwares criados para elaboração de modelos ecossistêmicos. Foi desenvolvido inicialmente para estimar biomassa/produção/consumo dos componentes do ecossistema (ECOPATH) (POLOVINA, 1984) e posteriormente, outras ferramentas adicionais foram incorporadas ao Ecopath, dentre elas, o Ecosim; no qual é possível fazer previsões futuras com base em uma serie temporal (WALTERS et al., 1997); Ecospace, que permite avaliar espacialmente a região modelada (WALTERS et al., 1999) e mais recentemente, “Value-Chain” que faz uma teia econômica dos recursos utilizados (TRAVERS et al., 2007; FULTON, 2010; HALOUANI et al., 2016b).

Atualmente o repositório global de modelos “Ecobase” conta com aproximadamente 571 modelos publicados de 1984 a 2014. A maior parte dos modelos ao longo destes últimos 30 anos foi aplicada para análise do funcionamento dos ecossistemas, da pesca e, mais recentemente, para poluição, aquicultura e Áreas Marinhas Protegidas, principalmente em sistemas marinhos tropicais no Oceano Atlântico norte e central (COLLETER et al., 2015).

Numa escala global, inúmeros estudos de modelagem trófica vêm sendo desenvolvidos (VILLANUEVA et al., 2006; COLL et al., 2008; TSAGARAKIS et al., 2010; ANGELINI e VAZ-VELHO, 2011; XU et al., 2011; DÍAZ-URIBE et al., 2012; LEGAGNEUX et al., 2012; GUO et al., 2013; DE MUTSERT et al., 2015; FRETZER, 2015; DUTTA et al., 2016; EDDY et al., 2016; HAAK et al., 2017). O declínio da biomassa pesqueira oceânica em todo o mundo foi analisado por Christensen et al. (2014), enquanto alguns modelos avaliaram a estrutura e o funcionamento dos ecossistemas (VILLANUEVA, 2015; TECCHIO et al., 2016) a partir da descrição e variação do fluxo trófico (NEIRA et al., 2014; SALCIDO-GUEVARA e ARREGUÍN-SÁNCHEZ, 2014; DE MUTSERT et al., 2015). Modelos tróficos também são utilizados para o desenvolvimento da aquicultura (KLUGER et al., 2015; IZQUIERDO-GOMEZ et al., 2016). A capacidade de suporte para a pesca também é objeto de muitos estudos com modelagem trófica (TOMCZAK et al., 2012; LONGO et al., 2015; GEERS et al., 2016). Finalmente, estes enfoques são úteis para a formulação de medidas de gestão, restauração e avaliação de impactos sobre os recursos e ecossistemas (PAULY et al., 2000; FORREST et al., 2015; ROSE et al., 2015; WOODWORTH-JEFCOATS et al., 2015; HALOUANI et al., 2016a).

No Brasil, o uso de modelos tróficos para descrição de ecossistemas aquáticos, apesar de ter apresentado um relativo aumento na última década, ainda é muito discreto e restrito principalmente a regiões lacustres, de rios e reservatórios (ANGELINI e PETRERE M., 1996; ANGELINI e AGOSTINHO, 2005; ANGELINI et al., 2010; GUBIANI et al., 2011; ROSA et al., 2014). O principal entrave para o desenvolvimento destes estudos é a carência de séries históricas e conhecimentos básicos sobre a ecologia e biologia dos ecossistemas e recursos vivos brasileiros (ANGELINI e GOMES, 2008). No entanto, apesar destas limitações, os ambientes marinhos e estuarinos também vêm sendo examinados, embora em menor proporção. Nascimento et al. (2011) avaliaram a estrutura trófica de comunidades demersais no sudeste do Brasil e em zonas de ressurgência (VASCONCELLOS e GASALLA, 2001; GASALLA e ROSSI-WONGTSCHOWSKI, 2004; VERA, 2010). Para o norte e nordeste do Brasil, os estudos em ambientes marinhos e estuarinos são ainda mais restritos. Freire et al. (2007, 2008) descreveram a região costeira do nordeste e avaliaram das políticas de pesca através de modelo trófico, Freire e Pauly (2010) avaliaram a diminuição do nível trófico das capturas no nordeste do Brasil, Wolff et al. (2000) geraram um modelo de fluxo trófico para o estuário do Rio Caeté no Pará e, mais recentemente, Xavier (2013) produziu um modelo com avaliação de ecossistema estuarino e recifal em Mamanguape (Paraíba). Não há nenhum modelo ecossistêmico de fluxo trófico publicado para a região costeira do estado de Pernambuco.

Com aproximadamente 187 km de extensão, o estado de Pernambuco engloba estuários com elevada biodiversidade, como na região norte do estado (Complexo estuarino do Canal de Santa cruz e o estuário do Goiana; VASCONCELOS FILHO e OLIVEIRA, 1999; RAMOS et al., 2011; LACERDA et al., 2014) e na região sul (os estuários de Suape e Rio Formoso; PAIVA e ARAÚJO, 2010; PEREIRA et al., 2010; BEZERRA et al., 2011). Estes estuários também têm grande importância sócio-econômica dada à relevância da pesca como fonte de renda e proteína (CPRH, 2001; BARBOSA et al., 2007; PINTO et al., 2015). Na Barra de Sirinhaém, no litoral sul de Pernambuco, 60% da população apresenta ligação direta ou indireta com atividade pesqueira (LIRA et al., 2010), sendo responsável pela maior frota camaroeira artesanal de arrasto entre os municípios costeiros do estado de Pernambuco (TISCHER e SANTOS, 2003). Além disso, apresenta um estuário com uma fauna diversificada (SILVA-JÚNIOR et al., 2016), destacando, entre outras espécies, o Camurim (*Centropomus* spp.), Xaréu (*Caranx* spp.), Bagres (*Sciades* spp. e *Cathorops* spp.), Marisco (*Anomalocardia brasiliana*), Tainhas (*Mugil* spp.), Carapebas e Carapicus (*Diapterus* spp. e *Eucinostomus* spp.).

A carência de informações do ponto de vista ecossistêmico das regiões estuarinas e marinhas do litoral de Pernambuco torna a elaboração de modelos matemáticos ecossistêmicos de extrema relevância do ponto de vista do manejo e ordenamento destes ambientes. A identificação das espécies-chaves, seu papel funcional no ecossistema, assim como a avaliação de cenários futuros, são apenas algumas das informações geradas por estes modelos, podendo contribuir para o uso sustentável destes biomas no Estado.

2. Objetivos

2.1 Geral

Avaliar a estrutura e dinâmica trófica do ambiente estuarino de Barra de Sirinhaém, litoral sul de Pernambuco.

2.2 Específicos

1- Descrever o fluxo de energia na teia trófica estuarina de Barra de Sirinhaém;

2- Identificar as principais fontes de energia para a ictiofauna através do Ecopath;

3- Identificar as espécies-chave e avaliar seu efeito de controle na rede trófica;

4- Descrever a estrutura e as interações na rede trófica;

5- Avaliar os efeitos da pesca na rede trófica;

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2. Artigo cientifico

Artigo científico a ser encaminhado a Revista
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4.1 Trophic structure in a neotropical estuary: the case study of Sirinhaém, Northeastern Brazil

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Introduction

Rivers, estuaries and coastal zones, intrinsically linked through transfers of water, nutrients and biota (O'BRIEN et al., 2016) are considered essential habitats for feeding, reproduction and growth of many aquatics organisms (ODUM and BARRET, 2007; CLOERN et al., 2014). Estuaries have a relatively low biodiversity (TECCHIO et al., 2016) and are considered as valuable areas of the world (COSTANZA et al., 2014) given that they provide many ecosystem services (BARBIER et al., 2011; BOEREMA and MEIRE, 2016). However, intensive anthropogenic activity, mainly the area on the river basin and nearby the estuaries, often affect water quality and aquatic biodiversity (VIANA et al., 2012) and consequently, changes the internal functioning of ecosystems (HOOPER et al., 2005).

The limitations of single-species approaches front the new global challenges arising from climate changes and ecosystem degradation (KALTENBERG and BENOIT-BIRD, 2016). In this context, ecosystem models, which take into account the interactions between the different components of the ecosystem, have been increasingly developed and applied worldwide to support the Ecosystem Approach to Fisheries (EAF) (GARCIA et al., 2003; CORRALES et al., 2015). Those models are a reality as a tool for ecosystem management, due to the capacity to include a wide range of processes across a wide range of scales, associated with quantitative and easy-to-interpret results (ROMAGNONI et al., 2015).

Amongst the family of models which takes into account the Ecosystem Approach to Fisheries (EAF), trophic models (PIKITCH et al., 2004) have arisen as promising to the decision makers in terms of management advice. Trophic models allow the evaluation of the energetic flux between the species, also describing the functional role of species and maturity of the ecosystems (CHRISTENSEN and PAULY, 1993). This family of models may also simulate scenarios like overfishing (CÁCERES et al., 2016; WANG et al., 2016), and also has been used to evaluate the mechanisms which regulate the growth and development of aquatic food webs (e.g. resource limitation, bottom-up and top-down control in food web) (ANGELINI et al., 2010; RUIZ et al., 2016). These models synthesize the comprehension of the systems, providing knowledge about the connectivity between the components, defining problems and making forecasts.

The Ecopath with Ecosim (EwE) is an ecosystem model based on the principle of energy balance and trophic flows in aquatic ecosystems. It is based on the information of biomass, diet, production and consumption rates of the main species of the ecosystems (PLAGÁNYI, 2007; HEYMANS et al., 2016). EwE is has been applied since 1984 and more than 433 studies have been published mainly focusing on tropical marine systems aiming at describing the trophic structure and fisheries-related issues (COLLETER et al., 2015). Although the number of EwE models has increased in the last two decades in Brazil, it is mainly restricted to lakes, reservoirs, rivers (GUBIANI et al., 2011; ANGELINI et al., 2013; ROSA et al., 2014) and marine systems (FREIRE et al., 2008; NASCIMENTO et al., 2011) in the south portion of the country. However, for estuarine ecosystems, only two studies are reported in Brazil, one of them (grey literature) in the northeastern part of the country (Wolff et al., 2000; Xavier 2013).

In Northeastern Brazil there are many estuaries (PAIVA and ARAÚJO, 2010), however, urbanization, mangrove degradation and overfishing have hampered these environments (Lessa et al., 2009). Elfes et al. (2014) applied the Ocean Health Index framework (HALPERN et al., 2012) to the Brazilian coastal ecosystems, and found lowest scores in this Brazilian region, given the high impact by sugar cane and others agribusiness industries (MELLO, 2009) which degrade rivers and consequently estuarine areas, impairing fisheries activities which are fundamental for local population (TISCHER and SANTOS, 2003).

This study aims to develop a trophic model, using the Ecopath approach in the Sirinhaém estuary, located within Protected Areas (APA): APA of Guadalupe and APA of Sirinhaém, both with no management plan. The main objectives of this study are: i) to describe the food web structure and trophic interactions; ii) to identify key species and evaluate the top-down or bottom-up effect; and iii) to evaluate the fishing effects on food web. Results can provide critical insights to evaluate the effective decision-making and policy actions to multispecies management and ecosystem conservation.

Materials and methods

Study area

The Sirinhaém River Estuary (SRE) is located in the southern coast of Pernambuco State (Northeast Brazil), 08°35'S and 035°02' W. It is characterized by a high density of mangrove (MAIA et al., 2006) and is located between the Marine Protected Area of Guadalupe and the Marine Protected Area of Sirinhaém (Fig. 1). The SRE, classified as a coastal plain estuary, have 9.5 km long, 350 m wide (increasing up to 800 m in the river mouth) with depths varying between 1.2 and 4.5 m (SILVA et al., 2011). The local climate is tropical, with rainfall of 20 – 450 mm. yr^{-1} (rainy season is between May and October, APAC, 2015), mean temperature of 29°C, pH range is between 5.8 and 8.5 and salinity between 0 and 36 (SILVA, 2009).

Ecopath with Ecosim model

The Ecopath model used in this study is based on 2 master equations: (1) used for definition of production and (2) for definition of consumption (CHRISTENSEN et al., 2008).

$$P_i = (B_i \times M2_i) + (P_i \times (1 - EEi)) + E_i + BA_i + Y_i \qquad (1)$$

$$Q_i = P_i + R_i + U_i \qquad (2)$$

Where P_i is the total production for each group *i;* B_i the biomass of each group *i;* $M2_i$ is the instantaneous predation rate for group *i*; EE_i is the ecotrophic efficiency (the fraction of production consumed, fished or exported out of the system) for each group *i*; E_i is the net migration rate (emigration–immigration); BA_i is the biomass accumulation rate for each group *i*; Y_i the total fishery catch rate of *i;* Q_i, P_i , R_i and U_i is respectively consumption, production, respiration and unassimilated food for each group *i*.

The overall equation (3) of the Ecopath model considers the biomass, production and consumption rate and the diet of the each group analyzed in the model (CHRISTENSEN and WALTERS, 2004):

$$B_i \times \left(\frac{P}{B}\right)_i = \sum_{j=1}^{n} B_j \times \left(\frac{Q}{B}\right)_j \times DC_{ji} + Y_i + E_i + BA_i + B_i \times \left(\frac{P}{B}\right)_i \times (1 - EE_i) \qquad (3)$$

P/B_i is the production rate of *i*, Q/B_j is the consumption rate of predator *j*, B_j is the biomass of the predating group *j* and DC_{ji} is the fraction of prey *i* in the average diet of predator j.

In the absence of information, BA*i* and E*i* were assumed as equal the 0 (COLL et al., 2006a).

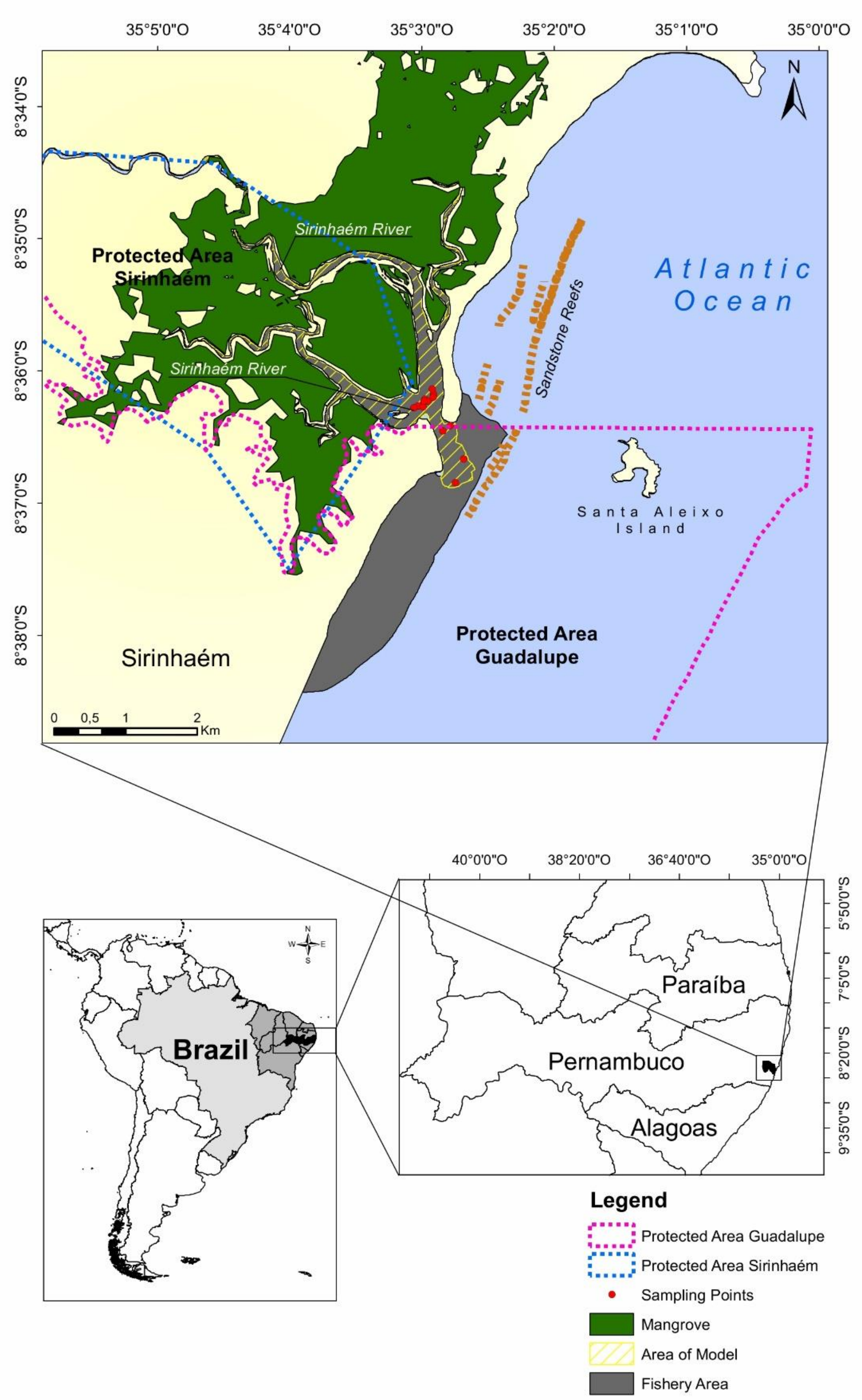

Figure 1 Estuary of Sirinhaém River, Northeastern Brazil, and the area of model with location of the sampling points carried out between 2013 and 2014

The Ecosim is a model of time-dynamic simulation that allows the evaluation of changes in food web to ecosystem level, based into initial conditions and parameter definition from the Ecopath model (CHRISTENSEN and WALTERS, 2004). The model uses a system of ordinary differential equations that derive from Eq.(3) (WALTERS et al., 1997, 2000).

$$\mathrm{dB}_i/\mathrm{dt} = \mathrm{g}_i \times \sum_j \mathrm{Q}_{ji} - \sum_j \mathrm{Q}_{ij} + \mathrm{I}_i - (M_i + F_i + e_i) \times Bi \qquad (4)$$

Where *dBi/dt* is the change in B of group *i* over time *t*, g_i is the net growth efficiency, Q_{ji} is the consumption rate of group *i* on group *j*, and Q*ij* is the consumption rate of group *j* on group *i*. I*i* is the immigration rate, M*i* is the non-predation natural mortality rate, F*i* is the fishing mortality rate, and e_i is the emigration rate. The consumption rates between prey *i* consumed by predator *j* are calculated based on the “foraging arena” theory, in which the prey biomass (B*i*) available to a predator is divided into vulnerable and non-vulnerable components, where the transfer rate between these two components is the vulnerability rate (v_{ij}), which determines the type of foodweb control (top- down, bottom-up or mixed). We used the Ecopath and Ecosim software (version 6.5.) in this study.

Model Components

Fish

For the model, fishes were captured quarterly, between March of 2013 and May of 2014 with a 250 m long, beach seine 2.5 m high and mesh size of 25 mm. Three sets were carried out for each sample. The swept area was calculated by tracking of trawl with GPS using the software *Image J.* Fish were stored in cool boxes with ice and then frozen for future analysis. The fish biomass was estimated through the sum of the individual weights of each group divided by the total dragged area (0.0787 Km^2), expressed in $t.km^2$. In order to minimize problems related to underestimation of fish assemblages due to gear selectivity, a catchability model (LAURETTA et al., 2013) was applied for the fish compartments (eq. 5 and 6).

$$p = \mathrm{q}L \times E \times A^{-1} \qquad (5)$$

$$\mathrm{N} = \mathrm{C} \times p^{-1} \qquad (6)$$

Where *p* is mean proportion of the population captured, qL is catchability coefficient, *E* is effort (total area dragged), *A* is habitat area (1.71 km^2), N is population abundance and C is the catch. The catchability coefficient (q*L*) was adapted for our fish community considering the same genus, body and/or fin format (see supplementary material - SOM 1).

The production (P/B) was obtained with two methods: when possible, we used linearized length converted catch curve to estimate the total mortality Z (CHAPMAN and ROBSON, 1960; PAULY, 1983) as P/B (ALLEN, 1971). For species where there is no fishery mortality (F) data, we used the natural mortality (PAULY, 1980) as P/B (eq. 7).

$$PB = M = k^{0.65} \times L_{\infty}^{-0.279} \times T^{0.463} \qquad (7)$$

Where M is natural mortality (year $^{-1}$), k is the growth coefficient (year $^{-1}$), L_{∞} is the asymptotic length (cm) and T is the mean water temperature (°C). The parameters k and L_{∞} were obtained from the literature or using the empirical equations of Le Quesne and Jennings (2012) and Froese and Binohlan (2000) respectively. T was obtained *in situ* and considered as the mean annual temperature, 27.8°C (See SOM 2).

The consumption (Q/B) (eq. 8) per biomass was obtained according to the following equation (PALOMARES and PAULY, 1998).

$$\text{Log } QB = 7.964 – 0.204 \times \log W\infty - 1.965 \times T' + 0.083 \times Ar + 0.532 \times H + 0.398 \times D \qquad (8)$$

Where W_{∞} is asymptotic weight (g), L_{max} is maximum length of the species, T\` is temperature in Kelvin, and Ar is aspect ratio of the caudal fin. W_{∞} was estimated using the equation $W_{\infty}=a \cdot L_{\infty}^{b}$, where *a* and *b* were based on Viana et al. (2016) and L_{∞} was obtained as described above. L_{max} was obtained from the literature. Photographic records of the caudal fin were taken for each species with *image J* software. Ar was hence calculated as $Ar = h^2/s$, where (h) is height of caudal fin and (s) is the surface area of the fin, extending to the narrowest part of the caudal peduncle (PALOMARES and PAULY, 1998), and obtained with software *image J*, through the photography of caudal fins. H and D represent the feeding type (h: 1 and d: 0 for herbivores; h: 0 and d: 1 for detritivores; h: 0 and d: 0 for carnivores). See SOM 4 for the parameters used to calculate the consumption (Q/B).

Diet composition

Diet information for the fish species of each compartment was primarily estimated from complementary studies of trophic guilds, obtained from stomach contents analyses of the study area (see SOM 6 for sources). When not available and for the other groups of the model, information were based on the literature. Table 3 shows the diet matrix used in the model.

Other compartments

For macrobenthos, we collected the sediment manually in three points of the estuary: channel, margin and river mouth. The biomass (Fiddler crabs, Polychaetes, Bivalves, Gastropods) was estimated using the relationship between the sample volume and the depth of the sediment where most of the benthic organisms are mainly distributed (following Xavier, 2013). In the present study, the depth of 10 cm is considered as the most probable area of distribution of the benthic fauna were the majority part of the interactions between the epifauna and the water (ROSENBERG, 2001). P/B of macrobenthos (Fiddler crabs, Polychaetes, Gastropods) was estimated based on the equation of Brey (1999), considering the maximum age for each group obtained from the literature and maximum body mass based on our data base. The Q/B was based on the equation proposed by Nichols (1974). All data of P/B and Q/B used in the model were obtained from literature. See SOM 5 for parameters, equations and references used.

The Phytoplankton and epiphyton biomass were obtained from the literature (Baltar et al., 1996; Silva, 2009), based on studies developed nearby the area. Zooplankton, microphytobenthos and the other groups of the macrobenthos (*Litopenaeus* spp and *Callinectes* spp.) were not sampled; therefore, the biomass was estimated from the model. The Detritus of the ecosystem was given following the equation proposed by Pauly et al. (1993).

For Phytoplankton, Epiphyton and Microphytobenthos, P/B and EE were obtained from the literature and the model, respectively. The P/B, Q/B and EE values of Zooplankton, Bivalves, Blue crabs and Shrimp were obtained from the literature.

Fishery landings

For this model, we considered the landings of the main fishery resources caught by the artisanal fleet landed in Sirinhaem (coast and estuary), from the years of 2000 and 2007 (IBAMA/CEPENE 2007). Although there is fishery statistics available for a longer time series, there are different methodologies of data collection. Landings from this period was standardized in terms of methods and data was considered as more accurate. Since the official landings are not divided by area (e.g. estuary and coast), we assumed different percentages of the total landing for the estuarine area, for each compartment explored by the fishery activity. We obtained this information based on literature and, logbooks and database available in the Laboratory (see SOM 6). The input landings data in the model was obtained by dividing the average annual catch of the estuarine area (t) by the fishing area (5.26 km^2).

Balancing of model and Network analysis

The Pedigree index was calculated to quantify the uncertainty related to each input value (B, PB, QB and diet matrix) in the model (CHRISTENSEN et al., 2005), ranging from 0 (low precision information) to 1 (data and parameters fully rooted in local data). For the evaluation of the equilibrium of the model, following Christensen and Walters (2004) and Christensen et al. (2008), when inconsistencies were detected, the initial input parameters were adjusted by a step-by-step approach for each group. Ecotrophic efficiency (EE) (which should be ≤ 1) is the first step taken into account for model diagnostic. The production to consumption ratio (P/Q) is supposed to be between 0.05 (lower P/Q values) and 0.3 (have higher P/Q values), with exception of the top predators (GUENETTE, 2014). For all groups, the respiration/assimilation (R/A) must be < 1 (DARWALL et al., 2010). In order to balance the groups, which presented EE >1, we reallocated the consumption to other preys that were part of the diet of the predators, as already applied by Albouy et al. (2010).

The transfer efficiency (TE) is the ratio between the sum of exports and flows predated by the next level and the throughput on the trophic level. Trophic interactions, estimated flows of detritus and the TE to upper trophic levels were described by diagram flow and “Lindeman spine” graphic (LINDEMAN, 1942), that synthetize all groups into a simple linear food web. The mixed trophic impact routine-MTI (ULANOWICZ and PUCCIA, 1990) and keystonness index (KS3) developed by Valls et al. (2015) were used, respectively, to analyze the response of both direct and indirect impacts of one group on another, and to identify keystone and dominant groups (groups that play an important role in the food web with relatively low biomass and a large relative impact, respectively). The top-down effect (td), as percentage contributions to the keystoneness for each species, was evaluated through the proportion of the negative values that contributed to the sum of overall effect of group *i* on all the other groups in the food web (LIBRALATO et al., 2006).

Also, in order to describe the ecological status (ODUM, 1969) and functioning of the Sirinhaém estuary, we obtained the Total System Throughput (TST), which is the sum of all flows of the system (consumption, exports, respiration, and flows to detritus). The total primary production/total respiration (TPP/TR) and the System Omnivory Index (SOI) were also obtained. TPP/TR describes the maturity of the ecosystem (ODUM, 1971) and values close to 1 are related to more mature systems. SOI is based on the variance of the TL of prey consumed by predators, ranging between 0 (full specialist consumers) and 1 (dominance of groups with high food plasticity, CHRISTENSEN et al., 2008). Connectance index (CI), that is a relation of the observed number of food links in the system relative to the number of possible links (GARDNER and ASHBY, 1970), was also obtained. Higher mature systems tends to have a greater connectivity than the less mature (CHRISTENSEN and WALTERS, 2004; CHRISTENSEN et al., 2005). Ascendency (A) and overhead (O) (ULANOWICZ, 1986), which correspond to a measure of system maturity and a measure of ecosystem stability, respectively (CHRISTENSEN, 1995), was estimated. Finn's cycling index (FCI) represent the proportion of recycled flows in the system (FINN, 1976), and can be used as a measure of ecosystem resilience (VASCONCELLOS et al., 1997). For more detailed description of these ecological indices, see Heymans et al. (2014).

Landing Analysis

One of the criteria that can be used to compare the ecological footprint of fishing activities is the Primary Production Required (PPR) to sustain fisheries (COLL et al., 2006b). This metric was mainly used to quantify the pressure of fishing on the ecosystems by calculating the PPR to sustain catches (PAULY and CHRISTENSEN, 1995; FABIO et al., 2016). Another EwE output expressed as function of the PPR is the Loss in secondary production index (L index) (Libralato et al., 2008), defined as (9):

$$L = -\frac{PPRi \times TE^{TLc-1}}{P_I \times \ln(TE)} \qquad (9)$$

Where P_1 = indicates the autotrophic production and detritus production by the food web from Calculated Net Primary Production (PP) + Flows to Detritus (FLDET), TE is the average efficiency of transfer between the TLs, PPRi and TLi are the Primary Production Required and TL for/of component i respectively. This method also allows the estimate of the probability of sustainability of the fishing (psust) (LIBRALATO et al., 2008). The reference values of the Lindex (L ± sd) for sustainable fishery, with confidence level of 75% and 95%, is $L_{75\%} = 0.021 \pm 0.013$ and $L_{95\%} = 0.007 \pm 0.007$ (COLL et al., 2008).

The indices that reflect the biodiversity are also important parameters for evaluating the effect of fisheries in the ecosystem. Kempton index (Q) expresses biomass species diversity of functional groups with TL≥3 (KEMPTON and TAYLOR, 1976), considered as an indicator of biodiversity evenness (AINSWORTH and PITCHER, 2006). Kempton index (Q) usually increases with the increasing biomass of high trophic level species and decreases with increase of the impacts under these species (e.g. fishing) (GOLDSWORTHY et al., 2013). Lower values of Q imply in a low evenness and richness and higher group dominance (ANGELINI et al., 2013).

To assess if the ecosystem can support fishing pressure, we created a set of fishing simulations for the SRE to assess the ecosystem changes induced by fishing and to verify the alterations of the Kempton and L indexes. It was performed 12 simulations, which represented 20-year of scenarios, increasing and decreasing fishing mortality for all species targeted by the fishery. The simulations were performed maintaining the initial Ecopath fishing mortality (F0) for the first 3 years of the simulation, then increasing or decreasing the fishing mortality in year 4 from F0 to F = F0× f (where f is a multiplier factor), maintaining the last value of F for the last 16 years of the simulation (Fig. 2). Thus, the simulations explored the effects of fishing mortality (F) ranging from 0 (no fishing) to 0.95 (SOM 7).

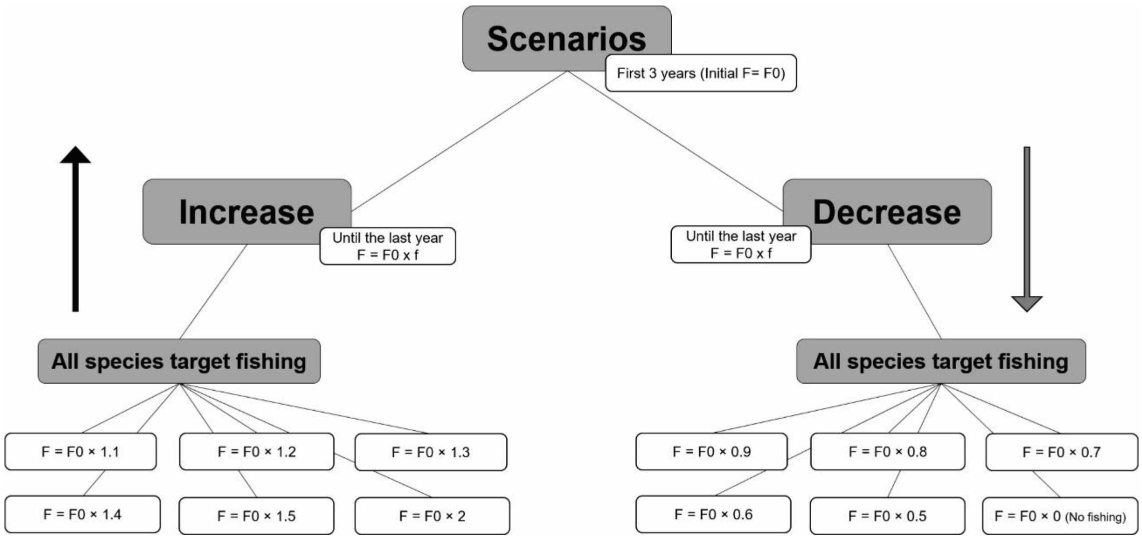

Figure 2. Scheme of the 12 simulations, which represent 20-year of scenarios with increasing and decreasing fishing mortality for all fishing target species in the Estuary of Sirinhaém River, Northeastern Brazil.

We used the Ecosim routine to simulate the scenarios. Usually the Ecosim vulnerability parameters are determined through the best fit considering the observed time series (e.g. Halouani et al., 2016; Wang et al., 2012). However, no time-series data were available in the present study to perform an authentic calibration; thus, we used the Ecosim default vulnerability parameters (Christensen and Walters, 2004; v = 2, which indicate a mixed food web control), and only the fishing mortality was changed given the different simulations, as proposed by Angelini et al. (2013). Therefore, the evaluation fishing pressure on food web was obtained from temporal changes in outputs of the Kempton Q indexes, L index, psust, mean trophic level of the cacth (mTLc) and biomass of the groups Fish and Invertebrates for the 20-years simulations.

Results

Basic estimation

Sirinhaém model included 26 trophic groups: three primary producers, zooplankton, six groups of macrobenthos, 15 groups of fishes and one detritus (Table 1). Fish groups were selected given the importance in abundance (% in weight of the samplings) (Fig. 3), landings relevance, position in water (surface or bottom) and trophic guilds (ELLIOTT et al., 2007; MOURÃO et al., 2014). Some fish components represent more than one species (Table.1).

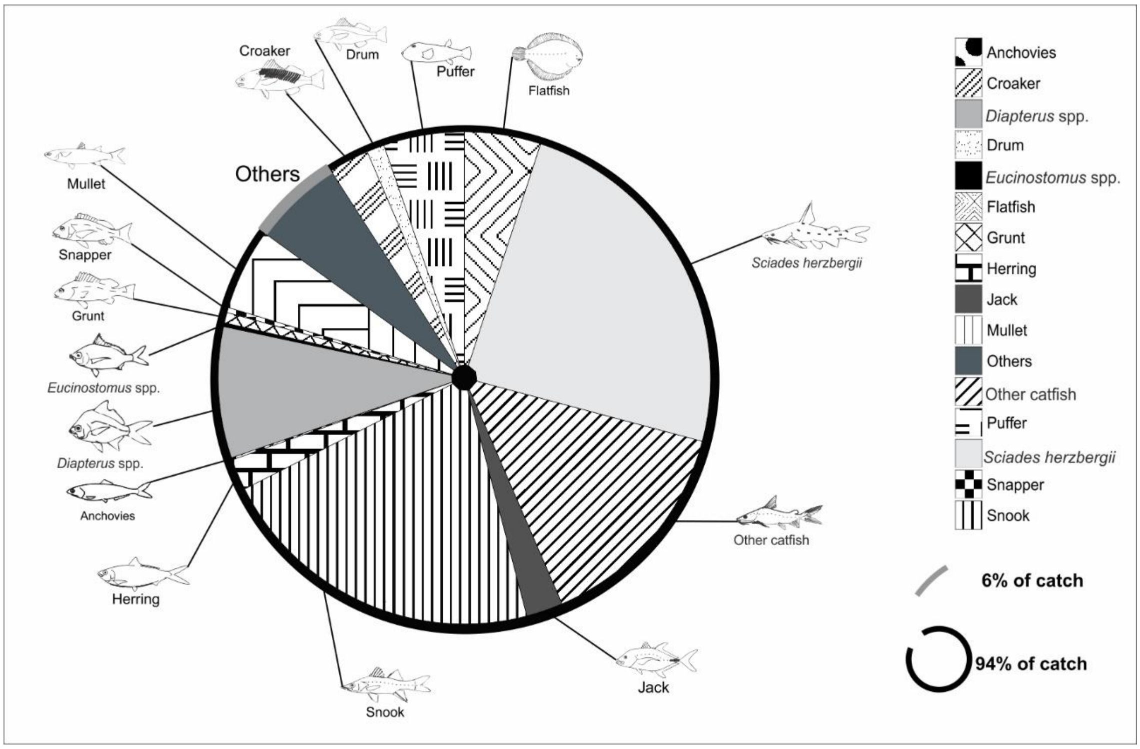

Figure 3 Fish composition, expressed in % in weight; captured with manual beach seine in the Estuary of Sirinhaém River, Northeastern Brazil.

Table 1 Taxonomic composition and trophic guilds of each compartment of the model in the Estuary of Sirinhaém River, Northeastern Brazil

	Group name	Family	Scietific name	Guilds
1	Phythoplankton	-	-	Productor
2	Zooplakton	-	-	Primary consumer
3	Epiphyton	-	-	Productor
4	Microphytobenthos	-	-	Productor
5	Fiddler crabs	Ocypodidae	*Uca* spp.	Deposit-feeders
6	Polychaetes	-	-	Deposit-feeders
7	Bivalves	Veneridae	*Anomalocardia brasilliana*	Filter-feeding
8	Gastropods	Neritidae	*Neritina virginia*	Scrapers
9	Blue crab	Portunidae	*Callinectes* spp.	Zoobentivorous
10	Shrimp	Peneidae	*Farfantepenaeus* spp.	Detritivore
11	Herring	Clupeidae	*Opisthonema oglinum*	Zooplanktivore
12	Anchovies	Engraulidae	*Cetengraulis edentulus* *Anchoa spinifer*	Zooplanktivore
13	Mullet	Mugilidae	*Mugil* spp	Omnivore/Detritivore
14	Flatfish	Achiridae	*Achirus lineatus* *Trinectes paulistanus*	Zoobenthivore
15	Puffer	Tetraodontidae	*Sphoeroides testudineus*	Opportunist/Omnivore
16	*Eucinostomus spp*	Gerreidae	*Eucinostomus argentus* *Eucinostomus gula*	Zoobenthivore
17	*Diapterus spp*	Gerreidae	*Diapterus auratus* *Diapterus rhombeus*	Opportunist/Omnivore
18	*Snapper*	Lutjanidae	*Lutjanus jocu*	Piscivorous/Zoobenthivore
19	*Sciades herzbergii*	Ariidae	*Sciades herzbergii*	Zoobenthivore/Carcinophago
20	Other Catfish	Ariidae	*Aspistor luniscutis* *Aspistor quadriscutis*	Omnivore
21	Drum	Sciaenidae	*Bairdiella ronchus* *Menticirrhus americanus*	Zoobenthivore
22	Grunt	Haemulidae	*Conodon nobilis* *Pomadasys crocro*	Piscivorous/Zoobenthivore
23	Croaker	Sciaenidae	*Micropogonias furnieri*	Opportunist/Omnivore
24	Snook	Centropomidae	*Centropomus undecimalis* *Centropomus parallelus*	Piscivorous/Zoobenthivore
25	Jack	Carangidae	*Caranx hippos* *Caranx lattus*	Piscivorous

The values of B, PB, QB and capture for all groups (Table 2) showed that only one component, Bivalves, represent more than a half of total biomass while total fish biomass represents 10% of total biomass, with fishing activities catching around 20% of this amount. High EE values were reported for most groups (e.g Polychaetes, Herring, Anchovies, *Eucinostomus* spp., Grunt, Croaker), mainly due to the high biomass of their main predators (e.g *Sciades herbergii* - B: 4.57 $t.km^{-2}$; Snook - B: 2.57 $t.km^{-2}$; *Diapterus spp* - B: 1.61 $t.km^{-2}$), however the EE of Snapper (EE: 0.06), Jack (EE: 0.15) and Bivalves (EE: 0.13) were considerably lower than other groups, since they are not heavily predated within this ecosystem (Table 2). The EE values of the groups targeted by fishing activities ranged between 0.3 – 0.35. Table 3 shows the diet matrix used in the model. The pedigree index and the measure of fit for SRE model were 0.34 and 1.754, respectively.

Table 2 Basics inputs and estimated outputs (bold) for the trophic groups for the model of the Estuary of Sirinhaém River, Northeast Brazil. TL: trophic level; B: biomass; P/B: production–biomass ratio; Q/B: consumption–biomass ratio; EE: ecotrophic efficiency; P/Q: production–consumption ratio; Y: landings; OI: Omnivory Index; PN: Path Number. Values in bold are estimated from the model.

	Group name	TL	B ($t.km^{-2}$)	P/B (per year)	Q/B (per year)	EE	P/Q	OI	PN	Y
1	Phytoplankton	1.00	2.22	652.71	-	**0.58**	**-**	-	-	-
2	Zooplankton	2.00	**1.01**	50.21	150.65	0.90	**0.33**	-	1	-
3	Epiphyton	1.00	1.37	153.31	-	**0.69**	**-**	-	-	-
4	Microphytobenthos	1.00	**4.43**	209.61	-	0.50	-	-	-	-
5	Fiddler crabs	2.03	16.00	7.30	31.89	**0.51**	**0.23**	0.0338	9	-
6	Polychaetes	2.16	2.87	2.91	17.26	**0.88**	**0.17**	0.1366	7	-
7	Bivalves	2.00	90.00	2.00	9.00	**0.14**	**0.22**	-	2	-
8	Gastropods	2.01	17.60	2.65	38.83	**0.42**	**0.07**	0.0095	3	-
9	Blue crab	2.24	**3.74**	2.00	8.00	0.76	**0.25**	0.1869	34	-
10	Shrimp	2.30	**4.29**	2.81	26.90	0.95	**0.10**	0.2112	25	-
11	Herring	2.50	0.32	1.97	18.34	**0.83**	**0.11**	0.25	2	-
12	Anchovies	2.00	0.02	2.23	90.39	**0.80**	**0.02**	-	1	-
13	Mullet	2.02	**2.03**	1.88	43.15	0.35	**0.04**	0.0151	5	0.971435
14	Flatfish	2.97	0.91	3.16	13.43	**0.32**	**0.24**	0.2439	81	-
15	Puffer	2.72	0.87	3.17	11.06	**0.53**	**0.29**	0.2618	50	-
16	*Eucinostomus* spp.	2.89	0.03	1.33	12.84	**0.81**	**0.10**	0.2114	57	-
17	*Diapterus* spp.	2.72	1.61	2.90	10.61	**0.22**	**0.27**	0.2609	39	0.2962
18	Snappers	3.16	0.07	0.33	6.43	**0.06**	**0.05**	0.0465	391	-
19	*Sciades herzbergii*	2.85	4.57	1.38	9.94	**0.24**	**0.14**	0.1568	74	0.89676
20	Other Catfish	2.90	1.80	1.13	12.50	**0.41**	**0.09**	0.2076	124	0.16835
21	Drum	3.15	0.15	1.73	9.44	**0.47**	**0.18**	0.041	80	-
22	Grunt	3.25	0.12	0.93	8.79	**0.87**	**0.11**	0.073	311	-
23	Croaker	2.21	0.43	0.21	6.90	**0.97**	**0.03**	0.1972	32	-
24	Snook	3.20	2.57	1.68	5.70	**0.18**	**0.29**	0.2492	2054	0.78123
25	Jack	3.28	0.23	0.47	6.66	**0.15**	**0.07**	0.1862	1123	-
26	Detritus	1.00	2.76	-	-	**0.49**	**-**	-	-	-

Table 3 Diet composition matrix for the functional groups in the Estuary of Sirinhaém River, Northeast Brazil. Were: the numbers represented the groups of the model.

Prey		Predactors																					
		2	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25
1	Phytoplankton	1.0000		0.1087	0.8000			0.1439	0.5000	1.0000	0.1539												
2	Zooplankton			0.1522			0.0161	0.2009	0.5000		0.0154	0.0061	0.0233	0.2269	0.4129				0.0631		0.0072	0.0568	0.0864
3	Epiphyton		0.1500			0.0366	0.0368				0.4618			0.0422	0.0590								
4	Microphytobenthos			0.3261		0.5843	0.2666	0.1499			0.2309	0.1516		0.0422	0.0590								
5	Fiddler crabs							0.0179				0.0400	0.0500	0.0005		0.4231	0.8170	0.6511	0.4044	0.0100		0.2498	0.1640
6	Polychaetes							0.0050				0.2600		0.4600	0.2000				0.0073				
7	Bivalves		0.0291				0.1558	0.0270					0.1810					0.0005		0.0100	0.0521	0.0145	0.0200
8	Gastropods			0.0109		0.0094	0.0644	0.0454				0.0600	0.2277	0.0219	0.0012	0.0020	0.0015	0.0227	0.0005	0.0100	0.0500	0.0285	0.0062
9	Blue crab											0.0172	0.1922	0.0844		0.1759	0.0030	0.0424	0.0506			0.1566	0.0330
10	Shrimp						0.0052					0.4171			0.0590	0.3602	0.0006	0.0500	0.4560	0.8837	0.0780	0.1000	0.3926
11	Herring															0.0070	0.0008			0.0300		0.0310	0.0042
12	Anchovies																			0.0060		0.0023	
13	Mullet																					0.0250	0.0002
14	Flatfish																			0.0030		0.0550	0.0722
15	Puffer																	0.0493		0.0030		0.0170	0.0650
16	*Eucinostomus* spp																			0.0030		0.0019	0.0001
17	*Diapterus* spp															0.0089						0.0470	0.0200
18	Snappers																						0.0010
19	*Sciades herzbergii*															0.0005						0.0430	
20	Others Catfish															0.0005						0.0450	
21	Drum															0.0049				0.0013		0.0065	0.0150
22	Grunt																					0.0042	0.0220
23	Croaker																					0.0050	0.0100
24	Snook																						
25	Jack																					0.0011	
27	Detritus		0.8121	0.4022	0.2000	0.3698	0.4551	0.4100			0.1381	0.0480	0.3259	0.1220	0.2090	0.0170	0.1770	0.1841	0.0180	0.0400	0.8127	0.0890	0.0450
28	Import																					0.0210	0.0430
Total		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Food web structure and trophic analysis

Trophic structure

Trophic level (TL) varied from 1 (Primary production groups) to 3.28 (Jack) (Table 2), with mean (excluding primary production) of 2.57 (Fig. 4). Carnivorous fish such as Jack (3.28), Grunt (3.24), Snook (3.20), Snappers (3.16) and Drum (3.14) occupied the top trophic level. Except for primary producers and detritus, almost all the groups showed TL > 1 < 3 in the SRE ecosystem.

Although top predators (Snook and Jack) have a larger number of paths, their omnivory index (OI) values were lower than the observed for Diapterus and Puffer (TL = 2.72) showing that omnivory could happen with a fewer number of paths and in a lower trophic level.

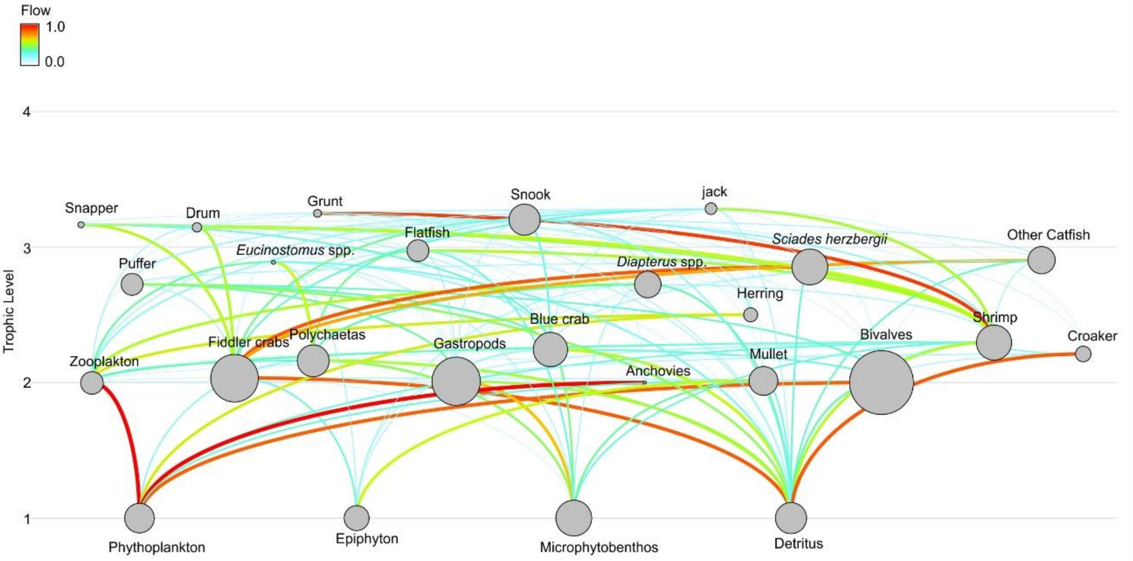

Figure 4. Schematic diagram of energy flow representing the food web structure of the Estuary of Sirinhaém River, Northeastern Brazil. The color lines denote energetic flow from are group to another (Red – High flow and White – Low flow). Different sizes of the circles indicate the different biomass (t km−2) of the components in trophic levels 1, 2, 3 and 4.

Transfer efficiencies

The Lindeman spine (Fig. 5) shows discrete trophic levels and highlights that 60.5% of the whole energy and matter (1,450 t km^{-2} y^{-1}) flows through the grazing food chain, while only 39.5% (945.4 t km^{-2} y^{-1}) flows in the detritus-based food chain. The TE was higher in a grazing food chain when compared to the detritus-based food chain (Table 4). Only 6.40% of the PP flowed to the trophic level II, the remaining returned to the detritus. The largest TE was observed between trophic level III and IV (9.82%) while the largest proportion of biomass of the system and contribution to detritus was concentrated in the TL II and TL I (138 t km^{-2} and 1,138 t km^{-2} y^{-1} respectively) (Fig. 5). The mean transfer efficiency was 6.06% from detritus and 7.18% from the primary producers, while the average trophic transfer efficiency for the entire system as a whole was 6.79% (Table 4)

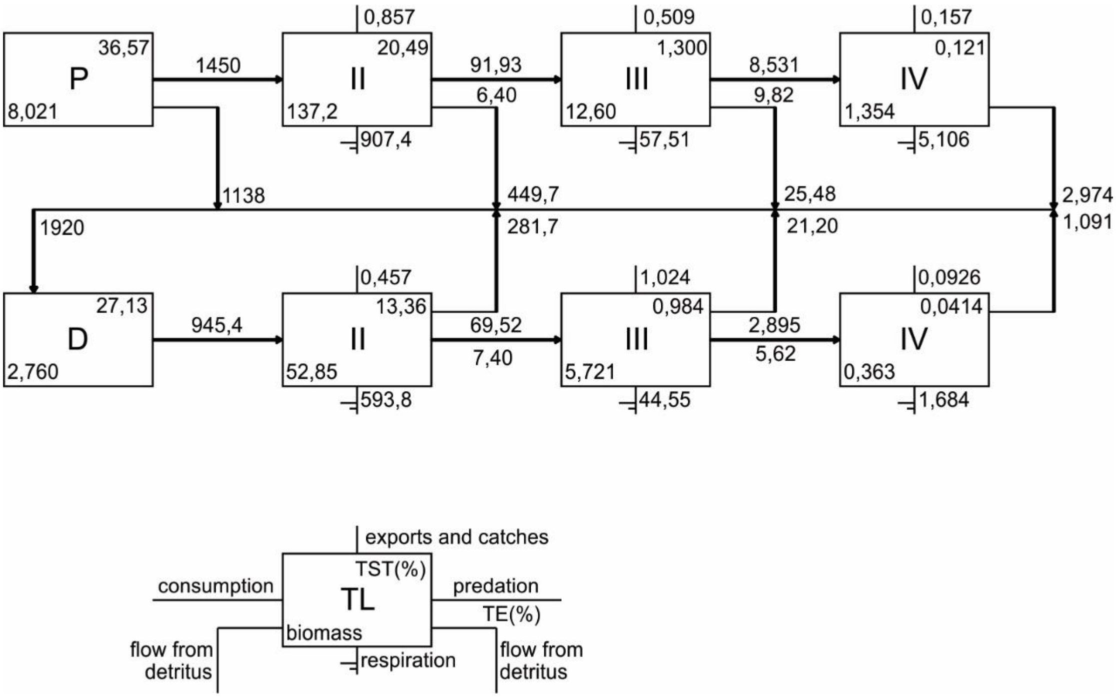

Figure 5. Trophic flows of the Estuary of Sirinhaém River, Northeastern Brazil by integer trophic levels (TL) in the form of the Lindeman spine. P and D represent the primary production and detritus, respectively, while the values in boxes indicate the biomass, import and percentage of total system throughput (TST) for each trophic level (TL). The values above and below arrows exhibit the efficiency of energy transfer (TE) through each trophic level.

Mixed trophic impacts (MTI) and Keystone species

The result of MTI exhibited both direct and indirect impacts of all groups of the system, for example, an increasing of 20% in the Snook biomass would have negative effects (red blocks) upon most of the groups in SRE ecosystem, but positive effects (blue blocks) on Snapper and Polychaetas (Fig. 6). However, fishery showed relatively strong negative effects on Snook and Mullet group (MTI= -0.647 and MTI= -0.325), but was beneficial for most groups, except the Snapper (MTI= -0.629). Because of a high biomass and predation rate, Bivalves and Gastropods presented direct negative impact in Phytoplankton (MTI= -0.437) and Microphytobenthos (MTI= -0.454) respectively. Other relevant impacts are caused by high predation rates. However, the highest negative or positive effect was observed in Snapper group as a result of the changes in Jack biomass (e.g. increase or decrease biomass of the Jack group – negative effect (MTI= -0.974) and positive effect (MTI= 0.913) in biomass of the Snapper, respectively) (Fig. 6).

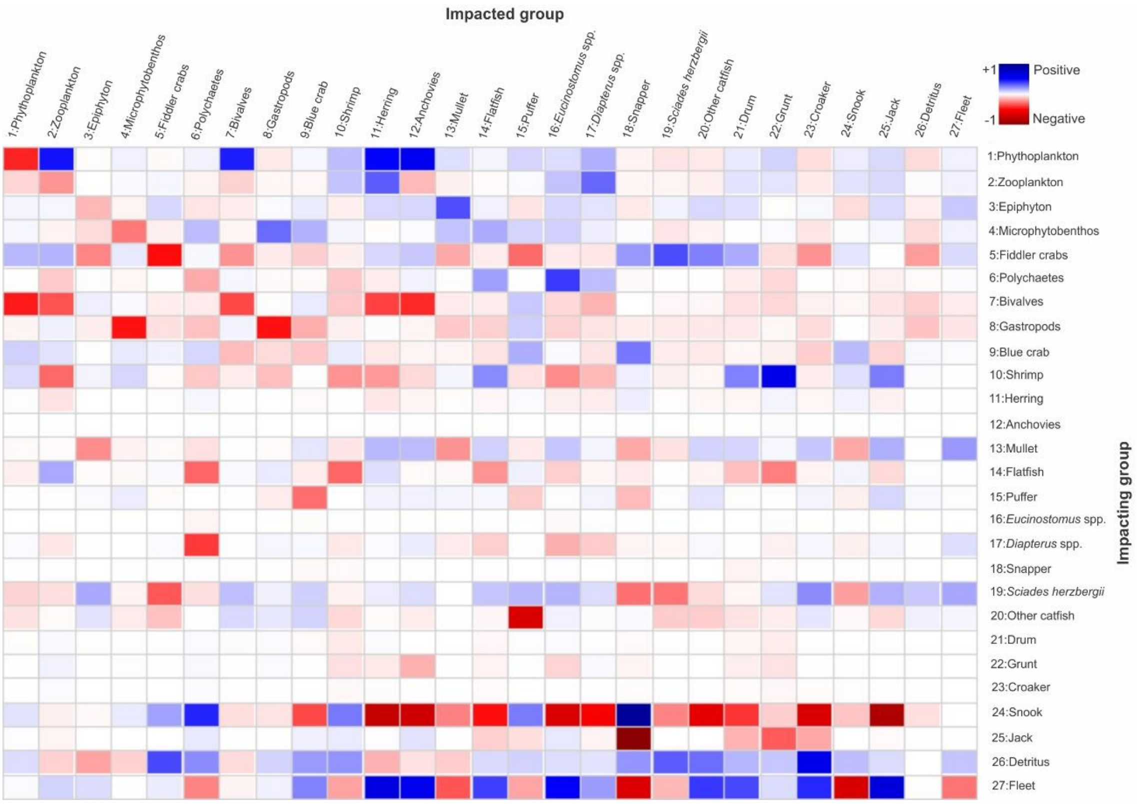

Figure 6. Mixed trophic impact (MTI) of the Estuary of Sirinhaém River, Northeastern Brazil. The color box shows negative (red) or positive (blue) impacts on the functional groups. The intensity of the color in the box are proportional to the degree of the impacts and its values range from −1 to +1.

Two groups of top predator show high KS3 rank values, Jack (KSi= 1.32; TL = 3.28) and Snook (KSi= 1.29; TL= 3.20), followed by the Phytoplankton (KSi= 0.99; TL= 1) and Flatfish (KSi= 0.94; TL= 2.97) (Fig. 7a). High values of Top-down effect (td) were associated with the Snook (td= 75%) and Jack (td= 99%), while Shrimp and Phytoplankton showed larger values of bottom-up effect (bu), (bu= 65% and bu= 84%, respectively). Only the Snook showed lower relative biomass and a higher impact in food chain compared to others groups (Fig. 7b). According to the MTI results and using the keystone species index, the Snook can be considered as the most important group in terms of relative total impact in the Estuary of the Sirinhaém River, although, Phytoplankton and Shrimp (Fig. 4) are important vectors for energetic transfer from food chain to top predators.

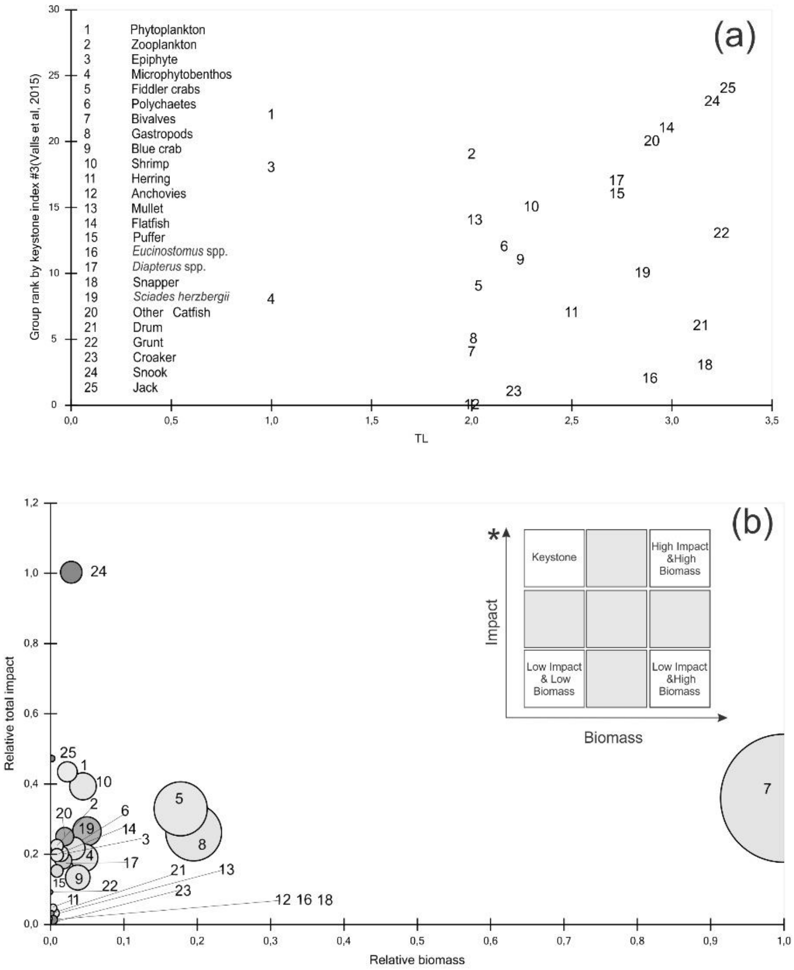

*Figure 7. (a) Keystone index (axis Y) and relative total impact (axis X) of each compartment of the Estuary of Sirinhaém River, Northeastern Brazil. (b) Relationship between relative total impact (axis Y) and relative biomass (axis X). Number and circle size indicates the name and percentage relative biomass of each group. * Conceptual identification of keystone species in food-web (VALLS et al., 2015).*

Ecosystem properties and indicators

The summary statistics and flow indices of the SRE ecosystem is given in Table 4. The total system throughput was 7,083 t km^{-2} y^{-1}, 36% and 23% were due to consumption (2,575 t km^{-2} y^{-1}) and respiratory process (1,610 t km^{-2} y^{-1}), respectively. A total of 14% was derived from export (978 t km^{-2} y^{-1}) and 27% of detritus (1,920 t km^{-2} y^{-1}), as a result of the backflows in the ecosystem. The ratio of total primary production to total respiration (TPP/TR) was 1.6 and the ratio of total primary production to total biomass (TPP/TB) was 16.25. The sum of all production (TP) was 3.038 t km^{-2} y^{-1}, and the total net primary production (TNPP) and the net system production (NSP) were 2.587 t km^{-2} y^{-1} and 977 t km^{-2} y^{-1}, respectively. The mean trophic level of the catch was estimated as 2.66, and the gross efficiency (catch/net primary production) was 0.001 in the SRE ecosystem. Overall, in the SRE ecosystem, the values of flow indices, i.e., CI and SOI, were estimated as 0.245 and 0.12 respectively; while the FCI and Finn’s mean path length (FML) calculated by the model were 5.79% and 2.73, respectively (Table 4). The values of ascendancy and overhead were 27.89% and 72.11% respectively (Table 4).

Analysis of Fisheries

Overall, the scenarios (increase and decrease of the fishing effort), caused a small variation in the different attributes analyzed (Biomass, L index, psust, Kempton’s and TL catch). Compared with the baseline scenario, the total biomass of the groups in the ecosystem showed an increase of 0.07 – 1.04% over scenarios of increased fishing and reduction of 0.06 - 0.53% with the decrease fishing. The biomass of fish was reduced with elevation of fishing; however, invertebrates, primary production (PP) and detritus biomass increased; the opposite was registered with the decrease of the fishing effort (Fig. 8).

The Total Primary Production Required (TPPR) for maintenance of production of the groups in ecosystem ranged from 8,535 to 9,086 t km^{-2} y^{-1}. The relation between TPPR and P1, showed the highest value of 63% into the worst scenario of increased fishing (F0x2), while the lower value was obtained with the simulation without fishing (58%) (Table. 5).

*Table 4. Ecosystem attributes, ecological and flow indicators of the food web structure of the Estuary of Sirinhaém River, Northeastern Brazil. * Rate of the Sum of all consumption, exports, respiratory flows and flows into detritus, in relation the total system throughput.*

Parameters	Value		Units	
Ecossystem properties				
Sum of all consumption (TC)*	2,575.322 (0.364)		t.km-2.y-1	
Sum of all exports (TE)*	978.046 (0.138)		t.km-2.y-1	
Sum of all respiratory flows (TR)*	1,610.220 (0.227)		t.km-2.y-1	
Sum of all flows into detritus (TD)*	1,920.327 (0.271)		t.km-2.y-1	
Total system throughput (TST)	7,083.915		t.km-2.y-1	
Sum of all production (TP)	3,037.931		t.km-2.y-1	
Mean trophic level of the catch (TLc)	2.669		-	
Gross efficiency (catch/net p.p.)	0.001		-	
Calculated total net primary production (TNPP)	2,587.893		t.km-2.y-1	
Net system production (NSP)	977.673		t.km-2.y-1	
Total biomass (excluding detritus) (TB)	159.273		t.km-2	
Total catch (Tc)	3.114		t.km-2.y-1	
Ecossystem maturity				
Total primary production/total respiration (TPP/TR)	1.607		-	
Total primary production/total biomass (TPP/TB)	16.248		-	
Total biomass/total throughput (TB/TST)	0.022		y-1	
Food web structure				
Connectance Index (CI)	0.245		-	
System Omnivory Index (SOI)	0.119		-	
Finn's cycling index (FCI)	5.799		% of total throughput	
Finn's mean path length (FML)	2.737		-	
Ascendancy (A)	27.89		%	
System overhead (O)	72.11		%	
Model reability				
Ecopath pedigree index	0.343		-	
Measure of fit (t*)	1.754		-	
Transfer efficiency				
		TL		
	II	III	IV	V
Producer	6.400	9.823	5.907	4.835
Detritus	7.403	5.624	5.358	4.742
All flows	6.796	8.014	5.767	4.821
Proportion of total flow originating from detritus: 0.42				
Transfer efficiencies (calculated as geometric mean for TL II-IV)				
From primary producers: 7.18%				
From detritus: 6.06%				
Total: 6.79%				

Table 5. Estimates of ecological indicators for maintenance of production of the groups in the Estuary of Sirinhaém River, Northeastern Brazil.

	Present situation	Increased fishing	Decreased fishing
	F0	F0x2	No fishing
P1 = PP + Flow to Detritus	4,508.23	4,522.57	4,493.55
Total Primary Production Required – TPPR (t.km-2.y-1)	7,264.48	7,140.23	7,728.39
P1/TPPR (%)	62	63.3	58.1
L' index	0.0054	0.01357	-
Fishery sustainability – psust	0.94	0.867	-

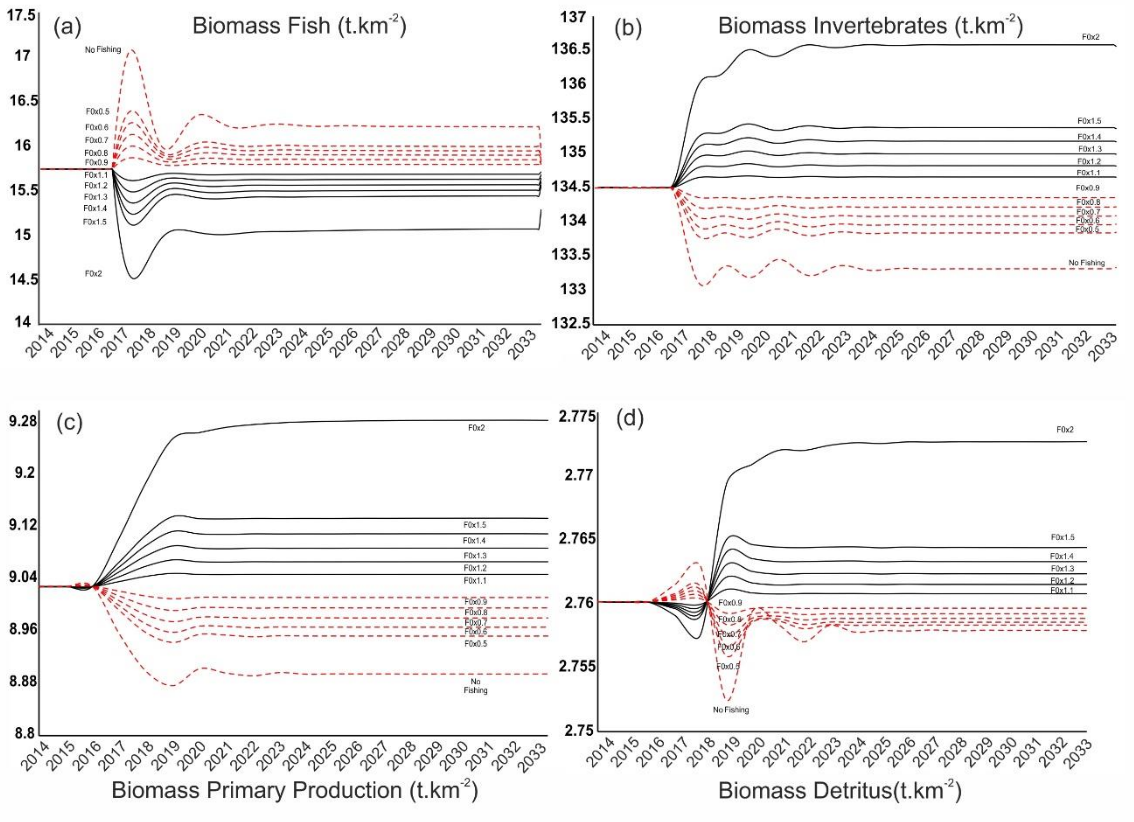

Figure 8. Changes in the biomass from Ecosim model, for different organisms (a. Fish, b. Invertebrates, c. Primary production – PP and d. Detritus) between 2014 and 2033 of the Estuary of Sirinhaém River, Northeastern Brazil. The black solid line and red dash line correspond to the simulations of increase and decreased of the fishing mortality, respectively.

The increased fishing mortality in the simulations raised the L index values, but even with the increased L index values, the fishery remains with sustainable levels, i.e, according to the index P_{sust} (Fig. 9a). When the sustainability of fishery for each target resource of the fishery is analyzed, the mullet group showed the higher reduction of probability to be sustainably fished with the increased fishing mortality, decreasing below of 70%, being the threshold of sustainability of fishery (psust = 75%). In other groups, the (psust) values remained over than 75%, with the lowest recorded for the Snook group (psust = 86%) (Fig 9b).

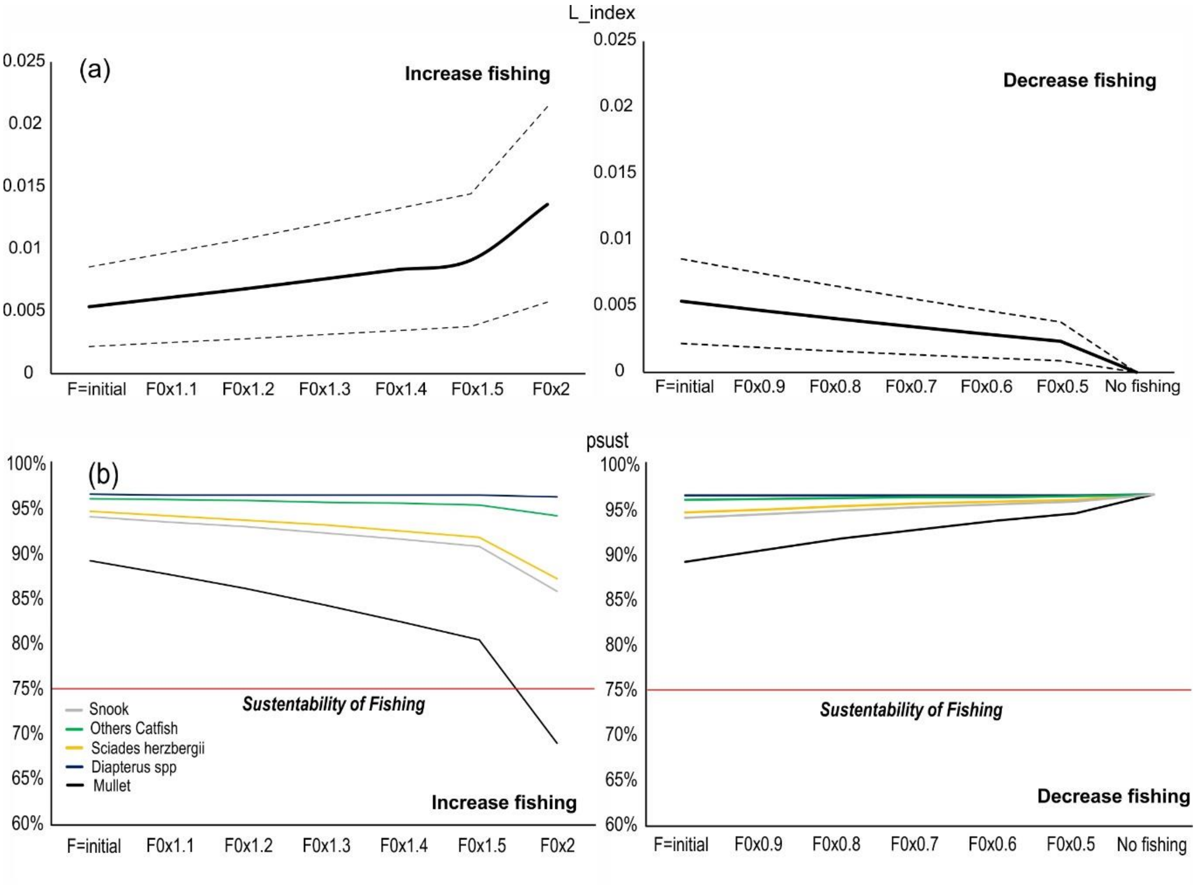

Figure 9. Ecological indicators (a) L index and (b) estimated probability to be sustainably fished (psust) based on the increase and decrease values of fishing mortality of the Estuary of Sirinhaém River, Northeastern Brazil.

The biodiversity indicator, the Kempton’s Q index, showed increase and decrease in response to increase and reduction of fishing mortality respectively (Fig. 10a), indicating that fishing activity improved the biodiversity evenness in ecosystem. It was observed a significant elevation or reduction of Q index after the initial periods with the fishing constant, in the simulations of increase and decrease fishing mortality respectively, followed of a stabilizing until the end of the simulation period (Fig. 10a). The lower level of evenness is attained when fishing mortality is reduced to 0 (No fishing), indicating the importance of these resources into the control of the food web, mainly the snook group, considered as a key species in the system.

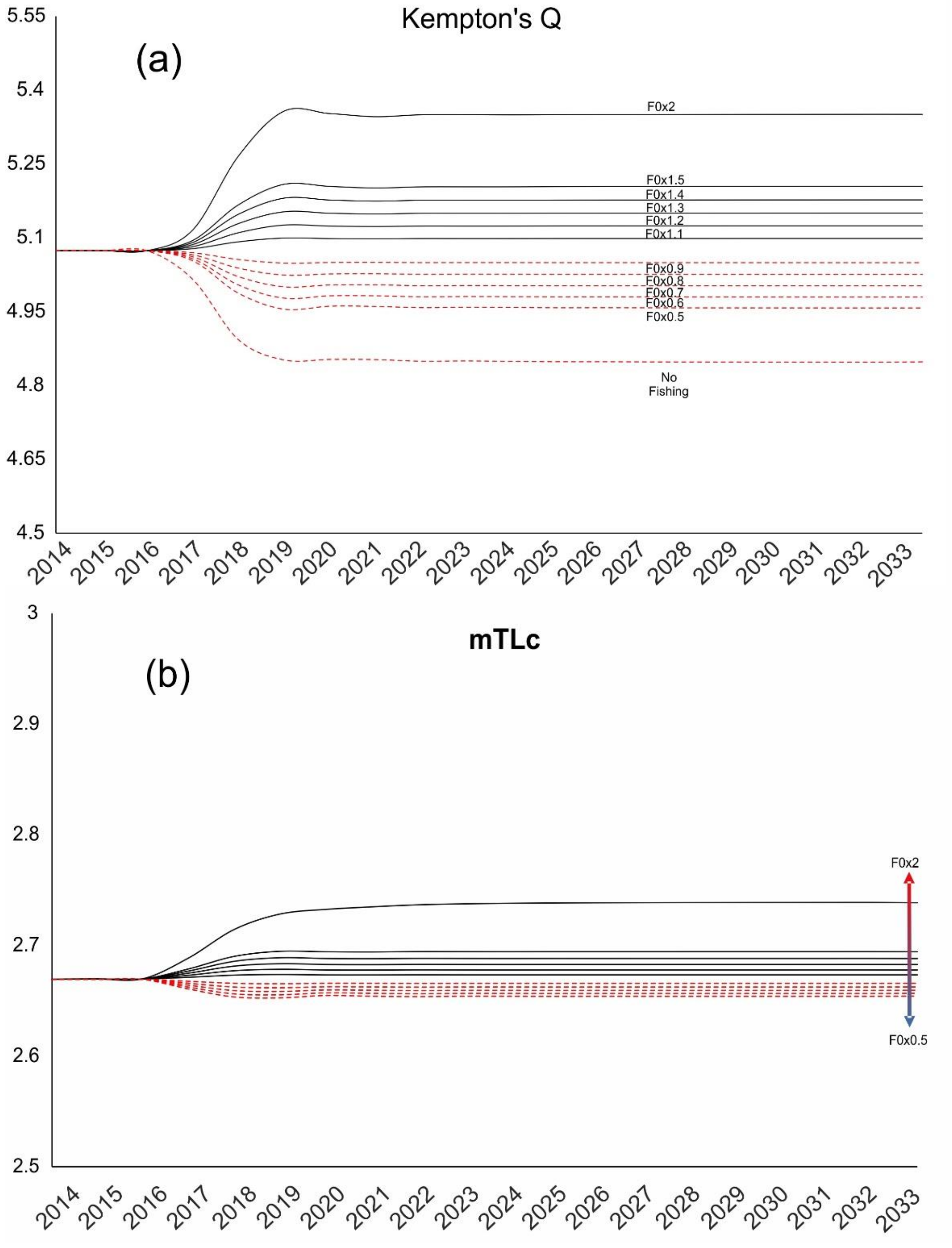

Figure 10. (a) Kempton's index and (b) mean trophic level catch (mTLc) estimated from variation of the values of fishing mortality between 2014 and 2033 of the Estuary of Sirinhaém River, Northeastern Brazil. The black solid line and red dash line correspond to the simulations of increased and decreased fishing mortality, respectively.

There was a general increasing trend of mTLc with Q index over time with the increasing of the fishery and a decreasing trend in response to the reduction of the fishing mortality (Fig. 10b). In general, the simulations with high fishing pressure resulted in the increased of removal of the species/group of low trophic level (mullets (TL: 2.01)), which probably would cause a reduction in mTLc. However, the increase of the fishing mortality, also caused the increase of catch of the species/group of high trophic level (e.g. Snook (TL: 3.20); *Sciades herzbergii* (TL: 2.85) and Others Catfish (TL: 2.90)), elevate the mTLc (Fig. 10b).

Discussion

Food web structure in SRE ecosystem

Estuarine ecosystems are classified as ideal habitats in tropical, subtropical and temperate regions for numerous marine and freshwater resources, constituting an essential system (PREMCHAROEN and PATHOM, 2014; COUILLARD et al., 2017); but also susceptible to a number of anthropogenic interference, for example, through industrial activities and fishing (BLABER, 2011; VIANA et al., 2012; ECOUTIN et al., 2014), which may cause changes in the food chain (BLABER, 2013).

In this study, it was developed a trophic model for a tropical estuary in Brazil. Estuarine trophic models are extremely rare in the tropics. This is mainly due to the absence of historical data and/or basic information on ecology and biology of the ecosystems and living resources, which is the case of Brazil (ANGELINI e GOMES, 2008). This model has strong and weak points. An intensive data collection of the main consumers was carried out. The diet of the main consumers was also obtained for the studied area. These were the main strong points. However, other groups, such as the primary production and zooplankton, were obtained from the literature. Yet, the pedigree value, which evaluate the robustness of the model, was within the a range established by Morissette et al. (2006), who evaluated the pedigree values of over 150 Ecopath models published (values between 0.164 and 0.675). The application of the catchability model was also a weak point of our model. Catchability, which is a key parameter in fishery stock assessment, is a concept in fishery biology which reflects the efficiency of a particular fishery. Its quantitative magnitude is expressed by the catchability coefficient, which relates the biomass abundance to the capture or fishing mortality (ARREGUIN-SANCHEZ, 1996). This parameter is used given the problematic of the underestimation of fish biomass due to gear selectivity in the Ecopath models (SIMON and RAFFAELLI, 2016; BENTORCHA et al., 2017). Some authors have chosen to, in the absence of catchability coefficients to correct biomass estimates, attributing maximum values to ensure mass-balance (CORRALES et al., 2017). In our study, we used the approach proposed by Lauretta et al. (2013). Given the absence of some information, we adopted the coefficient considering the similarity of our species with those of the author, considering the alike genus, body and/or fin format. Nevertheless, the occasional misuse of this parameters (or the decision of not using it) may cause the sub or overestimation of the biomass of some compartments. Finally, as a transitory habitat, estuaries are considered as open environments, and subject to variations of others habitats through recruitment, and a complex trophic structure, including apex predators (MAHONEY and BISHOP, 2017). Hence, given the difficulty of estimating the known connectivity amongst coastal areas, the assumption that there was no net migration with the model is here assumed as for other estuarine ECOPATH models (PATRÍCIO and MARQUES, 2006; HAN et al., 2016). Inclusion of a biomass accumulation factor and migration factor in the general Ecopath equation distinguishes Ecopath modelling as an ‘energy continuity’ approach rather than a strictly ‘steady-state’ approach (PATRÍCIO and MARQUES, 2006).

Most of the biomass in SRE was derived from invertebrates, followed by the fishes, mainly Mullets, Catfishes and Snook that showed higher biomass. The high biomass of invertebrates in tropical systems, particularly in estuaries, are important components as supporters of food web, providing links between primary producers and higher trophic levels (MCQUAID and GRIFFITHS, 2014; SHEAVES et al., 2016), while the relatively high biomass of the top predators can produce a predation pressure on forage species through the top-down control in the food web (WASSERMAN et al., 2013; DU et al., 2015).

The trophic level value of groups on SRE was lower when compared to those observed by Villanueva (2015) in estuarine ecosystems of the Senegal in the African. Dimension of the habitat, evenness, changes in the flux of nutrients, fishery and diet are only a few of the aspects that influence the estimates of the trophic level (PASQUAUD et al., 2010; XIA and TABETA, 2016). The omnivory index (OI) was low compared to that obtained in other estuarine systems (TECCHIO et al., 2015); indicating prey specialization for the majority of the groups. Usually, in tropical environments (e.g. estuaries), most of the fish species tend to be generalists or opportunists (KROETZ et al., 2016; PEREIRA et al., 2017), a consequence of the high biodiversity in these systems (PEREIRA et al., 2012). Low values of Omnivory Index (OI) may be an effect of a reduction of the trophic niche for functional groups with lower or similar TLs, both consequently causing reduction of the OI of their predators. The diversity of prey can be another important factor for the low values of OI. Sirinhaém showed lowest levels of functional diversity and species richness when compared to others estuarine zones of Pernambuco, Northeastern Brazil, which could be associated to the local environmental geomorphology (SILVA-JÚNIOR et al., 2016). This region has a large inner region and a narrow outermost area (SILVA et al., 2011); that can influence the flux of water, transport of larvae, nutrients and organisms between the coast and the estuary (LAYMAN et al., 2007), contributing for the lowest levels of functional diversity and species richness.

The % Total System Throughput-TST was high in TL I, where most is due to PP. The mean transfer efficiency (TE: 6.79) was relatively low compared to the theoretical value of 10% assumed by Lindeman (1942), indicating low global transfer efficiency. However, it is within the range reported in various estuarine trophic models (DE MUTSERT et al., 2012; LERCARI et al., 2015; ABDUL and ADEKOYA, 2016; HAN et al., 2016). Phytoplankton, Macroalgae, Microphytobenthos, Periphyton, and Seagrasses can be highly productive, and an important source of carbon for estuarine ecosystems (HYNDES et al., 2014). In our study, the larger contribution of PP to the food web is related to the importance of PP for secondary consumers as Bivalves, Fiddler crabs and Gastropods. The growth rate of primary production is determined by temperature, nutrient concentrations and light availability, and the rate of biomass change is determined by the balance between rates of growth and mortality including consumption by grazers as the bivalves, considered strong regulators of biomass primary production (CLOERN et al., 2014).

MTI analysis showed functional groups which were impacted negatively by Snook, and positively by fishery activity. Negative impacts can be associated to prevailing top-down effects whereas the positive ones can be associated to bottom-up effects (CHEN et al., 2015). The fishery activity causes a reduction in Snook biomass and consequently decrease the pressure of predation on their prey. Based in KS3 of the Ecopath, it was possible to observe that the Snook is a keystone species in SRE, presenting a high impact in the food web. Keystone species have strong influence on the abundance of other species and ecosystem dynamics (LIBRALATO et al., 2006; VALLS et al., 2015). Snooks are considered an important resource in SRE, both biologically (MERIGOT et al., 2016; SILVA-JÚNIOR et al., 2016) and economically (IBAMA, 2008). In a top-down control system, the top predators determine the bulk of the lower TLs through direct and indirect effects, as the predation and changes in their biomass, respectively (DINEEN and

ROBERTSON, 2010; TESTA et al., 2016). It was observed that some groups had a negative impact on themselves (e.g. Fiddler crabs and Gastropods), reflection of increasing cannibalism and competition for resources (COLL et al., 2006b; CHRISTENSEN et al., 2008).

Analysis of fisheries in the ecosystem

The scarcity of local information affects not only the region of SRE, but also all Brazilian territory and other underdeveloped countries, for instance Senegal (COLLÉTER et al., 2012). Other limitations include the lack of time-series of landing data, which hamper the calibration of the ecosystem dynamics, representing a limitation to the development of the Ecosim (HEYMANS et al., 2016). Thus, our simulations need to be considered with caution and as a first approximation of the effects of the changes of the fishery on the food web, making necessary complementary studies to analyze more precisely the effect of the fishing in ecosystem.

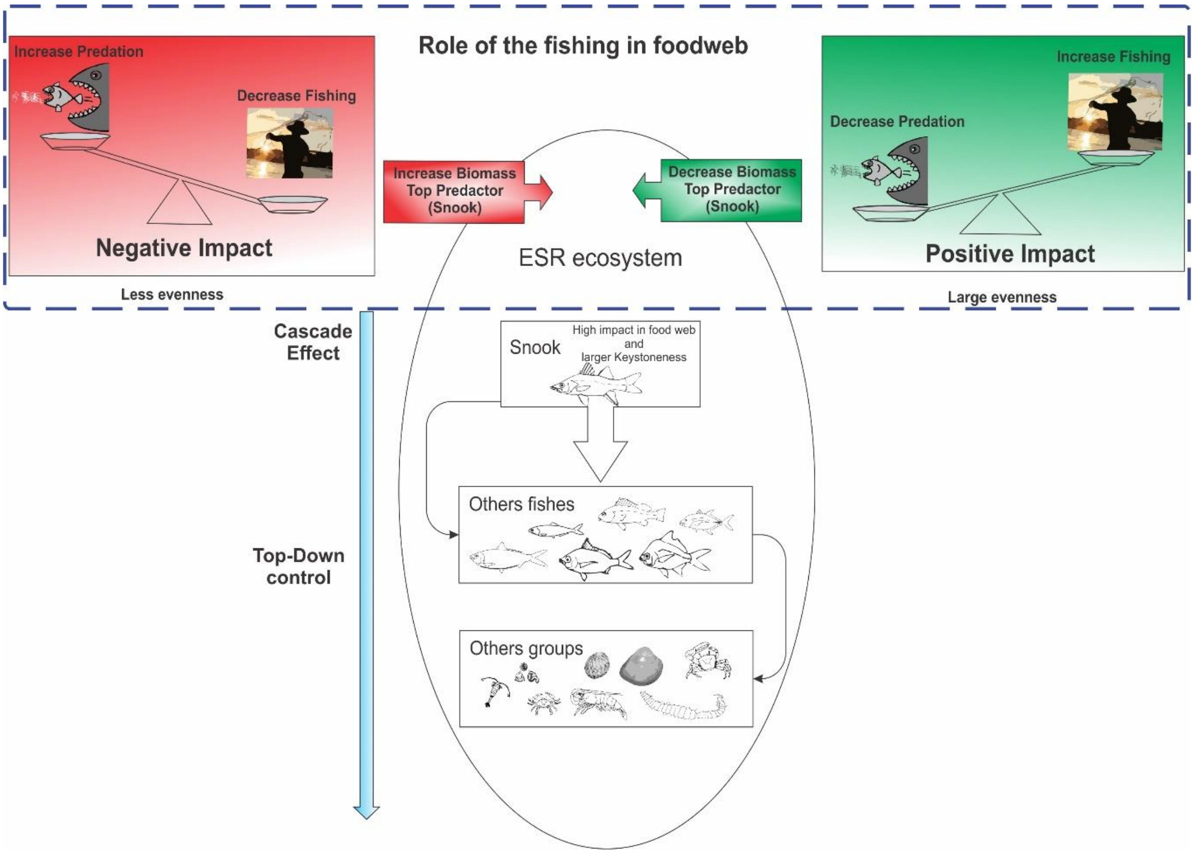

Figure 11. Conceptual model of the fishing effect in the Estuary of Sirinhaém River, Northeastern Brazil.

Compared with the basic model, the total primary production required (TPPR) to sustain the biomass production of the food web organisms, reduces with the increase of fishing, and increase with the decrease of fishing. This variation is the response of changes into the total biomass of fish in the ecosystem, for example, the higher biomass production of fish (decrease of fishing), require a higher TPPR. The Total Primary Production Required (TPPR) can return to detritus and enhance the recycling caused by the predatory species in the higher TLs, which results in values of TPPR that almost twice are higher than the P1 (Calculated Net Primary Production + Flows to Detritus) (ANGELINI et al., 2013). The reduction of TPPR can suggest a more efficient cycling of resources and a healthier state of the system, which can be associated with lower costs to ecosystem maintenance (HORNBORG et al., 2013).

A meta-analysis of the 51 classified models was conducted in order to develop the Psust index (LIBRALATO et al., 2008), a probability measure of sustainability, used in our study. Our results, shows that, in general, the increase of the effect of fishing in the simulations resulted in an increase of the mean L index values, nevertheless, the highest values of the L index that were obtained, were not sufficient to modify the probability of sustainable fishery (P_{sust}). When each resource caught by the fishery activity is observed, the mullets showed values below sustainability. The increase of fishing intensity over time decrease their probability of being sustainably fished (HEYMANS et al., 2014). In terms of diagnostic of a changing intensity of the fishery, we may considerer this results with caution. This index may be used with different data: (1) mass-balance models, (2) calibrated dynamic ecosystem models, and (3) landings data sets. In the case of (1), which is our study case, the Lindex and psust were designed to be estimated for models not used for defining reference values and results should be evaluated in terms of coherence with partial information available (LIBRALATO et al., 2008). Inconsistencies which may emerge between available information and the results of this index may be attributed by the high degree of uncertainty embedded in the reconstruction of historical ecosystems (HEYMANS, 2003). Hence, given the fragility of the estuarine environment and the known impacts of the ESR, information regarding others sources and models, should be considered in order to corroborate with the results here observed.

The Kempton index and mean trophic level catch increased and decreased in response to the increase and reduction of fishing mortality respectively, indicating that the fishery has an important role on evenness of the ecosystem. This results are distinct from those found by Wang et al. (2016) in Pearl River Estuary - China; Goldsworthy et al. (2013) in the Great Australian Bight - Australia; Lynam and Mackinson (2015) in North Sea and Ainsworth et al. (2011) in Northeast Pacific. Various factors can influence the increase of the mean trophic level catches (mTLc), as for example, the increase of the biomass of high trophic level stocks, the change in the fishing fleet (e.g. expansion of the fishing area) (KLEISNER et al., 2014), while the decrease in mTLc can be caused by an increase of the contribution of low-trophic-level species to landings rather than depletion of top predators (ESSINGTON et al., 2006). Our results can be explained by the little presence of species with key role in system (only snook), which implies in a low evenness and higher snook dominance. Therefore, the snook reduction due to fishing causes an increase of Kempton index and consequently a higher evenness in the ecosystem.

Ecosystem health and comparison with other estuarine systems

Based on ecological theories, Odum (1969) attested that the system omnivory index (SOI) and connectance index (CI), total system throughput (TST), ascendency (A), relative ascendency (A/C), System overhead (O), Finn’s cycling index (FCI), the ratio of total primary production to respiration (TPP/TR) and total primary production to biomass (TPP/B) are important indicators to measure the complexity, stability and maturity of the system (VASCONCELLOS et al., 1997; TOMCZAK et al., 2013; HEYMANS et al., 2014). The mature ecosystem tends to have TPP/TR of nearly 1 and a low TPP/TB value (CHEA et al., 2016).

In this study, the ecological indicators which evaluate the state of the ecosystem were, in general, similar the other tropical ecosystems (see Tables 4 and 6), with some key differences. For example, the TPP/TB and overhead in our model was high, while FCI value was relatively low. The higher values of TPP/TB for the SRE are associated principally with low total biomass in the system compared to other systems. The high predatory cycling index value of the overhead, suggests a trend towards a more mature ecosystem and with a larger capacity of resilience (FINN, 1976; HEYMANS and BAIRD, 2000; GUO et al., 2013). The high overhead value in SRE, together with the different ecological indicators analysed, indicated that, SRE is in process of development with a good degree of stability. Estuaries are dynamic ecosystems (MEIRE et al., 2005) characterized by integrating marine, coastal and fluvial systems through water flows, sediment, dissolved substances and organisms, constituting a socio-ecological system (NICOLODI et al., 2009), with ecosystem services (BOEREMA and MEIRE, 2016).

Even within a single habitat or ecosystem unit, the concept of food webs implies in connectivity (SHEAVES, 2009). Because of the high dynamic of ecosystem and the constant environmental changes, most estuaries are considered as immature systems, i.e, in state of development (TECCHIO et al., 2015) and requiring management strategies to the maintenance for the equilibrium state (PALLERO FLORES et al., 2017).

Consumption, exports, respiratory flows, detritus flows, CI and SOI rate also were very similar amongst estuarine ecosystems (see table 4 and 6), even though these indicators are influenced by the size of the system. CI and SOI are the important indices used to describe the food web feature, high values reflecting the high diversity of diet composition while low values indicating a linear food web pattern rather than a web-like structure (HEYMANS et al., 2004; CHEA et al., 2016). TST total sum of flows within the system are here considered as an indicator of ecosystem size (RUIZ et al., 2016), therefore having positive relation with system size. Tropical estuaries and brackish ecosystems are constantly exposed to both long-term trends and rapid environmental changes (VILLANUEVA, 2015). These system are characterized by complex food webs (SCHARLER and BAIRD, 2005; BIRD et al., 2016). The levels of organization of the fish structure in these ecosystems, can also be complex due to seasonal variations, which can cause prey limitations and modification in the use of the habitat (CLOERN and JASSBY, 2012; MORRIS et al., 2015).

In our study case, the SRE shows a typical trophic structure of tropical estuarine system, with a large contribution of phytoplankton to the food web. Mixed Trophic Impact (MTI) analysis, Keystoneness index and the fishing showed that the snook cause a cascade effect (Top-Down), indicating that this species is a keystone species. Snook has a high socio-economic relevance for the region, although with an unregulated fishery in the region. The SRE is subject to a high impact mainly by sugar cane and other agribusiness industries, regardless of begin is located within two Marine Protected Areas (MPA), but with no management plan. Protected Areas are considered one of the best alternative to conservation of the ecosystem, protecting a representative portion of the environment through spatial closures to extractive practices such as the fisheries (MORMEDE et al., 2017). However, its effectiveness may be hampered if there is a lack of dialogue and communication between local and scientific knowledge to the decision-marking (GERHARDINGER et al., 2009), ending up with a MPA without management plan, and hence without effectiveness.

This study is an important contribution to trophic modelling in tropical estuarine ecosystem, being the starting point for others studies of Ecosystem Approach to Fisheries (EAF) in the region, based in an existing model. Ecopath with Ecosim (EwE) was among the first ecosystem-level simulation model to be freely accessible contributing to its worldwide uptake and popularity as a key tool for the ecosystem-approach to fisheries and marine resources (VILLASANTE et al., 2016). This approach not only requires a thorough understanding of the impact of fishing on ecosystem functioning and of the ecological processes involved, but also quantitative tools such as ecosystem models to provide useful information and predictions in support of management decision (OLIVEROS RAMOS et al., 2017). Also, it is widely considered to be a strategy for achieving sustainable delivery of marine ecosystem services (FRANCIS et al., 2011).

The present study is a first trophic model of the region and should be improved with complementary and more accurate information, which can help to understand the effect of fishery on the environments with more precision. Also, incorporating additional tools to the current model, such as Ecospace (ABDOU et al., 2016), which allows an spatial evaluation of the model and, more recently, “Value-Chain”, an economical “chain” of the resources (HALOUANI et al., 2016b), would allow useful insights of the effects of various management policies and the possible trade-offs at the ecosystem level.

Table 6. Ecological indicators of ecosystem health, in different tropical estuarine systems. For comparison of the different models, Sum of all consumption, exports, respiratory flows and flows into detritus were divided for Total system throughput.

Attributes	Mamanguape Estuary[1]	Caete Estuary[2]	Sine-Saloum estuary[3]	Gambia River estuary[4]	Cameroon Estuary[5]	Ogun State coastal estuary[6]	Bolong de Bambuong[7]	
Coordinates	06º46’20”S 34º56’00”W	0°58'18"S 46°56'52"W	13°59'57"N 16°37'5"W	13°46'20"N 16°05'05"W	3°54'53"N 8°34'26"E	6°29'09"N 4°06'28"E	13°39'34"N 16°31'23"E	
Geographic zone	Tropical	Tropical	Tropical	Tropical	Tropical	Tropical	Tropical	
Size (Km^2)	-	220	543	654	1,750	26	4.7	
Depth (m)	4-8	<10	10-25	3-15	50	-	0-15	
Temperature (°C)	28	25.5-26.7	25	27	20.5	-	-	
Salinity (PSU)	1-37	-	45-130	38-45	-	-	36-140	
River flow range ($m^{-3}.s^{-1}$)	-	-	-	4.5-1500	-	-	-	
Number of groups	24	19	37	41	26	14	31	
Main Objective	Evaluate maturity and keystones species in ecossystem, and the relation between estuarine and reef zones, and possibles environmental impacts	Holistic picture of the Caete mangrove ecosystem	Describe the system structure and functioning of ecossystem (less exploited)	Describe the system structure and functioning of ecossystem (highly exploited system)	Describe and quantify structure and function of ecosystem, and mangrove biomass changes effects	Provides a snapshot of the estuarine community and mass balance fluxes using the Ecopath model	Assess the impact of a MPA on the entire trophic network using trophodynamic models	
Period of model	2011-2012	1970-1990	1991	2001	2010-2015	-	2003	2006-2008
Ecossystem properties								
Sum of all consumption (TC) - $t.km^{-2}.y^{-1}$	0.409	0.355	0.189	0.509	0.361	0.164	0.331	0.310
Sum of all exports (TE) - t.km-2.y-1	0.051	0.207	0.346	0.032	0.178	0.366	0.188	0.206
Sum of all respiratory flows (TR) - t.km-2.y-1	0.223	0.090	0.077	0.254	0.205	0.069	0.205	0.190
Sum of all flows into detritus (TD) - t.km-2.y-1	0.295	0.348	0.389	0.205	0.257	0.401	0.276	0.295
Total system throughput (TST) - t.km-2.y-1	7,764	10,559	27,938	2,585	18,615	34,385	6,138	5,867
Sum of all production (TP) - t.km-2.y-1	2,806	3,555	13,074	1,036	893	17,110	2,773	2,659
Mean trophic level of the catch (TLc)	2.42	2.08	2.59	3.10	-	-	2.73	3.13

Gross efficiency (catch/net p.p.)	0.003	0.085	0.001	0.004	-	-	-	-
Calculated total net primary production (TNPP) - t.km-2.y-1	2,127	3,134	11,815	658.51	7,105	-	2,411	2,321
Net system production (NSP) - t.km-2.y-1	-	558	9,677	81.54	3,294	12,598	1,151	1,207
Total biomass (excluding detritus) (TB) - t.km-2	4,227	13,132	260.96	53.39	184	181	132	127
Total catch (Tc) - t.km-2.y-1	7.46	268	12.06	3.36	-	-	-	-
Ecossystem maturity								
Total primary production/total respiration (TPP/TR)	1.23	3.31	5.52	1.12	1,865	6.33	1.91	2.08
Total primary production/total biomass (TPP/TB)	0.50	0.24	45.27	13.83	39	82.61	18.29	18.22
Total biomass/total throughput (TB/TST) - y-1	0.56	1.24	0.009	0.021	0.010	0.005	0.021	0.022
Food web structure								
Connectance Index (CI)	0.26	0.23	0.25	0.19	0.30	0.327	0.34	0.34
System Omnivory Index (SOI)	0.13	0.11	0.14	0.15	0.143	0.288	0.15	0.15
Finn’s cycling index (FCI) - % of total throughput	24.80	17.90	2.68	14.99	2.000	1.700	3.54	3.66
Finn’s mean path length (FML)	3.57	3.40	2.36	3.50	1.710	2.290	2.50	2.50
Ascendancy (A) - %	30.80	27.40	37.50	23.10	35	42.30	25.50	27.20
System overhead (O) - %	68.90	69.60	62.50	76.90	65	57.70	74.50	72.80

1- Xavier (2013); 2- Wolff et al. (2000); 3 and 4 - Villanueva (2015); 5 - Simon and Raffaelli (2016); 6 - Abdul and Adekoya (2016); 7 - Colléter et al. (2012).

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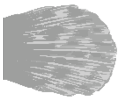
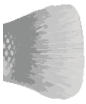
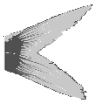
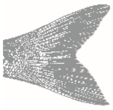
3. Considerações Finais

O Modelo Ecopath do Estuário do Rio Sirinhaém é considerado o primeiro modelo ecossistema estuarina no estado de Pernambuco, se tornando um ponto de partida para outros estudos de estrutura trófica com foco na gestão do ecossistema da região, tendo como base o nosso modelo. O ecossistema Sirinhaém, mostrou uma estrutura trófica semelhante a outros sistemas estuarino tropicais, com uma grande contribuição para a rede trófica vinda do Fitoplâncton. O camurim mostrou-se um componente muito importante no ecossistema, demostrando ter um efeito forte sobre os outros componentes. Este efeito está associado aos impactos provenientes da predação causando o conhecido efeito cascata “Top-Down”. Mesmo considerando que o SRE tem alto impacto principalmente pela cana-de-açúcar e outras indústrias de agronegócio, os índices de maturidade do ecossistema indicaram que o sistema estuarino de Sirinhaém está em desenvolvimento, estável e tem boa capacidade de resistência a perturbações externas, mas necessitando de proteção, pois mesmo estando localizado entre duas áreas marinhas protegidas, ambas não apresentam nenhum plano de manejo. A carência de informações biológicas básicas para região de Sirinhaém foram os principais entraves encontrada para elaboração do modelo, reforçando a necessidade de estudos de base para estruturação do modelo. Entretanto, os critérios de avaliação do modelo foram satisfatórios indicando que o mesmo foi bem-sucedido em sua aplicação. Por fim, estudos futuros são necessários, principalmente aqueles que ajudem a compreender o efeito da pesca no ecossistema de Sirinhaém com maior precisão.

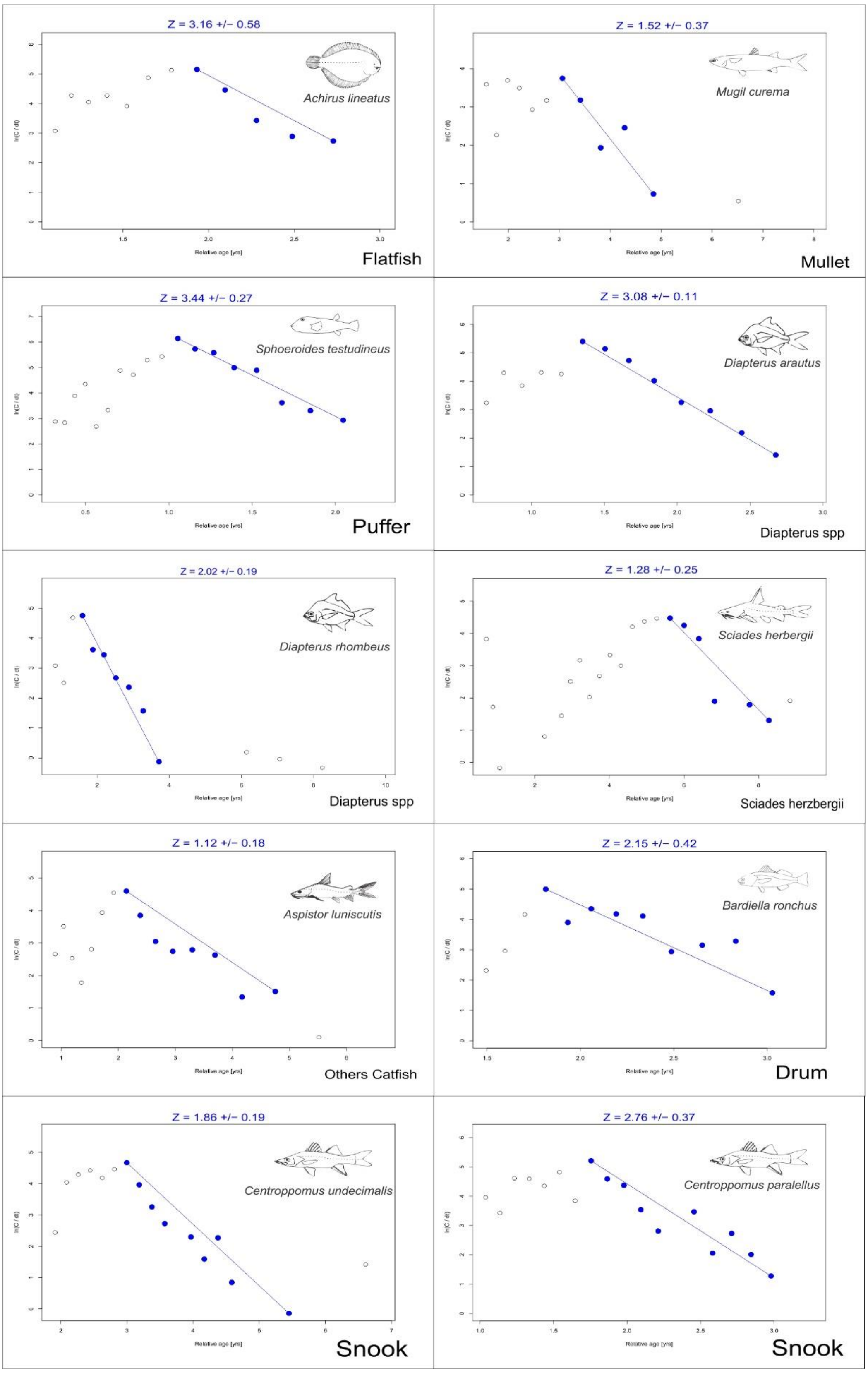
4. Anexos

SOM 1: Biomass (t.km^{-2}) of fish before and after the application of the catchability model (based on Lauretta et al., 2013). q*L* is the selected catchability coefficient for each species based in morphology of the caudal fin. *p* is the mean proportion of the population captured by the fishing gear *p* = (q*L*E)/A*, where *E* is effort (total area dragged), *A* is habitat area.

	B (t.km^{-2}) Before	q*L*	*p*	B (t.km^{-2}) After	Caudal fin form	Similar group in catchability model
Herring	0.01216	0.814	0.037	0.3244	A	*Menidia beryllina*[1]
Anchovies	0.00084	0.814	0.037	0.022	A	*Menidia beryllina*[1]
Flatfish	0.02268	0.540	0.025	0.91222	B	*Trinectes maculatus*[2]
Puffer	0.03144	0.784	0.036	0.8709	C	*Fundulus* spp.[3]
Eucinostomus spp.	0.00089	0.688	0.031	0.02821	D	*Eucinostomus harengulus*[4]
Diapterus spp.	0.05106	0.688	0.031	1.61209	D	*Eucinostomus harengulus*[4]
Snapper	0.00197	0.582	0.027	0.07367	E	*Lagodon rhomboides*[5]
Sciades herzbergii	0.14318	0.688	0.031	4.5735	D	*Eucinostomus harengulus*[4]
Other Catfish	0.05698	0.688	0.031	1.79891	D	*Eucinostomus harengulus*[4]
Drum	0.00401	0.582	0.027	0.14974	D	*Lagodon rhomboides*[5]
Grunt	0.00277	0.582	0.027	0.1173	D	*Lagodon rhomboids*[5]
Croaker	0.01569	0.784	0.036	0.4346	C	*Fundulus* spp.[3]
Snook	0.08108	0.688	0.031	2.5598	D	*Eucinostomus harengulus*[4]
Jack	0.00716	0.688	0.031	0.22606	D	*Eucinostomus harengulus*[4]
Morphology of the caudal fin						
	A	B	C	D	E	

SOM 2: Parameters and references used for estimation of P/B. L_{max} is the maximum length captured of the species (cm); $L\infty$ is the asymptotic length (cm) and k is growth coefficient. * $L\infty$ and k were estimated and based on empirical relationships; $logL\infty=0.044+0.9841*log(Lmax)$ (Froese and Binohlan, 2000) and $k=2.15*L\infty^{-0.46}$ (Le Quesne and Jennings, 2012), respectively.

Group name	L_{max} (cm)	L_∞(cm)	k	Reference
Herring	-	33.7	1.2	Lessa et al. (2008)
Anchovies*	-	15.67	1.05	Souza-Conceicao and Schwingel (2011)
Mullet	-	38.01	0.36	Santana et al. (2009)
Flatfish*	15.3	16.21	0.59	Viana et al. (2016)
Puffer	-	29.5	0.77	Tzeek-Tuz et al. (2012)
Eucinostomus spp	-	28.31	0.61	Silva et al. (2014)
*Diapterus spp**	42.3	44.1	0.24	Elliff et al. (2013)
Snapper	-	77.22	0.11	Rezende and Ferreira (2004)
Sciades herzbergii	-	51.05	0.15	Araújo-Junior et al. (2006)
Others Catfish*	36.5	38.15	0.40	Viana et al. (2016)
Drum*	-	36.6	0.48	Louis (1985)
Grunt*	36.6	38.25	0.40	Cervigón et al. (1992)
Croaker	-	60	0.05	Santos (2015)
Snook	-	140.8	0.07	Mendonça (2004)
Jack*	124	127.1	0.23	Cervigón et al. (1992)



SOM 3: Linearized length converted catch curve to estimate the total mortality (Z ± SE) (CHAPMAN and ROBSON, 1960; PAULY, 1983) for the main compartments caught by local fishery. Source of data: Bioimpact database from 2011 – 2015.

SOM 4: Parameters used as input for the estimation of the annual food consumption/biomass ratio (Q/B) of fish group. W_∞ is the asymptotic weight, obtained from equation $W\infty = a \cdot L_\infty^b$, where "a" is the regression intercept; "b" is the regression slope (see Viana et al., 2016); H and D represent the feeding type (h: 1 and d: 0 for herbivores; h: 0 and d: 1 for detritivores; h: 0 and d: 0 for carnivores); and Ar is aspect ratio of the caudal fin, **Ar = h^2/s, where (h) is height of caudal fin and (s) is the surface area of the caudal fin, extending to the narrowest part of the caudal peduncle (based on Palomares and Pauly, 1998).

Group name	a	b	W_∞ (g)*	H	D	h(mm)	s(mm^2)	Ar**
Herring	0.0081	3.01	321.10	0	0	38.45	362.39	4.1
Anchovies	0.0036	3.30	31.62	1	0	15.06	73.06	3.56
Mullet	0.011	2.98	565.6	1	0	74.97	2030.53	2.77
Flatfish	0.0096	3.27	86.74	0	0	19.58	394.08	1.07
Puffer	0.0213	2.93	431.4	0	0	31.92	608.19	1.77
Eucinostomus spp.	0.008	3.15	299.64	0	0	24.47	327.25	2.16
Diapterus spp.	0.009	3.16	1413.9	0	0	38.75	543.80	2.82
Snapper	0.0156	3	7183.1	0	0	34.71	641.08	1.93
Sciades herzbergii	0.0059	3.11	1209.8	0	0	51.46	1151.53	2.31
Others Catfish	0.004	3.26	572.4	0	0	46.43	838.93	2.71
Drum	0.005	3.33	804.2	0	0	36.47	842.67	1.6
Grunt	0.0096	3.14	894.81	0	0	22.54	378.86	1.34
Croaker	0.0144	3.02	3375.8	0	0	48.04	1611.08	1.5
Snook	0.0083	2.91	14842.6	0	0	57.44	1606.55	2.08
Jack	0.0126	2.97	22370.8	0	0	46.02	669.58	3.33

SOM 5: Input data and references by compartment for the the Estuary of Sirinhaém River, Northeastern Brazil. B: biomass; P/B: production per unit of biomass; Q/B: consumption rate per unit of biomass; EE: ecotrophic efficiency.

Group name		Original value	Reference	Observations
1	Phytoplankton			
	B	9.44 mg.Chl a.m^{-3} (2.22 t. Km^{-2})	Silva (2009)	Conversion of Chl a units for Wet weight (KASPRZAK et al., 2008)
	P/B	19.48 mgC.m^{-3}.h^{-1} (652.71 year^{-1})	Silva (2009)	Conversion of C units for Wet weight (PAULY and CHRISTENSEN, 1995)Considered depth of euphotic zone equal to 0.85 m and 12 hours light per day
	EE		Estimation from ecopath	
2	Zooplankton			
	B		Estimation from ecopath	
	P/B	50.21 year^{-1}	Albouy et al (2010); Angelini and Vaz-Velho (2011); Chea et al (2016); Chen et al (2015); Villanueva (2015)	Data corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	Q/B	150.65 year^{-1}	Albouy et al (2010); Angelini and Vaz-Velho (2011); Chea et al (2016); Chen et al (2015); Villanueva (2015)	Data corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	EE	0.9	Albouy et al (2010); Angelini and Vaz-Velho (2011); Chea et al (2016); Chen et al (2015); Villanueva (2015)	Data corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	Diet		Kleppel et al (1996); Schnetzer and Steinberg (2002); Schwamborn (1997)	
3	Epiphyton			
	B	1.37 t. Km^{-2}	Baltar (1996)	
	P/B	153.31 year^{-1}	Baltar (1996)	
	EE		Estimation from ecopath	
4	Microphytobenthos			
	B		Estimation from ecopath	
	P/B	23.49 gC.m^{-2} (209.61 year^{-1})	Spilmont et al (2009); Underwood and Kromkamp (1999)	Conversion of C units for Wet weight (PAULY and CHRISTENSEN, 1995)

	EE	0.50	Wolff et al (2000)	
5	Fiddler crabs			
	B	16 t. Km^{-2}	Estimates from our samples data	
	P/B	7.3 year^{-1}	Brey (1999); Koch and Wolff (2002); Koch et al (2005)	Estimation of P/B from equation Brey (1999) based in maximum age by species Koch et al (2005) and maximum body mass of the samples data
	Q/B	31.89 year^{-1}	Nichols (1974)	Estimation of Q/B from relationship Nichols (1974) based in our samples data
	EE		Estimation from ecopath	
	Diet		Koch and Wolff (2002); Nordhaus (2004)	
6	Polychaetes			
	B	2.87 t. Km^{-2}	Estimates from our samples data	
	P/B	2.91 year^{-1}	Brey (1999); Otegui et al (2012); Santos (1994); Souza and Borzone (2007)	Estimation of P/B from equation Brey (1999) based in maximum age by species Otegui et al (2012); Santo (1994); Souza and Borzone (2007) and maximum body mass of the samples data
	Q/B	17.26 year^{-1}	Nichols (1974)	Estimation of Q/B from relationship Nichols (1974) based in our samples data
	EE		Estimation from ecopath	
	Diet		Opitz (1996)	
7	Bivalves			
	B	90 t. Km^{-2}	Estimates from our samples data	
	P/B	2 year^{-1}	Opitz (1996)	Data of P/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	Q/B	9 year^{-1}	Opitz (1996)	Data of Q/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	EE		Estimation from ecopath	
	Diet		Resgalla and Piovezan (2009)	

8	Gastropods			
	B	17.6 t. Km^{-2}	Estimates from our samples data	
	P/B	2.65 year^{-1}	Absalao et al (2009); Brey (1999)	Estimation of P/B from equation Brey (1999) based in maximum age by species Absalao et al (2009) and maximum body mass of the samples data
	Q/B	38.83 year^{-1}	Nichols (1974)	Estimation of Q/B from relationship Nichols (1974) based in our samples data
	EE		Estimation from ecopath	
	Diet		Blanco and Scatena (2007); Da Cunha Lana and Guiss (1991); Opitz (1996)	
9	Blue crab			
	B		Estimation from ecopath	
	P/B	2 year^{-1}	Christensen et al (2009); Walters et al (2008)	Data of P/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	Q/B	8 year^{-1}	Christensen et al (2009); Walters et al (2008)	Data of Q/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	EE	0.76	Wolff et al (2000)	
	Diet		Chalegre (2008); Oliveira et al (2006)	
10	Shrimp			
	B		Estimation from ecopath	
	P/B	2.81 year^{-1}	Opitz (1996)	Data of P/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	Q/B	26.9 year^{-1}	Opitz (1996)	Data of Q/B corrected for differences of temperature with the Opitz equation (OPITZ, 1996)
	EE	0.95	Albouy et al (2010); Du et al (2015); Zetina-Rejón et al (2015)	
	Diet		Branco et al (2001); Moriarty and Barclay (1981); Newell et al (1995)	

11	Herring			
	B	0.32 t. Km^{-2}	Estimates from our samples data	
	P/B	1.96 year^{-1}	*	
	Q/B	18.34 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Chaves and Vendel (2008)	
12	Anchovies			
	B	0.02 t. Km^{-2}	Estimates from our samples data	
	P/B	2.23 year^{-1}	*	
	Q/B	90.39 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Gay et al (2002); Krumme et al (2008); Sergipensel et al (1999)	
13	Mullet			
	B		Estimates from ecopath	
	P/B	1.88 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	43.15 year^{-1}	**	
	EE	0.35	Colléter et al (2012) and Simon and Raffaelli (2016)	
	Diet		Vasconcelos Filho et al (2009)	
14	Flatfish			
	B	0.91 t. Km^{-2}	Estimates from our samples data	
	P/B	3.16 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	13.43 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Duarte and Andreata (2003); Vasconcelos Filho et al (2010)	
15	Puffer			
	B	0.87 t. Km^{-2}	Estimates from our samples data	
	P/B	3.17 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)

	Q/B	11.06 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Chi-Espínola and Vega-Cendejas (2013); Santos and Rodriguez (2011); Vasconcelos Filho et al (1998)	
16	*Eucinostomus spp*			
	B	0.03 t. Km^{-2}	Estimates from our samples data	
	P/B	1.33 year^{-1}	*	
	Q/B	12.84 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Denadai et al (2012); Ramos et al (2014); Vasconcelos Filho et al (2010); complementary study Bioimpact	
17	*Diapterus spp*			
	B	1.61 t. Km^{-2}	Estimates from our samples data	
	P/B	2.9 years^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	10.73 years^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Chaves and Otto (1998); Denadai et al (2012) Complementary study Bioimpact	
18	*Snapper*			
	B	0.07 t. Km^{-2}	Estimates from our samples data	
	P/B	0.33 year^{-1}	*	
	Q/B	6.42 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Monteiro et al (2009)	
19	*Sciades herzbergii*			
	B	4.57 t. Km^{-2}	Estimates from our samples data	
	P/B	1.38 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)

	Q/B	9.94 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Giarrizzo and Saint-Paul (2008) and complementary study bioimpact	
20	Others Catfish			
	B	1.8 t. Km^{-2}	Estimates from our samples data	
	P/B	1.13 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	12.5 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Bomfim (2014)	
21	Drum			
	B	0.15 t. Km^{-2}	Estimates from our samples data	
	P/B	1.73 year^{-1}	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	9.43 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Vendel and Chaves (1998); Complementary study Bioimpact	
22	Grunt			
	B	0.12 t. Km^{-2}	Estimates from our samples data	
	P/B	0.93 year^{-1}	*	
	Q/B	8.78 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Complementary study Bioimpact	
23	Croaker			
	B	0.43 t. Km^{-2}	Estimates from our samples data	
	P/B	0.21 year^{-1}	*	
	Q/B	6.90 year^{-1}	**	
	EE		Estimation from ecopath	
	Diet		Bessa et al (2015); Freret and Vanderli (2003)	
24	Snook			

	B	2.57 t. Km^{-2}	Estimates from our samples data	
	P/B	1.68 $year^{-1}$	Z=P/B from Allen (1971)	Estimation of Z from Linearized length converted catch curve (PAULY, 1983)
	Q/B	5.70 $year^{-1}$	**	
	EE		Estimation from ecopath	
	Diet		Complementary study Bioimpact	
25	Jack			
	B	0.23 t. Km^{-2}	Estimates from our samples data	
	P/B	0.46 $year^{-1}$	*	
	Q/B	6.66 $year^{-1}$	**	
	EE		Estimation from ecopath	
	Diet		Silvano (2001)	
26	Detritus	2.76 $year^{-1}$	Pauly et al (1993)	Estimation of biomass from equation Pauly et al. (1993) based on the production primary data

*$M=PB = k^{0.65} \cdot L_{\infty}^{-0.279} \cdot T^{0.463}$ based on the empirical equation of Pauly (1980); **$\log QB = 7.964-0.204 \cdot \log W_{\infty}-1.965 \cdot T'+0.083 \cdot Ar+0.532 \cdot H+0.398 \cdot D$, based on the empirical relationship of Palomares and Pauly (1998)

Conversion: Chl *a*/Wet weight= 0,5% (KASPRZAK et al., 2008); gC to Wet weight= 1:9 (PAULY and CHRISTENSEN, 1995); Wet weigth to kJ= 11:2.5 (MILLS, 1980) ; Dry weight to Wet weight= 1:5 (PARSONS et al., 1977)

SOM 6: Total, mean and biomass of artisanal fishery landings in the Estuary of Sirinhaém River, Northeastern Brazil, from 2000 to 2007 (t) for each compartments.

Groups	2000	2001	2002	2003	2004	2005	2006	2007	/Year	t/Km^2/Year
Snook	4.9	4.9	8	10.35	6.48	5.47	7.17	7.52	6.85	0.7812 [a]
***Sciades herzbergii/* ***Other Catfish*	7.5	8.6	12.4	10.18	7.67	11	16.34	15.95	11.21	1.0651[b]
Diapterus spp.	1.7	2	1.9	1.79	1.52	1.57	2.73	2.37	1.95	0.2961[c]
Mullet	8.5	9.4	9.7	9.9	5.87	7.94	9.7	7.12	8.51	0.9714[d]

Letters indicate the caught percent in estuary: a- 60%; b- 50%; c- 80%; d- 60%.

** The total caught percent in estuary, 84% is *Sciades herzbergii* and 16% is *Others Catfish.*

SOM 7: Input data for two scenarios with 12 simulations that represent the increase and decrease of fishing mortality for all fishing target species in the Estuary of Sirinhaém River, Northeastern Brazil. Where: F is the fishing mortality rate; F0 = initial mortality/cacth in ecopath model; Fx = F0 × f (where f is a multiplier); Catch ($t*Km^{-2}*Year^{-1}$).

		Mullets	*Diapterus* spp.	*S. herzbergii*	Other Catfish	Snook
Increase						
F0	Catch	0.971	0.296	0.897	0.168	0.781
	F0	0.479	0.184	0.196	0.094	0.304
F0x1.1	Catch	1.069	0.326	0.986	0.185	0.859
	F1	0.526	0.202	0.216	0.103	0.334
F0x1.2	Catch	1.166	0.355	1.076	0.202	0.937
	F2	0.574	0.220	0.235	0.112	0.364
F0x1.3	Catch	1.263	0.385	1.166	0.219	1.016
	F3	0.622	0.239	0.255	0.122	0.395
F0x1.4	Catch	1.360	0.415	1.255	0.236	1.094
	F4	0.670	0.257	0.275	0.131	0.425
F0x1.5	Catch	1.457	0.444	1.345	0.253	1.172
	F5	0.718	0.276	0.294	0.140	0.455
F0x2	Catch	1.943	0.592	1.794	0.337	1.562
	F6	0.957	0.367	0.392	0.187	0.607
Decrease						
F0x0.9	Catch	0.8743	0.2666	0.8071	0.1515	0.7031
	F7	0.4307	0.1654	0.1765	0.0842	0.2732
F0x0.8	Catch	0.7771	0.2370	0.7174	0.1347	0.6250
	F8	0.3828	0.1470	0.1569	0.0749	0.2428
F0x0.7	Catch	0.6800	0.2073	0.6277	0.1178	0.5469
	F9	0.3350	0.1286	0.1373	0.0655	0.2125
F0x0.6	Catch	0.5829	0.1777	0.5381	0.1010	0.4687
	F10	0.2871	0.1102	0.1176	0.0562	0.1821
F0x0.5	Catch	0.4857	0.1481	0.4484	0.0842	0.3906
	F11	0.2393	0.0919	0.0980	0.0468	0.1518
F0x0 (Nofishing)	Catch	0	0	0	0	0
	F12	0	0	0	0	0

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